A WATER RESOURCES TECHNICAL PUBLICATION

ENGINEERING AND SCIENCE MONOGRAPH NO. 43

AQUATIC ECOLOGY STUDIES OF TWIN LAKES, COLORADO 1971-86

Effects of a Pumped-Storage Hydroelectric Project on a Pair of Montane Lakes

UNITED STATES
DEPARTMENT OF THE INTERIOR

BUREAU OF RECLAMATION

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16. ABSTRACT

Twin Lakes, located in the mountains of central Colorado, are a pair of dimictic, oligotrophic, montane lakes of glacial origin. They were the site of limnology and fishery studies for 15 years, from 1971 through 1986. The purpose of these studies was to assess effects of construction and operation of the Mt. Elbert Pumped-Storage Powerplant and the new Twin Lakes Dam on the aquatic ecology of the lakes.

Raised lake levels (behind the new dam) changed the morphometry of the lakes. The changed morphometry affected summer stratification patterns in the upper lake, while powerplant discharge impacted stratification in the lower lake. A trophic upsurge in production occurred immediately following raising lake levels and may have masked or offset decreases in production resulting from increased turbidity and more rapid flushing of the lower lake. Species changes were noted in the phytoplankton assemblage and the density of algae decreased in the postoperational period. Benthic biomass increased while zooplankton densities decreased significantly in the postoperational period. Populations of *Mysis* decreased in the postoperational period, and based on 1989 trawls, seemed to have been adversely impacted by powerplant entrainment and increased flushing rates.

Twin Lakes has supported a trophy lake trout fishery and a primary concern was maintaining that fishery after the pumped-storage powerplant began operating. Results of the studies indicate a greater risk of winterkill for aquatic life in the lakes may occur if the powerplant does not operate during severe winter months. Habitat availability for lake trout may have shifted from the lower to the upper lake in the postoperational period.

Because studies at Twin Lakes were discontinued before a new equilibrium state was reached, and recruitment of lake trout to the spawning population takes 6 to 8 years, any chronic impacts of project operation on the lake trout fishery are still unknown. The opportunity for managing the lakes as high quality sport fisheries still exists, but further study to determine the status of longer-lived fish species is needed to provide information for management decisions.

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Effects of a Pumped-Storage Hydroelectric Project on a Pair of Montane Lakes

Environmental Sciences Section
Division of Research and Laboratory Services
Bureau of Reclamation
Denver Office
Denver CO 80225

Colorado Cooperative Fishery Research Unit
Colorado State University
Fort Collins CO 80523

May 1993

UNITED STATES DEPARTMENT OF THE INTERIOR

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BUREAU OF RECLAMATION

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The authors acknowledge the strong support provided by the managers and supervisors of all of the cooperating organizations to accomplish the studies. We particularly wish to recognize the continuous encouragement and support of L. O. Timblin, Jr. (retired, Reclamation scientist) and William McCormick of the Bureau of Reclamation, as well as the efforts of numerous colleagues from all of our agencies who worked on this massive project. First among these colleagues is Dr. Dale A. Hoffman, who began the project and set the cooperative effort in motion.

Special recognition also goes to Dr. Elmer Remmenga who provided expertise for the data reduction and statistical analysis of the more than 14 years Twin Lakes limnological data base. We express our appreciation to Ms. Barbara Knopf who spent a year toiling over the individual chapters and managed to bring some sense and continuity to the multiple-authored monograph. Richard Walters of the Bureau of Reclamation prepared the final draft for publication.

Finally, we acknowledge the fine work of scientists before us such as John Wesley Powell, David Starr Jordon, and Chauncy Juday who provided an historical description of the ecology of Twin Lakes at the turn of the 20th century.

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Introduction — Chapter 1

J. J. Sartoris and J. W. Yahnke

Bureau of Reclamation
Research and Laboratory Services
and Earth Sciences Divisions
Denver, Colorado

THE STUDIES

Twin Lakes, Colorado, were the subjects of cooperative limnology studies (1971-85) and fishery studies (1971-86). The impetus for these investigations was the construction and operation (on the lower lake) of the Mt. Elbert Pumped-Storage Powerplant—a feature of the U.S. Department of the Interior, Bureau of Reclamation's Fryingpan-Arkansas Project. Researchers from three agencies worked together to determine and quantify the effects of this hydropower development on the aquatic ecology of Twin Lakes.

The lead agency and source of funding for the cooperative studies was the Bureau of Reclamation. Reclamation's scientists and technicians from the Applied Sciences Branch in the Research and Laboratory Services Division carried out comprehensive limnological studies of Twin Lakes. The sport fishery was the focus of study for CDOW's (Colorado Division of Wildlife) researchers. A series of specialized investigations of topics ranging from fish entrainment at the powerplant to the biological history of recent lake sediments was done by researchers and graduate students from the CCFWRU (Colorado Cooperative Fishery and Wildlife Research Unit) at Colorado State University, Fort Collins.

LAKES AND THEIR SURROUNDINGS

Twin Lakes, at latitude 39°05′ N. and longitude 106°20′ W. in Lake County, Colorado, are located 24 km southwest of the historic mining town of

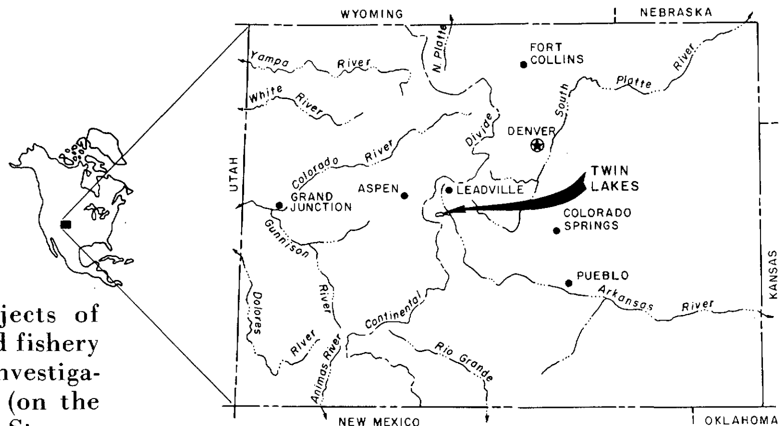

FIGURE 1-1.—*Location map of Twin Lakes, Colorado.*

Leadville (fig. 1-1). The lakes were formed originally as two separate but connected impoundments when moraines—deposited by Pleistocene glaciers—dammed Lake Creek, which is a tributary of the Arkansas River flowing from the eastern slope of the Sawatch Range (fig. 1-2). Lake Creek's watershed covers an area of 238 km^2 (fig. 1-3) at an altitude range of 1641 meters, from the summit of Colorado's highest peak (Mt. Elbert, 4399 m above mean sea level) to the creek's confluence with the Arkansas River (2578 m). The lakes' maximum water surface elevation is 2804 m.

Located in the Southern Rocky Mountains Physiographic Province of North America, the upper Arkansas River Valley is the northern end of the Rio Grande Rift (Chronic, 1980). The valley is bounded on the west by the Sawatch Range, which is the Continental Divide in this area, and on the east by the lower Mosquito Range. Both the Sawatch and Mosquito ranges rose as a single, wide anticline during the Laramide Orogeny in the early Tertiary Period. The rift was superimposed on this structure during the Miocene and Pliocene epochs, and the present Arkansas River flows generally south along the down faulted valley (Chronic, 1980).

FIGURE 1-2.—*Preproject westerly view of Twin Lakes, Colorado.*

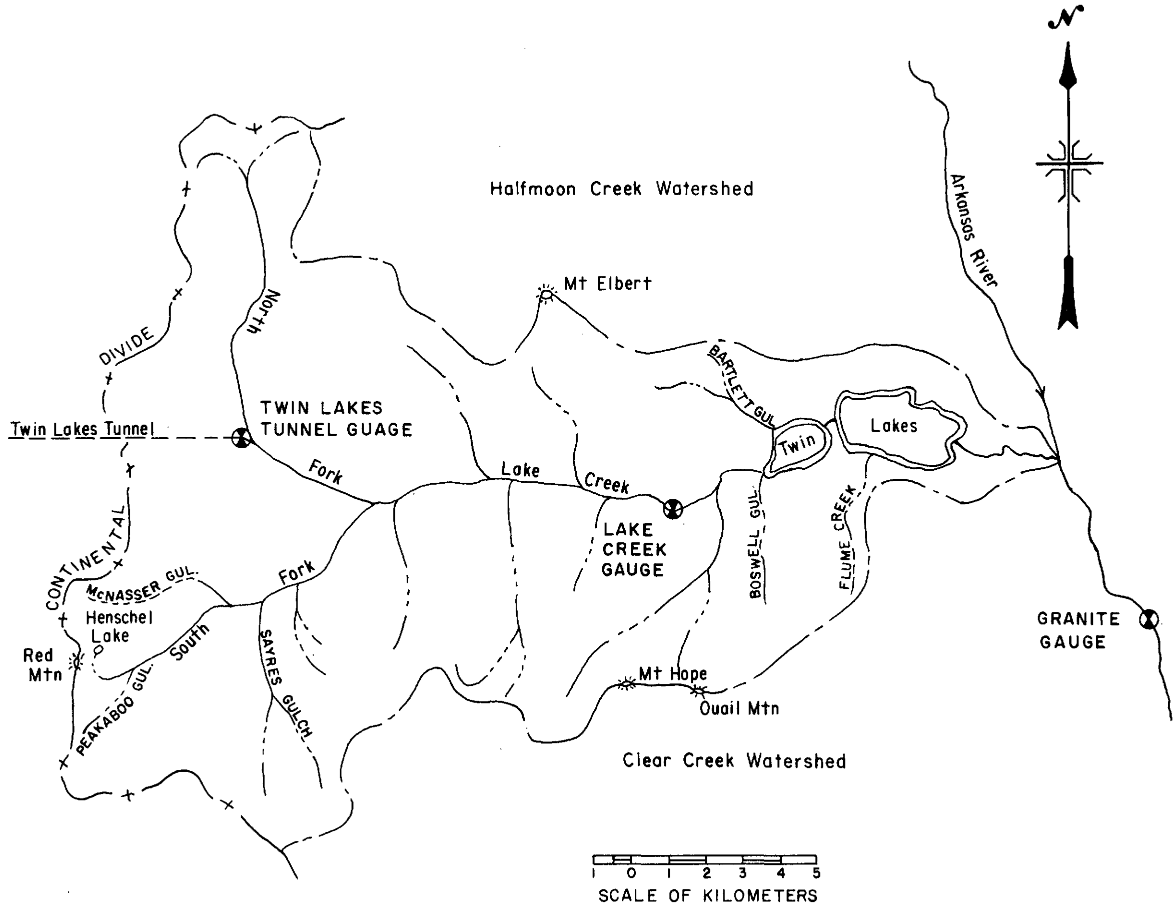

FIGURE 1-3.—*Lake Creek watershed, Colorado.*

Subsequent erosion of the two new mountain ranges filled the valley with alluvial deposits. During the Pleistocene glaciation, these alluvial deposits were reworked by ice pushing down from the Sawatch Range resulting in glacial outwash deposits being added to the valley fill. Twin Lakes were impounded by terminal (Lower Twin Lakes) and recessional (Upper Twin Lakes) moraines of the Lake Creek Glacier (Chronic, 1980). Thus, the lakes lie in deposits of glacial till, whereas most of the Lake Creek watershed drains the crystalline rocks of the Sawatch Range.

The climate of the upper Arkansas River Valley in the Twin Lakes area is cool and semiarid. Climatological data for Leadville, which is 244 m higher than Twin Lakes and 24 km away, are summarized in table 1-1.

Three life zones are represented in the Lake Creek watershed: alpine, subalpine, and montane (Pennak, 1966; Moenke, 1971; Weber, 1972). The area immediately around Twin Lakes lies within the Montane Life Zone (fig. 1-2). On the south side of the lakes, the steep slopes are heavily timbered with Douglas Fir (*Pseudotsuga menziesii*) and some lodgepole pine (*Pinus contorta*) (USDI, 1975). The slopes on the south side of the upper lake also have extensive patches of aspen (*Populus tremuloides*)

Table 1-1.—Climatological data, Leadville, Colorado.

Precipitation, millimeters	
Mean annual	469.4
January mean	33.5
July mean	69.5
Mean annual pan evaporation	1219.2
Temperature, degrees Centigrade	
Mean annual	2.3
January mean	-7.8
July mean	13.8
Mean frost-free growing season	
June 14 to Sept. 7, days	85
Solar radiation, gm-cal/cm^2/day	
Mean annual	425
January mean	240
July mean	650

(U.S. Geological Survey, 1970; Ruffner and Bair, 1978) established in old burn areas. North and east of the lower lake the vegetation consists of sagebrush (*Artemisia* spp.) and grass with a few scattered ponderosa pine (*Pinus ponderosa*)—in sharp contrast to the south shore (fig. 1-2). The ponderosa pines form dense stands north of Upper Twin Lakes; willows (*Salix* spp.) are thick in the Lake Creek Valley west of the lakes.

The morphometry of Twin Lakes changed when the dam on Lake Creek, about 762 m downstream of the lower lake outlet, was completed in October 1983. This earthfill structure raised the maximum water surface elevation in both lakes 2 m; it transformed the pair of connected lakes into a single reservoir having two distinct basins (figs. 1-4, 1-5). Table 1-2 summarizes the preproject and present morphometry of Twin Lakes.

Table 1-2 and a comparison of figures 1-4 and 1-5 show that the raised lake levels had the greatest impact on the morphometry of the upper lake. The main expansion of the upper lake was to the west, inundating a generally flat, marshy floodplain. The net effect of this expansion was to increase the lake's surface area to a greater extent than the volume was increased, and consequently, the upper lake's mean depth was actually decreased by 1.1 m. A more detailed discussion of area-capacity relationships of Twin Lakes is presented in chapter 2.

FRYINGPAN-ARKANSAS PROJECT

Reclamation's Fryingpan-Arkansas Project diverts water from the Western Slope (west of the Rocky Mountains' Continental Divide) to the southeastern plains of Colorado (fig. 1-6). Water is collected from the headwaters of the Fryingpan—a tributary to the Roaring Fork, which is a tributory to the Colorado River—and transported through a tunnel under the Continental Divide to the Arkansas River.

Western Slope features of the project include Ruedi Dam and Reservoir and the North and South Side Collection Systems. Ruedi Reservoir, on the Fryingpan River, stores replacement water to satisfy established Western Slope water rights, as well as some additional water for other western Colorado users. The North and South Side Collection Systems (on the headwater streams of the Fryingpan and Roaring Fork rivers, respectively) consist of several small diversion structures connected by conduits and tunnels to the Boustead Tunnel, which conveys the water to the Eastern Slope and to the upper Arkansas River Valley.

