LIMNOLOGICAL CHARACTERISTICS, NUTRIENT LOADING AND LIMITATION, AND POTENTIAL SOURCES OF TASTE AND ODOR PROBLEMS IN STANDLEY LAKE, COLORADO

by David K. Mueller and Barbara C. Ruddy

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CONVERSION FACTORS AND RELATED INFORMATION

Multiply	*By*	*To obtain*
acre	0.4047	hectare
acre-foot (acre-ft)	1,233	cubic meter
centimeter (cm)	2.54	inch
foot (ft)	0.3048	meter
inch (in.)	25.4	millimeter
kilogram (kg)	2.2046	pound, avoirdupois
liter (L)	0.2642	gallon (US)
microgram (µg)	0.00000003527	ounce, avoirdupois
mile (mi)	1.609	kilometer

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

$$°F = 9/5(°C) + 32.$$

Milligrams per liter (mg/L), used in this report for nutrient concentrations, is equivalent to grams per cubic meter and, at typical concentrations in natural waters, to parts per million.

The following terms and abbreviations also are used in this report:
- cubic micrometer per milliliter (μm^3/mL)
- micrograms per liter (µg/L)
- milligrams per kilogram (mg/kg)
- organisms per liter (orgs/L)

National Geodetic Vertical Datum of 1929 (NGVD of 1920): A geodetic datum derived from a general adjustment of the first-order level nets of both the United States and Canada, formerly called "Sea Level Datum of 1929."

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ABSTRACT

Standley Lake is a reservoir that provides water to several suburban cities in Jefferson County, northwest of Denver, Colorado. In 1988, a taste and odor problem developed in water supplied from the lake. In 1989, a study was begun to determine the nutrient availability in the lake, the potential nutrient limitation to algal biomass, and the occurrence and potential sources of compounds that cause taste and odor problems in water withdrawn from the lake. Physical, chemical, and biological water-quality data were collected during June 1989 through October 1990 from sites in the lake, its primary inflows, and its outflow.

Standley Lake was thermally stratified from June through September each year of the study and was well mixed during October-December and March-May. Nutrient concentrations were low in the photic zone. Total nitrogen generally was less than 0.3 milligram per liter and total phosphorus always was less than 0.02 milligram per liter. Concentrations were higher in the hypolimnion, particularly during stratification. Maximum total nitrogen was at least 0.5 milligram per liter during both years of the study, and maximum total phosphorus exceeded 0.05 milligram per liter during both years. Phytoplankton biovolume and chlorophyll *a* concentration were variable throughout the open-water season. The maximum chlorophyll *a* concentration was 6.1 micrograms per liter.

Estimated nutrient loads for October 1989 through September 1990 were 13,400 kilograms of nitrogen and 1,490 kilograms of phosphorus from surface-water inflows and 13,300 kilograms of nitrogen and 687 kilograms of phosphorus from the bottom sediment. Most of the flux from the bottom sediment occurred during stratification. Most of the nitrogen loading from all sources and the phosphorus loading from the bottom sediment was in a dissolved form that is readily available for algal uptake. However, only about 20 percent of the phosphorus in the inflow water was in the available form of orthophosphate.

A series of nutrient-limitation experiments was done to evaluate the algal response to phosphorus and nitrogen additions. Algal response was measured by changes in chlorophyll *a* concentration and phytoplankton biovolume. In every experiment, addition of both nitrogen and phosphorus was required to substantially increase algal biomass.

No taste and odor problems developed in Standley Lake during the study period, and no organic compounds that create earthy or musty odors, such as geosmin or 2-methylisoborneol (MIB), were detected in any water samples from the lake, the inflows, or the outflow. Algae cultured from these water samples failed to produce earthy or musty odors. Few actinomycetes bacteria were detected in water samples, and none were recovered from Standley Lake bottom-sediment samples.

INTRODUCTION

Standley Lake is a reservoir[1] that provides domestic water to the suburban cities of Northglenn, Thornton, and Westminster in Jefferson County, northwest of Denver, Colo. (fig. 1). In 1988, a taste and odor problem developed in water

[1]Although it is a reservoir, the word lake is used in its name. For consistency, it is referred to as a lake throughout this report.

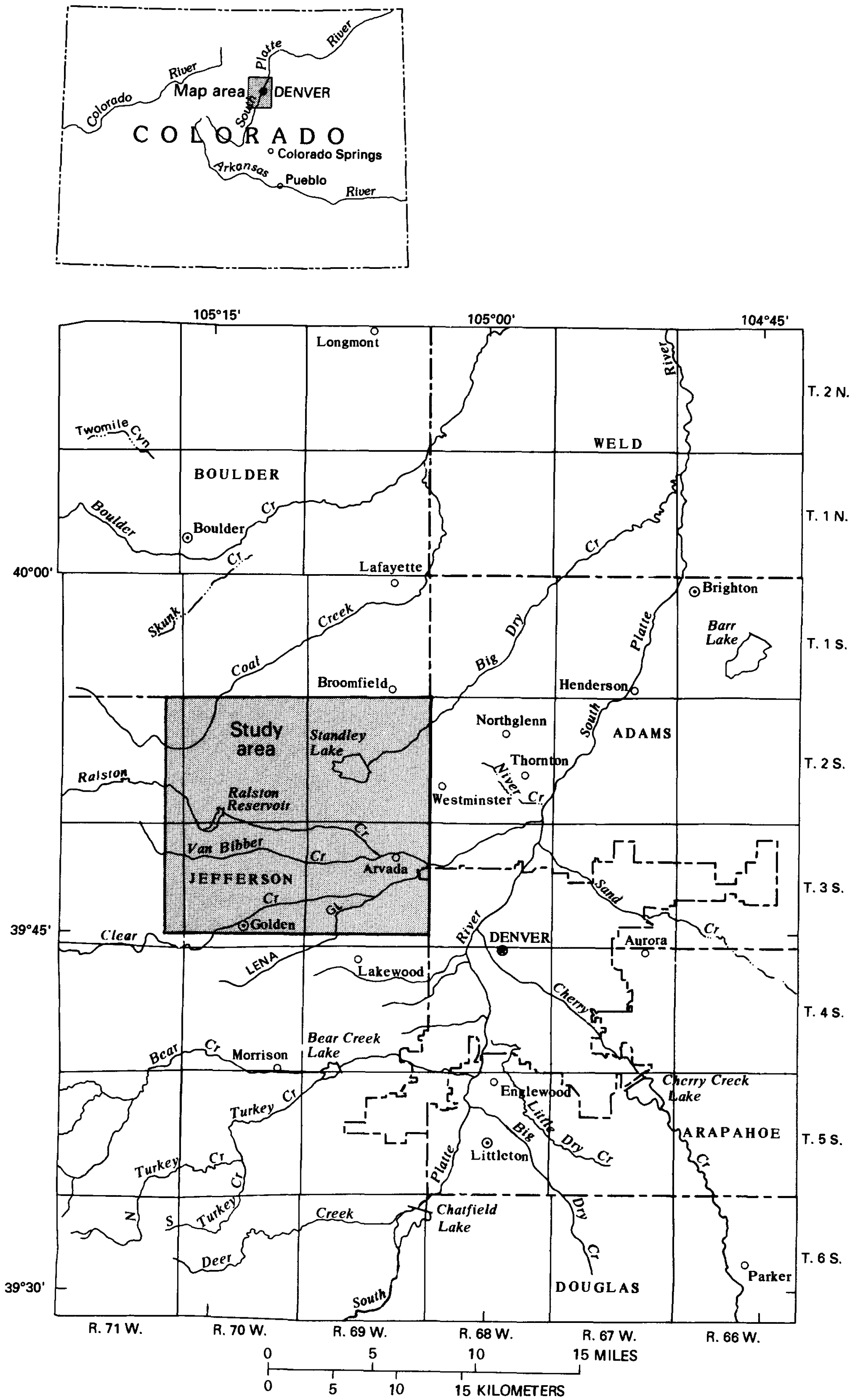


Figure 1.--Location of the study area.

supplied from the lake. This problem occurred in October and November, after the fall turnover, when the water in the lake became completely mixed. The cities suspected that the taste and odor problem was related to phytoplankton growing in the lake, and that this growth could be controlled by regulating the phosphorus concentration in the lake. The Colorado Water Quality Control Commission was petitioned by these cities to set a phosphorus standard for the lake. In a hearing before the Commission, the standard was opposed by the cities of Arvada and Golden and by Jefferson County. These entities potentially could be required to meet phosphorus-control regulations under the standard. They contended that previous studies were not conclusive in determining the source of the taste and odor problem or the factors limiting phytoplankton growth in the lake. The Commission deferred a decision on the standard and suggested that all the parties work together to develop the information necessary to determine appropriate standards for the protection of Standley Lake as a water-supply source. The parties requested that the U.S. Geological Survey conduct a study to provide this information.

In 1989, the U.S. Geological Survey, in cooperation with the cities of Arvada, Golden, Northglenn, Thornton, and Westminster and Jefferson County, began a study of Standley Lake. The objectives of this study were as follows:

1. Determine the limnological characteristics of Standley Lake regarding nutrients, phytoplankton, and zooplankton, and processes that could affect nutrient availability, algal growth, or the occurrence of taste and odor problems. These processes include stratification, turnover (mixing of the entire water column), and development of anoxic conditions.
2. Quantify the nutrient loading to Standley Lake from inflow sources and the retention of nutrients in the lake. Determine whether internal cycling from bottom sediment in Standley Lake may be a substantial source of nutrients to the lake water.
3. Determine whether the availability of phosphorus limits algal growth during periods when algae suspected of causing taste and odor problems may be present in Standley Lake.
4. Determine the compounds responsible for taste and odor problems in the water delivered from Standley Lake and the potential sources of these compounds to water in the lake.

Objectives 1-3 were accomplished by the U.S. Geological Survey. Objective 4 was accomplished by Drexel University, Philadelphia, Pa., using water and biota samples provided by the U.S. Geological Survey; results of this part of the study are presented in two reports from Drexel University (I.H. Suffet and Djanette Khiari, Drexel University, written commun., 1991; Patricia Cascallar and Wesley O. Pipes, Drexel University, written commun., 1991).

Purpose and Scope

This report presents the results of the study of Standley Lake and its inflows and outflow, including nutrient loading to the lake, the potential nutrient limitation to algal biomass, and occurrence and potential sources of compounds that cause taste and odor problems in water withdrawn from the lake. Samples were collected from several locations in Standley Lake, from three surface inflows to the lake, and from the outflow to the city of Westminster's Semper Water Treatment Plant (fig. 2). Sampling began on June 21, 1989, and concluded on October 30, 1990. The data from sample analyses and onsite measurements made at the time of sampling are presented in a separate report (Ruddy and others,1992). Although the U.S. Department of Energy's Rocky Flats plant is located in the Standley Lake watershed (fig. 2), the effects of drainage from the plant, other than nutrients in Woman Creek, are not considered in this report.

Description of the Study Area

Standley Lake is located in the city of Westminster, a suburb of Denver, in northeastern Jefferson County, Colo. It is a reservoir formed by an earthen dam on Big Dry Creek (fig. 2). Storage of water in the lake began in 1910. The lake originally was used to supply water for irrigation, but as suburban development replaced farmland in the delivery area, much of the water use shifted to domestic supply. During 1963-66, the lake was

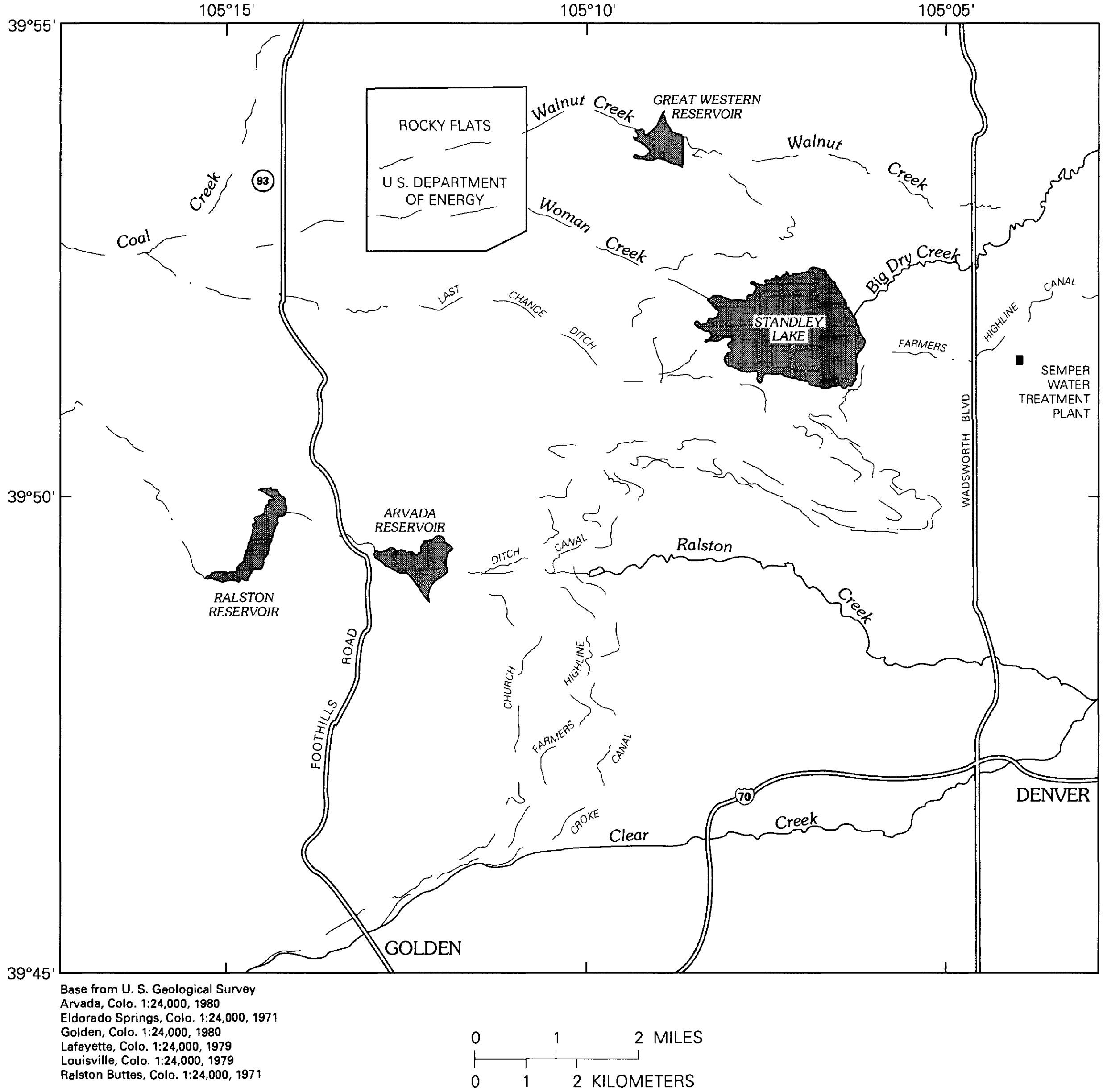


Figure 2.--Location of Standley Lake and the surrounding area.

enlarged to create more capacity for municipal users; however, the full capacity of the lake was not usable until 1981.

At its full-pool elevation, Standley Lake has a capacity of about 43,000 acre-ft and a surface area of about 1,200 acres (Richard P. Arber Associates, 1982). Mean depth is about 36 ft and maximum depth is 96 ft, based on the original land surface. Mean annual inflow to the lake during 1981-89 was about 37,000 acre-ft (Daniel Strietelmeier, city of Westminster, written commun., 1990). Hydraulic retention during this period, calculated as the ratio of capacity to inflow, was about 1.2 years.

Inflows and Outflow

Natural inflow to Standley Lake is intermittent. Most of the inflow is imported by canals from Clear Creek, to the south, or from Coal Creek, to the west. Water from Clear Creek is diverted into three canals in the vicinity of Golden (fig. 2). The Farmers Highline and Croke Canals flow approximately parallel to each other and deliver water to Standley Lake through a common channel near the southern end of the dam. The Church Ditch conveys water around the western side of Standley Lake. Water from this ditch can be diverted into the lake through the Last Chance Ditch channel or through Woman Creek. Water from Coal Creek also can be diverted into Standley Lake through the Last Chance Ditch.

Each canal delivers water to Standley Lake only during certain times of the year. The normal season for diversions into the Farmers Highline Canal is March 20 through November 11. Water from the canal is delivered to Standley Lake during most of this period, although flow may bypass the lake for several days to provide water for irrigation downstream. The Croke Canal diverts water during November 12 through March 19 and during peak runoff in June. All diverted water is delivered to Standley Lake. The season for Church Ditch diversions is April 1 through October 31, but deliveries to Standley Lake normally are not made during this entire period. The season for Last Chance Ditch diversions is November 1 through April 20, depending on the availability of water in Coal Creek. No water was diverted into Last Chance Ditch during November or December 1989. Water also may be diverted into the ditch during peak runoff in May. All diverted water is delivered to Standley Lake.

Monthly inflows to and outflows from Standley Lake during the study period (June 1989 through October 1990) are listed in table 1. Volumes were combined for each inflow location. Inflow through the Last Chance Ditch channel is referred to as Last Chance and Church Ditches, because both ditches can provide inflow to the lake at this location. The Woman Creek inflow represents intermittent natural runoff and deliveries from the Church Ditch. For the entire study period, about 90 percent of the total inflow was delivered by the Farmers Highline and Croke Canals. The total inflow for water year 1990 (October 1989-September 1990) was about 37,000 acre-ft, which is about equivalent to the 1981-89 mean annual inflow.

Outflow from Standley Lake is controlled by an outlet structure located several hundred feet west of the dam at a depth of 72-86 ft below the full-pool water surface. From this outlet, water may be delivered to Big Dry Creek and to municipal water treatment plants serving the cities of Northglenn, Thornton, and Westminster. Outflows during the study period are listed in table 1. About 82 percent of the outflow was delivered to the treatment plants.

History of Inflow Operation

During the past two decades, several revisions have been made in the operation of inflows to Standley Lake. These operational revisions had the potential to cause changes in the nutrient dynamics within the lake.

Beginning about 1950, sanitary-sewage effluent from the city of Golden and the Adolph Coors Company brewery and brewery-process effluent were discharged to Clear Creek upstream from the Croke Canal diversion (Daniel Strietelmeier, city of Westminster, written commun., 1990). During the winter, these effluents could have been diverted into the canal and delivered to Standley Lake. In 1972, in an effort to improve the water quality of Clear Creek, the brewery-effluent discharge point was changed, and the entire effluent stream was discharged into the

Table 1.--Monthly inflows to and outflows from Standley Lake during the study period

[Data from the city of Westminster]

Month	Inflow (acre-feet)				Outflow (acre-feet)			
	Farmers Highline and Croke Canals	Last Chance and Church Ditches	Woman Creek	Total	Big Dry Creek	Northglenn and Thornton	Westminster	Total
1989								
June	10,969	238	459	11,666	428	1,067	1,477	2,972
July	3,545	28	450	4,023	1,325	1,812	2,665	5,802
August	1,445	85	431	1,961	1,626	1,617	1,815	5,058
September	1,679	11	0	1,690	180	1,505	1,348	3,033
October	1,750	0	0	1,750	283	1,538	988	2,809
November	1,705	0	0	1,705	0	997	557	1,554
December	2,165	0	0	2,165	0	999	559	1,558
1990								
January	2,119	12	9	2,140	0	986	538	1,524
February	1,616	12	7	1,635	0	933	440	1,373
March	2,841	161	196	3,198	0	1,046	478	1,524
April	1,691	689	213	2,593	0	1,133	570	1,703
May	2,935	511	114	3,560	449	1,766	1,199	3,414
June	11,809	228	768	12,805	2,325	1,901	2,137	6,363
July	2,391	110	565	3,066	741	1,114	1,463	3,318
August	444	100	185	729	1,107	1,670	1,833	4,610
September	1,489	31	81	1,601	795	1,322	1,281	3,398
October	1,668	0	5	1,673	0	1,106	690	1,796
Total	52,261	2,216	3,483	57,960	9,259	22,512	20,038	51,809

Croke Canal during winter diversions (November-April). In 1982, winter diversions into the Croke Canal ceased, and the municipalities using water from Standley Lake began negotiating with the city of Golden and the Adolph Coors Company to move their discharge points downstream from all diversions into the lake. This move was completed in 1988, and winter diversions into the Croke Canal resumed. Since that time, no brewery effluent or Golden/Coors sewage effluent has entered Standley Lake.

Also, before 1989, the common practice was to flush the canals directly into Standley Lake at the beginning of each diversion season (Daniel Strietelmeier, city of Westminster, written commun., 1990). This flushing carried a large amount of sediment, trash, and organic material that had accumulated in the canals while they were not in use.

Beginning in March 1989, the canals have been flushed into nontributary stream channels for at least 24 hours prior to the initial delivery to the lake each season. During this same time, diversion of storm runoff from streams into the canals was halted.

Previous Investigations

A water-quality and eutrophication study of Standley Lake was initiated in 1981 and was carried out primarily by Richard P. Arber Associates of Denver. In a series of reports to the funding agencies, primarily the cities of Thornton and Westminster, the investigators presented the results of various phases of the study.

During the first phase of the study (January 1981-September 1982), the major conclusions were related to trace metals, trophic status, nutrient limitation, and inflow loading (Richard P. Arber Associates, 1982). Arsenic, lead, and mercury concentrations were determined to increase in outflow from the lake during certain times of the year, and release from the bottom sediment was the suspected source. The lake was considered to be mesotrophic, based on chlorophyll *a* concentrations, phytoplankton density and composition, light transparency (Secchi-disk depth), and dissolved-oxygen (DO) profiles (Richard P. Arber Associates, 1982, p. 3-71 through 3-73). Hypolimnetic DO concentrations commonly were less than 2 mg/L during the late summer and early fall, but no anoxic conditions were recorded. Based on ratios of ambient inorganic-nitrogen to orthophosphate, nitrogen was identified as the limiting nutrient during the spring (N:P = 4.7), with a shift to phosphorus limitation during late summer (N:P = 10.1) and fall (N:P = 13.1) (Richard P. Arber Associates, 1982, p. 3-47). Discharges from the Adolph Coors Company to the Croke Canal were identified as a potentially substantial source of nutrient loading, which could contribute to a deterioration of water quality in the lake (Richard P. Arber Associates, 1982, p. 6-2).

The second phase of the study focused on sources of pollution in the Standley Lake watershed (Black and Veatch, 1983; Richard P. Arber Associates, 1984a) and on the effect of metals and organic constituents in the lake (Richard P. Arber Associates, 1984b and 1984c). The effects from pollution sources surveyed at the time of the study (1983) were not considered to be serious. The greatest concern was about effects from the city of Golden/Adolph Coors Company wastewater-treatment-plant discharges upstream from and into the Croke Canal. The effect of trace-metal contamination from mining activities in the Clear Creek basin was a secondary concern. The study also identified the potential for increased nonpoint-source pollution, including nutrient loading, from future urbanization of the watershed. The survey of metals in the lake indicated seasonal releases of manganese from the bottom sediment during periods of low hypolimnetic DO concentrations (Richard P. Arber Associates, 1984b, p. 60). Releases of arsenic, and possibly chromium and lead, were suspected to be associated with the release of manganese. Mercury and selenium were determined to occasionally exceed the State drinking-water standards. Management of nutrient inflows was suggested to control further degradation, primarily increased algal growth and decreased DO concentrations. The survey of organic constituents focused on total organic halides (TOX) and the potential for formation of trihalomethanes (THM) during the drinking-water treatment process. The Golden/Coors discharges were identified as a substantial source of TOX loading (Richard P. Arber Associates, 1984c, p. 44). Algal growth in the lake was suspected to be related to production of THM precursors (Richard P. Arber Associates, 1984c, p. 35-36) and also to be related to taste and odor problems that occurred following fall turnover in the lake (Richard P. Arber Associates, 1984c, p. 22-24). Again, management of nutrient inflows was suggested to control algal growth.

The third and fourth phases of the study extended the data base through 1985 (Richard P. Arber Associates, 1986 and 1987). Seasonal periods of low DO concentrations in the hypolimnion occurred each year, and an anoxic condition was recorded during September and October 1985. Releases of manganese from the bottom sediment, in association with low DO concentrations, and periods of elevated concentrations of mercury and selenium continued to occur. Concern for potential exceedence of State water-quality standards for arsenic and THM also continued. Chlorophyll *a* concentrations were less than expected based on consideration of nutrient concentrations, indicating that empirical models may not be appropriate for determining algal response to nutrient loading. Laboratory bioassays indicated that algal growth was limited by phosphorus and nitrogen. Peak chlorophyll *a* concentrations occurred in the fall, following turnover (Richard P. Arber Associates, 1987, p. iii). Peak phytoplankton densities occurred in the spring and late fall, and diatoms were the dominant species. However, the phytoplankton samples were collected from the lake outflow, which was withdrawn through a lake-bottom outlet, and may not have represented actual algal populations in the lake. Bottom sediment analyses indicated a tendency for fine particulates to migrate and accumulate in the deeper areas of the lake. Concentrations of cadmium, lead, and mercury in the sediment were considered to be somewhat elevated, but concentrations of organic carbon, nitrogen, and phosphorus were not considered to be extremely high in comparison to other reservoirs and lakes (Richard P. Arber Associates, 1986, p. 85).

The final report on the Standley Lake Water Quality Program (Richard P. Arber Associates, 1988a) presented data collected during 1986 and results of an empirical modeling study relating inflow phosphorus loading, in-lake phosphorus concentration, and algal response, represented by chlorophyll *a* concentration. Control of phosphorus loading was recommended to prevent trophic deterioration in the lake. An additional report (Richard P. Arber Associates, 1988b) suggested methods for controlling phosphorus inflow from point and nonpoint sources.

The city of Westminster continued to collect data on Standley Lake during 1987-89. Some of these data were used in testimony before the Colorado Water Quality Control Commission during hearings to consider implementation of a phosphorus standard for the lake. However, no additional reports were prepared by the city before initiation of the U.S. Geological Survey study in June 1989.

Data Collection

A data-collection program was designed to meet the four objectives of the study. Sampling and measurement methods generally were based on standard techniques (Guy and Norman, 1970; Edwards and Glysson, 1988; Britton and Greeson, 1989; Ward and Harr, 1990) but also included special techniques adapted for this study. Data-collection sites are described in table 2 and are shown in figures 3 and 4. A complete description of data-collection methods, along with tabulated measurements and laboratory analyses, is presented in a separate data summary (Ruddy and others, 1992).

The limnological characteristics of Standley Lake regarding nutrients, biota, and processes that could affect the trophic status of the lake were determined by in-lake monitoring at sites L1 and L2 (fig. 3). These sites were chosen to represent the main body of the lake and to be consistent with sampling locations used in previous studies. Onsite measurements included Secchi-disk transparency and depth profiles of water temperature, pH, dissolved oxygen, and specific conductance. Samples were collected weekly, biweekly, or monthly, depending on the season. Samples were collected from within the photic zone and near the bottom at both sites and from an intermediate depth at site L1 during times when the lake was thermally stratified. Samples were analyzed for concentrations of nutrients and trace constituents. In addition, samples collected from within the photic zone were analyzed for concentrations of chlorophyll *a* and densities and biovolumes of phytoplankton. A vertically integrated sample was analyzed for densities of zooplankton.

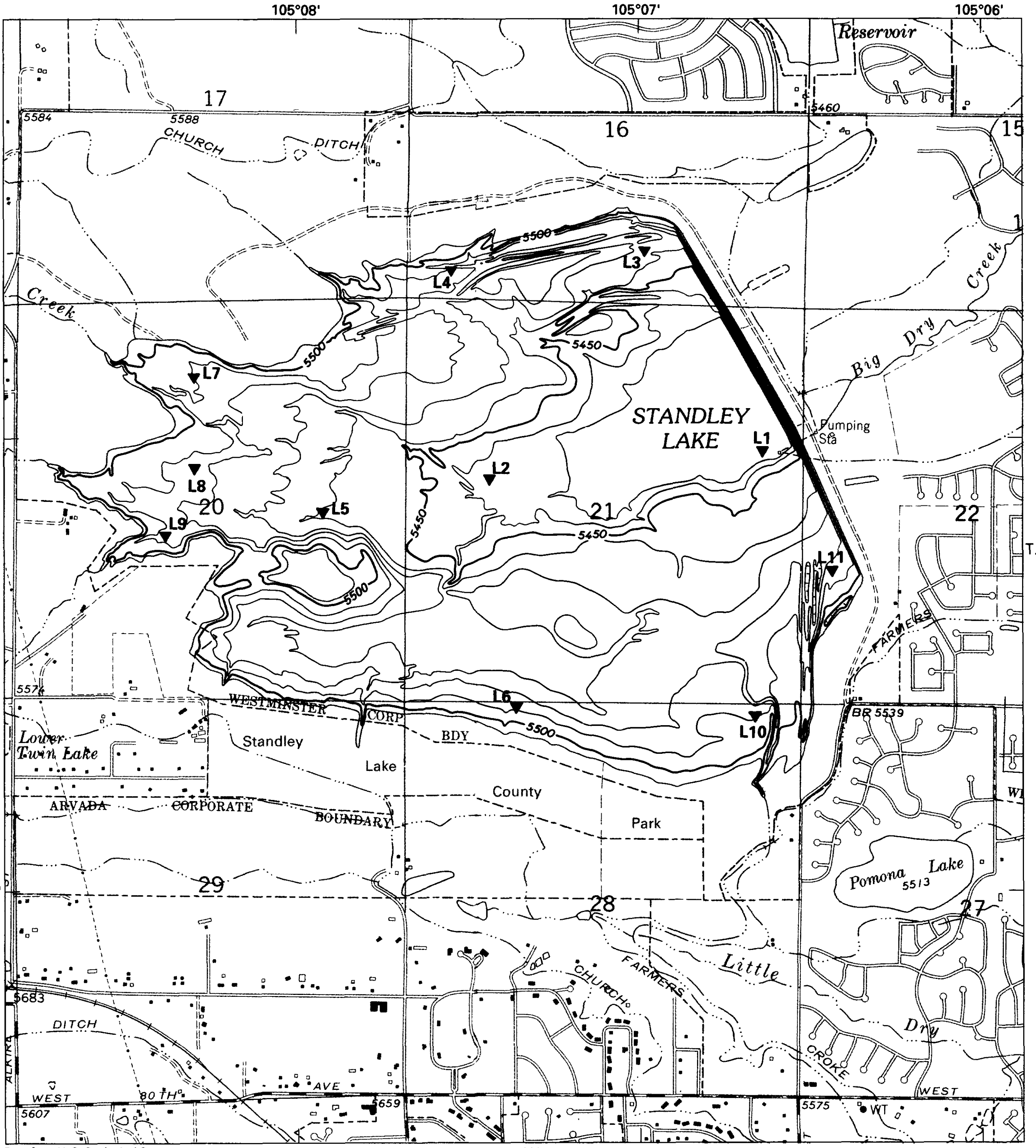
Samples from various types of substrate materials were collected from the lake to determine the relative abundance of periphyton algae taxa. Periphyton includes the entire micro-organism community that attaches to or lives upon submerged solid surfaces (Britton and Greeson, 1989). Samples were collected three times each year of the study, in August, September, and October, at various locations in the lake. In 1989, samples were collected from artificial substrates (polyethylene strips) that were suspended in the water column and from submerged natural objects, such as rocks or plant material. The species composition on the artificial and natural substrates differed so much that the data from the artificial substrates were considered not representative of natural conditions and were not used. In 1990, samples were collected from the surface of the unconsolidated bottom sediment, which is more characteristic of the substrate available in most of the lake.

Data for estimation of nutrient loading and retention were collected at three inflow sites (IO1, IO2, and IO3) and one outflow site (IO4) (fig. 4). Daily streamflow at each inflow site was measured in one or more Parshall flumes by the Farmers Reservoir and Irrigation Company, and data were provided by the city of Westminster (Daniel Strietelmeier, city of Westminster, written commun., 1989 and 1990). Inflow and outflow water samples were collected weekly, biweekly, or monthly, depending on the season. Additional samples were collected from inflow sites during or immediately following major rainfall and snowmelt events. Outflow samples were collected from the raw-water intake to the city of Westminster's Semper Water Treatment Plant, and the analytical

Table 2.--Description of sampling sites used for the study

[Identification number is latitude and longitude of the site with a sequence number of 00 at the end; see figures 3 and 4 for site location]

Site number	U.S. Geological Survey identification number	Site name
	LAKE SITES	
L1	395159105063200	Standley Lake near dam
L2	395150105072300	Standley Lake near center
L3	395224105065700	Standley Lake near spillway
L4	395218105073600	Standley Lake (north side)
L5	395148105080000	Standley Lake near island
L6	395125105072700	Standley Lake (south side)
L7	395208105082900	Standley Lake near Woman Creek inlet
L8	395153105083800	Standley Lake (west side)
L9	395144105083100	Standley Lake near Last Chance Ditch inlet
L10	395124105063900	Standley Lake near Farmers Highline and Croke Canal inlet
L11	395148105062200	Standley Lake near boat ramp
	INFLOW AND OUTFLOW SITES	
I01	395111105064100	Farmers Highline and Croke Canals
I02	395119105090600	Last Chance and Church Ditches
I03	395216105084500	Woman Creek and Church Ditch
I04	395131105041500	Semper Water Treatment Plant



Base from U. S. Geological Survey
Arvada, Colo. 1:24,000, 1980
Golden, Colo. 1:24,000, 1980
Lafayette, Colo. 1:24,000, 1979
Louisville, Colo. 1:24,000, 1979

0 — 1 MILE

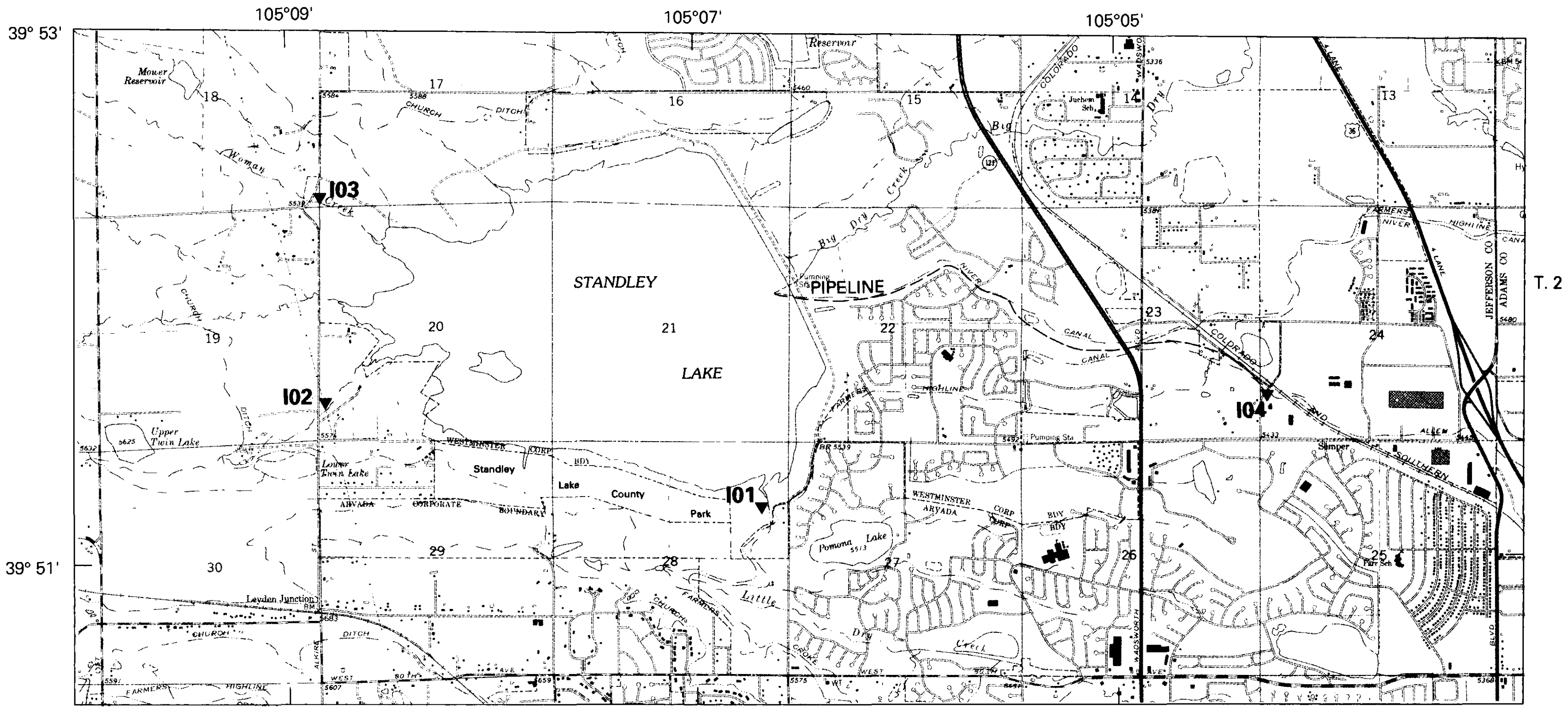
0 — 1 KILOMETER

EXPLANATION

L6▼ SAMPLING SITE AND NUMBER--See table 2 for description

—5500— BATHYMETRIC CONTOUR--Contour interval 10 feet. National Geodetic Vertical Datum of 1929

Figure 3.--Location of sampling sites in Standley Lake.



Base from U. S. Geological Survey
Arvada, Colo. 1:24,000, 1980
Golden, Colo. 1:24,000, 1980
Lafayette, Colo. 1:24,000, 1979
Louisville, Colo. 1:24,000, 1979

0 1 MILE

0 1 KILOMETER

EXPLANATION

102▼ SAMPLING SITE AND NUMBER--*See* table 2 for description

Figure 4.--Location of inflow and outflow sampling sites.

results were assumed to be representative of all outflows. Onsite measurements made at the time of sampling included water temperature, pH, dissolved oxygen, and specific conductance. Samples were analyzed for concentrations of suspended solids and nutrients. Samples collected at inflow sites also were analyzed for densities and biovolumes of phytoplankton and for concentrations of total organic carbon.

Bottom sediment and interstitial pore-water samples were collected to determine the potential flux of constituents to the lake water. Sediment was collected from about the upper 4 cm of the lake bottom for analysis of solid-phase chemical composition. Samples were collected at sites L1-L10 (fig. 3) in July 1989 when the lake was thermally stratified. Interstitial pore-water chemistry was determined for water extracted from discrete sections of sediment cores. Cores were obtained by using a gravity-driven coring device operated from the lake surface. The cores were extruded in a nitrogen atmosphere to minimize oxidation of chemical species. Sections of the core were removed at 1-cm intervals, and the pore water was extracted by centrifuging. Water from the bottom of the lake was collected at the same time for use in determining constituent gradients between the pore water and the lake. Sediment cores were collected in August and October 1989 at sites L1 and L2; in May 1990 at sites L1, L2, L4, L5, L8, and L10; and in August 1990 at sites L1, L2, and L5 (fig. 3).

A series of in-lake enclosure experiments was used to test the algal response to phosphorus and nitrogen additions. The experiments were done at site L3 (fig. 3), where the water depth was about 25-30 ft. Two types of enclosures were used: (1) Small, completely sealed enclosures referred to in this study as microcosms (Wurtsbaugh and others, 1985; Morris and Lewis, 1988; Dodds and Priscu, 1990); and (2) large enclosures that were open to the atmosphere, referred to in this study as mesocosms (Bloesch and others, 1988; French and others, 1988; Elser and others, 1990). The microcosms were 10-L cubitainers suspended in the lake at one-half the Secchi-disk depth. The mesocosms were polyethylene tubes about 3 ft in diameter, open at the top, and extending about 15 ft into the lake. Microcosm experiments were done in July, August, September, and October 1989, and in August and October 1990. The mesocosm experiment was done once, in October 1990. Construction, installation, and sampling of the microcosms and mesocosms is described in a separate report (Ruddy and others, 1992).

Lake and inflow water samples were analyzed for organic compounds known to cause taste and odor problems. Algae and bacteria from water and bottom-substrate samples were grown in enrichment cultures to identify potential sources of taste- and odor-causing compounds. Water samples, which were used for analysis of organic compounds and for extraction of phytoplankton algae for enrichment cultures, were collected from deep and shallow sites in the lake and from inflows. Lake-water samples were collected from within the photic zone and from near the lake bottom. Bottom-substrate samples, which were used for extraction of periphyton algae and bacteria, were collected from shallow lake sites. In 1989, water and phytoplankton samples were collected at sites L1, L2, L3, L10, IO1, and IO4, and periphyton samples were collected at sites L3, L7, L9, L10, and L11. In 1990, water and phytoplankton samples were collected at sites L1, L2, L3, L8, L10, L11, IO1, and IO4, and periphyton samples were collected at sites L7, L10, and L11.

Acknowledgments

The authors extend their appreciation to the many people in the cooperating city and county governments who helped guide the Standley Lake study: Scott Daniels, James McCarthy, and Sterling Schultz from the city of Arvada; Gregory Moore from the city of Golden; Kipp Scott from the city of Northglenn; Annette Barnard and James Gindelberger from the city of Thornton; Sharon Bernia, Kelly DiNatale, and David Kaunisto from the city of Westminster; and Jeanne Jacobis and Mary Lynn Tucker from Jefferson County. In addition, we would like to thank Daniel Strietelmeier, of the city of Westminster, for providing information on lake and canal operations and data on inflows and outflows, and Leslie Cooper, of Standley Lake Park, for providing access to the lake and assistance in establishing experimental installations.

Sharon Campbell, of the U.S. Bureau of Reclamation, coordinated the analysis of biota samples and provided useful advice and sampling equip-

ment throughout the study. I.H. Suffet and Wesley Pipes, of Drexel University, provided guidance on collection of samples for taste and odor identification, and performed chemical and biological analyses on these samples.

LIMNOLOGICAL CHARACTERISTICS OF STANDLEY LAKE

During the study period, Standley Lake was thermally stratified from June through September each year and was well mixed during October-December and March-May. Profiles of temperature and dissolved oxygen (fig. 5) indicate that conditions were almost identical at both sampling sites (L1, near the dam, and L2, near the center of the lake). A thermocline had begun to develop between a depth of about 10 and 20 ft when sampling was initiated in June 1989. The thermocline became deeper as the surface water warmed during July. By mid-August, the epilimnion was about 25 ft deep, and the hypolimnion, extending from a depth of about 40 ft to the bottom of the lake, was entirely anoxic. Stratification began to break down following a period of snow and cold weather during September 9-13, but the bottom water remained anoxic until the lake became completely mixed in early October. The lake remained isothermal and well aerated through December and also during March-May 1990. No data were collected during the period of intermittent and unstable ice cover, to determine whether stratification occurred in the winter. Development of summer stratification in 1990 followed the same pattern seen in 1989. A thermocline became established during June and deepened in July. By mid-August, an anoxic hypolimnion had once again become established between a depth of 40 ft and the bottom. September was warmer in 1990 than in 1989; therefore, the hypolimnion diminished more gradually. The bottom water remained anoxic until the lake became isothermal and then completely mixed sometime between October 2 and 10. Even though weather conditions varied, the periods of stratification and hypolimnetic anoxia were almost identical in 1989 and 1990. Fall turnover, when the lake became completely mixed, was within a few days of the same date in both years.