Water diverted through the Boustead Tunnel is stored in Turquoise Lake, a reservoir on the Lake Fork of the Arkansas River. From Turquoise Lake, project water flows 17 km through Mt. Elbert Conduit south to the Mt. Elbert Forebay of the Mt. Elbert Pumped-Storage Powerplant. Along the way, additional project water from Halfmoon Creek (fig. 1-6) is intercepted by the Halfmoon Diversion Dam and diverted into the conduit.

Mt. Elbert Forebay is located on the lateral moraine to the north of Twin Lakes, 137 meters above the Mt. Elbert Pumped-Storage Powerplant, on the northwest corner of the lower lake. This powerplant consists of two pump-turbine units having a generating capacity of 100 megawatts each. Mt. Elbert Forebay water is released down the penstocks and through the turbines to generate electric power during hours of peak demand. In the off-peak periods, power from the grid is used to drive the pump-turbines in reverse to pump water from Twin Lakes up into Mt. Elbert Forebay. Water needed to meet irrigation, municipal, and industrial demands in southeastern Colorado is released through Twin Lakes Dam, down Lake Creek, and into the Arkansas River (fig. 1-6). Pueblo Reservoir, on the lower Arkansas River, is the terminal storage facility for the Fryingpan-Arkansas Project.

Table 1-2.—Morphometric data, Twin Lakes, Colorado.

	†Preproject conditions			Present conditions*		
	Upper lake	Lower lake	Both lakes	Upper lake	Lower lake	Both lakes
Maximum water surface elevation, m			2802			2804
Maximum surface area, ha	263	737	1000	381	742	1123
Maximum volume, $m^3 \times 10^6$	41.1	112.6	153.7	55.3	118.4	173.7
Maximum depth, m	28.0	27.1		30.2	29.3	
Mean depth, m	15.6	15.3		14.5	16.0	
$\overline{Z}/Z_{max}$	0.556	0.565		0.480	0.546	
Shoreline length, km	6.3	10.8	17.0	10.7	13.7	24.4
Shoreline development index	1.09	1.12		1.54	1.42	

* Present conditions are from January 1984 to the present.
† Preproject relates to the completion of the dam on Lake Creek, i.e., prior to Twin Lakes Dam.

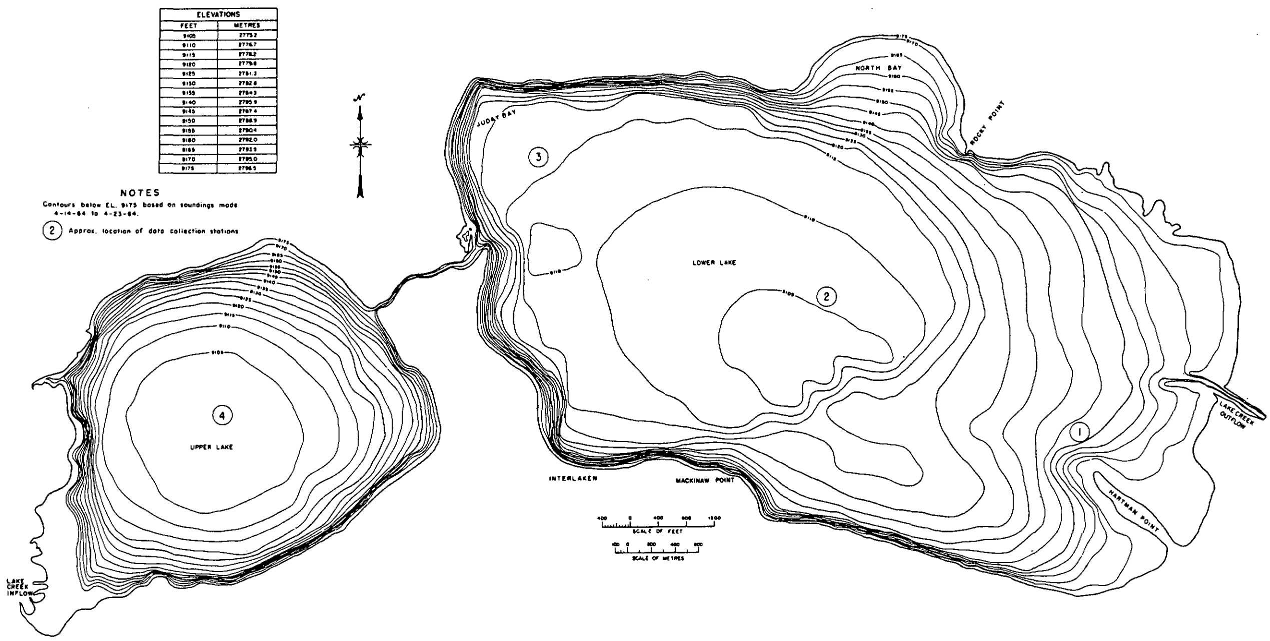

FIGURE 1-4.—*Preproject bathymetric map of Twin Lakes, Colorado.*

The project features at Twin Lakes were built and brought into operation over a period of about 12 years. Powerplant construction began in 1972; the first 100-MW pump-turbine unit went on-line in September 1981. The second unit completed testing and began regular operation in August 1984. Mt. Elbert Forebay construction began in 1975 and was completed in 1977. However, in 1979-80 Mt. Elbert Forebay was drained, and a chlorinated polyethylene liner was installed to eliminate seepage problems. Seepage problems also beset the Twin Lakes Dam—which was started in 1978 and finally closed in October 1983. Water levels in Twin Lakes reached their new operating elevation behind this dam in January 1984. As noted earlier, the new maximum water surface elevation is 2 m higher than it was before the new dam was completed (table 1-2).

The postoperational period (project period) includes 2 years of transitional data:

- 1982, wherein the powerplant had considerable testing upon the first unit before increasing the lakes' elevation, and
- 1983, during which the powerplant was nonfunctional throughout most of the growing season.

Because the lakes' elevation was increased in late 1983, only 1984 and 1985 (through Sept.) represent the postoperational data base. (See also ch. 5: Introduction.)

HISTORICAL BACKGROUND

The first mention of Twin Lakes in scientific literature occurs in progress reports of the U.S. Geological and Geographical Survey of the Territories for the years 1873 and 1874 (Hayden 1874, 1876). Hayden (1876) described the lakes and presented morphometric information based on a water surface elevation of 9182 feet (2799 m) above mean sea level. At that elevation, Hayden reported maximum depths of 79 and 75 feet (24.1 and 22.9 m) for the upper and lower lakes, respectively. Hayden (1876) also noted that the deepest point in the lower lake was near its upper end, toward the morainal deposit separating the two lakes. Hayden's depth data were apparently based on a total of about 20 soundings in both lakes—combined. Three physical features that have since disappeared were mentioned: (1) a waterfall on Lake Creek at the point where it emerges from the canyon above Twin Lakes, (2) an extensive marshy meadow on the west shore of the upper lake, and (3) a difference in elevation between the upper and lower lakes of about 2 meters.

By 1889, when D. S. Jordan visited the area during his seining expedition through Colorado and Utah, the waterfall above the lakes had been destroyed by blasting, and placer mining operations had created a turbidity problem in the main stem of the Arkansas River (Jordan, 1891). Jordan reported that Twin Lakes contained suckers (*Catostomus*

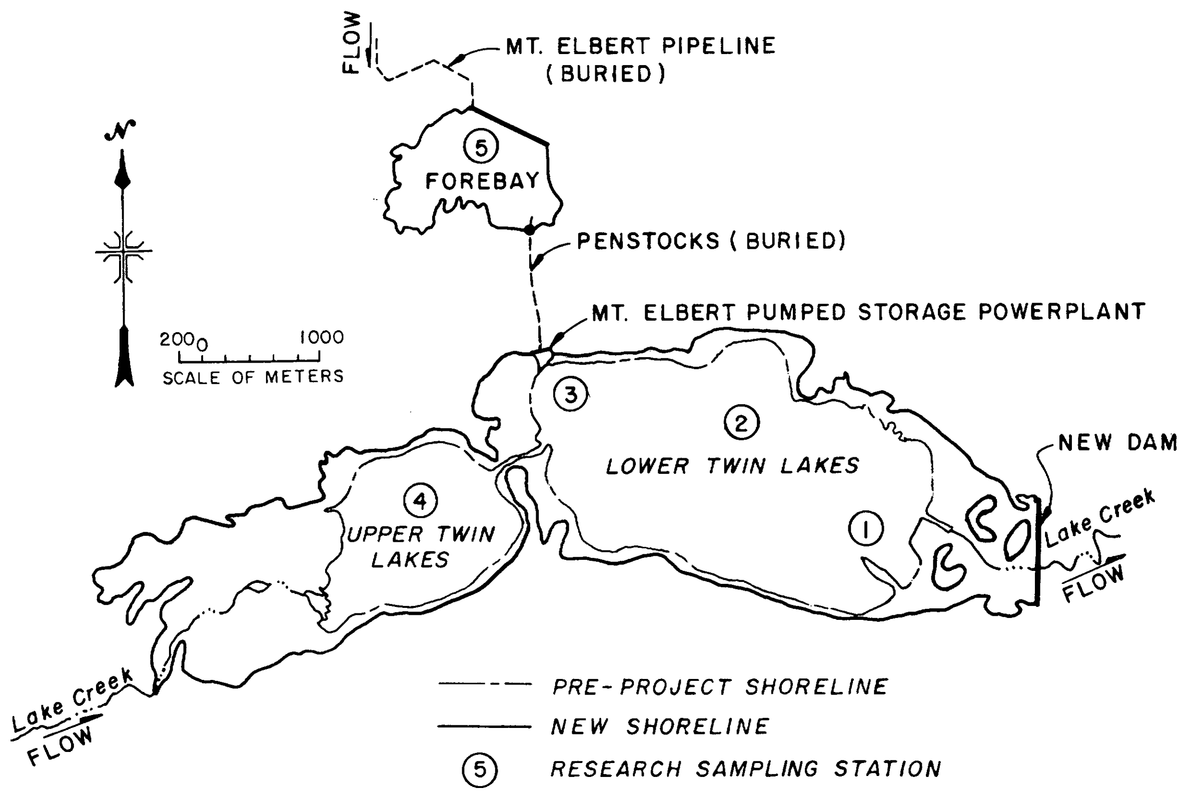

FIGURE 1-5.—*Present Twin Lakes shoreline with noted project features.*

spp.), minnows (*Cyprinidae*), and two species of native trout, one of which—the yellow-fin (*Salmo mykiss macdonaldi*)—was described as a new species indigenous to the lakes (Jordan and Evermann, 1890).

Additional soundings of Twin Lakes were made by Powell (1891) in 1889. The surface elevations of the lakes were reported as 9194 and 9200 feet (2802 and 2804 m) above mean sea level for the lower and upper lakes, respectively. Powell (1891) also indicated that the natural fluctuation was most likely no more than 2 feet (0.6 m) annually. On the basis of 86 soundings of the lower lake and 44 of the upper lake, Powell prepared a generalized bathymetric map of Twin Lakes. On the basis of 10-foot contours (3 m), each lake showed depths in excess of 80 feet (24.4 m), but the actual maximum depths were not reported.

Powell (1891) also installed eight gauging stations in the upper Arkansas River Valley, and published a discussion of the hydrology of the basin. This discussion was primarily confined to the seasonal distribution of flow since only 1 year of data (1890) was available for seven of the eight sites, including one located immediately below the outlet of Twin Lakes. A longer record was available for a gauge that had several locations generally described as being between Canon City and Pueblo. The data for this gauge were collected between 1885 and 1888 by the state engineer and during 1889-90 by Powell's party. These data show the vagaries of the year-to-year water yield of the basin, and documented the need to augment the river in the latter part of July and during August of most years for successful irrigation of crops in the lower Arkansas River Valley.

Powell (1891) proposed developing Twin Lakes as a storage reservoir. This proposal was based on the topographic/bathymetric survey of the lakes and adjacent lands, the gauge data mentioned above, and extrapolation of precipitation data from Leadville, Colorado, to the drainage basin above the town. Briefly, the plan involved constructing a dam downstream from the outlet of the lakes to raise the water surface elevation to 9240 feet (2816 m). Active storage was to be provided between elevations 9190 and 9232 feet (2801 and 2814 m), with the 8 feet (2.4 m) above the latter elevation being used to store floods in lieu of constructing a spillway. This proposal would result in 103,500 acre-feet (0.13×10^9 m^3) of active reservoir capacity. By excavating the channel between the two lakes, they could be operated as a single reservoir. Storage would include the natural flow of Lake Creek as well as water diverted from the Arkansas River, which would be conveyed to the reservoir by a canal from a diversion dam located near Hayden Station, 13 km south of Leadville. An estimated 206,400 acre-feet (0.25×10^9 m^3) of annual runoff was estimated to be available for storage.

By 1901, the Twin Lakes Reservoir and Canal Company had constructed a 5.6-m high dam across the natural outlet of the lower lake and excavated a new outlet canal about 1 km to the north of that dam (Juday, 1906). This configuration increased the potential maximum vertical fluctuation of the lower lake to 7.8 m, and that of the upper lake to 5.8 m.

Chauncey Juday (1907) carried out a limnological survey of Twin Lakes for the Bureau of Fisheries (now U.S. Fish and Wildlife Service) during the summers of 1902 and 1903. Although chemical data were not reported, Juday's physical observations were generally similar to those of these 1971-86 studies. However, Juday's biological data indicated that diversity and perhaps abundance of zooplankton and fish were greater than in recent years. Four new species of fish, including the Mackinaw trout (*Salvelinus namaycush*), had been introduced since Jordan's 1889 visit, whereas the yellow-fin trout had become extremely rare. Juday wrote that, during late summer, a placer mining diversion frequently dried up Lake Creek above Twin Lakes, thus stranding large numbers of brook trout (*Salvelinus fontinalis*) and greatly reducing the inflow to the lakes. Juday (1906) indicated that it was proposed to dredge the channel connecting the two lakes so as to equalize their fluctuation; however, this apparently was not accomplished during the period he worked at Twin Lakes.

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Juday (1906) made 94 soundings of the upper lake and 85 of the lower during July 8-22, 1902. The maximum depths reported were 74 and 82 feet (22.6 and 25.0 m) in the lower and upper lakes, respectively. However, water surface elevation was not given; it was simply stated that the upper lake was at about its normal stage, while the lower lake was about 1 foot lower than usual. Because the lakes had only been regulated for about 1 year, *normal* and *usual* are a little difficult to define precisely. On the basis of his 1902 soundings, plus additional ones made in 1903, Juday (1907) prepared a hydrographic map showing 10-foot bottom contours. This map was once again not tied to a benchmark elevation, although every indication is that the lakes were full. The water surface was shown as being immediately adjacent to the road from Twin Lakes Village to Granite, Colorado. However, the maximum depths on the map are—as reported in Juday (1906)—at 74 and 82 feet. Thus, the water surface must have been well below maximum on the 1907 map; because Hayden (1876) had reported a maximum depth of 76 feet (23.2 m) before construction of the dam across the outlet to Lake Creek.