Light transparency of the water at the surface of Standley Lake, measured by using a 20-cm-diameter, black-and-white Secchi disk, on the dates of sample collection is plotted in figure 6. Again, there is little difference between the sampling sites L1 and L2. In 1989 and 1990, transparency increased following stratification in June and decreased following turnover in October. During August and September, transparency was variable and was generally greater in 1990 than in 1989. There was no consistent relation of transparency to suspended-solids concentration or phytoplankton biovolume in the photic zone (fig. 7). The coefficient of determination (R^2) for a multiple-regression model of transparency with suspended-solids concentration and phytoplankton biovolume as independent variables was 0.20. This result indicates that, even in combination, variations in suspended solids and phytoplankton account for only about 20 percent of the variation in transparency.

Two sets of measurements were made in 1989 to determine whether mechanical mixing from wind or motorboats could affect the stability of the lake during stratification. If stability deteriorated, nutrient availability in the epilimnion could have been subsequently affected. The first set of measurements was made on August 18 and 21. An extended period of strong wind occurred on August 19. At site L1 (near the dam), the depth of the epilimnion increased from 25 ft before the windy period to 35 ft afterwards, and the transparency increased from 58 to 92 in. The depth of the anoxic hypolimnion was not affected. No substantial changes occurred at site L2 (near the center). Concentrations of chemical constituents essentially were identical on both days at both sites. The second set of measurements was made on September 1 and 5. Over the intervening Labor Day weekend, 372 boats entered Standley Lake Park (Leslie Cooper, city of Westminster, oral commun., 1989). Again, the depth of the epilimnion increased at site L1 from about 35 ft on September 1 to about 40 ft on September 5. The depth of the hypolimnion at site L1 and the entire profile at site L2 were unaffected. Transparency increased at both sites from 79 to 100 in. at site L1 and from 89 to 101 in. at site L2. The only substan-

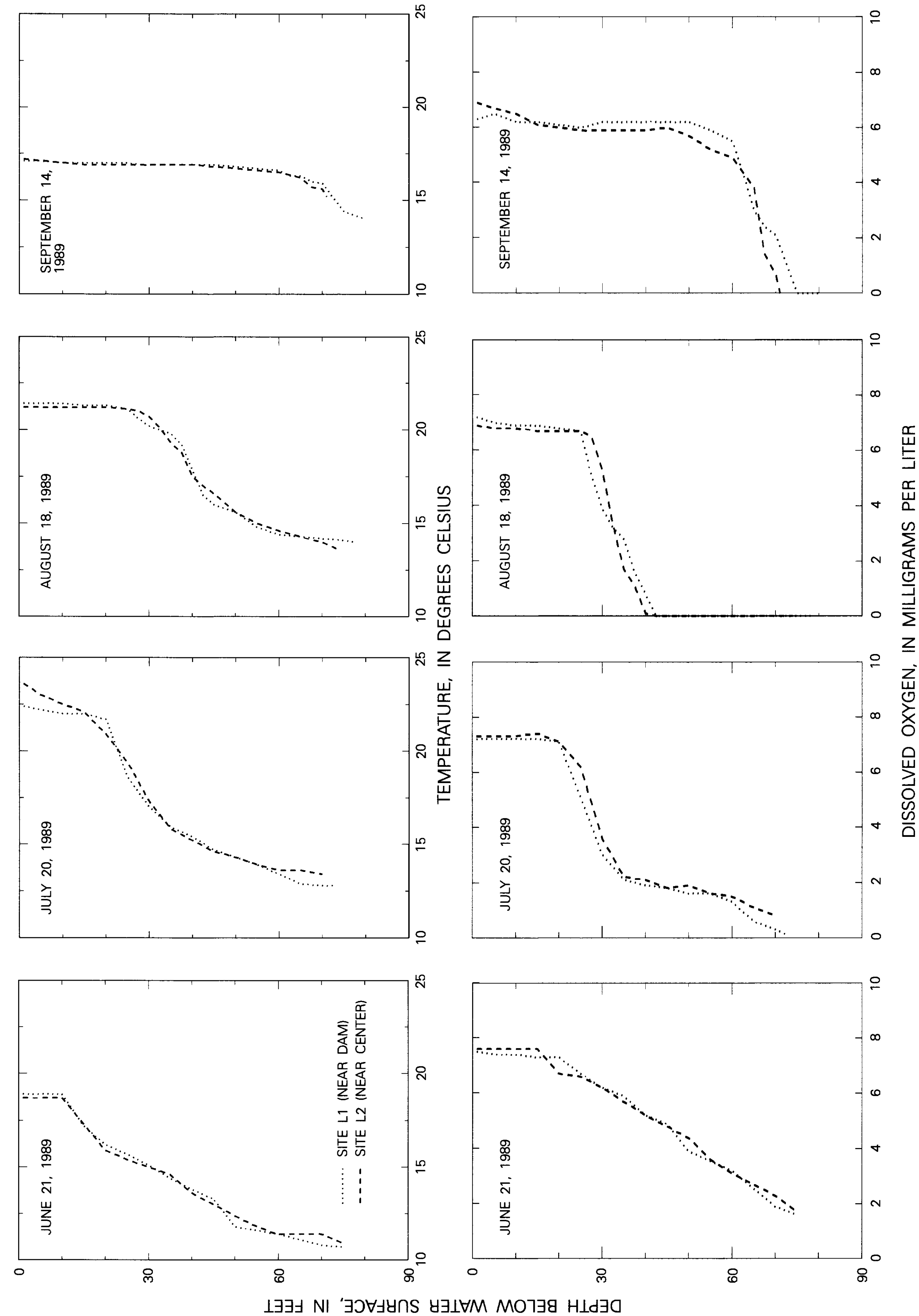


Figure 5.--Profiles of temperature and dissolved oxygen in Standley Lake on selected sampling dates.

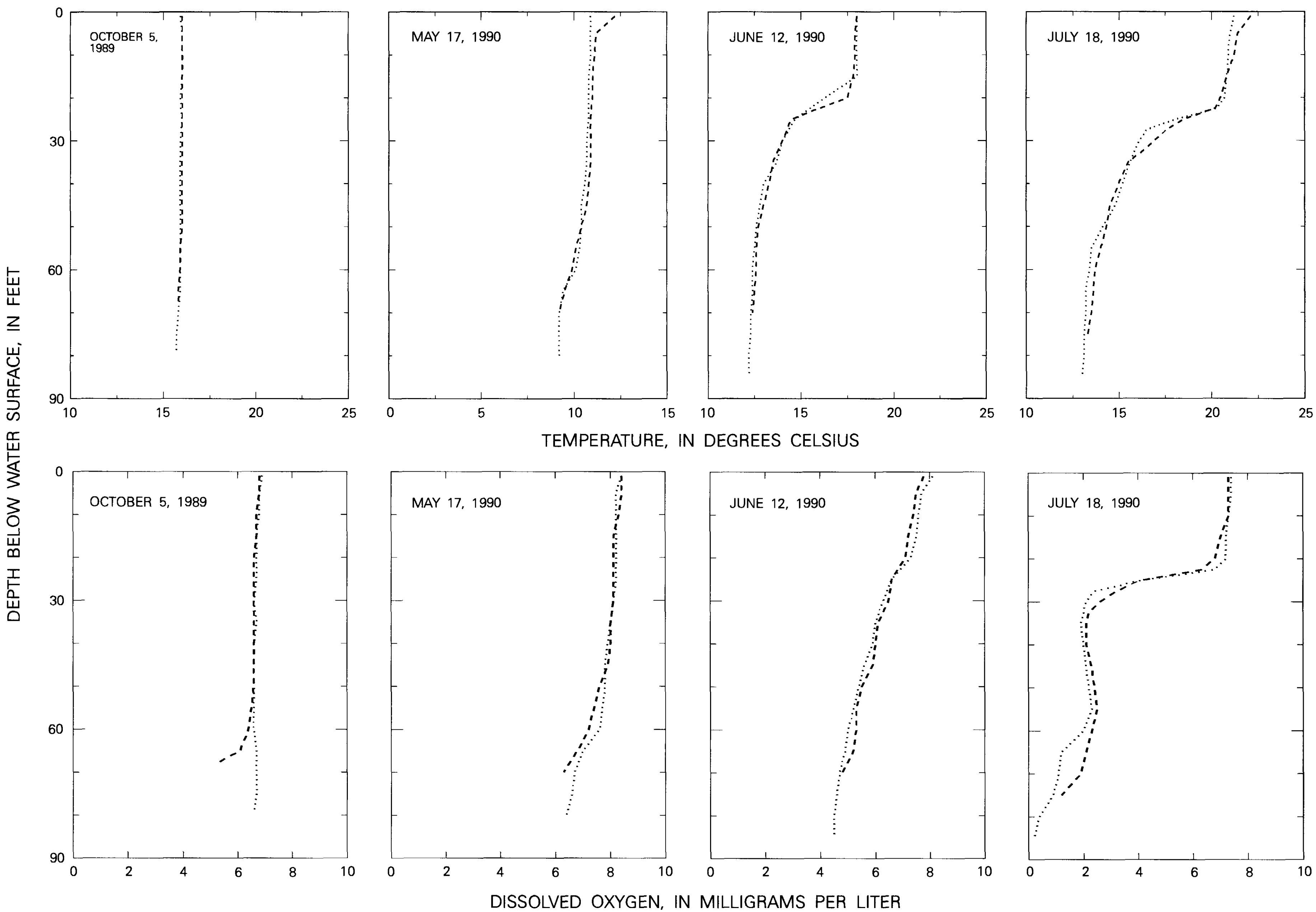


Figure 5.--Profiles of temperature and dissolved oxygen in Standley Lake on selected sampling dates.--Continued

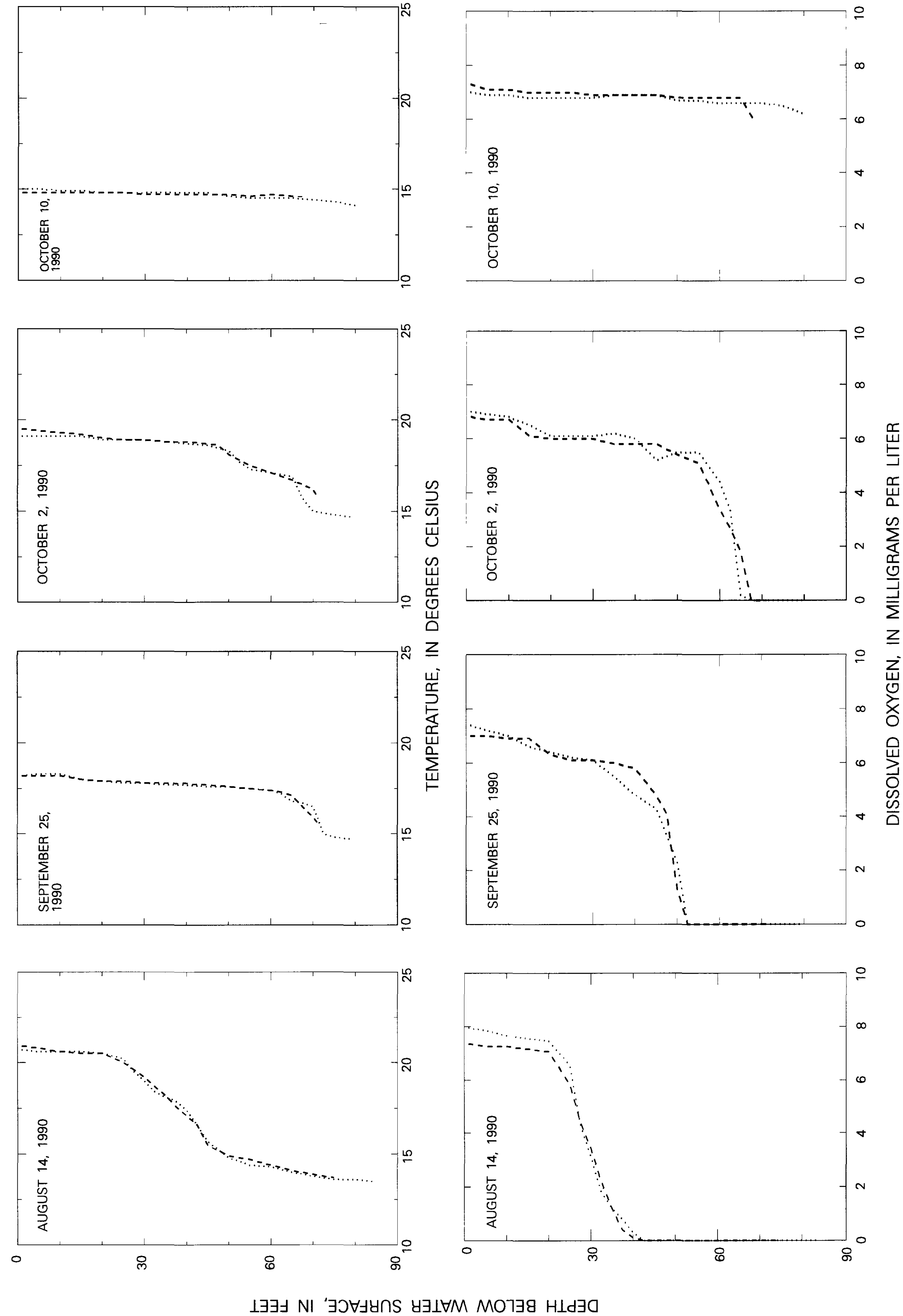


Figure 5.--Profiles of temperature and dissolved oxygen in Standley Lake on selected sampling dates.--Continued

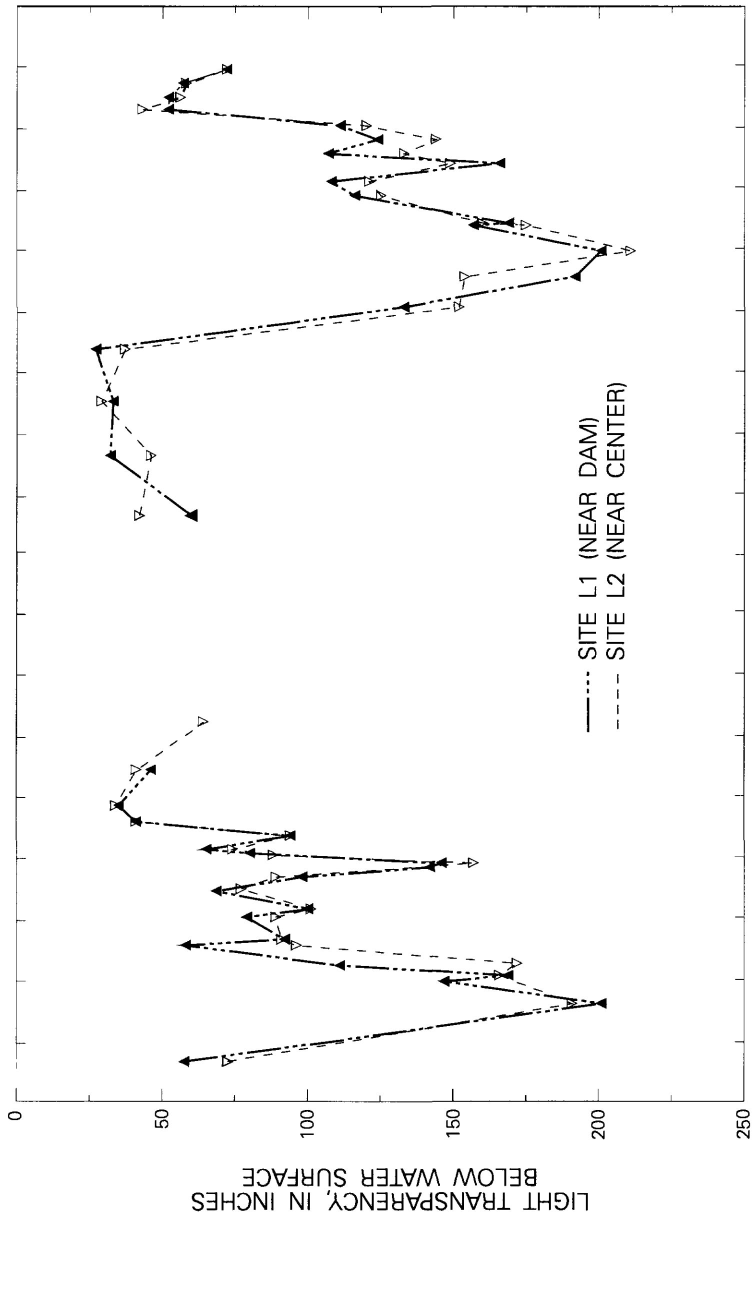


Figure 6.--Light transparency (Secchi-disk depth) in Standley Lake.

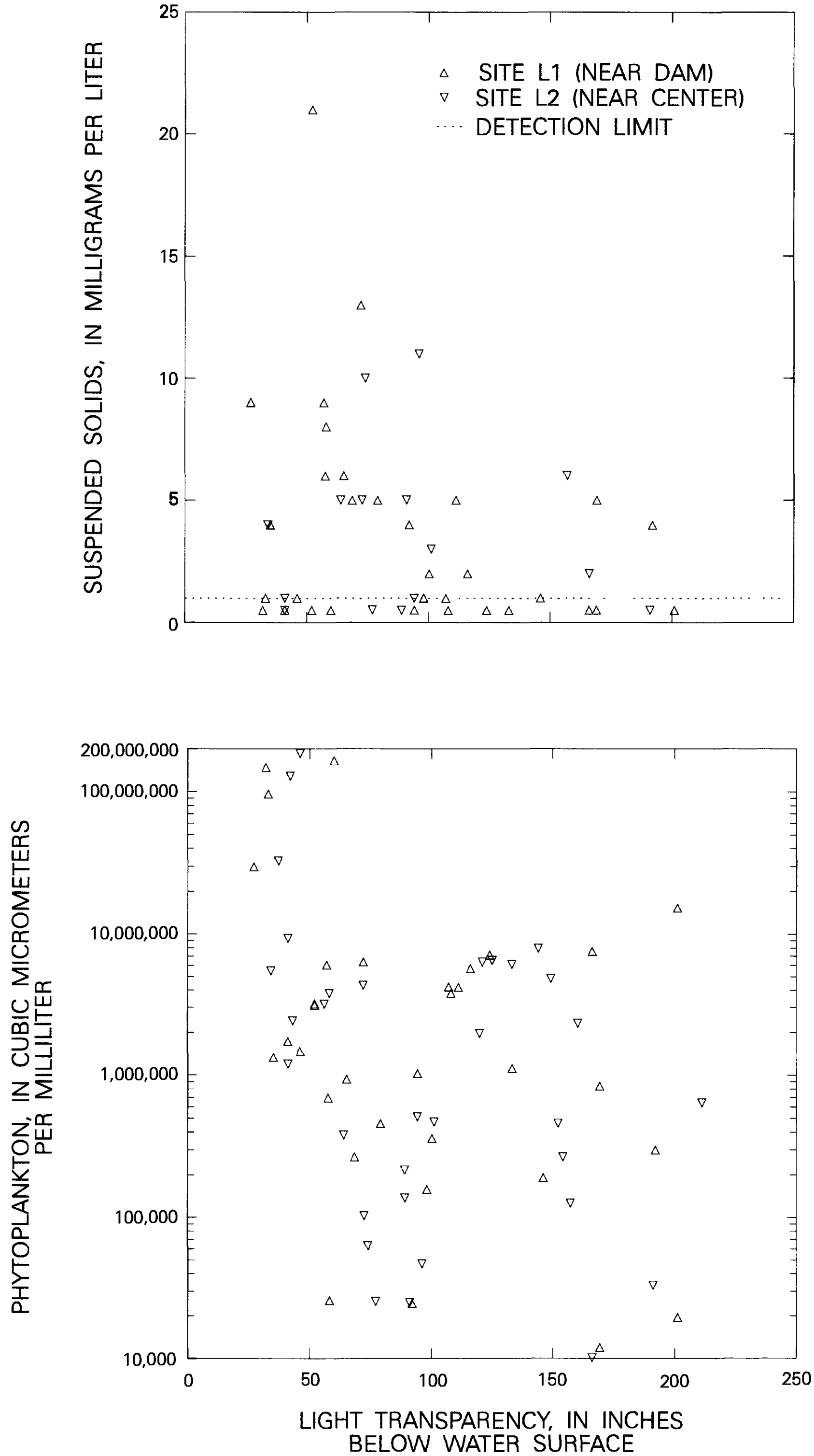


Figure 7.--Relations of light transparency to suspended-solids concentration (1989) and phytoplankton biovolume (1989-90) in Standley Lake.

tial changes in concentrations of chemical constituents were increases in phosphorus, iron, and manganese in the bottom-water sample from site L2; however, it seems unlikely that these changes could be due to any activity on the surface of the lake because of the stable stratification at that site. Overall, no evidence indicated that wind or motorboat activity could routinely break down the stratification in Standley Lake and introduce additional nutrients into the photic zone.

Nutrient Concentrations

Total nitrogen and total phosphorus concentrations measured in surface-water (photic zone) samples at sites L1 (near the dam) and L2 (near the center) are plotted in figure 8. Concentrations of both these nutrients in the photic zone were very low throughout the study period. Concentrations of total nitrogen exceeded 0.2 mg/L in only two samples, and often were less than analytical detection (0.1 mg/L). Concentrations of total phosphorus were always less than 002 mg/L.

Concentrations of total nitrogen and total phosphorus in bottom-water samples generally were higher and more variable than concentrations in the photic zone (fig. 8). Maximum concentrations of total nitrogen were at least 0.5 mg/L during both years of the study, and maximum concentrations of total phosphorus exceeded 0.05 mg/L both years. Concentrations in bottom-water samples were highest during periods of hypolimnetic anoxia and then decreased sharply after fall turnover. This pattern could indicate a release of nutrients from the bottom sediment during anoxic conditions or decomposition of organic matter settling from the photic zone. In either instance, concentrations would increase in the hypolimnion during stratification. Following turnover, when the lake becomes completely mixed, the nutrients would be dispersed through the water column, and concentrations in the bottom water would be diluted.

Algae and Zooplankton

The data collected on planktonic algae (phytoplankton) and zooplankton in Standley Lake included concentrations of chlorophyll *a*, an indicator of phytoplankton biomass; phytoplankton biovolume, computed from species density counts; and zooplankton density. These data for site L1 (near the dam) are listed in table 3. Dominant phytoplankton and zooplankton groups in each sample also are listed. Chlorophyll *a* concentrations, phytoplankton biovolumes, and zooplankton densities for sites L1 and L2 (near the center) are plotted in figure 9.

Comparison of the chlorophyll *a* and phytoplankton-biovolume data in table 3 and figure 9 indicates no consistent relation. The pairwise, linear (Pearson) correlation coefficient between these data was less than 0.03, and the rank (Spearman) correlation coefficient was less than 0.02. This result, which indicates neither linear nor nonlinear correlation, is not unusual. The chlorophyll *a* content of algal biomass varies with species composition, and even within single species the chlorophyll *a* content can vary due to external factors, such as nutrient availability or the intensity of solar radiation (Nicholls and Dillon, 1978). Large fluctuations in biovolume may not cause similar fluctuations in chlorophyll *a* concentration if the species composition changes or environmental factors change.

For samples collected during 1989, the extreme values of chlorophyll *a* concentration and phytoplankton biovolume occurred about simultaneously, but phytoplankton biovolume had a larger relative spread. Minimum values occurred in July and early August. Values then fluctuated during August and September as a succession of small blooms occurred, including the blue-green alga *Merismopedia*, the diatom *Melosira*, and small cryptomonads. Maximum values occurred in October and November after fall turnover. Biovolume minimums were associated with greater abundance of green algae, and biovolume maximums were associated with diatoms and cryptomonads.

In the spring of 1990, phytoplankton biovolume was dominated by a large and persistent bloom of the diatom *Asterionella*. Biovolume decreased during May through mid-July, but generally remained larger than in 1989 throughout the summer. Biovolume did not increase after fall turnover in 1990 as it had in 1989. The concentrations of chlorophyll *a* were no larger in 1990 than in 1989. Concentrations during the spring bloom were not elevated in proportion to biovolumes.

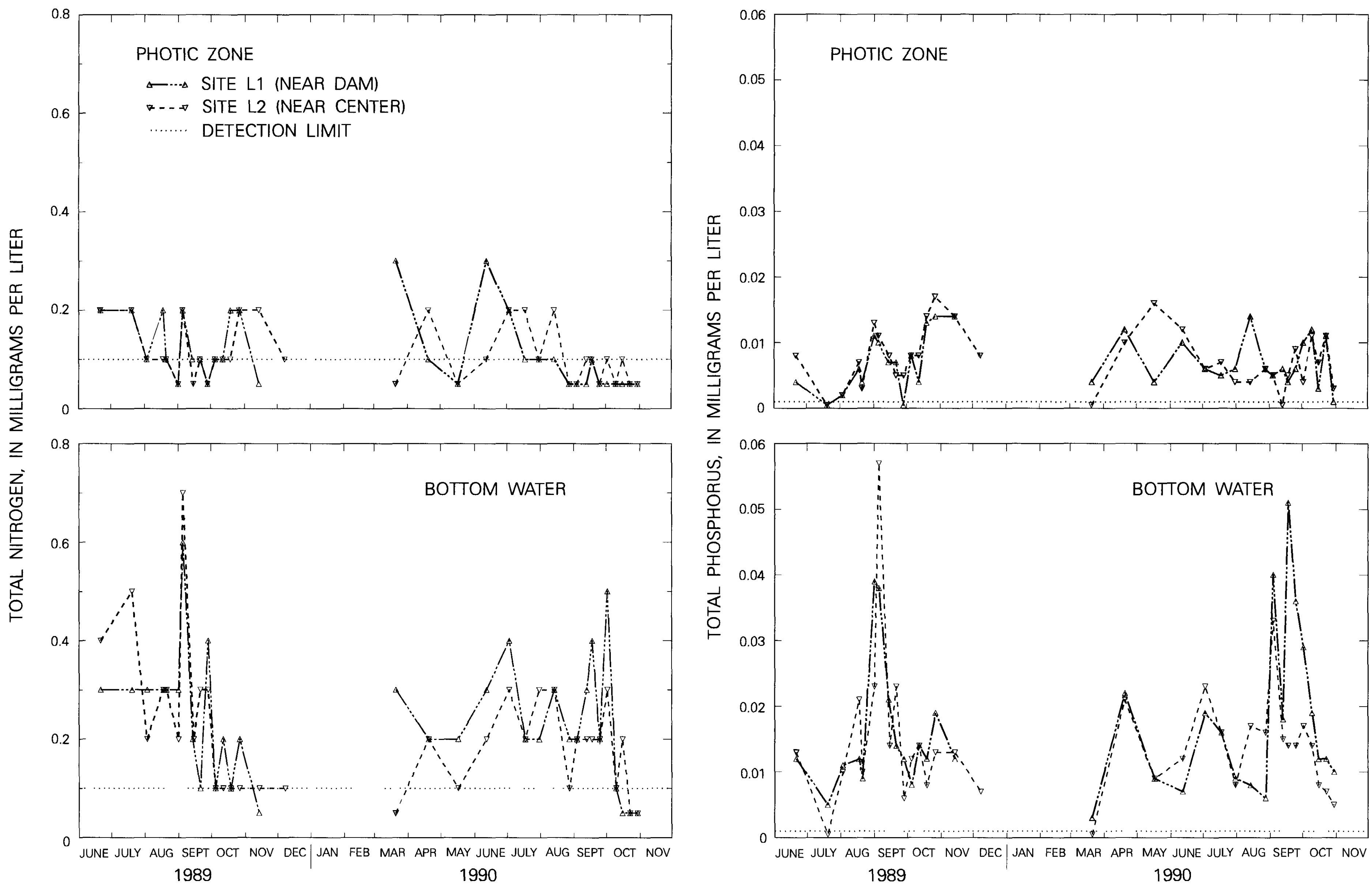


Figure 8.--Total nitrogen and total phosphorus concentrations in the photic zone and bottom water of Standley Lake. (Values less than the detection limit are plotted at one-half the detection limit.)

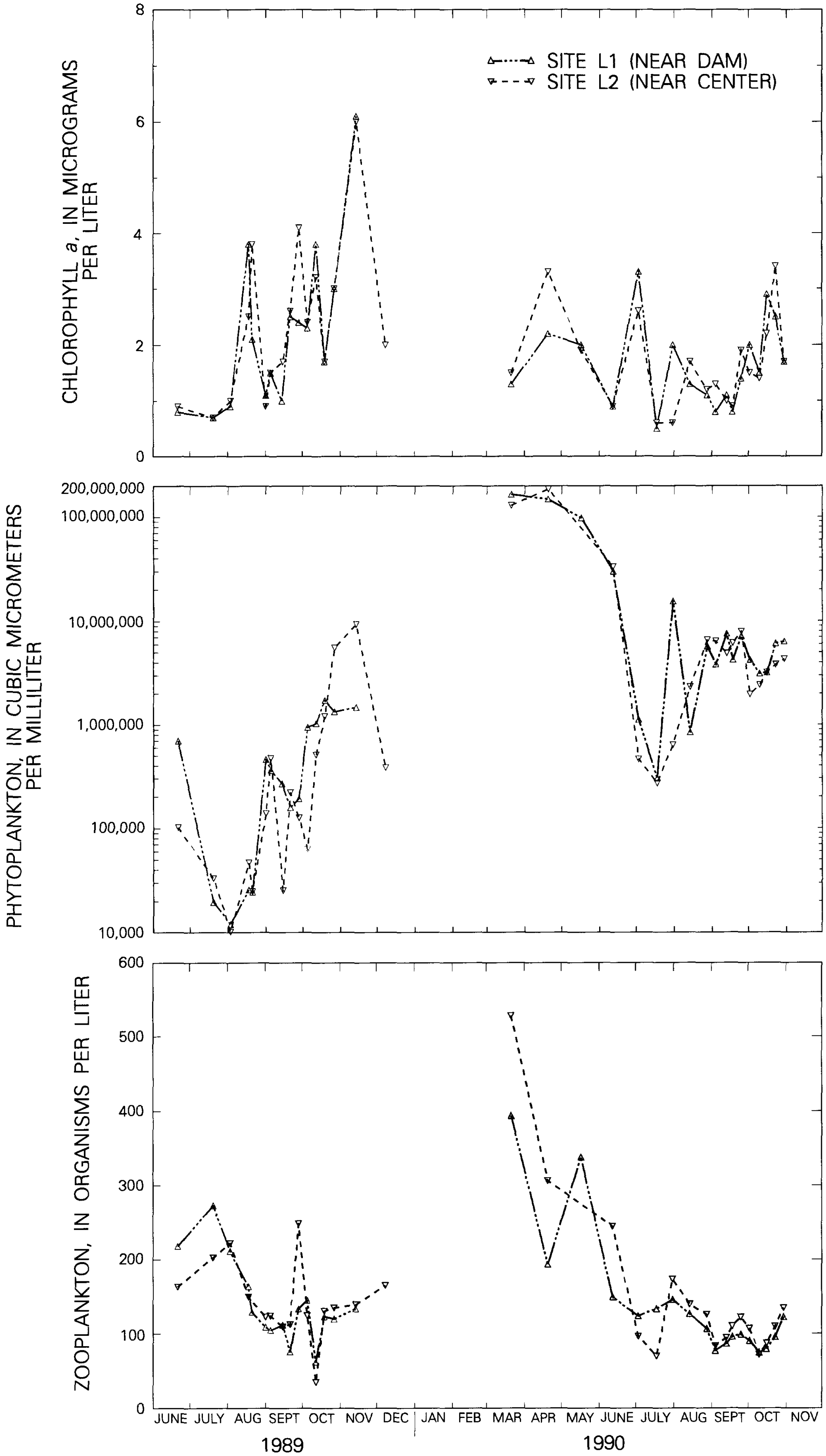


Figure 9.--Chlorophyll *a* concentrations, phytoplankton biovolumes, and zooplankton densities at sites L1 and L2 in Standley Lake.

Table 3.--Chlorophyll *a* concentrations, phytoplankton biovolumes and dominance, and zooplankton densities and dominance at site L1 in Standley Lake

[See fig. 3 for site location; µg/L, micrograms per liter; µm³/ml, cubic micrometers per milliliter; orgs/L, organisms per liter; D, diatoms; G, green algae; Df, dinoflagellates; BG, blue-green algae, C, cryptomonads; R, rotifers; Co, copepods]

Date	Chlorophyll *a* (µg/L)	Phytoplankton		Zooplankton	
		Biovolume (µm³/mL)	Dominant group(s)[2]	Density (orgs/L)[3]	Dominant (group(s)[4]
1989					
June 21	0.8	700,000	D	220	R
July 20	0.7	20,000	D, G, Df	270	R
August 3	0.9	12,000	D, Df	210	Co
August 18	3.8	25,000	BG	160	Co
August 21	2.1	24,000	BG, D	130	Co
September 1	1.1	450,000	BG, C	110	Co
September 5	1.5	360,000	BG, C	100	Co
September 14	1.0	270,000	C	110	Co
September 21	2.5	160,000	C	76	Co, R
September 28	2.4	190,000	C	130	R, Co
October 5	2.3	940,000	D	140	R
October 12	3.8	1,000,000	BG, D, C	60	R
October 19	1.7	1,700,000	D	120	R, Co
October 27	3.0	1,300,000	C	120	R, Co
November 14	6.1	1,500,000	D	130	R, Co
December 8[1]	2.0	380,000	D, C	160	R, Co

Table 3.--Chlorophyll *a* concentrations, phytoplankton biovolumes and dominance, and zooplankton densities and dominance at site L1 in Standley Lake--Continued

Date	Chlorophyll *a* (µg/L)	Phytoplankton		Zooplankton	
		Biovolume (μm^3/mL)	Dominant group(s)[2]	Density (orgs/L)[3]	Dominant (group(s)[4]
1990					
March 21	1.3	160,000,000	D	390	R
April 20	2.2	150,000,000	D	190	R
May 17	2.0	97,000,000	D	330	R
June 12	0.9	29,000,000	D	150	R, Co
July 3	3.3	1,100,000	D,G	120	Co, R
July 18	0.5	300,000	G, D	130	Co, R
July 31	2.0	16,000,000	D	140	R, Co
August 14	1.3	840,000	D	130	Co, R
August 28	1.1	5,700,000	D	110	Co, R
September 4	0.8	3,800,000	D	77	Co, R
September 13	1.1	7,500,000	BG, D	85	Co, R
September 18	0.8	4,300,000	D	96	Co
September 25	1.4	7,100,000	D	98	Co
October 2	2.0	4,200,000	D	90	Co
October 10	1.5	3,100,000	D, BG	75	Co
October 16	2.9	3,200,000	D, BG	77	Co
October 23	2.5	6,100,000	D, BG	94	R, Co
October 30	1.7	6,300,000	D	120	R, Co

[1]No sample was collected at site L1 on this date. The listed data are from a sample collected at site L2, near the center.

[2] Dominant phytoplankton groups are defined as groups having the largest biovolume or having a biovolume at least 50 percent as large as the largest biovolume.

[3]Does not include species with less than one organism per liter in the sample.

[4]Dominant zooplankton groups are defined as groups having the largest density or having a density at least 50 percent as large as the largest density.

Zooplankton did not seem to have a substantial effect on phytoplankton biovolume. The zooplankton population was similar in both years of the study, from June through October, in terms of density and dominance, but phytoplankton biovolume was much larger the second year (fig. 9, table 3). Also, the decrease in zooplankton density in the spring of 1990 occurred before the decrease in phytoplankton biovolume, rather than afterwards, which could occur if the phytoplankton were responding primarily to grazing pressure. The zooplankton were likely responding to grazing pressure from fish. During April through June 1990, when the zooplankton population generally was decreasing, Standley Lake was stocked with almost 2 million walleye fry, about 77,000 brown trout, mackinaw, and wiper fingerlings, and about 67,000 adult rainbow trout (Philip Goebel, Colorado Division of Wildlife, written commun., 1991). Zooplankton population decreases in 1989 also occurred after fish stocking.

The distribution of five phytoplankton groups in selected samples from site L1 (near the dam) are shown in figure 10. The general pattern of algal succession was similar in both years of the study, even though the population was much larger the second year. Diatoms composed most of the biovolume in the spring. Green algae became a large fraction when the biovolume decreased each July. In 1989, the dinoflagellate *Ceratium hirundinella* also was abundant. Blue-green algae became dominant in August 1989 and in September 1990. Following a cold period in mid-September 1989, the total biovolume decreased and cryptomonads became dominant. After fall turnover in both years, *Melosira* blooms reestablished diatom dominance. Also in both years, blooms of blue-green algae occurred in mid-October. However, no large accumulations of algae were observed either in the lake or on the shoreline at any time during sample collection.

The occurrence of abundant populations of periphyton are summarized in table 4. Phytoplankton abundance also is listed for comparison. Many of the taxa listed were abundant in only one matrix, either on the bottom substrates or in the water column. Taxa detected exclusively on the bottom substrates included the green alga *Spirogyra* and the blue-green alga *Oscillatoria*. Both of these have been associated with odor problems (Palmer, 1977; Mallevialle and Suffet, 1987), but *Spirogyra* is associated with grassy odors, rather than the musty odors that have been a problem in Standley Lake. *Oscillatoria* is notable for producing organic compounds that create musty odors (Mallevialle and Suffet, 1987). Other taxa that were most often abundant on bottom substrates were the diatoms *Achnanthes, Asterionella, Fragillaria*, and *Melosira*, and the blue-green alga *Lyngbya*. Of these, *Achnanthes* and *Lyngbya* are not associated with odor problems, and the odor-producing species of *Asterionella (A. gracillima)* was never identified in samples from Standley Lake. The odor-producing species of *Fragillaria (F. construens)* was the most abundant species of *Fragillaria* on bottom substrates, both solid and sediment. *F. construens* can cause musty and geranium odors in water (Palmer, 1977). *Fragillaria* also was abundant in the water column during both years of the study, but the abundant species was *F. crotonensis*, which is not suspected not suspected of causing odor problems.

Algae abundant exclusively in the water column (phytoplankton) included the green alga *Chlamydomonas*, the diatom *Stephanodiscus*, various unidentified cryptomonads, the blue-green alga *Aphanizomenon*, and the dinoflagellate *Ceratium* (table 4). All of these have been associated with taste and odor problems in water, and in particular, musty odors have been associated with *Chlamydomonas*, *Anabaena*, and *Aphanizomenon* (Palmer, 1977; Mallevialle and Suffet, 1987). However, the maximum densities of these three algae in Standley Lake were all less than 1 percent of the critical limits for odor production, as reported by Mallevialle and Suffet (1987). The odor-producing diatom *Melosira* also was abundant in the water column, but at less than 4 percent of the critical limit.

NUTRIENT LOADING TO STANDLEY LAKE

Nutrient loading to Standley Lake was analyzed to determine the amounts of nitrogen and phosphorus supplied to the lake from external and internal sources. The primary external loading sources were assumed to be the surface-water

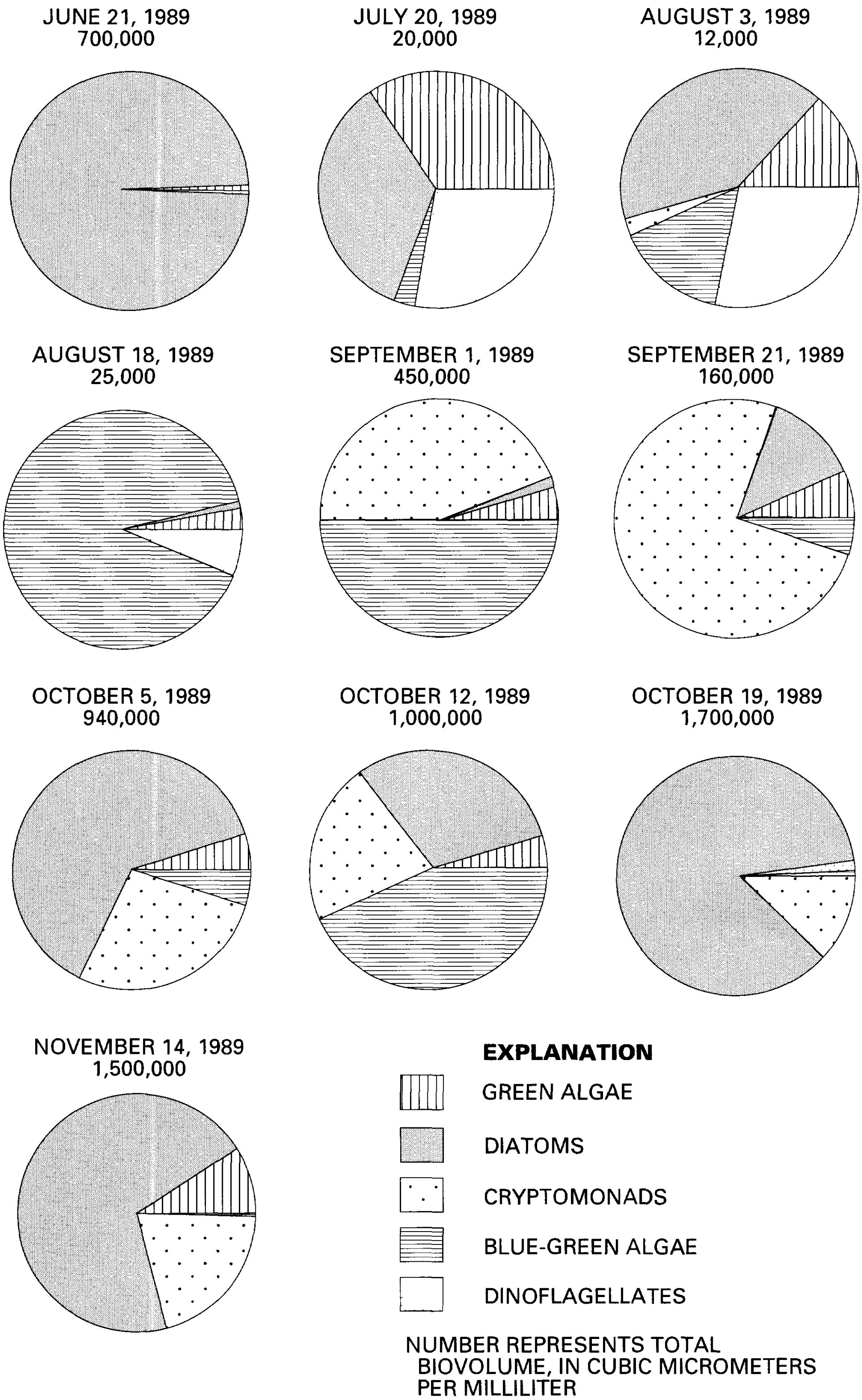


Figure 10.--Distributions of biovolume for five phytoplankton groups in selected samples from the photic zone at site L1 in Standley Lake.