Several important changes took place in the 55 years between Juday's work and the next known limnological study at Twin Lakes. The most important of these, from a hydrologic perspective, was the completion of the Twin Lakes Tunnel in 1935 (Ubbelohde et al., 1971). This tunnel conveys water from Grizzley Reservoir, which is located on Lincoln Gulch in the Roaring Fork River Basin on the Western Slope, to the North Fork of Lake Creek. Data presented (USDI, 1975) show that the tunnel diversions have increased the average annual discharge of Lake Creek by about 42 percent. Another major change during this interim period was that the stream connecting the upper and lower lakes was dredged so that both lakes would fluctuate essentially as one (Nolting, 1968). The marshy meadow west of the upper lake was mentioned for the last time by Juday (1906, 1907); by 1971 that area was largely an eroded floodplain. In 1957, W. D. Klein, of the Colorado Department of Game and Fish, introduced the *Mysis* shrimp into the lower lake from Clearwater Lake, Minnesota, on an experimental basis.

D. H. Nolting (1968), of the Colorado Department of Game, Fish and Parks, did some limnological work at Twin Lakes between 1958 and 1961 as part of a larger study of the lake (Mackinaw) trout in Colorado. Nolting's physical and chemical data are in general agreement with those obtained in 1971-86, but his biological data differ in that he reports the presence of zooplankters of the genus *Daphnia* and the longnose dace (*Rhinichthys cataractae*) in Twin Lakes. Nolting also reported that, by 1968, there was still no indication that Klein's experimental *Mysis relicta* introduction had been successful.

L. M. Finnell, Colorado Division of Wildlife, began fishery investigations on the entire Fryingpan-Arkansas Project in 1970. In his first progress report, Finnell (1972) reported that the success of the Twin Lakes *Mysis* introduction became evident in 1969, and that shrimp transplants out of Twin Lakes into other lakes and reservoirs began in 1970.

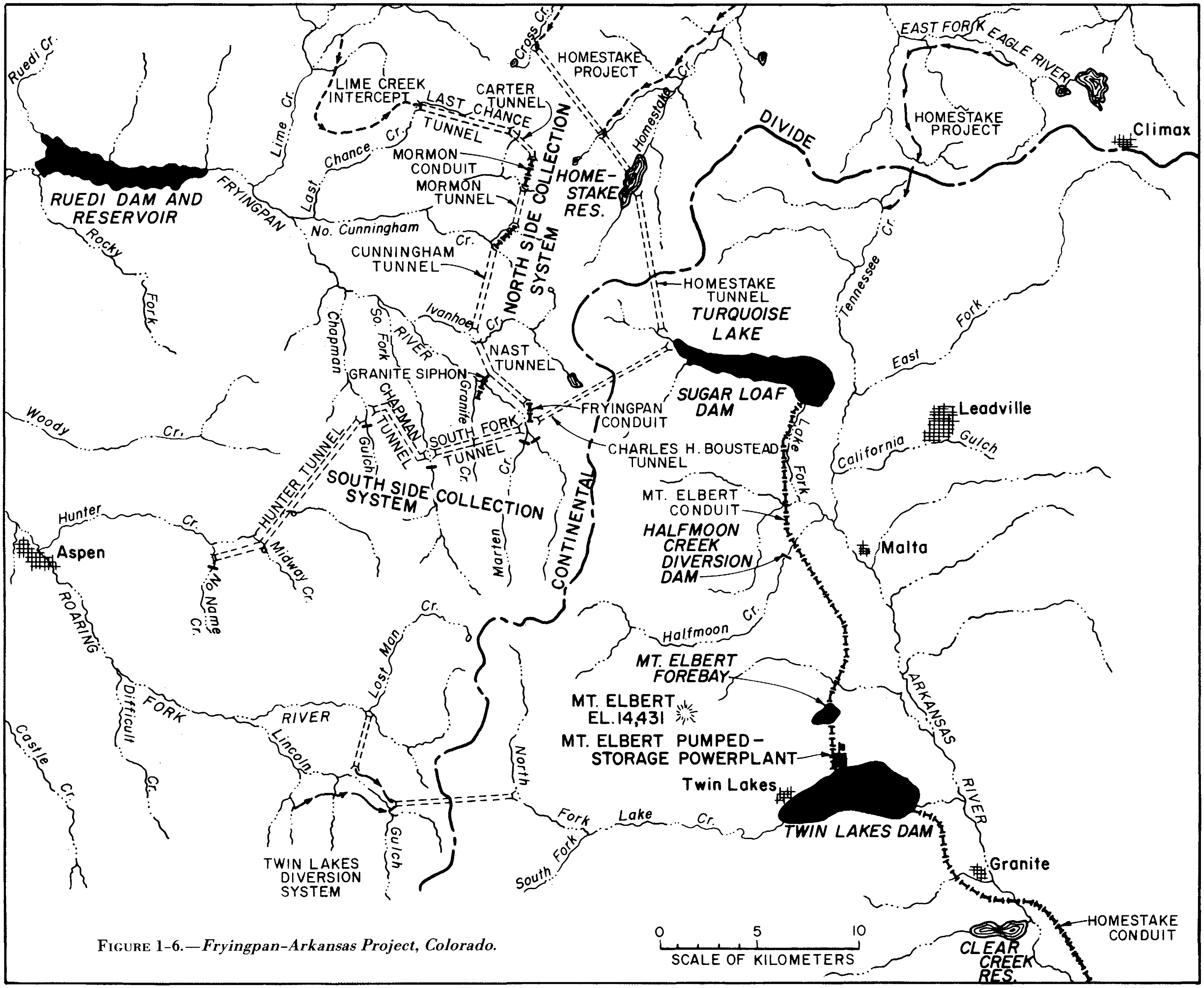

FIGURE 1-6.—*Fryingpan-Arkansas Project, Colorado.*

Hydrology of the Twin Lakes System

Chapter 2

J. W. Yahnke

Bureau of Reclamation
Earth Sciences Division
Denver, Colorado

INTRODUCTION

This chapter documents the changes in the hydrology of Twin Lakes as a result of the development of the Bureau of Reclamation's Fryingpan-Arkansas Project. The project, as currently authorized, is briefly described in chapter 1. A full description is available in the Final Environmental Statement—Fryingpan-Arkansas Project (USDI, 1975).

The Colorado Division of Water Resources operates a gauge on Lake Creek. The gauge is located 2 km (1.2 miles) above the waterline of Twin Lakes (Duncan et al., 1985). During the 36-year period of record that includes 1947-84, excluding 1959 and 1963, the mean annual discharge at the Lake Creek gauge has been 0.15×10^9 m^3 (121,700 acre-feet; Duncan et al., 1985). The additional diversions conveyed through the Mt. Elbert Conduit total 0.16×10^9 m^3 (131,000 acre-feet). Thus, the volume of water entering Twin Lakes could more than double as a result of operating Mt. Elbert Conduit and direct flow generation at the powerplant.

METHODS

Flow data were not gathered during the Twin Lakes study; rather, they were acquired from a variety of existing sources. Operational data were taken from Reclamation records. These data are summarized annually in the Fryingpan-Arkansas Project AOP (Annual Operating Plan) which is usually released in April. The summary data that are pertinent to the Twin Lakes studies include:

- Twin Lakes contents,
- Discharge data for Lake Creek,
- Discharge data for the Mt. Elbert Conduit,
- Amount of water passed during pumping,
- Amount of water passed for power generation,
- Power generation at Mt. Elbert Pumped-Storage Powerplant, and
- Releases below Twin Lakes Dam (USDI, 1975).

The AOP also summarizes the operation of the Twin Lakes Tunnel since Reclamation began operating Twin Lakes. Records before that time were provided by the Twin Lakes Reservoir and Canal Company.

The Lake Creek gauge data, published by USGS (U.S. Geological Survey), are stored in the USGS Water Data Storage and Retrieval System known as WATSTORE (Shope et al., 1975). Data on three gauging stations (fig. 1-3) were retrieved from WATSTORE using Bureau of Reclamation's data link (Main, 1981). The gauging stations included:

- Lake Creek above Twin Lakes (USGS gauge Sta. 07084500),
- Arkansas River at Granite, Colorado (Sta. 07086000), and
- Halfmoon Creek near Malta, Colorado (Sta. 07083000), at Halfmoon Creek Diversion Dam.

Each of these gauging stations has a reasonably long period of record. The retrieved WATSTORE data were therefore used to put the 1977-85 study period in better historical perspective. For these periods of record, stations are: Arkansas River gauge—75 years, Lake Creek gauge—37 years; and Halfmoon Creek—39 years.

Data on transbasin diversions to the Upper Arkansas River Basin were taken from USGS annual water data reports because those data were not located in the WATSTORE data base. Only the major diversions contributed significantly to the discharge of the Upper Arkansas River, and those were the only diversion data included in this study. Data include the Busk-Ivanhoe Tunnel diversions (Sta. 09077500) and Homestake Tunnel (Sta. 090771600). Data on diversions by the Wurtz, Columbine, and Ewing ditches were also available, but have not been used. The Busk-Ivanhoe Tunnel began diverting in 1925, but only records of 1947-85 have been used. The entire records for the Homestake (1967-85) and Boustead (1972-85) tunnels were used.

The WATSTORE retrievals included mean annual statistics and a flow duration analysis. Additional statistical analyses were performed on Reclamation's CYBER 875 mainframe computer using SPSS-X (anonymous, 1986).

RESULTS AND DISCUSSION

Basin Hydrology

The gauge data that were retrieved from the USGS data base are summarized in table 2-1. The three gauges are located in the Upper Arkansas River Basin near Twin Lakes (fig. 1-3). The Lake Creek gauge represents the inflow record for Twin Lakes in the absence of the development of the Fryingpan-Arkansas Project. The gauge record includes the natural flow of Lake Creek plus diversions through the Twin Lakes Tunnel. The gauge record spans a period of 39 years from 1947-85; however, the retrieval included 37 years. The years 1959 and 1963 were not included in the retrieval because they did not satisfy the criterion for missing days, i.e., less than 30 days of missing data per year.

The year 1977 ranks last, out of the 37 years of record, for the Lake Creek gauge (table 2-1). The next lowest mean annual discharge for the Lake Creek gauge is 3.1 m^3/s (79 ft^3/s), which appears in the record for both 1950 and 1954. Comparison of these 3 years (1950, 1954, and 1977) shows that 1977 exhibited a far lower water yield than either of the next lowest years. The years 1950 and 1954 are followed closely by 1981 in the long-term gauge record. Thus, the 9-year study period includes 2 of the years with the lowest water yield in the total gauge record.

As noted previously, 1963 was not included in the retrieved data set. At other gauges in the Upper Arkansas River Basin, 1963 ranked among the years with the lowest mean annual discharge. However, the ranking—with respect to 1950 and 1954—varies at the other gauges, and the ranking of 1963 in the Lake Creek gauge record is unclear. Because the major change in the hydrology of the Twin Lakes system involves an increase rather than a decrease, in the hydraulic loading to Twin Lakes, years having a low mean annual discharge will remain within the historic range—whereas years having higher yields will expand the upper bound of the range. This phenomenon can be illustrated with the data currently available for the Granite gauge on the Arkansas River.

Table 2-2 shows a breakdown of gauge records at various sites in the Upper Arkansas River Basin as well as the major transbasin diversions. The periods shown reflect periods related to water resource development.

Table 2-1.—Mean annual discharge and comparative rankings of water years for selected gauges in the Upper Arkansas River Basin.

	Lake Creek above Twin Lakes				Arkansas River near Granite				Halfmoon Creek near Malta			
	Mean annual discharge		Ranking		Mean annual discharge		Ranking		Mean annual discharge		Ranking	
	ft^3/s	m^3/s	Study period	Total record	ft^3/s	m^3/s	Study period	Total record	ft^3/s	m^3/s	Study period	Total record
Preproject period												
1977	79	2.2	9	37	219	6.2	9	73	14	0.4	9	39
1978	209	5.9	1	6	463	13.1	7	16	32	0.9	6	13
1979	176	5.0	5	18	538	15.2	5	6	33	0.9	5	11
1980	143	4.0	7	27	545	15.4	4	4	34	1.0	4	10
1981	111	3.1	8	34	342	9.7	8	48	20	0.6	8	37
Postproject period												
1982	198	5.6	3	9	492	13.9	6	10	31	0.9	7	34
1983	204	5.8	2	7	647	18.3	2	2	38	1.1	2	5
1984	150	4.2	6	25	687	19.5	1	1	49	1.4	1	1
1985	178	5.0	4	16	573	16.2	3	3	38	1.1	3	6

Table 2-2.—Mean annual flow for Upper Arkansas River Basin and Transcontinental (Rocky Mountains) Divide Tunnels by development period from daily data.

Site	1947-66		1967-71		1972-76		1977-81		1982-85					
	m^3/s	ft^3/s	m^3/s	ft^3/s	m^3/s	ft^3/s	m^3/s	ft^3/s	m^3/s	ft^3/s	χ^2	$P>\chi^2$	†ρ	‡$P>\rho$
Busk-Ivanhoe	0.17	6.1	0.27	9.4	0.24	8.4	0.26	7.7	0.26	9.0	11.8	0.019	0.487	0.001
Homestake Tunnel		0	0.97	34	1.19	42	0.90	32	0.80	28	28	< .00001	.719	< .0005
Boustead Tunnel		0		0	1.30	46	1.60	57	3.39	120	37.2	< .00001	.846	< .0005
Total diversions to Turquoise Lake	0.17	6.1	1.23	44	2.73	96	2.72	96	4.45	157	31.4	< .00001	.894	< .0005
Halfmoon Creek	0.83	29.2	0.80	28.4	0.70	24.8	0.75	26.6	1.10	39.0	7.4	.115	.097	.278
Lake Creek at gauge	4.66	165	5.61	198	4.70	166	4.07	144	5.17	182	4.6	.333	.090	.297
Twin Lakes Tunnel	1.50	53	2.04	72	1.84	65	1.92	68	1.92	68	8.8	.067	.770	< .0005
Arkansas River at Granite Gauge	11.0	388	12.3	434	12.9	455	11.9	421	17.0	600	11.5	.022	.437	.003
Arkansas River—less diversions	9.3	329	9.0	318	8.3	294	7.3	257	10.6	374	3.6	.464	-.074	.327

† Kruskal-Wallis "Oneway Analysis of Variance" based on five periods shown.
‡ Spearman correlations between mean annual flow and year (1947-85).

• The 1947-66 period is a baseline condition in which the Busk-Ivanhoe Tunnel diverted into Turquoise Lake and the Twin Lakes Tunnel diverted into Twin Lakes. Both are tunnels above the Granite gauge which is used as a benchmark.

• The second period (1967-71) coincides with the initial diversions through the Homestake Tunnel; the diversions were increased in 1967 from 7.5×10⁶ to 77.0×10⁶ m³ in 1971 (6,100 to 62,500 acre-ft).

• The third period coincides with the initial diversions through the Boustead Tunnel in 1972.