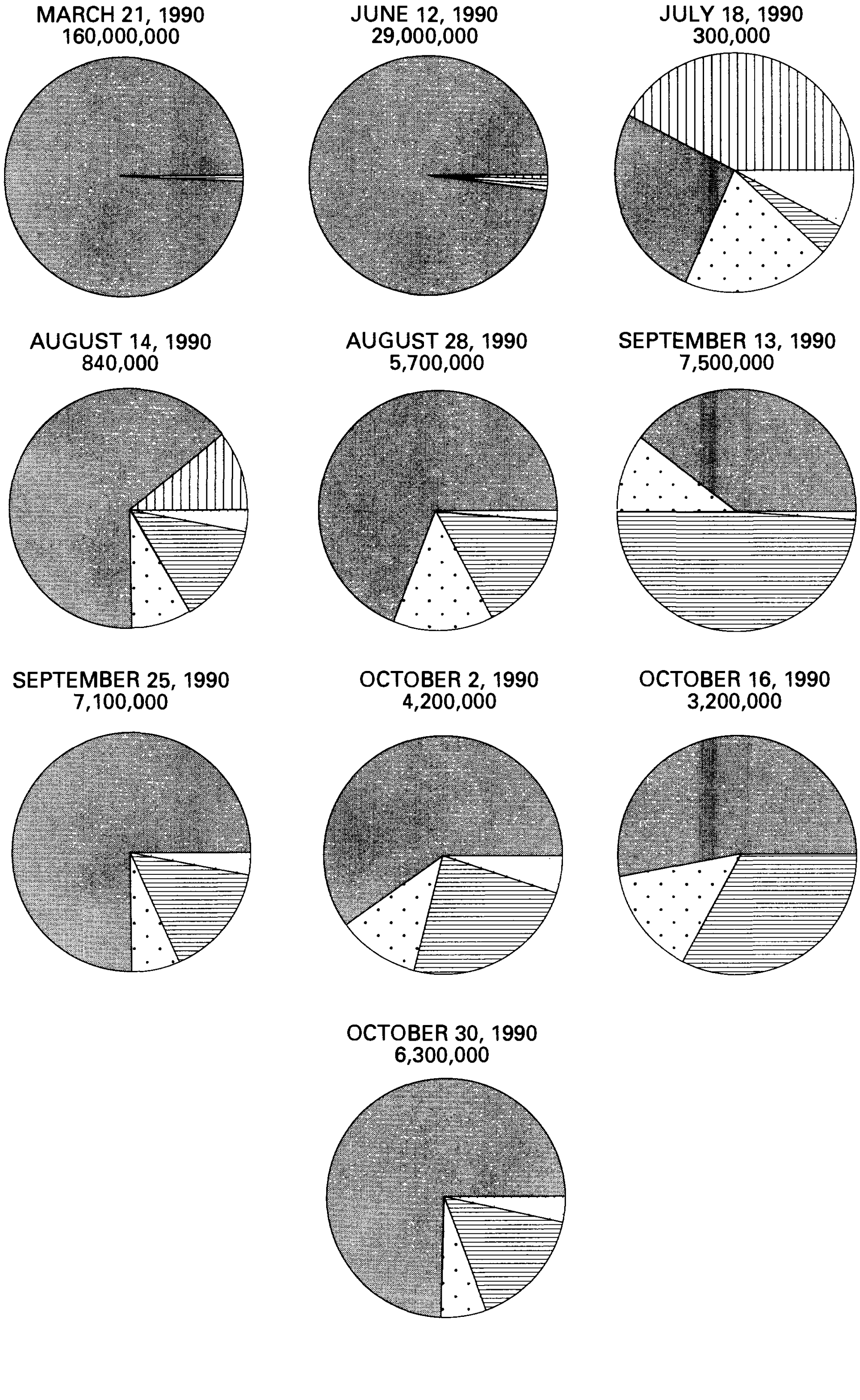


Figure 10.--Distributions of biovolume for five phytoplankton groups in selected samples from the photic zone at site L1 in Standley Lake.--Continued

Table 4.--Occurrences of abundant populations of algae taxa in Standley Lake, 1989 and 1990

[Data are compilation of samples collected at all phytoplankton and periphyton data-collection sites]

Group Genus[1]	Occurrences of periphyton abundance on bottom-sediment samples[2]		Occurrences of phytoplankton abundance in lake-water samples[3]	
	1989	1990	1989	1990
Green algae				
Chlamydomonas	0	0	0	11
Mougeotia	2	0	1	0
Pithophora	2	0	0	0
Spriogyra	7	0	0	0
Staurastrum	0	0	3	0
Diatoms				
Achnanthes	1	11	1	2
Asterionella	0	8	4	33
Fragillaria	4	13	2	11
Melosira	1	6	13	19
Nitzschia	2	2	0	1
Stephanodiscus	0	0	0	10
Surirella	2	0	0	1
Cryptomonads	0	0	16	35
Blue-green algae				
Anacystis	0	0	0	2
Aphanizomenon	0	0	2	1
Gomphosphaeria	0	0	0	12
Lyngbya	1	9	1	0
Merismopedia	0	0	4	7
Oscillatoria	0	7	0	0
Dinoflagellates				
Ceratium	0	0	1	22
Total number of samples	13	27	31	35

[1]Only taxa with more than one occurrence are listed.
[2]Abundance defined by the analyst (1989) or as density greater than 2,000 cells per square centimeter (1990).
[3]Abundance defined as biovolume greater than 20,000 cubic micrometers per milliliter.

inflows. The primary internal source was assumed to be flux from the bottom sediment. In the following sections, loading from these two primary sources will be analyzed separately and then compared to indicate their relative importance to the overall nutrient availability in the lake.

Loading from Inflow Sources

Data for analysis of nutrient loading from inflow sources were collected at three sites (IO1, IO2, and IO3; fig. 4 and table 2). Periodic water samples for nutrient analysis were collected at all sites. Sampling methods and analytical results are presented in a separate report (Ruddy and others, 1992). Time-series plots of total nitrogen and total phosphorus concentrations are shown in figure 11. Data for sites IO2 and IO3 are combined on a single graph because the primary source of water at both sites is the Church Ditch. Daily streamflow was measured at each inflow site. During April-October 1990, daily measurements of temperature and specific conductance were made on the inflow from the Farmers Highline and Croke Canals (site IO1). About 90 percent of the total inflow to the lake during the study period came from this source (table 1).

The mass discharge of a constituent at a point in a surface-water channel commonly is called the constituent load. It is a function of the constituent concentration and the streamflow. Numerous methods have been proposed for estimation of constituent loads in surface channels for which periodic concentration and daily streamflow data are available. Among the simplest of these is the time-interval method (Scheider and others, 1979), in which the data record is divided into discrete intervals on the basis of the dates of constituent sampling. Generally these divisions are made at the midpoint between sampling dates. The constituent load during the time interval is estimated as the product of the concentration measured in the interval and the total streamflow during the interval. Time-interval loads can then be summed to estimate monthly and annual loads. This is the method that has been used in previous studies of nutrient loading to Standley Lake (Richard P. Arber Associates, 1987). The primary disadvantages of this method are: (1) The statistical uncertainty of the estimated loads cannot be determined and (2) censored values (less than detection) of constituent concentration in a time interval must be assigned an arbitrary value. These problems can be avoided if the method is modified so each time interval includes several samples. Then the mean concentration and its standard error can be calculated for each time interval, and the load is estimated as the product of the mean concentration and the total streamflow during the interval. The statistical uncertainty of this estimated load can be determined by using the standard error of the mean concentration. Censored values can be incorporated into the mean concentration by using techniques described by Helsel and Cohn (1988).

Many studies have determined that time-interval methods of load estimation generally are more biased and less precise than methods based on a relation between concentration and streamflow (for example, Westerdahl and others, 1981; Richards and Holloway, 1987; Young and others, 1988). In natural, unregulated channels, constituent concentrations generally can be approximated as a power function of streamflow, usually linearized to a relation between the logarithms of concentration and streamflow. However, factors other than streamflow may be more important in determining constituent concentrations in canals, such as those that provide the inflow to Standley Lake. Rainfall and snowmelt in the Standley Lake watershed, which were followed by changes in constituent concentrations in the canal water, often did not cause a change in the flow released from the canals to the lake. Sometimes precipitation in the Clear Creek watershed upstream from the canal headgates (fig. 2) also would affect constituent concentrations in the canals without changing flow. The maximum phosphorus concentration measured during the study in the inflow from the Farmers Highline and Croke Canals (site IO1) resulted from such a precipitation event. At other times, a change in demand for water in a canal could result in a large increase or decrease in inflow to the lake, but the constituent concentrations would be completely unaffected.

An analysis of linear and nonlinear (logarithmic and rank) correlations indicated generally weak relations between streamflow and nutrient concentrations in inflows to Standley Lake (table 5). Correlation coefficients were greater than

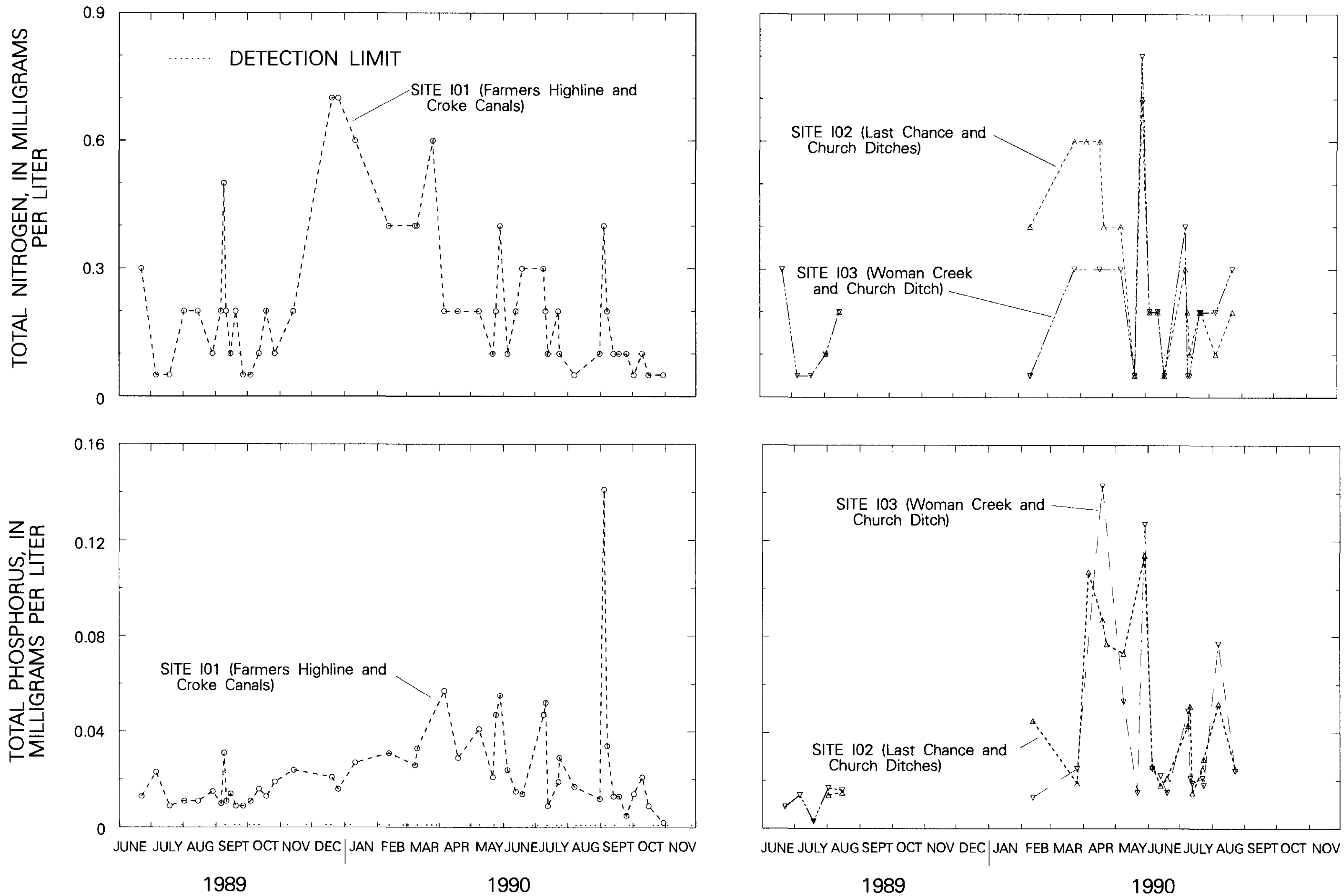


Figure 11.--Total nitrogen and total phosphorus concentrations at the inflow sites. (Values less than the detection limit are plotted at one-half the detection limit.)

Table 5.--Correlation of nutrient concentrations to streamflow and specific conductance at the inflow sites

[--, coefficient not applicable]

Site (see fig. 4)	Type of correlation	Coefficient for correlation between: Total nitrogen and streamflow	Total phosphorus and streamflow	Total nitrogen and specific conductance	Total phosphorus and specific conductance
IO1	Linear	0.09	0.01	0.48	0.06
	Logarithmic	0.33	0.11	--	--
	Rank	0.45	0.19	0.25	0.20
IO2	Linear	0.65	0.79	--	--
	Logarithmic	0.35	0.63	--	--
	Rank	0.46	0.70	--	--
IO3	Linear	-0.08	-0.12	--	--
	Logarithmic	0.17	0.06	--	--
	Rank	-0.15	-0.03	--	--

0.50 for only one inflow site (Last Chance and Church Ditches, site IO2). Coefficients for the correlation between total phosphorus concentrations and streamflow at the other two sites were all less than 0.20. Because about 90 percent of the total inflow volume came from the Farmers Highline and Croke Canals, daily specific conductance was monitored at site IO1. However, the correlations between nutrient concentrations and specific conductance at this site also were weak (table 5). Because no consistently strong relation was determined between nutrient concentrations and any daily-measured property, relational methods of load estimation were considered inappropriate for this study.

The procedure used to estimate nutrient loading to Standley Lake was a modification of the time-interval method. The time-series plots shown in figure 10 were analyzed to stratify the study period into time intervals on the basis of changes in nutrient concentrations. A single set of time intervals was defined for each inflow site, on the basis of total nitrogen and total phosphorus concentrations. Data for sites IO2 and IO3 were combined because the number of samples from either site was limited, and the primary source of flow at both sites was the Church Ditch. In some instances, an obvious change in concentration occurred following a change in canal operation. For example, the large increase in total nitrogen concentration at site IO1 in December 1989 occurred after the source of flow was shifted from the Farmers Highline Canal to the Croke Canal. In such instances, the boundary of the time interval was set at the date of the operational change.

Four time intervals were identified for site IO1 and three time intervals were identified for sites IO2 and IO3. The data for sites IO2 and IO3 were combined to compute the means and standard errors for the two summer/fall time intervals when the Church Ditch was the primary source of flow at both sites. For the winter/spring time interval, means and standard errors were computed separately at each site. During this time interval, the Last Chance Ditch provided much of the flow at site IO2, while the flow at site IO3 was primarily natural runoff. Differences in total nitrogen concentrations between the two sites were particularly obvious during February through early May (fig. 11). The mean concentrations and standard errors of total nitrogen and total phosphorus concentrations for each time interval are listed for site IO1 in table 6 and for sites IO2 and IO3 in table 7. The distributions of total nitrogen and total phosphorus concentrations within each time interval are shown in figure 12.

Nutrient loading to the lake from each site during each time interval was estimated using the mean concentrations for the time interval and the daily flows during the time interval. Because concentrations of total nitrogen occasionally were less than analytical detection (0.1 mg/L), mean values were computed by using censored-data techniques (Helsel and Cohn, 1988). Total nitrogen and total phosphorus loads were computed on a daily basis and then summed by month. Monthly inflow loads for the study period estimated by the mean method are listed in table 8. For comparison, load values computed by the midpoint method used in previous studies of Standley Lake also are listed. Although mean-method and midpoint-method estimates differed for individual months, the total loads estimated for the study period generally were within 10 percent. Therefore, estimates based on the time-interval mean concentration are comparable to estimates in previous studies.

To determine the retention of nutrients in Standley Lake, the monthly outflow loads were estimated using the mean method, as described for the inflow loads. Total nitrogen and total phosphorus concentrations in the raw-water delivery to the Semper Water Treatment Plant (site IO4) are shown in figure 13. Mean concentrations and standard errors for the designated time intervals are listed in table 9; data distributions within the time intervals are plotted in figure 14. Comparison of estimated inflow and outflow loads, computed by using the mean method, indicates 4,600 kg of nitrogen and 690 kg of phosphorus were retained in Standley Lake during the study period (table 8). This retention represents about 30 percent of the inflow load of nitrogen and about 41 percent of the inflow load of phosphorus.

Flux from the Bottom Sediment

The process of nutrient release from lake-bottom sediment has been studied since the 1940's. Release of nitrogen and phosphorus from bottom sediment can occur by a variety of mechanisms under a wide range of environmental conditions (Forsberg, 1989). In some instances, phosphorus release has been shown to be a substantial source of the in-lake supply such as in Lake Sammamish (Welch and Spyridakis, 1972), Shagawa Lake (Larsen and others, 1981), and Lake Sevan (Parparova, 1990). In the Standley Lake study, nitrogen and phosphorus fluxes from the bottom sediment were estimated for comparison to the inflow loading.

Several methods are available for estimation or measurement of nutrient flux. Sealed chambers, such as a benthic respirometer (Lantrip and others, 1987) or a box core (Carlton and Wetzel, 1988), can be installed in the field or set up in the laboratory to measure nutrient release, which is defined as the concentration increase in the overlying water during a period of time. Alternatively, the flux can be computed based on the concentration gradient measured between the pore water in the sediment and the overlying water in the lake. The gradient method was chosen for use in the Standley Lake study, because the data collection was less complicated than for the direct measurement methods. Seepage through the fine sediment at the bottom of the lake was assumed to be negligible; therefore, hydraulic gradients were not considered an impedance to constituent movement in the direction of concentration gradients.

Table 6.--Summary statistics for total nitrogen and total phosphorus concentrations in the inflow from the Farmers Highline and Croke Canals (site I01)

[Mean concentrations and standard errors in milligrams per liter; censored (less than detection) values were incorporated by using techniques described by Helsel and Cohn (1988)]

	Time interval			
	June 18, 1989 to November 12, 1989	November 13, 1989 to April 4, 1990	April 5, 1990 to July 11, 1990	July 12, 1990 to October 31, 1990
	TOTAL NITROGEN			
Number of samples	16	8	11	14
Mean	0.16	0.50	0.22	0.12
Standard error	0.029	0.063	0.026	0.025
	TOTAL PHOSPHORUS			
Number of samples	16	7	11	14
Mean	0.014	0.025	0.037	0.024
Standard error	0.0015	0.0022	0.0049	0.0093

Table 7.--Summary statistics for total nitrogen and total phosphorus concentrations in the inflows from the Last Chance and Church Ditches (site I02) and from Woman Creek and the Church Ditch (site I03) during selected time intervals

[Mean concentrations and standard errors in milligrams per liter; censored (less than detection) values were incorporated by using techniques described by Helsel and Cohn (1988)]

	Time interval			
	June 16, 1989 to September 19, 1989	January 2, 1990 to May 31, 1990		June 1, 1990 to September 19, 1990
	Sites I02 and I03	Site I02	Site I03	Sites I02 and I03
	TOTAL NITROGEN			
Number of samples	7	8	6	20
Mean	0.14	0.50	0.32	0.19
Standard error	0.036	0.050	0.10	0.18
	TOTAL PHOSPHORUS			
Number of samples	7	7	6	20
Mean	0.013	0.075	0.063	0.030
Standard error	0.0019	0.013	0.024	0.0036

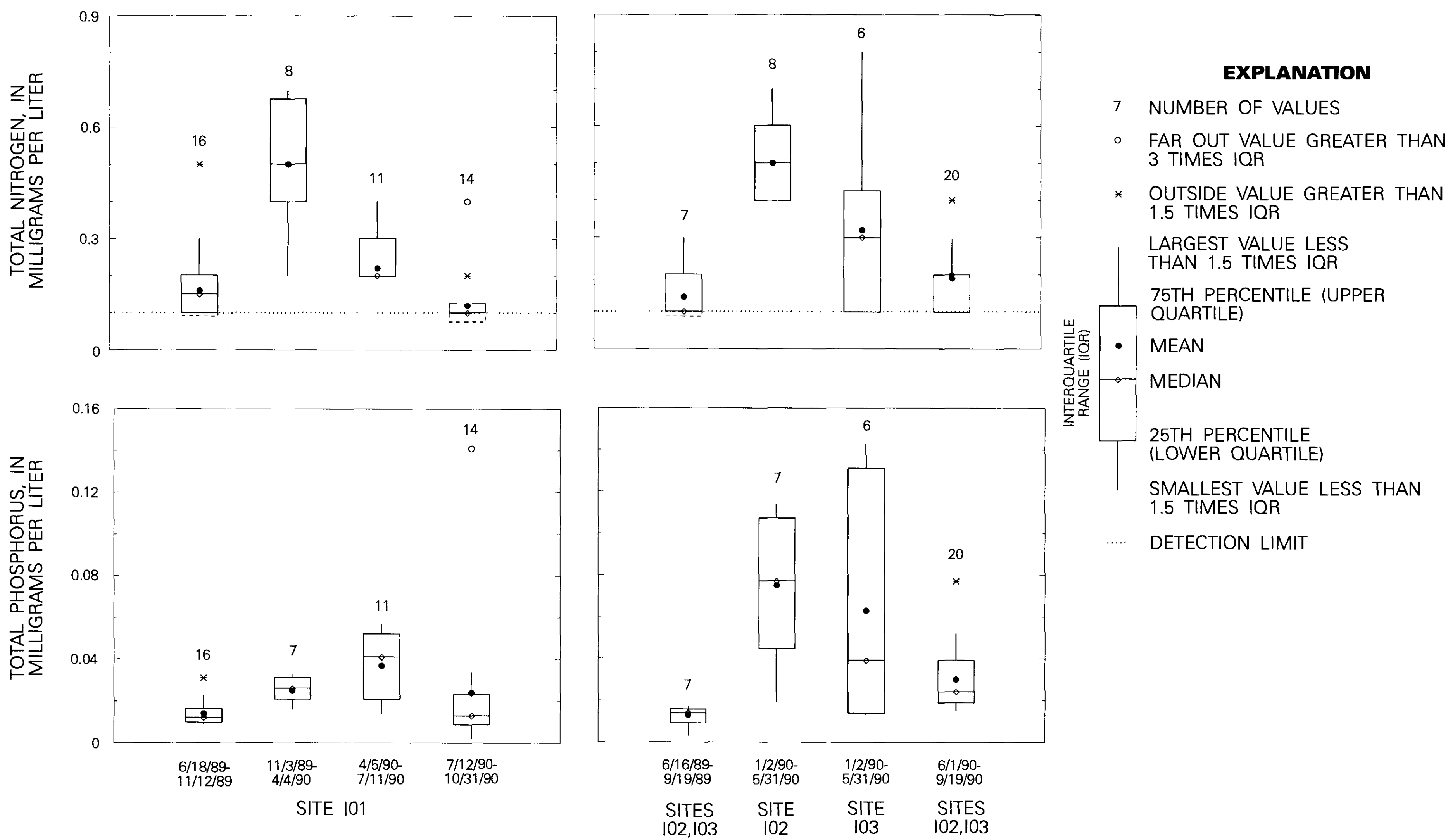


Figure 12.--Distributions of total nitrogen and total phosphorus concentrations at the inflow sites during selected time intervals.

Table 8.--Estimated monthly loads of total nitrogen and total phosphorus in Standley Lake inflows and outflow during the study period

[Values less than 1,000 are rounded to two significant figures; values greater than 1,000 are rounded to three significant figures]

Month	Constituent load, in kilograms per month									
	Farmers Highline and Croke Canals (site I01)		Last Chance and Church Ditches (site I02)		Woman Creek and Church Ditch (site I03)		Total inflow		Total outflow (site I04)	
	Mean method	Mid-point method	Mean method	Mid-point method	Mean method	Mid-point method	Mean method[1]	Mid-point method[2]	Mean method	Mid-point method
					TOTAL NITROGEN					
1989										
July	700	300	5.0	18	78	35	780	350	1,500	1,810
August	290	340	15	17	74	88	370	440	1,310	1,710
September	330	350	2.0	2.8	0.0	0.0	330	360	790	930
October	340	270	0.0	0.0	0.0	0.0	350	270	310	300
November	820	400	0.0	0.0	0.0	0.0	820	400	170	110
December	1,340	1,840	0.0	0.0	0.0	0.0	1,340	1,840	170	380
1990										
January	1,310	1,510	8.2	6.6	10	1.6	1,320	1,520	170	330
February	1,000	800	8.3	6.7	8.7	1.4	1,010	810	150	85
March	1,750	1,720	100	120	80	74	1,930	1,910	170	94
April	490	420	420	460	84	79	1,000	960	190	130
May	800	1,040	320	220	45	80	1,160	1,340	880	470
June	3,200	2,930	54	63	200	120	3,440	3,110	1,650	1,120
July	520	700	26	30	150	150	700	890	860	1,140
August	66	30	24	20	48	53	130	100	1,200	810
September	220	300	7.2	7.6	21	30	250	330	880	500
October	250	130	0.0	0.0	0.0	0.0	250	130	200	160
Total	13,400	13,100	990	970	800	710	15,200	14,800	10,600	10,100

Table 8.--Estimated monthly loads of total nitrogen and total phosphorus in Standley Lake inflows and outflow during the study period--Continued

Month	Constituent load, in kilograms per month									
	Farmers Highline and Croke Canals (site I01)		Last Chance and Church Ditches (site I02)		Woman Creek and Church Ditch (site I03)		Total inflow		Total outflow (site I04)	
	Mean method	Mid-point method	Mean method	Mid-point method	Mean method	Mid-point method	Mean method[1]	Mid-point method[2]	Mean method	Mid-point method
	TOTAL PHOSPHORUS									
1989										
July	61	69	0.46	0.75	7.2	5.9	69	76	130	110
August	25	20	1.4	1.5	6.9	8.7	33	31	110	85
September	29	25	0.183	0.21	0.0	0.0	29	525	67	69
October	30	32	0.0	0.0	0.0	0.0	30	32	42	41
November	45	49	0.0	0.0	0.0	0.0	45	49	23	33
December	67	51	0.0	0.0	0.0	0.0	67	51	23	21
1990										
January	65	70	1.2	0.74	2.0	0.41	68	71	23	20
February	50	60	1.2	0.75	1.7	0.35	53	61	20	14
March	88	130	15	3.8	16	6.2	120	140	23	8
April	78	78	64	77	17	32	160	190	25	29
May	130	170	47	55	8.9	13	190	240	76	110
June	540	290	8.5	9.1	28	20	580	320	140	140
July	92	100	4.1	4.4	21	20	120	120	74	56
August	13	9.0	3.7	4.3	6.9	14	24	27	100	80
September	44	63	1.1	0.95	3.0	2.4	48	67	75	90
October	49	22	0.0	0.0	0.0	0.0	49	22	27	29
Total	1,400	1,240	150	160	120	120	1,670	1,520	980	940

[1]Numbers are summation of mean method inflow values from Farmers Highline and Croke Canals, Last Chance and Church Ditches, and Woman Creek and Church Ditch.

[2]Numbers are summation of midpoint method inflow values from Farmers Highline and Croke Canals, Last Chance and Church Ditches, and Woman Creek and Church Ditch.

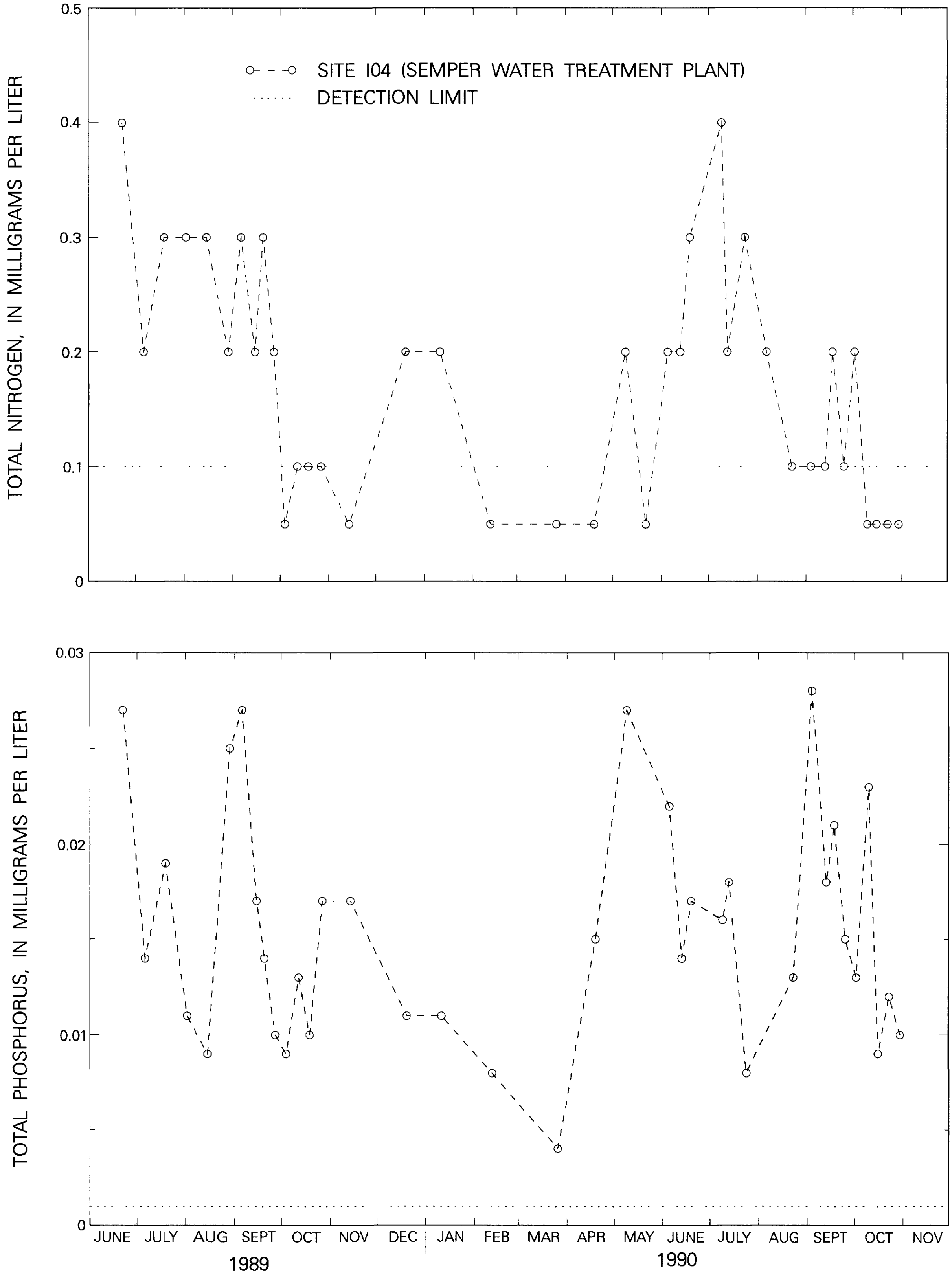


Figure 13.--Total nitrogen and total phosphorus concentrations at the outflow site. (Values less than the detection limit are plotted at one-half the detection limit.)

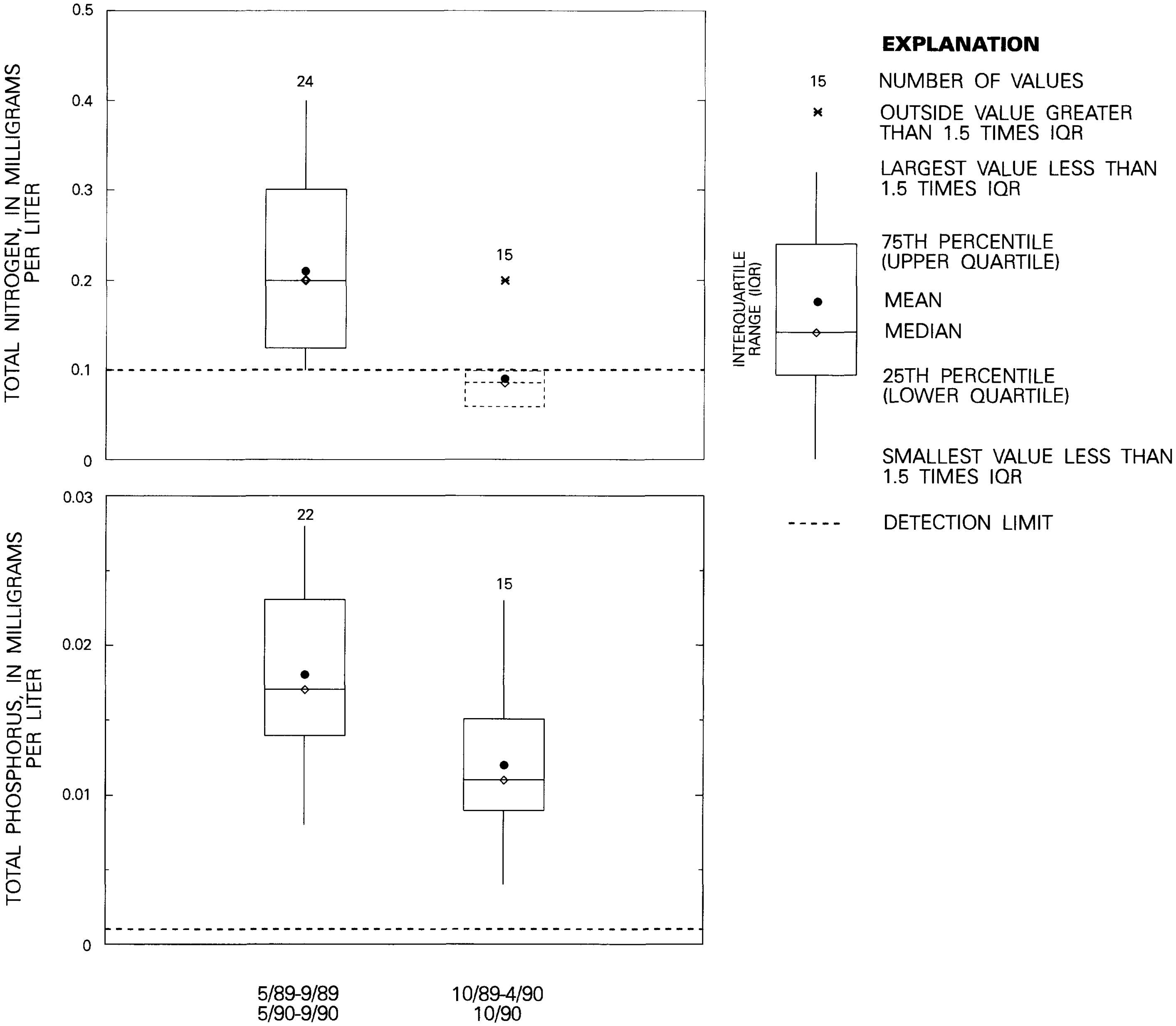


Figure 14.--Distributions of total nitrogen and total phosphorus concentrations at the outflow site during selected time intervals.

Table 9.--Summary statistics for total nitrogen and total phosphorus concentrations in the Standley Lake outflow (site 104) during selected time intervals

[Mean concentrations and standard errors in milligrams per liter; censored (less than detection) values were incorporated by using techniques described by Helsel and Cohn (1988)]

	Time interval	
	June-September 1989 and May-September 1990	October 1989-April 1990 and October 1990
	TOTAL NITROGEN	
Number of samples	24	15
Mean	0.21	0.090
Standard error	0.018	0.016
	TOTAL PHOSPHORUS	
Number of samples	22	15
Mean	0.018	0.012
Standard error	0.0013	0.0012

In the gradient method, mass flux across the sediment-water interface is assumed to be a diffusive process that can be described by Fick's law (Callender, 1982; Ullman and Aller, 1982):

$$J = -\Phi \ D_s \left(\frac{\partial C}{\partial x}\right) \qquad (1)$$

where

J = mass flux, in milligrams per square centimeter per month;

Φ = the porosity of the sediment;

D_s = the bulk-sediment diffusion coefficient, in square centimeters per month; and

$\frac{\partial C}{\partial x}$ = the concentration gradient, in grams per liter per centimeter (equivalent to mg/cm^4).

Monthly loading from the bottom sediment is estimated as the product of the computed flux and the contributing area in the lake.

The diffusion coefficient (D_s) is a function of the free-solution diffusion coefficient at infinite dilution (D_o), corrected for temperature, for the particular nutrient species. In this study, D_o for nitrogen was computed as:

$$D_{o_{(N)}} = m \ (0.398 \ T + 9.76) \qquad (2)$$

where

m = a factor to correct the units, in 10^6 seconds per month; and

T = the temperature, in degrees Celsius;

and D_o for phosphorus was computed as:

$$D_{o_{(P)}} = m \ (0.187 \ T + 2.66) \qquad (3)$$

(E.C. Callender, U.S. Geological Survey, written commun., 1989; after Li and Gregory, 1974). D_s for both nutrients was then computed as:

$$D_s = \frac{D_o}{\Phi F} \qquad (4)$$

where

F = the modified formation factor (Ullman and Aller, 1982).

The formation factor accounts for tortuosity, which is related to the flow path through the sediment and is a hindrance to diffusion. This factor has been estimated empirically as

$$F = \frac{1}{\Phi^n} \qquad (5)$$

where n varies from 1.3 to 5.4 depending on the sediment material. In this study, n was assumed to be 3.0, which is a reasonable value for mud and clay sediments (E.C. Callender, U.S. Geological Survey, written commun., 1989).

Sampling of bottom sediment and pore water in Standley Lake is discussed briefly in the "Data Collection" section of this report, and in more detail in a separate report (Ruddy and others, 1992). Sediment was sampled at 10 sites in July 1989 to characterize the distribution of nutrients and other chemical constituents in the bottom of the lake. The highest concentrations of nitrogen and phosphorus occurred at the two deep sites (L1, near the dam, and L2, near the center). The water level at these sites was more than 60 ft deep throughout the study. Water levels at the remaining eight sites were no more than 45 ft deep. Sediment from the two deep sites also had the highest concentrations of organic carbon, iron, and several trace elements. Total phosphorus concentrations at these two sites (630 mg/kg) were more than double the mean concentration at the other sites (about 260 mg/kg). Therefore, the potential for nutrient release was considered highest in the deeper areas of the lake, and sampling of pore water for flux computation was focused on the two deep sites.

Sediment cores were collected at sites L1 and L2 (fig. 3) in August and October 1989. The August samples were collected during a period of anoxia in the hypolimnion that was associated with thermal stratification. The October samples were collected under low-oxygen conditions just prior to turnover. Preliminary analysis of the pore water extracted from these cores indicated substantial nutrient flux into the lake-bottom water at these two sites. Based on these results, a third set of samples was collected in May 1990 during a period of high-oxygen concentration at the lake bottom. These samples were collected from sites L1 and L2 to evaluate potential nutrient flux under oxygenated water, and from four sites (L4, L5, L8, and L10; fig. 3) in shallower water to evaluate the areal extent of potential nutrient flux. A final set of samples was collected at sites L1, L2, and L5 in August 1990 to evaluate the annual variation in potential nutrient flux. In all instances, pore water was extracted from 1-cm increments of the sediment cores. Pore-water samples were extracted from the upper 10 cm of cores collected in 1989 and from the upper 4 cm of cores collected in 1990.

The dissolved nitrogen and dissolved phosphorus concentration gradients in sediment pore water sampled in May 1990 are plotted in figure 15. The steepest gradients from the sediment to the overlying water occurred at the three deepest sites (L1, L2, and L5). Phosphorus concentration had essentially no gradient between the upper 1 cm of sediment and the overlying water at the three shallowest sites (L4, L8, and L10). The gradient in nitrogen concentration at site L8 (near the west side) was only slightly less than the gradient at site L5 (near the island), but the sediment was almost dry below the upper 1 cm. On the basis of this evidence, the bottom area represented by sites L1, L2, and L5 was considered potentially contributory to nutrient flux. The depth of water at all these sites was at least 40 ft, which was the depth of the top of the anoxic hypolimnion during both years of the study. Therefore, the bottom area used to compute nutrient loading from benthic-flux estimates was defined to be the area below a depth of 40 ft at normal pool elevation. This area was determined to be 404 acres in a previous study (Hydrotriad, Ltd., 1981).

The nutrient-concentration gradients from the sediment pore water to the lake as determined from August and October samples (fig. 16) generally were steeper than the gradients determined from May samples (fig. 15). The concentration profiles in the upper 10 cm of sediment were fairly uniform for nitrogen, but very erratic for phosphorus. The large oscillations shown in the phosphorus profiles were unexpected, and might be attributed either to random contamination during extraction of the pore water and preparation of samples or to problems with laboratory analysis of small-volume samples. However, the phosphorus profile plotted from the mean concentrations for all August and October samples was reasonably uniform (fig. 17). In addition, the standard error of the mean concentration in the upper 1 cm of the sediment was small. Therefore, this mean was considered adequate for determining the gradient used in the flux computation. In order to maintain consistency in the analysis, all nutrient gradients required in flux computations were based on mean concentrations in the upper 1 cm of the sediment. Use of mean values also enabled determination of statistical uncertainty in the gradients and the resultant estimates of nutrient loading from the sediment. The mean values and standard errors of nutrient concentrations in the upper 1 cm of sediment for all samples from sites L1, L2, and L5 are listed in table 10. Separate values are listed for samples collected in May and for samples collected in August and October. The May values were used to compute nutrient flux during the winter and nonstratified periods, October through May. The August and October values were used to compute flux during the months of summer stratification, June through September. The October samples were collected early in the month and were considered more representative of the

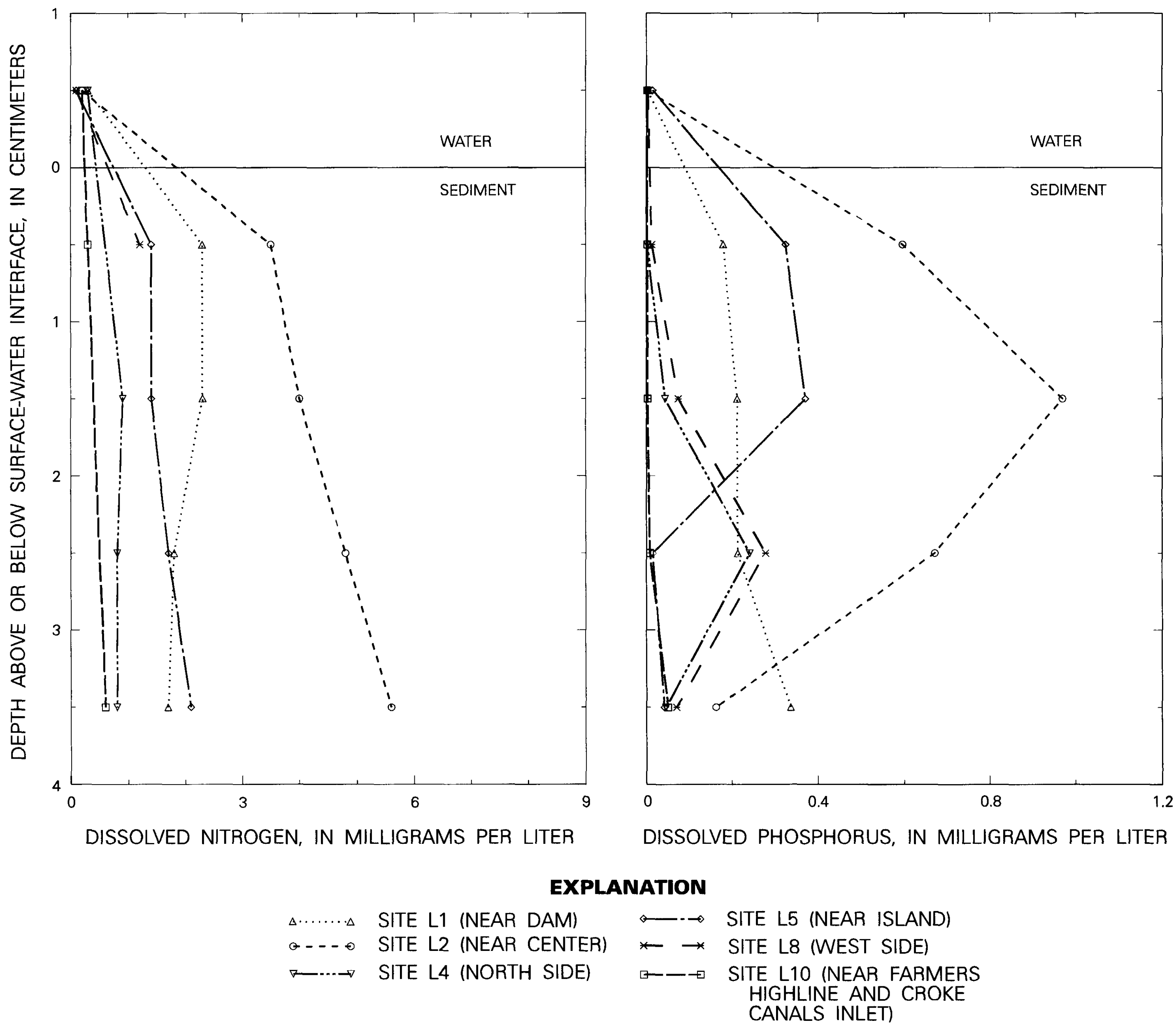


Figure 15.--Dissolved nitrogen and dissolved phosphorus gradients in the sediment pore water of Standley Lake, May 1990.