During its initial diversion period, Boustead Tunnel did not exhibit a pattern of gradual increase like the Homestake Tunnel. The diversions during this period began at 39.6×10^6 m^3 (32,070 acre-ft) in 1972; the range in annual diversions was 12.2×10^6 m^3 (9,930 acre-ft) through Boustead as compared to a range of 50.3×10^3 m^3 (40,810 acre-ft) through the Homestake Tunnel during its initial 5 years (1967-71) of operation. A further difference is that the smallest diversion through the Boustead Tunnel occurred in 1976 and coincided with no diversion through the Homestake Tunnel that year. In the case of Boustead, the reduced diversion was due to drought conditions in the contributing watershed; in Homestake's case, it was due to structural problems in the tunnel itself.

The next 5-year period (1977-81) should have reflected operation of the diversion tunnels at full capability. However, diversions did not occur through Homestake Tunnel in 1978 while permanent repairs were undertaken. In addition, the period included two drought years (1977 and 1981) which also reduced diversions—particularly through Boustead Tunnel (e.g., only 19.5×10^6 m^3 (15,800 acre-ft) were diverted during 1977).

The final period (1982-85) shown in table 2-2 consists of 4 rather than 5 years. This period comprises the testing and postoperational periods of the Mt. Elbert Pumped-Storage Powerplant at Twin Lakes. It also includes the periods of maximum diversion through the transcontinental divide tunnels into the Upper Arkansas River Basin. Water years 1983, 1984, and 1985 ranked 2d, 1st, and 3d, respectively, in the long-term gauge record for the Arkansas River at Granite (table 2-1). Table 2-2 also shows a χ^2 and P value. A statistically significant χ^2 indicates a significant difference in flow exists among the periods shown. A significant ρ indicates that any difference is significantly correlated with time.

Each of the three tunnels in table 2-2 has a highly significant correlation between their level of import and time on the basis of Spearman's ρ. The Granite gauge record similarly correlates with time, although not as significantly as the three tunnels (table 2-2). However, when the gauge record at Granite is adjusted by subtracting the transbasin diversions, the correlation between the gauge record and time becomes nonsignificant and even slightly negative (table 2-3). Thus, the relationship to time is a reflection of increasing levels of water resources development above the Granite gauge.

Hydrographs illustrating interrelationships are presented on figure 2-1. Before 1965, diversions to the Upper Arkansas Basin included the Busk-Ivanhoe Tunnel, Twin Lakes Tunnel, Wurtz Ditch, and Ewing Ditch. The two ditches have a relatively small capacity and are included within the Arkansas River flow in all the river data sets (fig. 2-1, table 2-2). The hydrograph shows that the correlation between the Arkansas River gauge record and time is governed by the study period (table 2-3), which begins in the extreme drought year of 1977 and coincides with the 1983-85 series of high-yield years (table 2-1, fig. 2-1). However, all diversions must be included in the data set for the correlation between the long-term record and the time to be statistically significant.

Table 2-3.—Pearson correlations (*r*) between mean annual flow of the Arkansas River and year by operational period.

	1947-76	1977-85	1947-85
Gauge record	0.284	†0.704	‡0.458
Less diversions above Turquoise Lake	-0.083	†0.662	-0.004
Less all major diversions	-0.196	†0.706	-0.093

†$0.01<P<0.05$ ‡$P<0.01$. Otherwise not significant.

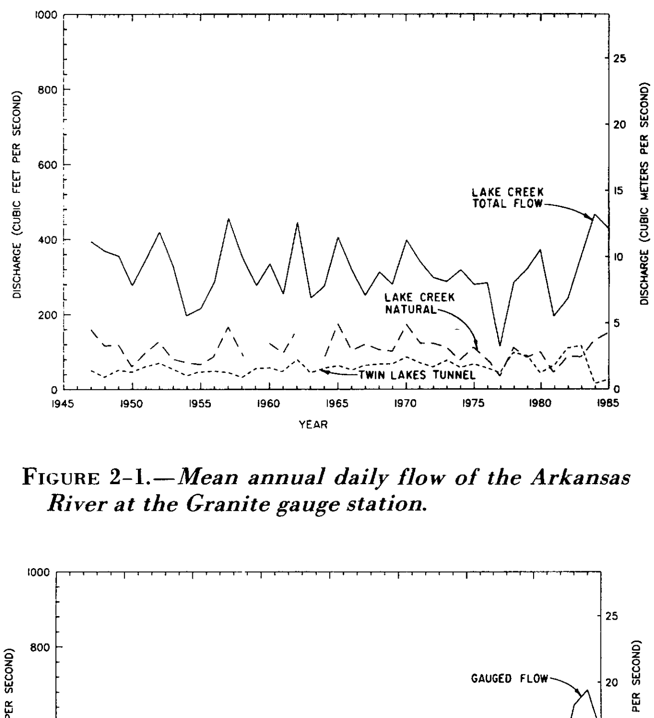

FIGURE 2-1.—*Mean annual daily flow of the Arkansas River at the Granite gauge station.*

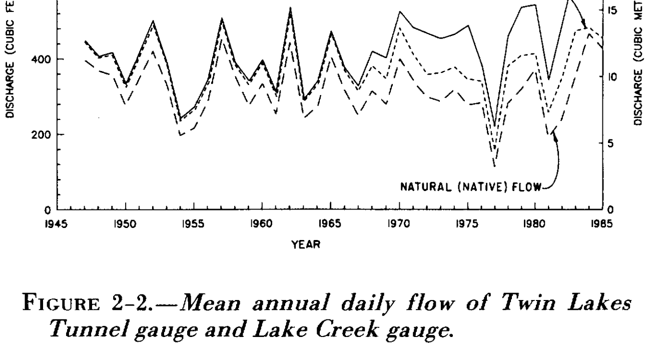

FIGURE 2-2.—*Mean annual daily flow of Twin Lakes Tunnel gauge and Lake Creek gauge.*

Figure 2-2 shows similar hydrographs for Lake Creek—both with Twin Lakes Tunnel (gauged) and without it (native or natural). Over time, the pattern that is exhibited by Lake Creek is rather different from that shown by the Arkansas River (figs. 2-1, 2-2, table 2-1). The peak years in the Lake Creek hydrograph are centered around 1970 (fig. 2-2); the similar period in the Arkansas River hydrograph becomes increasingly dominated by transbasin diversions (fig. 2-1). The differences in the patterns exhibited by the two sets of hydrographs can be further illustrated by the correlation matrix presented in table 2-4, which shows a breakdown among the various gauges similar to that of table 2-3.

During the prestudy period of record (1947-76), Twin Lakes Tunnel discharges correlate highly with those of the Busk-Ivanhoe Tunnel and those gauges that include diversions (table 2-4). Also, a highly significant correlation ($P<0.001$) exists with year—over the same period. Alternatively, the natural flow in Lake Creek correlates with each of the gauges regardless of whether the gauges include diversions or not (table 2-4). There is no significant correlation between the natural flow of Lake Creek and either the Busk-Ivanhoe diversions or year during the early period. The total gauged flow of Lake Creek correlates with Busk-Ivanhoe diversions but not with year (table 2-4). The presence of the Twin Lakes Tunnel discharge in the Lake Creek gauge record influences the correlations considerably. With the Twin Lakes' diversions included (fig. 2-2), the Lake Creek gauge correlates better with other gauges that are affected by transbasin diversions. Lake Creek natural flow correlates better with other gauges that are not affected by such imports. During the prestudy period, Twin Lakes Tunnel diversions were a significant component of the Lake Creek flow.

During 1977-85, correlations exhibited by the Twin Lakes diversions differed from those of the earlier period (1947-76). Although there was a correlation with the Busk-Ivanhoe Tunnel water diversions and the Lake Creek total flow, the correlations with the Arkansas River at the Granite gauge fell to near zero and all others became negative (table 2-4). The negative correlations were not statistically significant (table 2-4). However, the smallest annual diversions through the tunnel for the entire 1947-85 period occurred during 1984 and

1985 (fig. 2-2). These 2 years coincided with the later postoperational period of the Mt. Elbert Pumped-Storage Powerplant. The peak diversions through the tunnel over the entire period occurred in 1982 and 1983 (fig. 2-2). Those years coincided with the testing period for the powerplant. The net effect of the Twin Lakes Tunnel operation during the 1982-85 period was to offset or partially mask the increasing volume of water entering the Twin Lakes system (table 2-5). Table 2-5 presents a breakdown of the sources of inflow to Twin Lakes. For example, the volume entering Twin Lakes by way of the Mt. Elbert Conduit and the powerplant increased by over 60 percent in 1984-85 over what it had been in 1982-83; the total inflow to the Twin Lakes increased by about 10 percent. Consequently, the increased hydraulic loading to the Twin Lakes system was not as great as would have been expected had Twin Lakes Tunnel been operated to divert water to Twin Lakes at nearer to historical levels.

In conjunction with the operation of the Mt. Elbert Pumped-Storage Powerplant, a new dam was constructed at Twin Lakes. The characteristics of the enlarged lakes are shown in table 1-2. The lakes enlarged upon filling in January 1984. Thus, in addition to being subjected to increasing inflows, the postoperational period was affected also by the physical alteration of the lakes. The net effect was that the enlargement essentially offset the increased inflows in terms of the hydraulic residence time of the lakes (table 2-5). The most dramatic decrease

Table 2-4.—Pearson correlations (r) relative to mean annual discharge records for the Upper Arkansas River and Lake Creek Basin.

	Period of record	Busk-Ivanhoe Tunnel	Halfmoon Creek	Lake Creek natural flow	Lake Creek total flow	Arkansas River at Granite	Arkansas River less imports	Year
Twin Lakes	1947-76	†0.840	0.237	*0.407	†0.668	†0.613	0.196	0.551
Tunnel	1977-85	* .714	.134	- .291	* .602	.008	.304	.230
	1947-85	† .717	.075	.001	† .538	* .300	.093	* .328
Lake Creek	1947-76	.265	† .882		† .952	† .740	† .785	.061
natural flow	1977-85	.182	† .861		* .589	† .782	† .891	* .640
(at gauge)	1947-85	.177	† .779		† .844	† .568	† .804	.114
Lake Creek	1947-76	‡ .497	† .786	† .952		† .803	† .715	.241
total flow	1977-85	‡ .755	* .605	* .589		* .659	.487	.340
	1947-85	† .531	† .684	† .844		† .629	† .622	.078
Arkansas River	1947-76	.281	† .852	† .815	† .797	† .877	† .984	.083
with Twin Lakes	1977-85	* .593	† .953	† .853	* .718	† .982	† .945	* .662
Tunnel only	1947-85	* .370	† .870	† .800	† .761	† .844	† .965	.004

Note: † $\rho > \tau \leq 0.001$. ‡ $0.001 < \rho \leq 0.01$. * $0.01 < \rho \leq 0.05$. Otherwise not significant ($\rho > \tau > 0.05$).

Table 2-5.—Mean annual flow to Twin Lakes by source, mean lake volume, and hydraulic residence time.

Year	Twin Lakes Tunnel		Lake Creek natural flow		Mt. Elbert Powerplant		Total inflow		Twin Lakes mean volume		Hydraulic residence
	m^3/s	ft^3/s	m^3/s	ft^3/s	m^3/s	ft^3/s	m^3/s	ft^3/s	$m^3 \times 10^6$	acre-ft	time, year
1977	0.88	31.1	1.36	47.9	0		2.24	79	103.8	84.2	1.47
1978	2.02	71.5	3.89	137.5	0		5.92	209	122.2	99.1	0.66
1979	1.85	65.3	3.80	134.1	0		5.64	199	114.0	92.4	0.64
1980	0.88	31.1	3.15	111.1	0		4.02	142	127.4	103.3	1.01
1981	1.34	47.4	2.15	75.8	0.85	30.0	3.51	124	133.2	108.0	1.22
1982	2.26	79.8	3.34	117.9	3.38	119.5	8.98	317	122.4	99.2	0.43
1983	2.40	84.8	3.54	124.9	2.93	103.6	8.86	313	125.8	102.0	0.45
1984	0.34	12.1	3.92	138.3	5.45	192.5	9.72	343	156.6	127.0	0.47
1985	0.56	19.7	4.45	157.3	4.85	171.2	9.85	348	163.1	132.2	0.53

Note: Hydraulic residence time, in years, equals the ratio of total annual inflow to mean annual volume.

in the residence time occurred in 1982. What makes the change so dramatic was that the large imports through the Mt. Elbert Conduit coincided with larger than average diversions through the Twin Lakes Tunnel and followed the year with the fourth lowest mean annual discharge in the Lake Creek gauge record (table 2-1). The hydraulic residence time declined from 1.22 in 1981 to 0.43 in 1982. However, a similar change occurred between 1977 and 1978, with a decline from 1.47 to 0.66 (table 2-5). Nevertheless, the smallest hydraulic residence time of the study period is 1982. Even though the inflow increased during the years 1982-85, hydraulic residence time remained fairly constant (table 2-5) because of the Twin Lakes enlargement and the pool level that remained at near maximum capacity (fig. 2-3). The year-to-year changes in the hydraulic residence time during 1982-85 were relatively small.

Fryingpan-Arkansas Project Operations

At the beginning of this study in 1977, the Twin Lakes Reservoir and Canal Company was still operating Twin Lakes. In anticipation of operating the powerplant, the Bureau of Reclamation's Fryingpan-Arkansas Project owned 1.18×10^9 m^3 (9,602 acre-ft) of water in Twin Lakes (Reclamation, 1978:6). By 1978, the powerplant was still not in operation. Because of commitments to the Southeastern Colorado Water Conservancy District, 4.9×10^6 m^3 (4,000 acre-ft) had to be withdrawn from storage in Twin Lakes and project ownership declined to 2×10^6 m^3 (1,639 acre-ft) in Twin Lakes (Reclamation, 1979:6). Project water in Twin Lakes further decreased to 1.3×10^6 m^3 (1,018 acre-ft) by the end of 1979 (Reclamation, 1980:5). Mention is not made of the volume of Fryingpan-Arkansas Project water stored in Twin Lakes during 1980, but it was noted that 12.3×10^6 m^3 (10,000 acre-ft) was available in Turquoise Reservoir to be routed to fill Mt. Elbert Forebay through the Mt. Elbert Conduit for testing the pump-turbine in Mt. Elbert Pumped-Storage Powerplant (Reclamation, 1981:8). This was the status of the Fryingpan-Arkansas Project at the conclusion of the preoperational period of the study.

The Fryingpan-Arkansas Project assumed the operation of Twin Lakes on June 1, 1981 (Reclamation, 1982:9). However, several significant events occurred to the limnological study of Twin Lakes earlier in 1981. Mt. Elbert Forebay began to fill on November 17, 1980 (water year '81) and was completed with a pool of 13.6×10^6 m^3 (11,000 acre-ft) by June 23 (Reclamation, 1982:12). Powerplant testing began shortly afterwards at low power levels (fig. 2-4). The testing occurred after the water was drained from the area between the two dams; the second dam had been completed on June 17. The need to drain the area between the two dams developed because seepage through the new dam exceeded that which had been anticipated in the design (Reclamation, 1982:9). Thus, the old dam was returned to service and the seepage problem was evaluated. The area between the two dams remained dry through the end of 1981 (Reclamation, 1982:9). The maximum storage had been reached on July 1 (fig. 2-3) behind the old dam (Reclamation, 1982:12).

Twin Lakes' storage was restricted during 1982 (fig. 2-3) due to the continued operation of the old dam (Reclamation, 1983:13). Water that would have been stored in an enlarged Twin Lakes had to be conveyed to Pueblo Reservoir during the winter months to obtain capacity for storing spring runoff (fig. 2-3). At the same time, Mt. Elbert Pumped-Storage Powerplant was operating using 157.9×10^6 m^3 (128,000 acre-ft) of water to generate electricity (Reclamation, 1983:13, fig. 2-4). Over 25 percent of

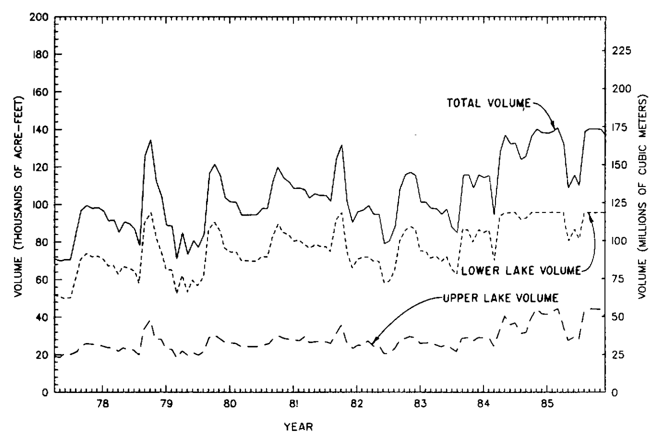

FIGURE 2-3.—*End of month volume at Twin Lakes.*

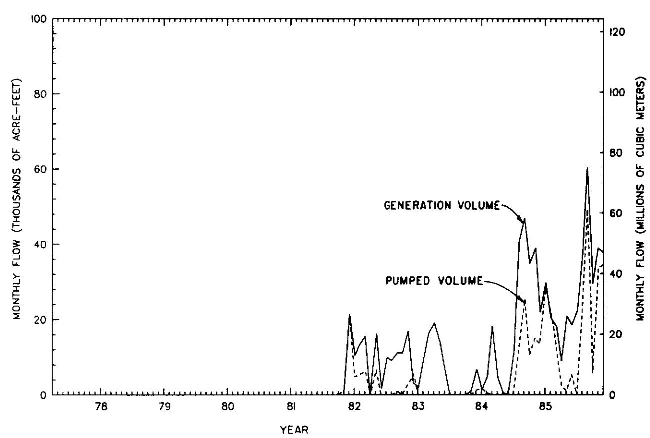

FIGURE 2-4.—*Mt. Elbert Pumped-Storage Powerplant operations indicating volumes of water for generation and pumping.*

the water used for power generation was pumped from Twin Lakes (Reclamation, 1983; and fig. 2-4).

During 1983, Twin Lakes operation was similar to that of 1982. The old dam still functioned and water was conveyed to Pueblo Reservoir during the winter to evacuate capacity (Reclamation, 1984:11). Mt. Elbert Pumped-Storage Powerplant generation was somewhat reduced during 1983; only 93.4×10^6 m^3 (75,690 acre-ft) passed through the turbines (Reclamation, 1984:11; fig. 2-4).

Twin Lakes' elevation was finally raised to its postproject normal operating level behind the new dam in January 1984 (fig. 2-3). Mt. Elbert Pumped-Storage Powerplant generated much more power during 1984 than in previous years (fig. 2-3). During 1984, 275.9×10^6 m^3 (223,680 acre-ft) passed through the turbines (Reclamation, 1985:11). Of the total water used for generation, 102.6×10^6 m^3 (83,180 acre-ft) were pumped from Twin Lakes (Reclamation, 1985:11; fig. 2-4). Twin Lakes was near capacity at the end of the year (Reclamation, 1985).

During 1985, Twin Lakes were operated basically full and bypassing all inflow (Reclamation, 1986:8; figs. 2-3, 2-5). During late March, some drawdown was required in preparation of being able to control releases at the safe channel capacity—below Twin Lakes Dam (Reclamation, 1986:8). By the end of May, Fryingpan-Arkansas Project reservoirs were virtually full, and Twin Lakes were allowed to fill into part of the reserved power pool and then drawn down to operating range by the first part of July (Reclamation, 1986:8). The Mt. Elbert Pumped-Storage Powerplant operated throughout the year (fig. 2-4). The 1985 operation was also the first in which water pumped from Twin Lakes was responsible for more power production than was flow-through water. Flow-through water available for

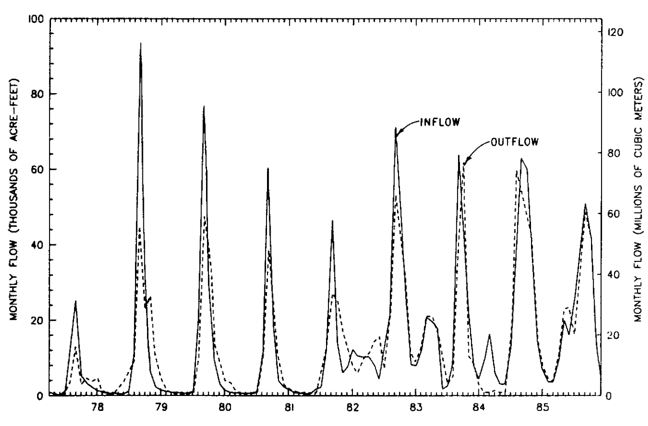

FIGURE 2-5.—*Total monthly inflow to Twin Lakes and releases to the Arkansas River.*

power generation was 137.8×10^6 m^3, in addition to 272.1×10^6 m^3 pumped from Twin Lakes (111,720 and 220,600 acre-ft) (Reclamation, 1986:8).

The *limnology* study concluded at the end of 1985, but the *fisheries* study continued to the end of 1986. *The 1986 operation of Twin Lakes was similar to that of 1985.* Twin Lakes operated throughout the year basically full and bypassing all inflow (Reclamation, 1987:8). The powerplant had 83.1×10^6 m^3 of flow-through water available along with 511.4×10^6 m^3) pumped from Twin Lakes (67,380 and 414,580 acre-ft; Reclamation, 1987:8).

Changes in the physical dimensions of Twin Lakes caused by its enlargement are summarized in table 1-2. The major change in dimensions occurred in the upper lake basin (table 1-2), where the area increased to over 140 percent of its previous size, while the lower lake area increased less than 1 percent. As a result, the mean depth (volume divided by area) of the upper lake decreased by 1.1 m because of the large expanse of inundated shallows. The mean depth of the lower lake increased by 0.7 m, which offset the decrease in the upper lake; thus, the Twin Lakes' mean depth increased by 60 mm.

Table 1-2 also presents the ratio of the mean depth to the maximum depth ($\overline{Z}/Z_{max}$). Hutchinson (1957) showed that this ratio is a simple measure of the complexity of basin form. Most lakes are within the range of 0.33 and 0.50; values greater than 0.5 are characteristic of lakes in glacially overdeepened valleys, among others (ibid.). Although he was discussing erosive processes, Hutchinson (1957) stated that any action of shore processes tends to decrease the value. In Twin Lakes' case, raising the dam crest has had such an effect (table 1-2) because the newly inundated area has all the characteristics of an extensive mudflat. Morphometrically, inundation of extensive shallow areas has a similar effect to beach formation in changing the $\overline{Z}/Z_{max}$ ratio.

Figure 2-3 shows the lakes' monthly volume and figure 2-5 shows the total amount of water entering Twin Lakes and released to the Arkansas River. During 1977-80, the inflow was characterized by a sharp peak caused by spring runoff and a low point for the remainder of the year (fig. 2-5). Beginning in 1981, a second, smaller peak occurred following the normal peak from runoff (fig. 2-5). The second peak was due to imports to Twin Lakes from the powerplant by way of the Mt. Elbert Conduit. Thus, the capability now exists to fill the Twin Lakes late in the year. The plot of volume shows that in 1984 and 1985 the lakes were essentially filled through the winter months (fig. 2-3). The increased volume during the winter of 1984 was due to the operation of the new dam.

The monthly operation of the upper and lower lakes is shown on figure 2-3. The volume of the upper lake appears to change proportionally much less than that of the lower lake during 1977-83 (fig. 2-3). Once the enlarged lakes began operating in 1984 the proportional changes in volumes of the upper and lower lakes appeared similar (fig. 2-3). Similarly, the changes in the area of the two lakes through the year are proportional in the period prior to enlargement (fig. 2-6). After enlargement, the upper lake showed greater areal fluctuation than the lower lake (fig. 2-6). The plots further exemplify the difference in the proportional effect of enlargement on volume and area of each of the lakes.

Heretofore, the random factors and other projects affecting the hydrology of the upper Arkansas River Basin have been discussed. Delays in construction and operation of Mt. Elbert Pumped-Storage Powerplant and Twin Lakes Dam were also summarized. The study design, for the Twin Lakes Limnology/Fishery Study, was based on operations studies conducted before constructing the Fryingpan-Arkansas Project. A comparison of the preconstruction studies and the actual operations during the Twin Lakes Limnology/Fishery Study is presented in table 2-6. The operations studies included the period from 1928 through July of 1965.

The historic operation was based on the Twin Lakes Reservoir and Canal Company's records of their operation of Twin Lakes. The "with the project" data are based on a simulated operation of the Fryingpan-Arkansas Project with full development of all related projects (table 2-6; see also ch. 2; Final Environmental Statement—Fryingpan-Arkansas Project, USDI, 1975).

The total inflow to Twin Lakes under the "with project" condition by source is shown in table 2-7.

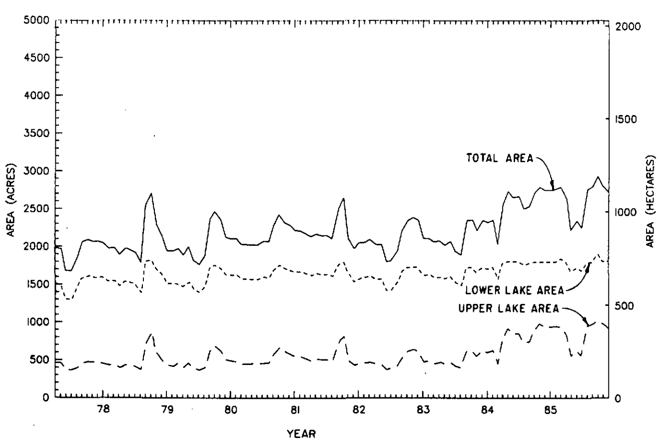

FIGURE 2-6.—*End of month surface area of Twin Lakes.*

The water assigned to Colorado Fuel & Iron Steel Corp. has since been sold to the Homestake Project and the City of Pueblo; water still will be available for diversion through Twin Lakes in the future. All inflows would be used for power generation at Mt. Elbert Pumped-Storage Powerplant.

Table 2-6.—Comparison of lakes': volume, inflow, and hydraulic residence time observed during this study and those from earlier operation studies (USDI, 1975).

	Operations Study 1928-65		Limnology Study 1977-85	
	Historic	With the project	Preoperation	Postoperation
		Twin Lakes		
Volume:				
acre-ft × 10^3	86.0	91.1	97.4	115.1
$m^3 \times 10^6$	106.1	112.3	120.1	142.0
Inflow:				
acre-ft × 10^3	122.2	304.4	113.4	239.4
$m^3 \times 10^6$	150.7	375.5	139.9	295.3
Residence time, days	257	109	314	176
		Upper Twin Lakes		
Volume:				
acre-ft × 10^3	23.8	25.1	26.4	31.3
$m^3 \times 10^6$	29.4	31.0	32.5	38.6
Inflow:				
acre-ft × 10^3	122.2	136.7	109.1	133.1
$m^3 \times 10^6$	150.7	168.6	134.6	164.2
Residence time, days	71	67	88	86
		Lower Twin Lakes		
Volume:				
acre-ft × 10^3	63.6	67.5	71.0	85.8
$m^3 \times 10^6$	78.5	83.2	87.6	105.8
Inflow:				
acre-ft × 10^3	122.2	304.4	113.4	239.4
$m^3 \times 10^6$	150.7	375.5	139.9	295.3
Residence time, days	190	81	229	131

Table 2-7.—Twin Lakes water sources and project average annual volumes (USDI, 1975).

	m^3	*acre-ft*
Fryingpan-Arkansas Project	83.3	69,200
Homestake Project	77.7	63,000
CF&I Steel Corporation	21.5	17,400
Busk-Ivanhoe Tunnel	8.9	7,200
Twin Lakes Reservoir and Canal Co.		
historic	44.4	36,000
long-term	62.3	50,500
Lake Creek natural flow	106.3	86,200
Lake Fork divertible natural flow	20.6	16,700

The Fryingpan-Arkansas preconstruction operation studies (table 2-6) projected that the average monthly volume of Twin Lakes would only increase by about 21.8×10^3 m^3 (5,000 acre-ft); however, the increase in the mean volume from preproject to the postproject operation was 21.8×10^6 m^3 (17,700 acre-ft) during the limnology study. The increase in the mean volume of the lakes was much greater than was projected by preconstruction operation studies.

The annual inflow to Twin Lakes was projected to increase by a factor of 2.5 based on preconstruction operation studies. The increase during the limnology study was a factor of 2.1, based on the data in table 2-6. The difference in project inflow "with the project" and that observed during the limnology study is due to the Homestake Project. During this limnology study, Homestake water was not diverted through Twin Lakes, and the preconstruction estimate of inflow is consequently high.

The major effect of the increased inflow is an increase in flushing of the reservoir. Increased flushing can be approximated by estimating the hydraulic time of water in the reservoir. Residence time is the period required to replace the entire volume of the lakes at a given inflow rate; estimates are shown in table 2-6. The residence time projected by the preconstruction operation studies decreases from an average of 257 to 109 days "with the project." The preoperation and postoperation residence times for the limnology study are 314 and 176 days, respectively. Preconstruction estimates indicate that a much shorter residence time could be expected in the long term; i.e., a much greater degree of flushing is probable.

Table 2-6 also shows a similar breakdown of mean volume, inflow, and hydraulic time for the upper and lower lakes—individually. These should be considered as being only semiquantitative and presented for comparative purposes only. These data assume that all the inflow from the powerplant is confined to the lower lake; backwater effects in the upper lake are ignored. The indication is that flushing would be much greater in the lower lake; at full development, flushing could be much greater.