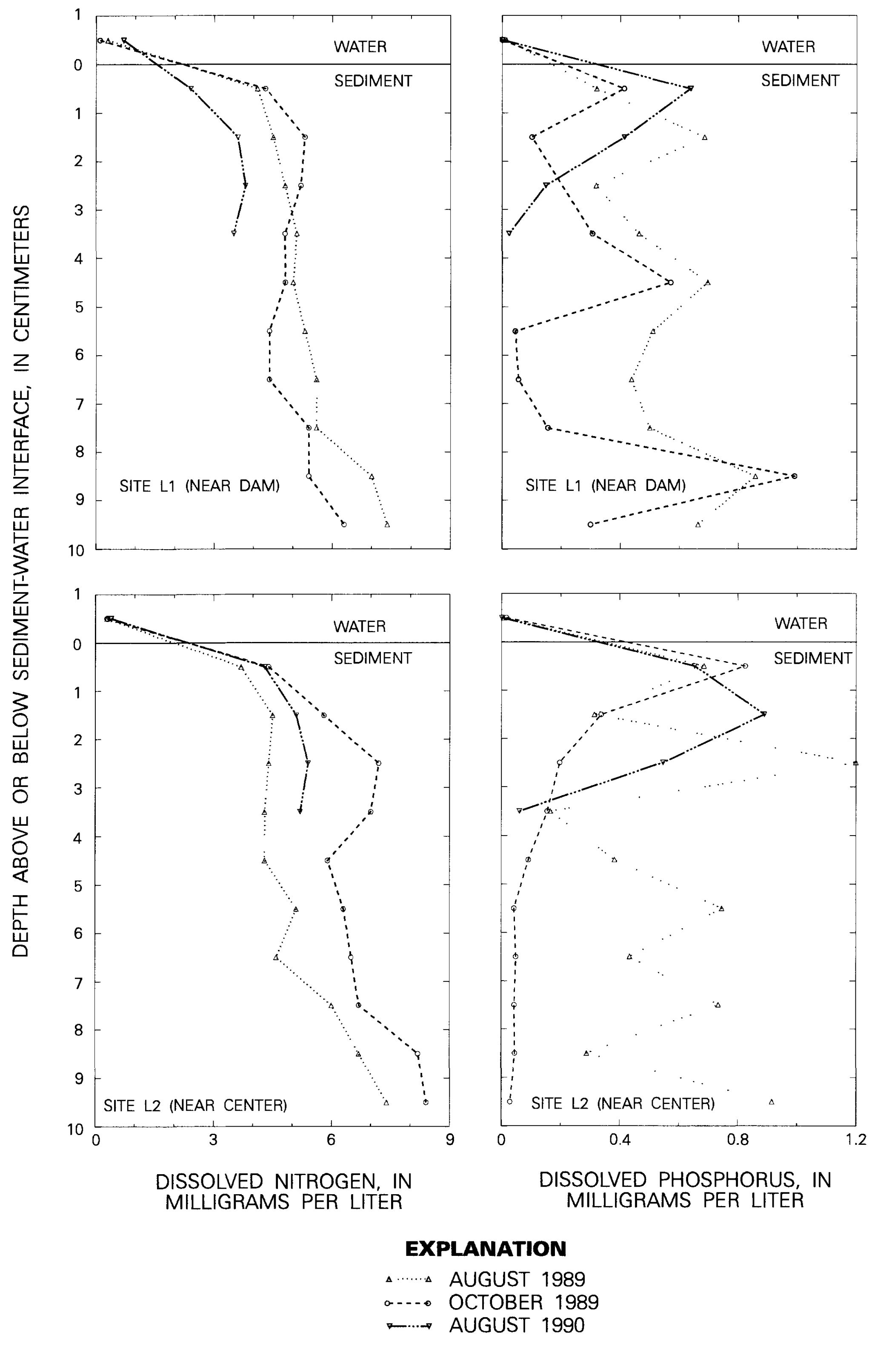


Figure 16.--Dissolved nitrogen and dissolved phosphorus gradients in the sediment pore water of Standley Lake, August and October 1989 and August 1990.

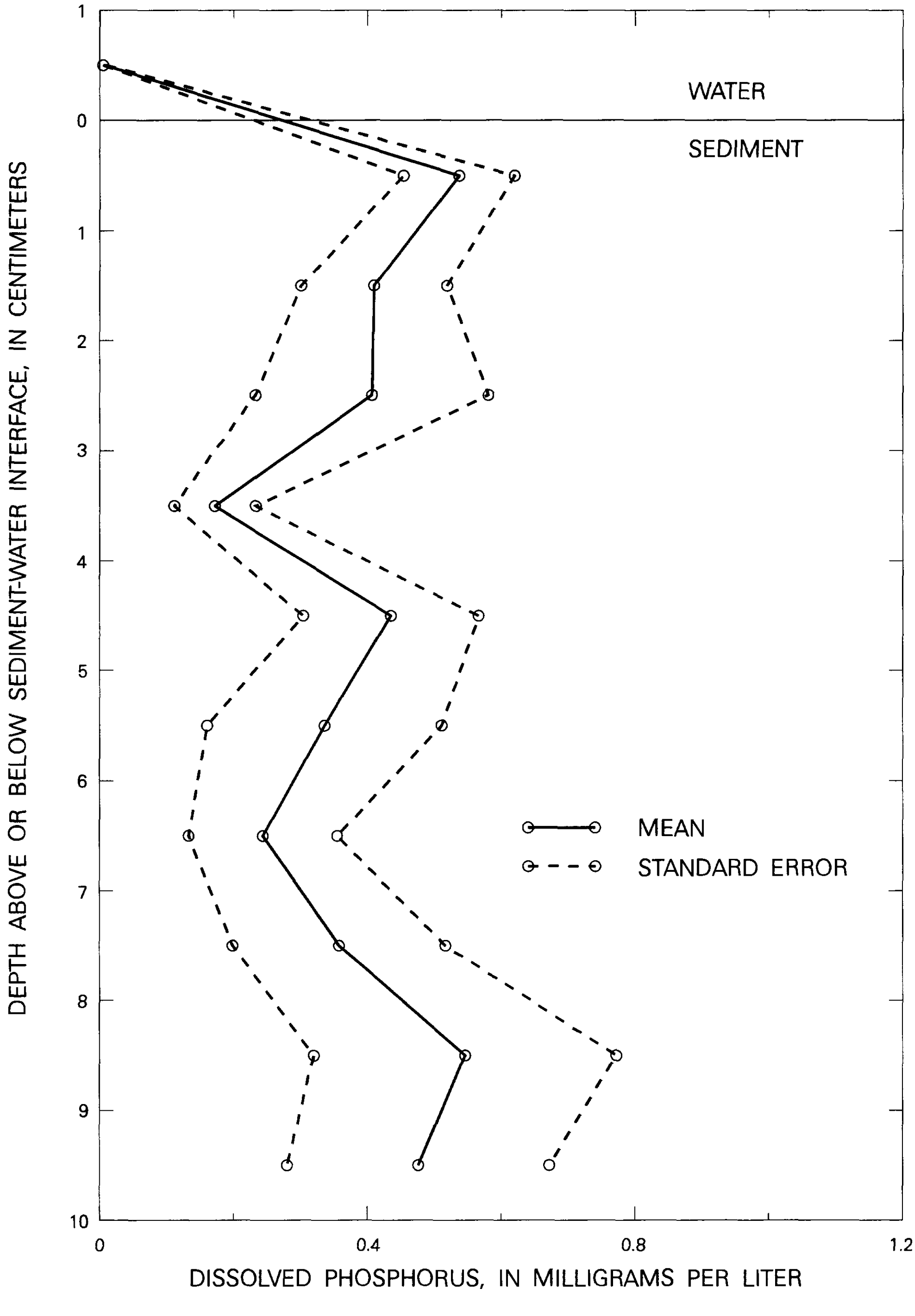


Figure 17.--Gradient of mean dissolved phosphorus concentrations in the sediment pore water of Standley Lake, August and October 1989 and August 1990.

stratified period, which ended with fall turnover during the first week of October in 1989 and 1990.

The data used to compute nutrient flux from the bottom sediment and the resultant monthly loadings are listed in table 11. Temperature and porosity are required for computation of the diffusion coefficient. Monthly temperature is the mean of all measurements at the bottom of the lake profiles during each month. No data were available for January and February; therefore, temperatures used for these months were based on December and March measurements. Porosity is the mean of all data from the upper 1 cm at sites L1, L2, and L5. Nutrient concentrations in the lake are mean values from analyses of bottom samples collected at sites L1 and L2 throughout the study. Data were combined for samples collected during October through August because the range of measured concentrations was small. Concentrations in samples collected during September generally were higher, particularly for phosphorus; therefore, separate means were computed for the September data.

Results of the flux and loading computations (table 11) indicate that the bottom sediment in Standley Lake provides a considerable amount of nitrogen and phosphorus to the overlying water. About 60 percent of the total loading occurs during the period of summer stratification (June-October) and probably contributes to the increase in nutrient concentrations in the hypolimnion.

To compare the loading from the bottom sediment to loading from the inflows, loads and standard errors were computed for a 1-year period (table 12). For the year, nitrogen loading from the bottom sediment was almost identical to loading from all inflow sources. However, the phosphorus loading from the bottom was less than 50 percent of the inflow loading, and the difference was statistically significant ($p < 0.001$).

The significance of the difference between inflow and bottom-sediment loads was determined on the basis of the statistical uncertainty in nutrient concentrations measured in both sources. For inflow sources, the standard error of estimated load in table 12 was computed using the standard errors of nutrient concentrations in the inflow water during defined time intervals (tables 6 and 7). For bottom sediment, the standard error of estimated load was computed based on the standard error of nutrient concentrations in the upper 1 cm of sediment pore water (table 10). Other possible errors, including uncertainties in streamflow measurement and laboratory analyses, were not considered. The total error for each source was computed as the square root of the sum of the individual squared errors. Other sources of error, such as uncertainty of the estimate of contributing area, were not considered.

The assumption that the primary sources of loading are surface inflows and flux from the bottom sediment was checked by comparing these load estimates with an estimate of annual atmospheric loading. Mean concentrations of total nitrogen (1.56 mg/L) and total phosphorus (0.08 mg/L) in wetfall precipitation in the Denver area were obtained from a 1980-81 urban storm-water study (Gibbs and Doerfer, 1982, p. 262-274). Precipitation amounts near Standley Lake during 1989-90 were obtained from measurements made at the Semper Water Treatment Plant (Sharon Bernia, City of Westminster, written commun., 1991). Based on precipitation over the full-pool area of the lake, annual atmospheric loading of nitrogen was estimated to be about 3,100 kg, or about 12 percent of the combined annual load from inflow and bottom-sediment sources (table 12). The annual atmospheric loading of phosphorus was estimated to be about 170 kg, or about 8 percent of the combined load from inflow and bottom-sediment sources. Although not negligible, atmospheric loading is small and within the error of load estimates from inflow and bottom-sediment sources. Therefore, the original assumption seems reasonable.

The relation between nutrient loading and algal growth is affected by several factors other than simply mass input to the lake. About 50 percent of the flux from the bottom sediment occurs during the summer while the lake is stratified; therefore, the nutrients are trapped in the hypolimnion and are unavailable to algae. At the time of fall turnover, the nutrient supply from the hypolimnion might mix into the photic zone, and if other conditions are right, cause an increase in algal growth. However, prior to turnover, some of the hypolimnetic nutrient supply might be flushed from the lake in bottom-level withdrawals. The chemical form of the nutrients also is important. Most of the nitrogen loading from all sources and the phosphorus loading from the bottom sediment

Table 10.--Summary statistics for dissolved nitrogen and dissolved phosphorus concentrations in the upper 1 centimeter of sediment pore water at sites L1, L2, and L5 in Standley Lake

[mg/L, milligrams per liter; censored (less than detection) values were incorporated by using techniques described by Helsel and Cohn (1988)]

Time of sampling	Number of samples[1]	Nitrogen (mg/L)		Phosphorus (mg/L)	
		Mean	Standard error	Mean	Standard error
May 1990	3	2.4	0.61	0.367	0.122
August 1989 and 1990 and October 1989	7	3.6	0.36	0.638	0.083

[1] Samples were collected from site L5 only in May and August 1990.

Table 11.--Monthly data used in the computation of nutrient flux from the bottom sediment and resultant nutrient loads

[Values rounded to no more than three significant figures]

Month	Temperature (degrees Celsius)	Porosity	Concentration (milligrams per liter)				Load (kilograms)	
			Nitrogen		Phosphorus			
			Lake	Pore water	Lake	Pore water	Nitrogen	Phosphorus
October	13.7	0.9	0.2	2.4	0.012	0.367	1,070	59
November	7.9	0.9	0.2	2.4	0.012	0.367	877	45
December	4.5	0.9	0.2	2.4	0.012	0.367	806	39
January	4.3	0.9	0.2	2.4	0.012	0.367	806	39
February	4.3	0.9	0.2	2.4	0.012	0.367	728	35
March	4.3	0.9	0.2	2.4	0.012	0.367	806	39
April	6.5	0.9	0.2	2.4	0.012	0.367	839	43
May	9.2	0.9	0.2	2.4	0.012	0.367	942	50
June	11.6	0.9	0.2	3.6	0.012	0.537	1,510	78
July	13.1	0.9	0.2	3.6	0.012	0.537	1,630	86
August	13.7	0.9	0.2	3.6	0.012	0.537	1,650	88
September	14.9	0.9	0.3	3.6	0.027	0.537	1,600	86
Total							13,300	687

Table 12.--Loads and standard errors of nitrogen and phosphorus from inflow and bottom-sediment sources to Standley Lake during a 1-year period

[Values rounded to no more than three significant figures]

Source Time interval	Nitrogen (kilograms)		Phosphorus (kilograms)	
	Load	Standard error	Load	Standard error
INFLOW				
Farmers Highline and Croke Canals				
10/01/89 - 11/12/89	356	65	31	3
11/13/89 - 04/04/90	6,260	793	317	28
04/05/90 - 07/11/90	4,790	566	806	107
07/12/90 - 09/30/90	438	91	88	34
Last Chance and Church Ditches				
01/02/90 - 05/31/90	861	86	129	22
Woman Creek and Church Ditches				
01/02/90 - 05/31/90	232	73	46	17
Last Chance and Church Ditches and Woman Creek				
06/01/90 - 09/19/90	477	45	75	9
INFLOW TOTAL[1]	13,400	988	1,490	119
BOTTOM SEDIMENT				
October - May	6,870	1,910	349	120
June - September	6,390	681	338	54
BOTTOM SEDIMENT TOTAL[1]	13,300	2,030	687	132

[1]Total standard error is the square root of the sum of the individual squared errors.

was in a dissolved form that is readily available for algal uptake. However, the available form of phosphorus (orthophosphate) averaged only about 20 percent of the total phosphorus concentration in inflow water samples.

Nutrient loading and availability also are dependent on conditions in the watershed or the area that contributes water to the canals upstream from Standley Lake. Loading values in this report pertain to conditions that existed during the period of data collection. If conditions change, loadings may be affected. Anthropogenic activities in areas that contribute runoff either directly to the lake or to the inflow canals could increase the loading of both nitrogen and phosphorus.

NUTRIENT LIMITATION TO ALGAE GROWTH IN STANDLEY LAKE

In-lake enclosure experiments, as described in the "Data Collection" section of this report, were used to evaluate the potential limitation of algal growth by the nutrients nitrogen and phosphorus. Small-enclosure (microcosm) experiments were done in 1989 and 1990 during summer stratification and immediately after fall turnover in October, which is the time when previous taste and odor problems had occurred. A large-enclosure (mesocosm) experiment was done concurrently with the microcosm experiment in October 1990. The mesocosm experiment was used to compare results of the simpler microcosm experiment to a more complex experiment that allowed for atmospheric exchange.

Four treatments of three replicates each were used for all experiments: (1) Control (no nutrient addition), (2) addition of nitrogen, (3) addition of phosphorus, and (4) addition of nitrogen plus phosphorus. For the July 1989 experiment, the additions were 1,000 μg nitrogen and 100 μg phosphorus per liter. For the August, September, and October 1989 experiments, the additions were 100 μg nitrogen and 10 μg phosphorus per liter. For the 1990 experiments, the additions were 200 μg nitrogen and 20 μg phosphorus per liter. Microcosms were incubated in the lake at one-half the Secchi-disk depth for 5 days and then were removed and sampled. Mesocosms were sampled on days 2, 4, and 8 after installation (day 1). The final samples (day 8) were used for comparison to the microcosm results. Algal response was determined by measuring the concentration of chlorophyll *a* and the phytoplankton biovolume in each enclosure.

The responses to nutrient additions, in terms of chlorophyll *a* concentration for each nutrient-limitation experiment, are shown in figure 18. For every experiment, the maximum response occurred in the nitrogen-plus-phosphorus treatments. Concentrations of chlorophyll *a* in each nitrogen-plus-phosphorus replicate were higher than in any other treatment replicate in every experiment. The concentrations in the single-nutrient treatments generally were not much different than in the control enclosures. The response, in terms of total phytoplankton biovolume, generally was similar to the chlorophyll *a* response, except for greater variability among replicates and, in some instances, less difference among treatments (fig. 19).

The statistical significance of differences among treatments was tested by using analysis of variance, and differences between individual treatments were evaluated by using Duncan's multiple-range test (SAS Institute, 1985). Duncan's test was selected because it minimizes the error of incorrectly determining that a difference between treatments is not significant. Separate analyses were made for data from each microcosm experiment and from the mesocosm experiment.

The results of these tests are listed for chlorophyll *a* concentration and total phytoplankton biovolume in table 13. The analysis of variance indicated at least one significant difference ($p \leq 0.1$) between treatments for both measures of algal response in every experiment except for phytoplankton biovolume in the October 1990 microcosms. The multiple-range test indicated that both measures of algal response in the nitrogen-plus-phosphorus treatments were significantly larger than in the control for every experiment.

Significant differences also were identified between the control and nitrogen treatments in the October 1989 and August 1990 microcosm experiments (chlorophyll *a*) and the October 1990 mesocosm experiment (chlorophyll *a* and phytoplankton). In all these instances, the algal growth increased owing to the addition of nitrogen, but was still less than the values attained with the addition of nitrogen plus phosphorus. The only significant difference between the control and phosphorus treatments was for concentrations of chlorophyll *a* in the September 1989 microcosm experiment, but in this instance the concentration in the phosphorus treatment was less than the concentration in the control (fig. 18).

The results of the concurrent microcosm and mesocosm experiments in October 1990 generally were similar. For these two experiments, the overall significance level of the analysis of variance was less than 0.01 for each comparison, except phytoplankton biovolume in the microcosms (table 13). For both experiments, the concentrations of chlorophyll *a* and the phytoplankton biovolume were significantly larger in the nitrogen-plus-phosphorus treatment than in the control. Neither measure of algal response in the phosphorus treatments was significantly different than in the

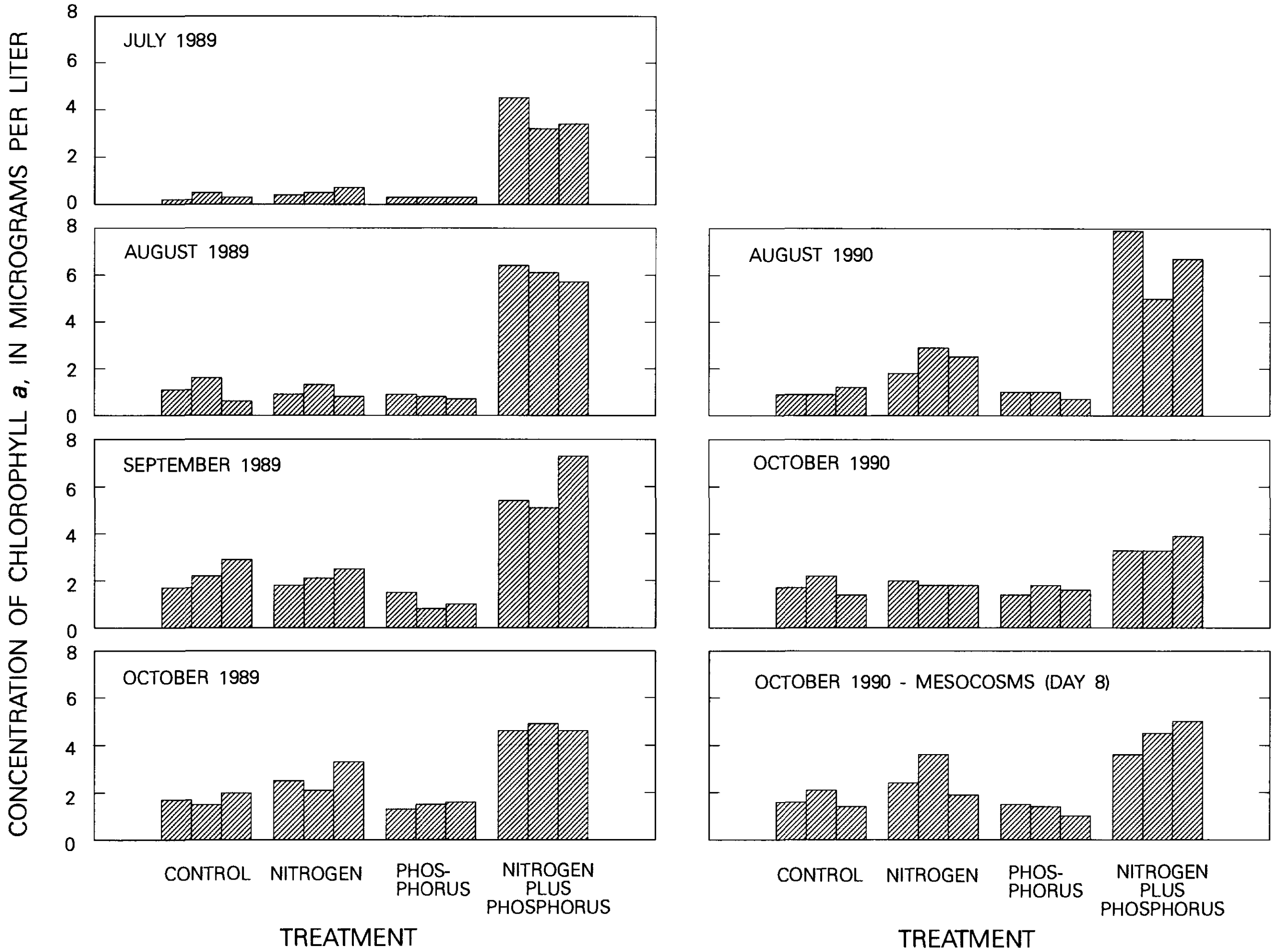


Figure 18.--Chlorophyll *a* concentrations in treatment replicates for nutrient-limitation experiments in Standley Lake, 1989-90.

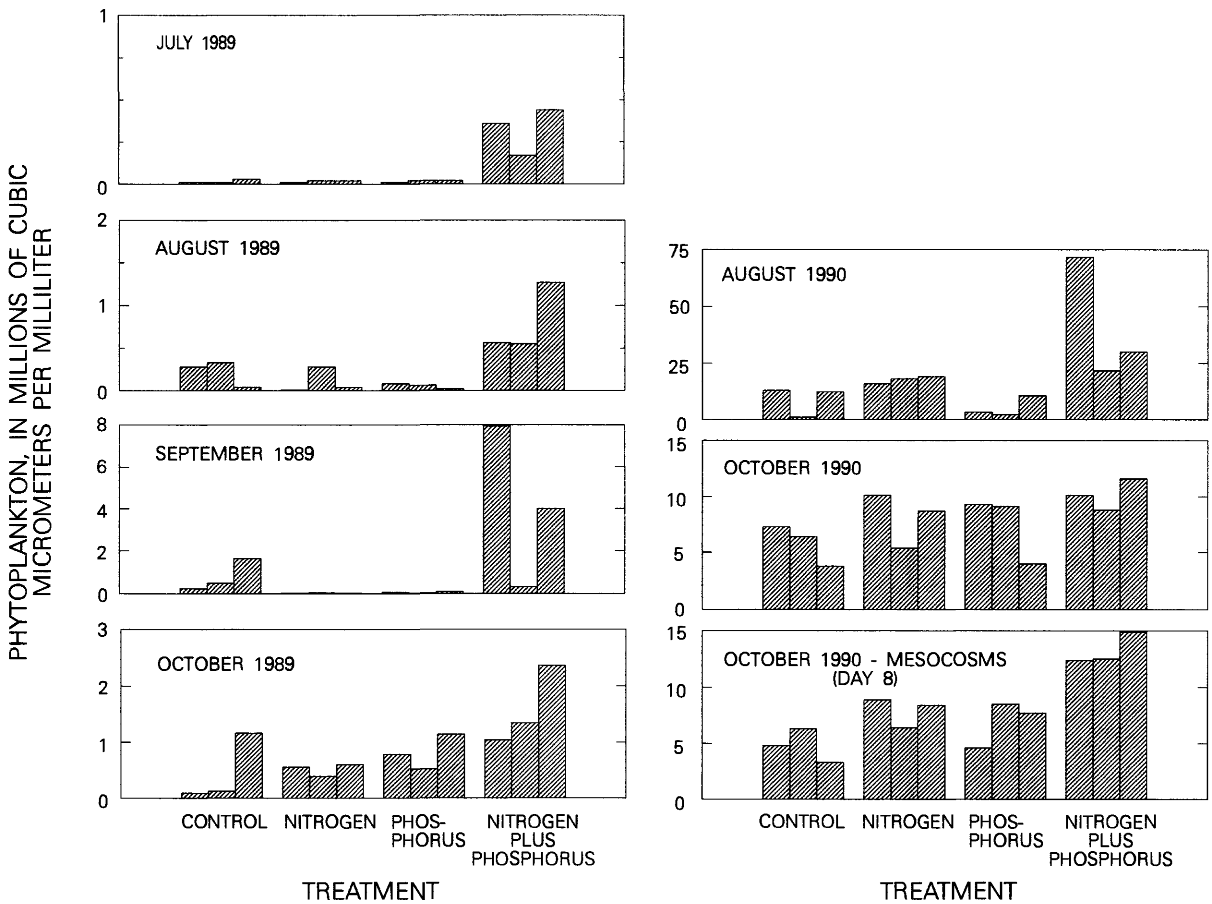


Figure 19.--Phytoplankton biovolumes in treatment replicates for nutrient-limitation experiments in Standley Lake, 1989-90.

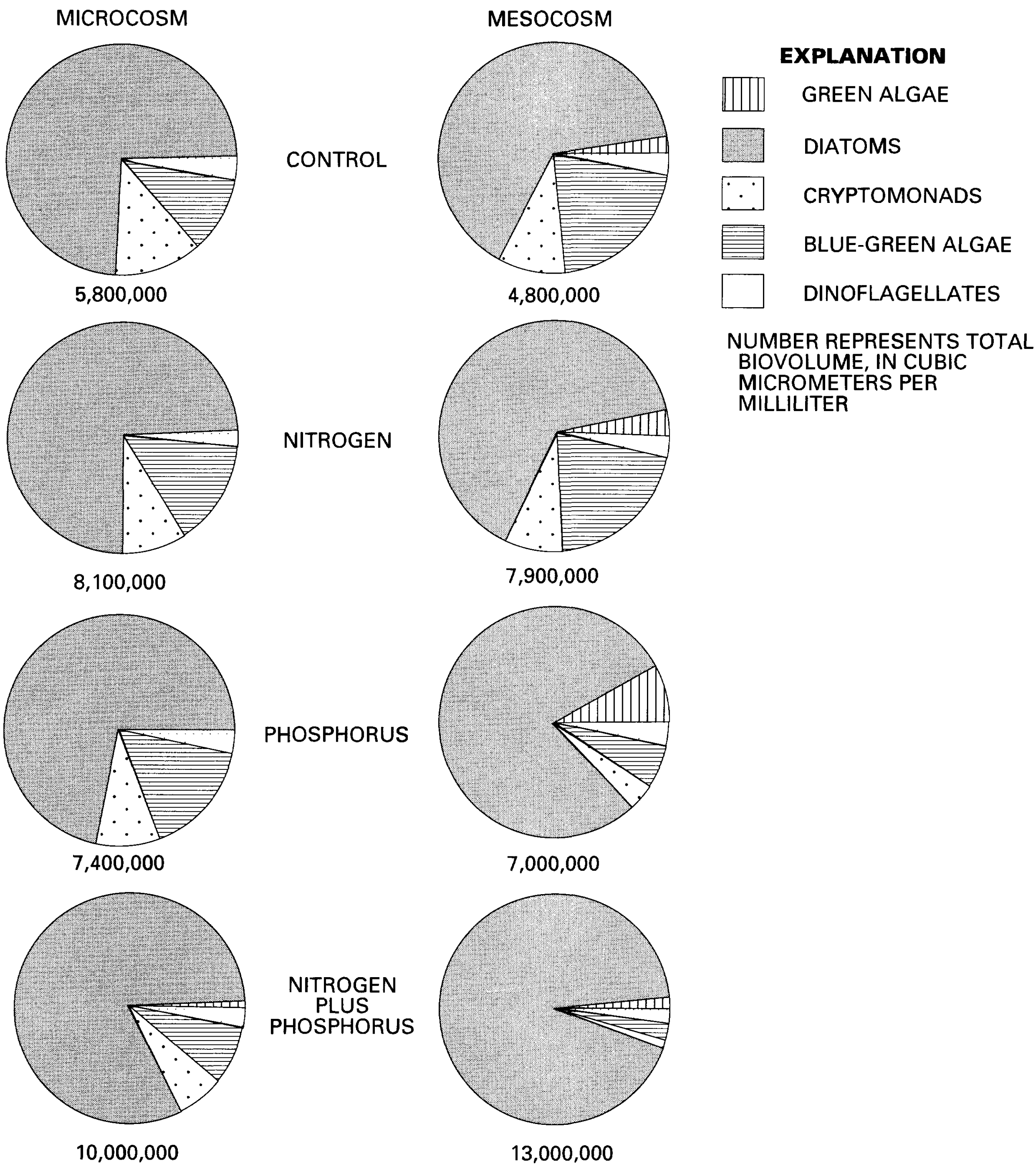


Figure 20.--Volumetric distribution of algal groups in the microcosm and mesocosm experiments, October 1990.

Table 13.--Results of analysis of variance and Duncan's multiple-range test of chlorophyll *a* concentrations and phytoplankton biovolumes among treatments for nutrient-limitation experiments in Standley Lake, 1989-90

[<, less than]

Year	Month	Response variable	Analysis of variance	Duncan's multiple-range test groups by treatment[1]			
			Significance level (p-value)	Control	Nitrogen addition	Phosphorus addition	Nitrogen-plus-phosphorus addition
			MICROCOSMS				
1989	July	Chlorophyll *a*	<0.01	B	B	B	A
		Phytoplankton	<0.01	B	B	B	A
	August	Chlorophyll *a*	<0.01	B	B	B	A
		Phytoplankton	0.02	B	B	B	A
	September	Chlorophyll *a*	<0.01	B	B and C	C	A
		Phytoplankton	0.10	B	B	B	A
	October	Chlorophyll *a*	<0.01	C	B	C	A
		Phytoplankton	0.08	B	B	B	A
1990	August	Chlorophyll *a*	<0.01	C	B	C	A
		Phytoplankton	0.05	B	B	B	A
	October	Chlorophyll *a*	<0.01	B	B	B	A
		Phytoplankton	0.21	B	A and B	A and B	A
			MESOCOSMS				
1990	October	Chlorophyll *a*	<0.01	C	B	C	A
		Phytoplankton	<0.01	C	B	B and C	A

[1]Groups are identified by letter. Treatments having the same letter are in the same group, and have no significant differences. Treatments having different letters are significantly different. The largest concentrations or biovolumes are in group A.

controls. However, significant differences between the control and nitrogen treatments occurred only in the mesocosms.

The volumetric proportions of algal groups in the microcosms and mesocosms also were similar (fig. 20). In every treatment, the populations were dominated by diatoms. The proportions of green algae were slightly larger in the mesocosms, particularly in the phosphorus treatment. Overall, the microcosm and mesocosm results were considered to be equally representative of conditions in Standley Lake.

POTENTIAL SOURCES OF TASTE AND ODOR PROBLEMS

A major taste and odor problem developed in drinking water supplied from Standley Lake in the period following fall turnover in 1988. The odor in the water was described as earthy-musty, one of the most common odor problems in drinking-water supplies and the most difficult to remove (Mallevialle and Suffet, 1987). An earthy-musty odor is commonly caused by the organic compounds geosmin and 2-methylisoborneol (MIB), which can be produced as metabolic or decay products of certain types of algae, particularly blue-green species, and actinomycetes bacteria.

The sampling program for this study was designed to determine the presence of taste and odor causing compounds in Standley Lake and to identify potential sources of these compounds. Samples were collected during August through October in order to evaluate conditions in the lake, its inflows, and outflow prior to and during the time when a taste and odor problem was likely to develop. Samples were analyzed for organic taste and odor causing compounds by using closed-loop stripping analysis (I.H. Suffet and Djanette Khiari, Drexel University, written commun., 1991). Algae and actinomycetes from the samples were grown in enrichment cultures to determine their potential for producing earthy-musty odors (Patricia Cascallar and Wesley O. Pipes, Drexel University, written commun., 1991).

No taste and odor problems developed at any time during the sampling period for this project (June 1989 through October 1990). Neither geosmin nor MIB was identified in any samples from Standley Lake, the inflows, or the outflow (I.H. Suffet and Djanette Khiari, Drexel University, written commun., 1991). However, MIB was determined in samples from Great Western Reservoir, about 2 mi northwest of Standley Lake (fig. 2), collected during a major taste and odor problem in October and November 1990. The strongest odors were detected in samples from near the bottom of the reservoir. Great Western Reservoir receives inflow from the Church Ditch, which also supplies water to Standley Lake.

None of the algae-enrichment cultures derived from Standley Lake samples developed earthy or musty odors, and none of the dominant algae in those cultures have been suspected of producing such odors (Patricia Cascallar and Wesley O. Pipes, Drexel University, written commun., 1991). Earthy odors in algae cultures derived from Great Western Reservoir water samples disappeared in less than 2 weeks, indicating that the odor-causing compounds were not produced by algae growing in the water at the time of sampling. Actinomycetes counts in water samples from Standley Lake were small, and most occurrences were in inflow samples rather than in the lake itself. Some actinomycetes cultures produced earthy or musty odors, but the conditions of culture were so different from those of the lake water that this result is not strong evidence of odor production by actinomycetes in the lake under natural conditions (Patricia Cascallar and Wesley O. Pipes, Drexel University, written commun., 1991). No actinomycetes could be cultured from bottom-sediment samples.

Overall, no evidence of a significant source of odor-causing compounds was determined from algae in the water of Standley Lake or from actinomycetes bacteria in the water or the bottom sediment. Even water collected from Great Western Reservoir during an odor event did not contain any biota that continued to produce odors in enrichment cultures. This condition could exist if the compounds causing the odor were derived from phytoplankton and persisted longer than the algae that produced them, or if the compounds were produced and released into the water column by periphytic organisms on the bottom of the reservoir. Periphyton samples were not collected from Great Western Reservoir, but several algae that can cause earthy or musty odors were identified in samples from Standley Lake. A periphyton source of odor-causing compounds is consistent with qualitative results that the strongest odors in Great Western Reservoir were in the shallow areas of the reservoir and in samples collected near the bottom. One potential source of odor-causing compounds in periphyton collected from Standley Lake is the blue-green alga *Oscillatoria*, which was abundant in samples of unconsolidated bottom sediment collected in shallow areas of Standley Lake in August through October 1990. *Oscillatoria*, along with other blue-green algae and actinomycetes bacteria, are the most common sources of earthy and musty odors in drinking water (Mallevialle and Suffet, 1987). The only other blue-green alga abundant in periphyton samples was *Lyngbya*, which is not suspected of causing odor problems. Actinomycetes bacteria could not be cultured from Standley Lake sediments.

Continued monitoring of the lake is necessary to improve understanding of the development of taste and odor problems. Routine monitoring of physical and chemical properties in the lake and inflows is advisable. Sampling of phytoplankton and periphyton algae also is important, particularly during the time period of historical taste and odor events.

SUMMARY

Standley Lake is a reservoir that supplies water to several suburban cities in Jefferson County, northwest of Denver, Colo. In 1988, a taste and odor problem developed in water supplied from the lake. The cities suspected that the taste and odor problem was related to phytoplankton growing in the lake, and that this growth could be controlled by regulating the phosphorus concentration in the lake. In 1989, a study was undertaken to determine the nutrient availability in the lake and the processes affecting nutrient availability, the potential nutrient limitation to algal growth, and the occurrence and potential sources of compounds that cause taste and odor problems in water withdrawn from the lake. Physical, chemical, and biological water-quality data were collected during June 1989 through October 1990 from sites in the lake, its primary inflows, and its outflow.

During the study period, Standley Lake was thermally stratified from June through September each year and was well mixed during October-December 1989 and March-May 1990. Nutrient concentrations were low in the photic zone: total nitrogen was generally less than 0.3 mg/L, and total phosphorus was always less than 0.02 mg/L. Concentrations were higher in the hypolimnion particularly during stratification. Maximum total nitrogen was at least 0.5 mg/L during both years of the study, and maximum total phosphorus exceeded 0.05 mg/L during both years. Transparency of the water at the surface of Standley Lake increased following stratification and decreased following the fall turnover. Phytoplankton biovolume and chlorophyll *a* concentration were variable throughout the open-water season. The maximum chlorophyll *a* concentration was 6.1 μg/L.

Nutrient loading to Standley Lake was analyzed to determine the amounts of nitrogen and phosphorus supplied to the lake from external and internal sources. The primary external loading sources were assumed to be surface-water inflows; the primary internal source was assumed to be flux from the bottom sediment. Estimated nutrient loads for October 1989 through September 1990 were 13,400 kg of nitrogen and 1,490 kg of phosphorus from surface-water inflows and 13,300 kg of nitrogen and 687 kg of phosphorus from the bottom sediment. Most of the flux from the bottom sediment occurred during stratification; therefore, the nutrients were trapped in the hypolimnion until fall turnover. Most of the nitrogen loading from all sources and the phosphorus loading from the bottom sediment was in a dissolved form that is readily available for algal uptake. Available phosphorus averaged about 20 percent of the total phosphorus in inflow water.

A series of nutrient-limitation experiments was done to evaluate the algal response to phosphorus and nitrogen additions. Small, completely sealed enclosures (microcosms) were used in experiments done in 1989 and 1990 during summer stratification and immediately after fall turnover, in October. A large, open enclosure (mesocosm) experiment was done concurrently with the microcosm experiment in October 1990. Four treatment groups were used for all experiments: (1) control (no nutrient addition), (2) addition of nitrogen, (3) addition of phosphorus, and (4) addition of nitrogen plus phosphorus. Algal response was measured by changes in chlorophyll *a* concentration and phytoplankton biovolume in each enclosure. In every experiment, addition of nitrogen plus phosphorus was required to significantly increase algal response.

Some of the phytoplankton and periphyton taxa in Standley Lake have been associated with taste and odor problems. Phytoplankton that are known to contribute musty odors to water, including *Chlamydomonas, Melosira, Anabaena,* and *Aphanizomenon* were abundant at times in water samples from the photic zone, although at densities much less than the critical limits for development of odor problems. *Melosira, Fragillaria construens*, and *Oscillatoria*, all of which can

produce musty odors, were abundant in periphyton samples.

No taste and odor problems developed in Standley Lake during the study period, and no organic compounds that create earthy or musty odors were detected in any water samples from the lake, the inflows, or the outflow. A musty odor problem occurred in Great Western Reservoir, about 2 mi northwest of Standley Lake, in October 1990. A known odor-causing compound, 2-methylisoborneol (MIB), was identified in water samples collected from this reservoir. Algae cultured from water samples from Standley Lake and Great Western Reservoir failed to produce earthy or musty odors. Few actinomycetes bacteria were detected in water samples, and none could be cultured in bottom-sediment samples collected from Standley Lake.

Continued monitoring of the lake is necessary to improve understanding of the development of taste and odor problems. Frequent measurement of physical properties and analyses of chemical and biological characteristics, including chlorophyll *a* concentration and phytoplankton densities, during the time period of historical taste and odor events, could provide the data needed to evaluate these problems and strategies for their remediation.

During 1989-90, nutrient concentrations in Standley Lake generally were low. Algal biomass, as measured by chlorophyll *a* concentrations and phytoplankton density and biovolume, also was small. However, anthropogenic activities in areas that contribute runoff either directly to the lake or to the inflow canals could cause an increase in algal abundance in the lake if they result in increased loading of both nitrogen and phosphorus.

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