The operation of Twin Lakes has been changed again since the conclusion of sampling in 1985. The Homestake Project now conveys its water from Turquoise Lake to its Trout Creek Pumping Plant by way of the Mt. Elbert Pumped-Storage Powerplant and Twin Lakes (Griswold, 1988). The exact timing of Homestake deliveries is unknown. The city of Colorado Springs has limited storage capacity; thus, their share probably reflects a classic demand pattern for cities east of the Rocky Mountains' Continental Divide. Aurora, Colorado, has capability to store Homestake water in Spinney Mountain Reservoir in South Park, Colorado; those deliveries can be made at any time of the year that operation dictates. The Homestake diversions during the Twin Lakes limnology study averaged 28.6×10^6 m^3 (23,180 acre-ft) per year during the 1977-81 period and 25.0×10^6 m^3 (20,285 acre-ft) per year during the 1982-85 period. Before Homestake diversions could reach the 77.7×10^6-m^3 (63,000-acre-ft) capacity—projected in the preconstruction operation studies—the Homestake II Project would have to be constructed. The project is in litigation because of the cumbrous permitting process (Griswold, 1988). At the present time, it is uncertain when Homestake II construction will begin.

* * * * *

Currents — Chapter 3

M. Maiolie and E. P. Bergersen

Colorado Cooperative Fish
and Wildlife Research Unit
Colorado State University
Fort Collins, Colorado

INTRODUCTION

Currents are important to the physical and chemical characteristics of lentic environments and to the distribution of planktonic organisms and those that feed on them. Changes in the currents of Twin Lakes were considered among the impacts on the physical environment from the enlargement of the lakes and from power generation. Increased velocity of the currents could affect the *Mysis relicta* population, which is known to be sensitive to turbulence (Gregg and Bergersen, 1980), disturb bottom sediments, and dislodge larval fish from the protection of some shores. Because currents from power generation occur throughout the winter, two additional effects were considered. Erosion and pitting may occur in the ice sheet, making parts of the lake dangerous for such recreational uses as ice fishing, and four-wheel drive and snowmobile races. Also, currents under the ice may be sufficiently strong to keep sediments suspended during the period of time when they normally settle to the bottom. Natural lake currents were documented by Maiolie (1987) and were compared to similar information obtained during power generation.

METHODS

Wind and currents were measured in August, September, October, and November 1979. Currents were measured under the ice in March and April 1980, and in January, February, March, and April 1981. Normal or background currents were compared to currents observed during power generation cycles in February 1982 and August 1984. Examples of the maps of currents were compared to currents observed during power generation cycles in February 1982 and August 1984. The examples—drawn from the measurements—are included here; a complete map set is presented in Bergersen and Maiolie (1981) and Maiolie (1987).

Wind Measurements

Wind data were gathered with a recording wind vane made by placing a Minolta time-lapse movie camera over a conventional wind vane. The camera was set to record the vane's position every 12 to 15 minutes so that one roll of 8-mm movie film recorded 1 month of wind directions. The vane and camera were placed on a building roof near the north shore of the lower lake. A street lamp 5 meters away provided light for nighttime exposures.

Wind direction and velocity plots (wind roses) were drawn to describe the wind direction from 16 compass points. One wind rose was drawn for each quarter of a day. Each "arm" segment of the wind rose radiating from the center represented one reading of the wind vane; the orientation of the wind rose arms indicated the direction from which the wind was blowing. Wind measurements were compared to current measurements to evaluate the influence of wind on current patterns.

Current Measurements

Surface currents.—Surface current measurements were based on movements of submerged drogues designed to move passively with the surface currents and to be affected minimally by the wind (fig. 3-1). Because the velocities of surface currents were based on the straight-line distance between recorded positions, they represent minimum estimates of current velocities. A generation cycle of the Mt. Elbert Pumped-Storage Powerplant began on

August 15, 1984, at 0950 and ended at 1514 hours. During that time, drogues were set at 20 locations on standard transects across Lower Twin Lakes. Position of each drogue was recorded by triangulation with a sextant from 0955 until 1632 hours.

Under-ice currents.—An ink tracing technique was developed to measure the velocity and direction of under-ice currents. A 2-m probe was constructed of polyvinyl chloride pipe 1.9-m diameter, and a measuring bar 100 mm long was attached to its lower end at a right angle. A 10-milliliter oral syringe was fitted into the probe, and a cap with an 0.8-mm hole in the tip was placed over the end of the syringe (fig. 3-2).

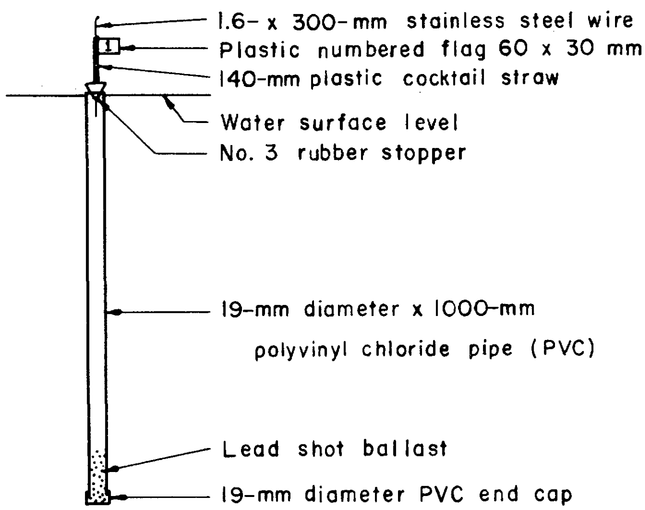

FIGURE 3-1.—*Surface current measuring device using submerged drogues.*

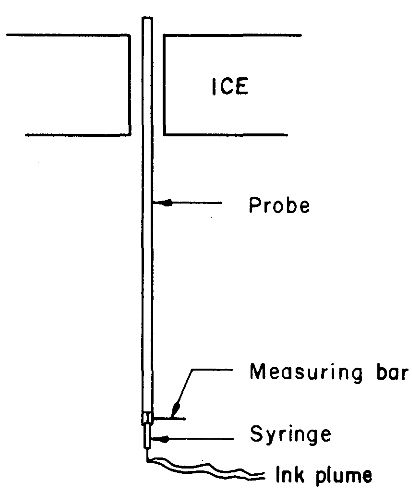

FIGURE 3-2.—*A 2-meter probe to measure velocity and direction of under-ice currents.*

When the probe was lowered through a hole in the ice 150 mm in diameter, a plume of white ink was allowed to stream passively from the syringe tip. Then, this plume was disrupted by moving the probe a few centimeters to one side. The break in the ink trail was timed as it moved the length of the measuring bar. A mean of three velocity measurements was obtained at each location. Current direction and location of each station were measured with a sextant. Water velocity and direction were measured at various depths between the ice and a depth of 2.6 m to note changes in current velocity and direction caused by the frictional influence of the ice cover. Four replicate measurements were made at five depths at each of three locations on the upper lake. Then, the 12 measurements at each depth were averaged.

RESULTS AND DISCUSSION

Upper Twin Lakes Circulation

Surface currents of Upper Twin Lakes are determined largely by the direction and velocity of the wind during ice-free periods (figs. 3-3, 3-4). Because this lake has a low shoreline development, an open exposure, and a small size, currents move nearly straight downwind. When the downwind shore is reached, currents move along the shore and their velocity decreases. The maximum surface current measured on the upper lake was 840 cm/min.

Bottom currents of Upper Twin Lakes were variable, although circulation patterns existed in August and November 1979 (figs. 3-5, 3-6). Winds were predominantly out of the southeast, but mountains on the southside of the lake blocked or slowed the wind in this area. Winds from the north were stronger and caused counterclockwise currents. Bottom currents along the southeast shore tended to be sufficiently strong to remove silt from the rock bottom and provide what appeared to be highly suitable lake trout spawning habitats.

Clockwise circulation predominated under the ice in the upper lake in April 1980 and January, February, and April 1981 (fig. 3-7). The average measured current velocity for these four months was 24, 33, 38, and 16 cm/min, respectively. A maximum velocity of 67 cm/min was observed in January 1981.

In March 1981, currents in the main gyre reversed direction (fig. 3-8). The movement of the gyre became counterclockwise, whereas circulation at the lake perimeter was clockwise. Current velocities dropped to an average of 13 cm/min, and the gyre center was near the south shore.

Convection currents, created by differential heating and cooling of the water, were probably

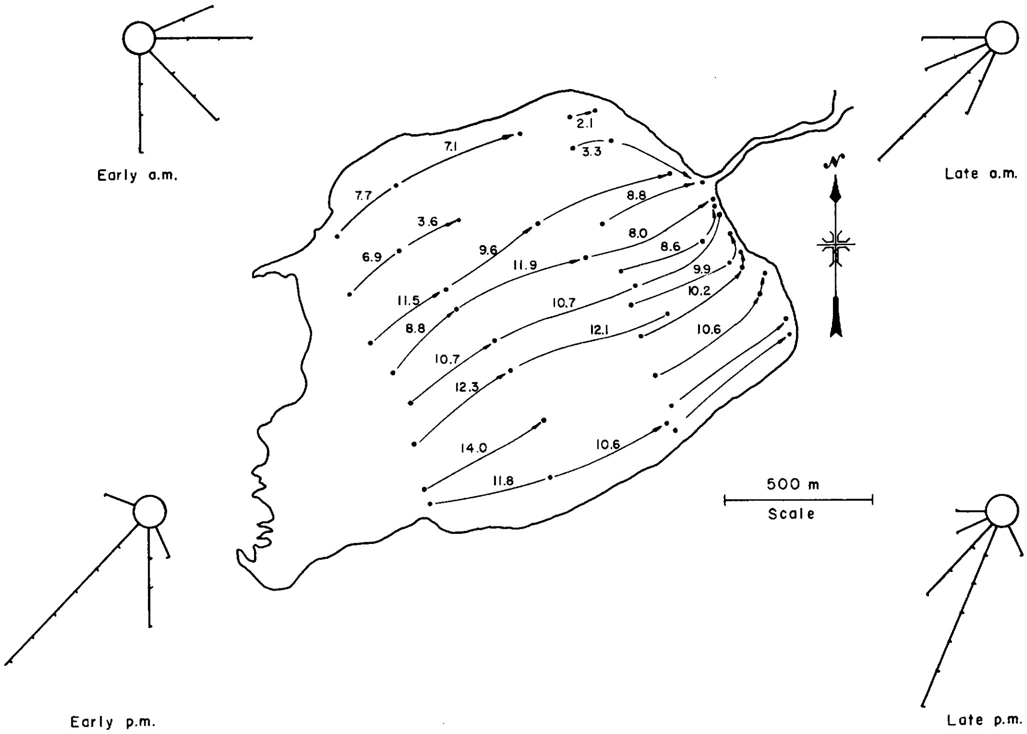

FIGURE 3-3.—*Surface current direction and velocity (centimeters per second) in Upper Twin Lakes, August 13, 1979. Wind roses show wind direction and velocity (meters per second) on the day of current sampling.*

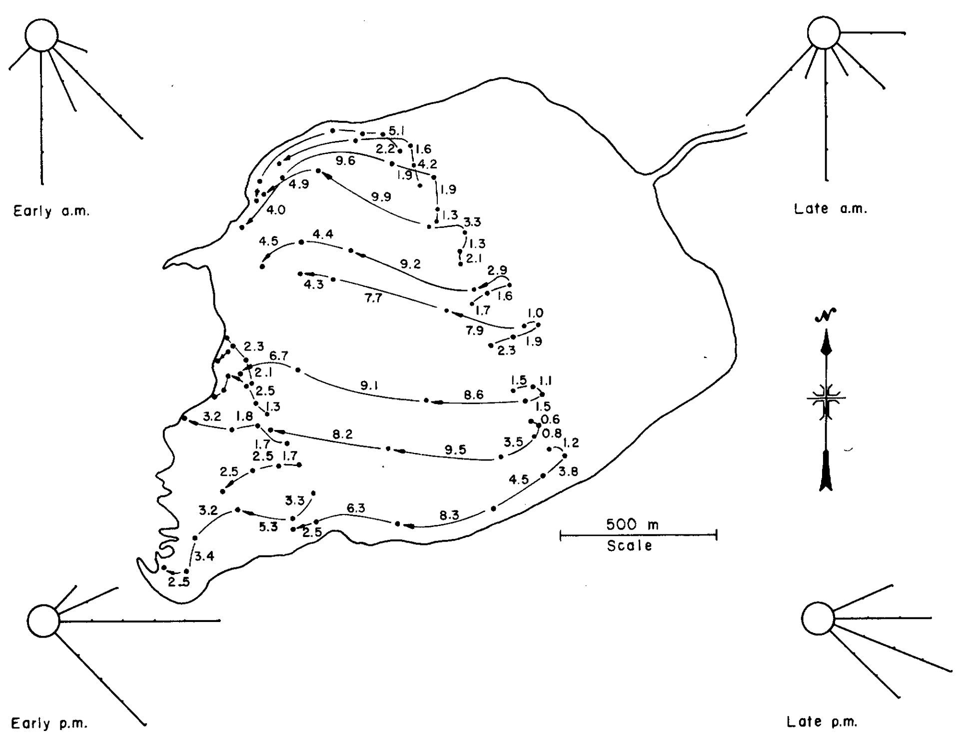

FIGURE 3-4.—*Surface current direction and velocity (cm/s) in Upper Twin Lakes, September 17, 1979. Wind roses show wind direction and velocity (m/s) on the day of current sampling.*

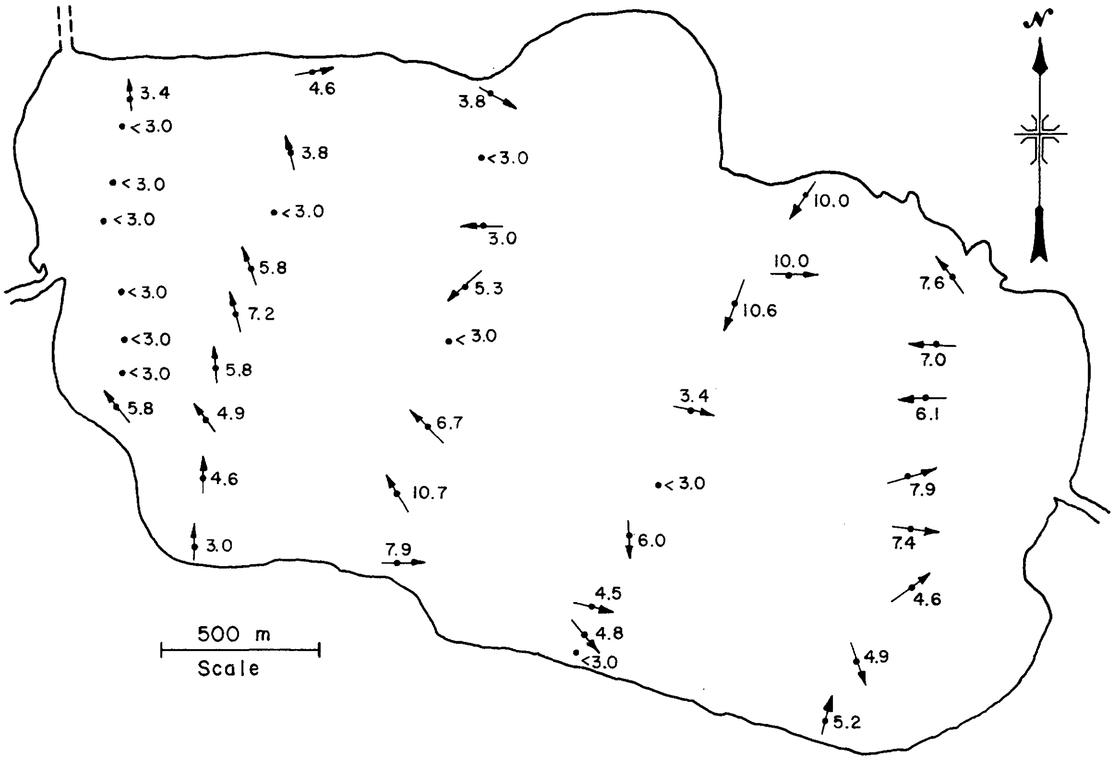

FIGURE 3-5.—*Direction and velocity (cm/s) of bottom currents in Lower Twin Lakes, September 11 and 12, 1979.*

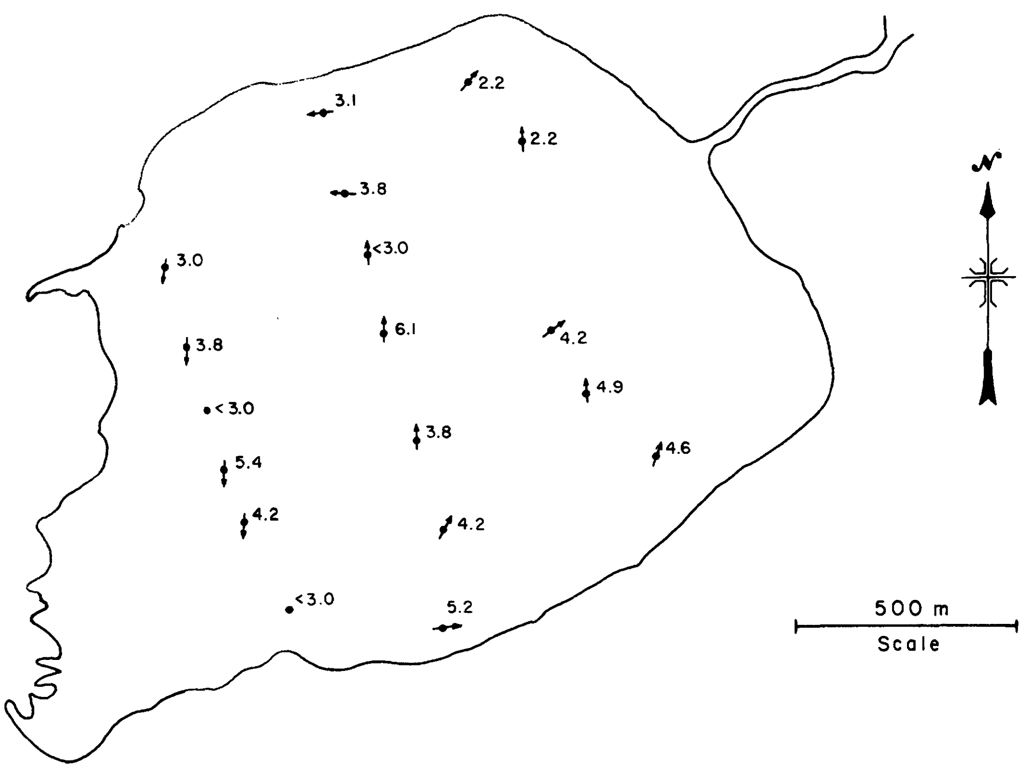

FIGURE 3-6.—*Direction and velocity (cm/s) of bottom currents in Upper Twin Lakes, November 26, 1979.*

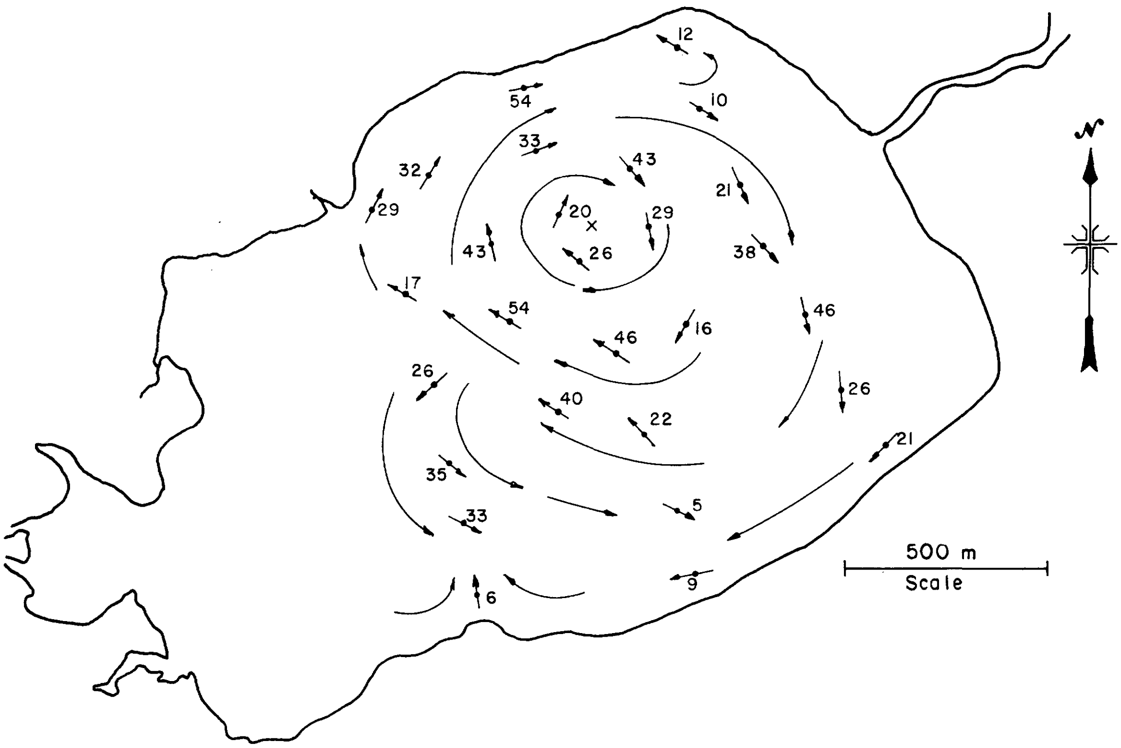

FIGURE 3-7.—*Under-ice current direction and velocity (cm/min) in Upper Twin Lakes, February 2, 1981.*

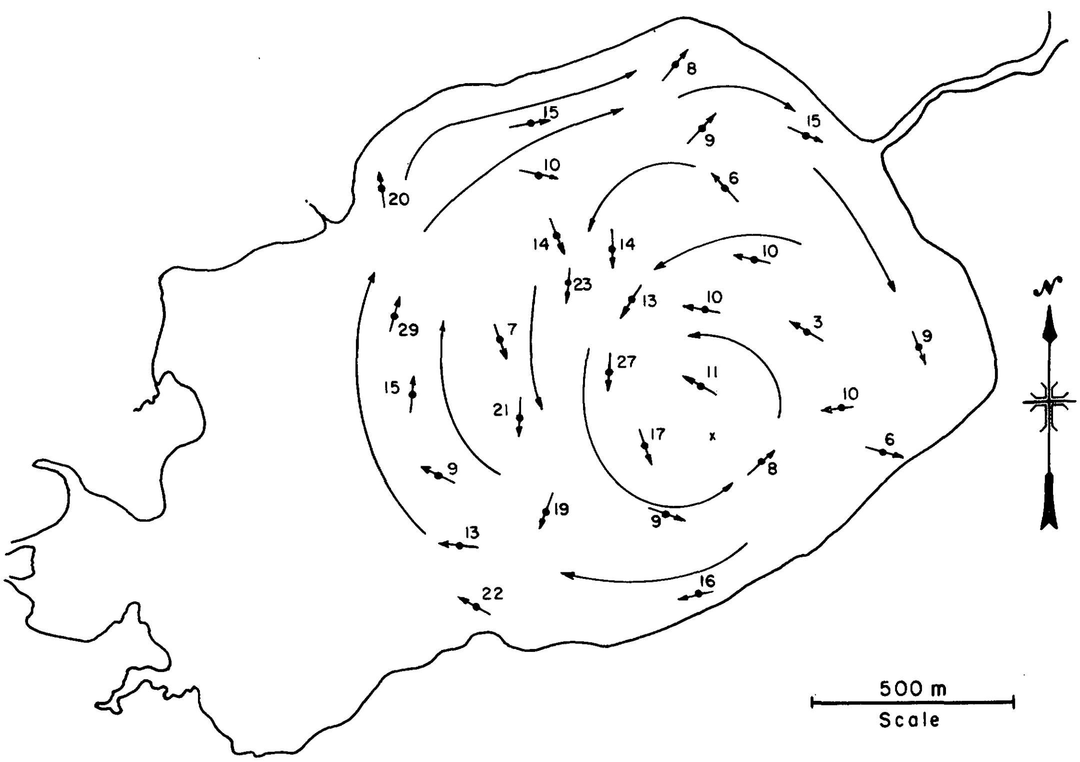

FIGURE 3-8.—*Under-ice current direction and velocity (cm/min) in Upper Twin Lakes, March 10, 1981.*

responsible for some of the under-ice water movement. Heat from the littoral sediments would create density differences between near shore and offshore waters, causing clockwise rotation at the perimeter and counterclockwise rotation in the middle. This pattern was duplicated by Fultz (1960) *in vitro* and was observed in March 1981 in the upper lake when the lake temperature was greater than 4 °C. Furthermore, temperatures increased 2 °C/m into the sediments under shallow water, and only 0.6 °C/m into the sediments in deeper parts of the lake, which supports the hypothesis regarding differential heating between littoral and profundal sediments.

Lower Twin Lakes Circulation

Surface currents on Lower Twin Lakes were more variable than those on Upper Twin Lakes because the wind patterns were more variable over the lower lake (figs. 3-9, 3-10). The north bay, Mackinaw Point, and the raised land south of the connecting channel between the lakes protected areas of the lake from direct exposure to the wind. For example, Mackinaw Point's east side was protected from west winds, and eddies frequently formed on the lee side. South of the channel, raised land with tree cover protected the channel's lee side.

West winds drive surface currents strongly downwind on the north side of the lake. Currents then tend to travel back upwind in the protected area north of Interlaken. The maximum observed current velocity was 822 cm/min.

Bottom currents proved to be variable and were not clearly related to wind patterns (figs. 3-11, 3-12). An obvious pattern could not be established for the lake as a whole. Strong currents were measured on Rocky, Hartman, and Mackinaw Points. Current velocities from 600 to 660 cm/min were recorded on Rocky Point in September and November. Hartman Point was found to have current velocities of 420 cm/min converging on its tip. In September and October, current velocities were measured off Rocky Point as high as 642 and 420 cm/min, respectively. Currents in these areas were likely to have kept sediments flushed out of rocks—providing good spawning substrate. Indeed, Walch (1980) also identified Mackinaw and Hartman Points as likely lake trout spawning areas. Also, Rocky and Mackinaw Points commonly received high fishing pressure, which indicated that lake trout congregated here at times other than the spawning season. The currents and presence of rocks were probably attractive to the lake trout.

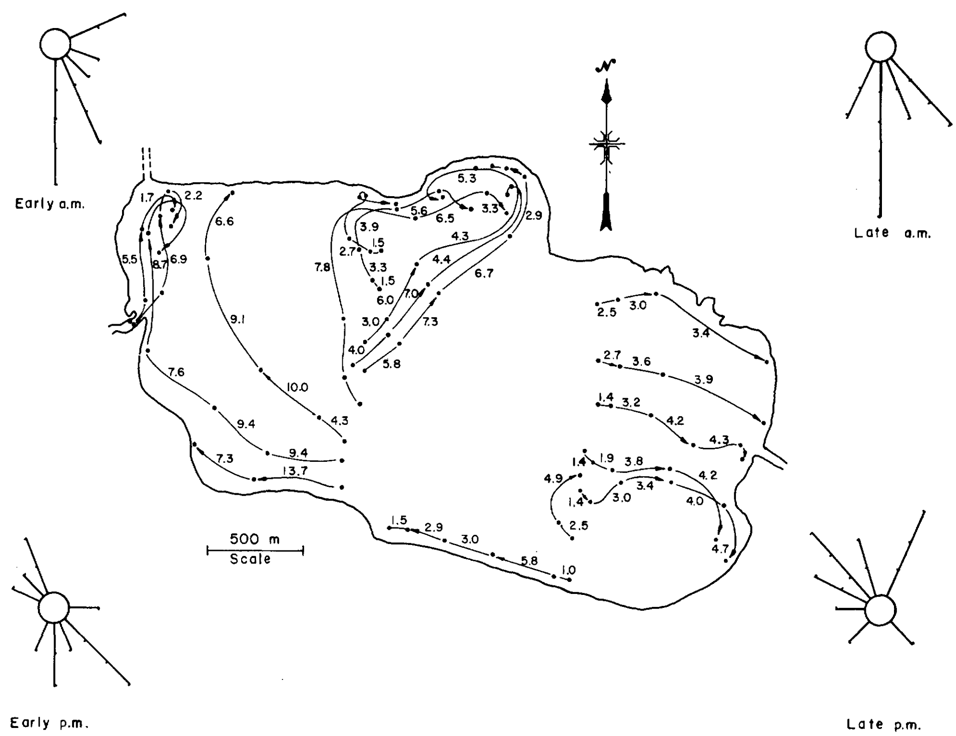

FIGURE 3-9.—*Surface current direction and velocity (cm/s) in Lower Twin Lakes, September 7, 1979. Wind roses show wind direction and velocity (m/s) on the day of current sampling.*

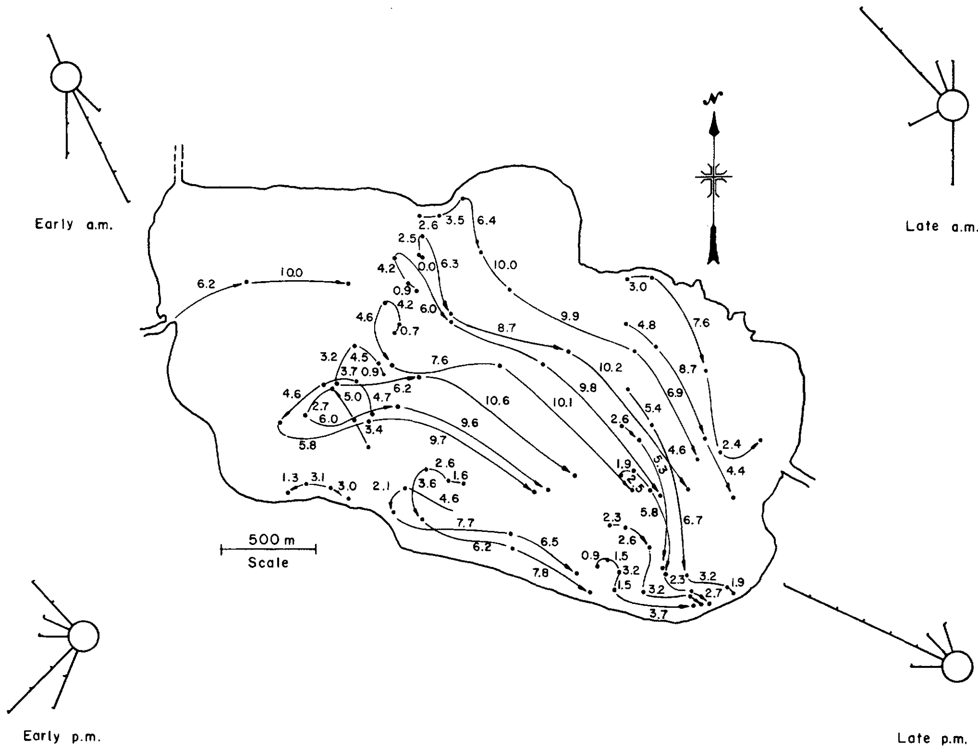

FIGURE 3-10.—*Surface current direction and velocity (cm/s) in Lower Twin Lakes, October 1, 1979. Wind roses show wind direction and velocity (m/s) on the day of current sampling.*

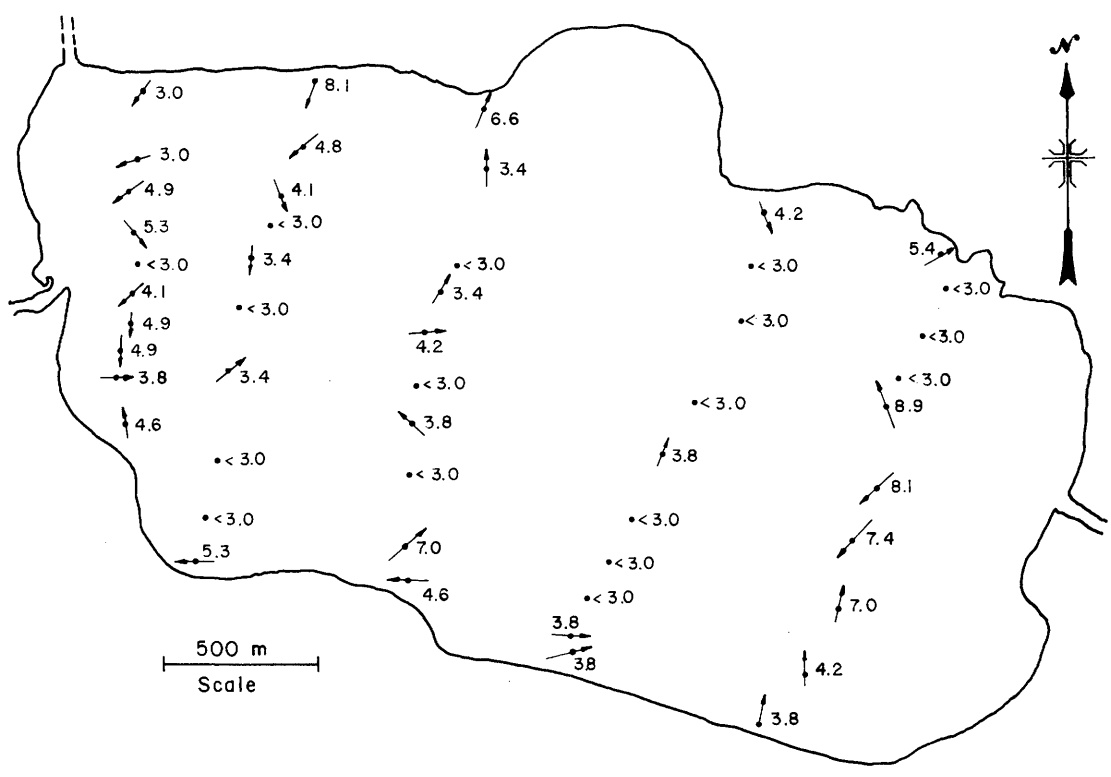

FIGURE 3-11.—*Direction and velocity (cm/s) of bottom currents in Lower Twin Lakes, October 2, 3, and 4, 1979.*

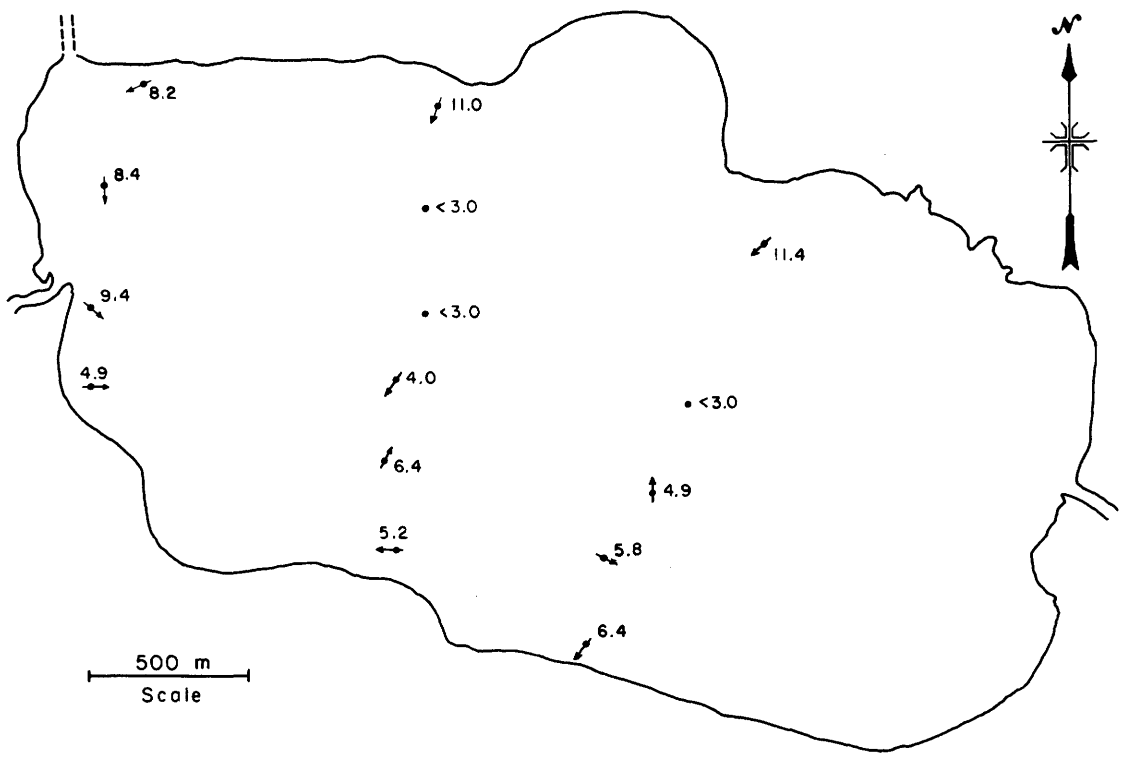

FIGURE 3-12.—*Direction and velocity (cm/s) of bottom currents in Lower Twin Lakes, November 14, 1979.*

In August 1979, strong currents moving to the northeast entered the lower lake through the channel between the lakes. This water did not continue along the bottom, but flowed into the epilimnion at some distance off the bottom where a temperature similar to that of the inflow was present. Current velocities as high as 912 cm/min were recorded in this area. These currents were the strongest of any observed in the two lakes.

In April 1980, under-ice currents in the lower lake were characterized by opposing pulses of water; a complete 180° change in direction occurred in a matter of minutes. As a result, distinct circulation patterns were difficult to identify. In January and February 1981, a general circulation pattern was discernible, but water movement was much more complicated in the lower lake than in the upper lake and included many small gyres (fig. 3-13). Circulation patterns were unstable and changed monthly. Average measured current velocity for January and February was 32 cm/min. Maximum measured velocity was 100 cm/min in January. In March, the current pattern changed to a central clockwise gyre (fig. 3-14). Current speeds were reduced greatly, with an average measured velocity of 13 cm/min. The eastern end of the lake was calm.

Irregular contours and irregular heating and cooling were suspected of causing the variable current patterns under the ice and the numerous small gyres. Pulsed currents moving in various directions were more common and variable than those in the upper lake. This finding suggested that surface area was a factor in their occurrence, which in turn, implicated differential pressure by wind on the ice cover as being responsible for some water movement in the winter (Verber, 1964).

Generation Cycle

Winter.—During the February 1982 generation cycle, strong flows were injected into the lower lake from the northwest corner. These currents dissolved the ice cover for about 500 m along the western shore of the lake. An unusually large and strong counterclockwise gyre formed in the western two-thirds of the lower lake (fig. 3-15). The eastern-most side of this gyre was delineated by the 18-m depth contour where the flat plain of the lower lake rises to the eastern shore. Though currents were measured at the 2-m depth, the apparent influence of bottom contours on currents patterns indicate that the gyre extended through the entire water column. Current velocity under the ice cover was measured as high as 174 cm/min. Average measured current velocity was 83 cm/min.

On the eastern third of the lake, several small gyres were found with velocities ranging from 13

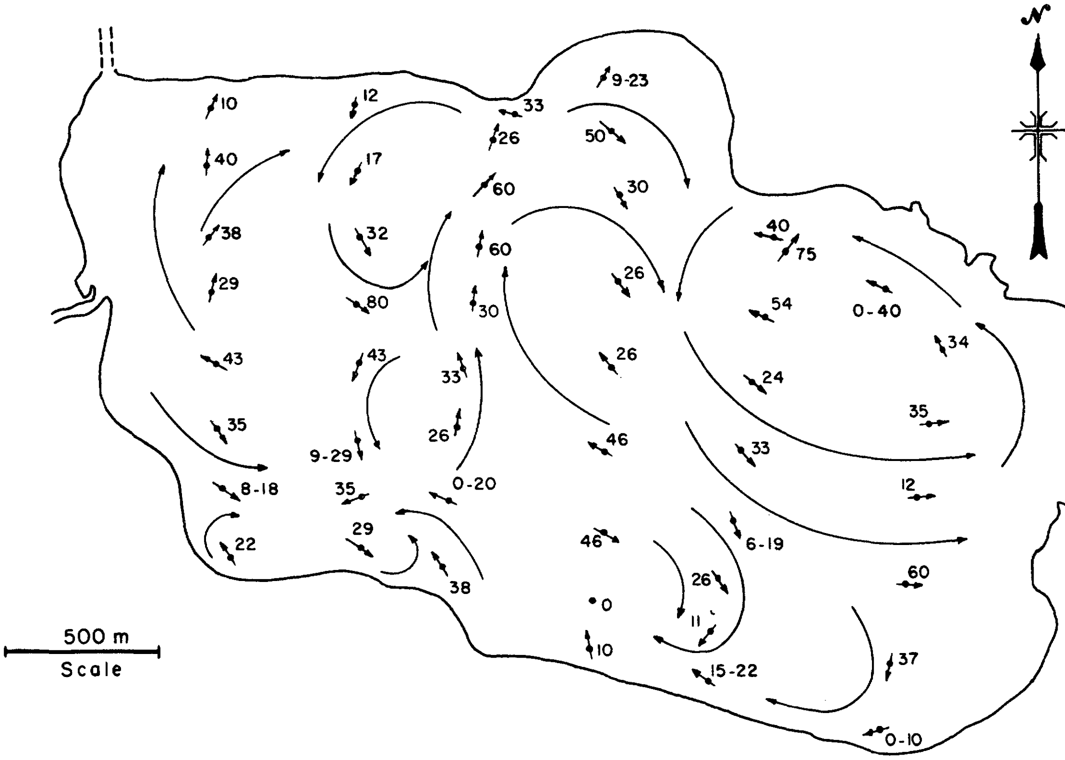

FIGURE 3-13.—*Under-ice current direction and velocity (cm/min) in Lower Twin Lakes, February 18, 1981.*

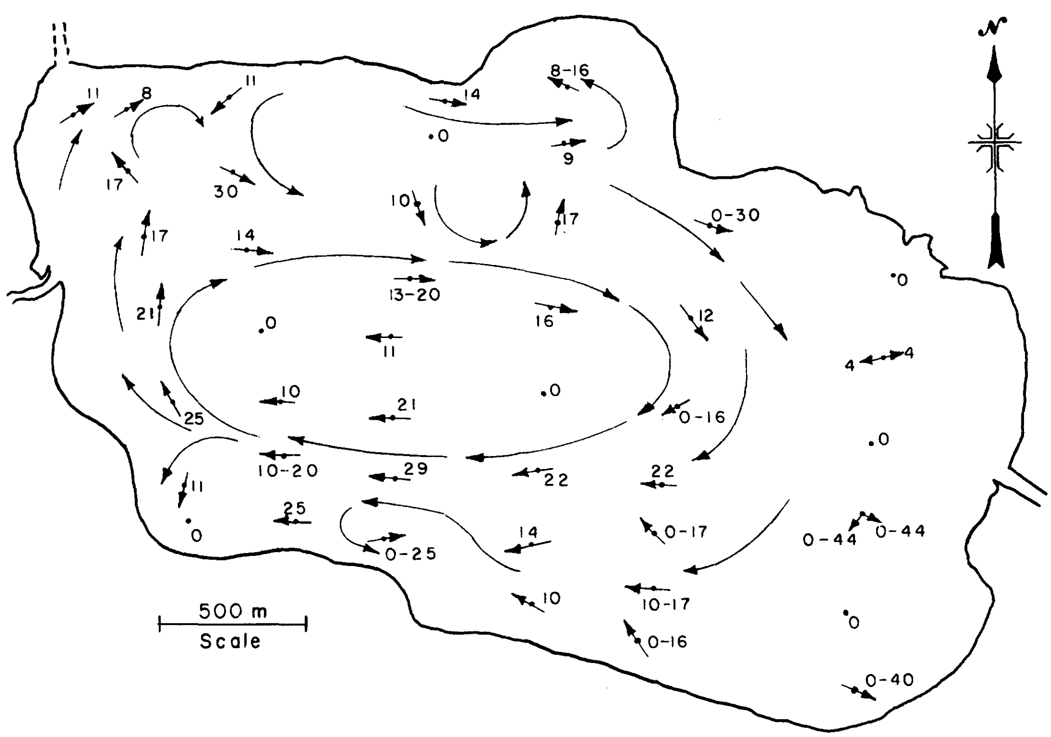

FIGURE 3-14.—*Under-ice current direction and velocity (cm/min) in Lower Twin Lakes, March 17, 18, and 19, 1981.*

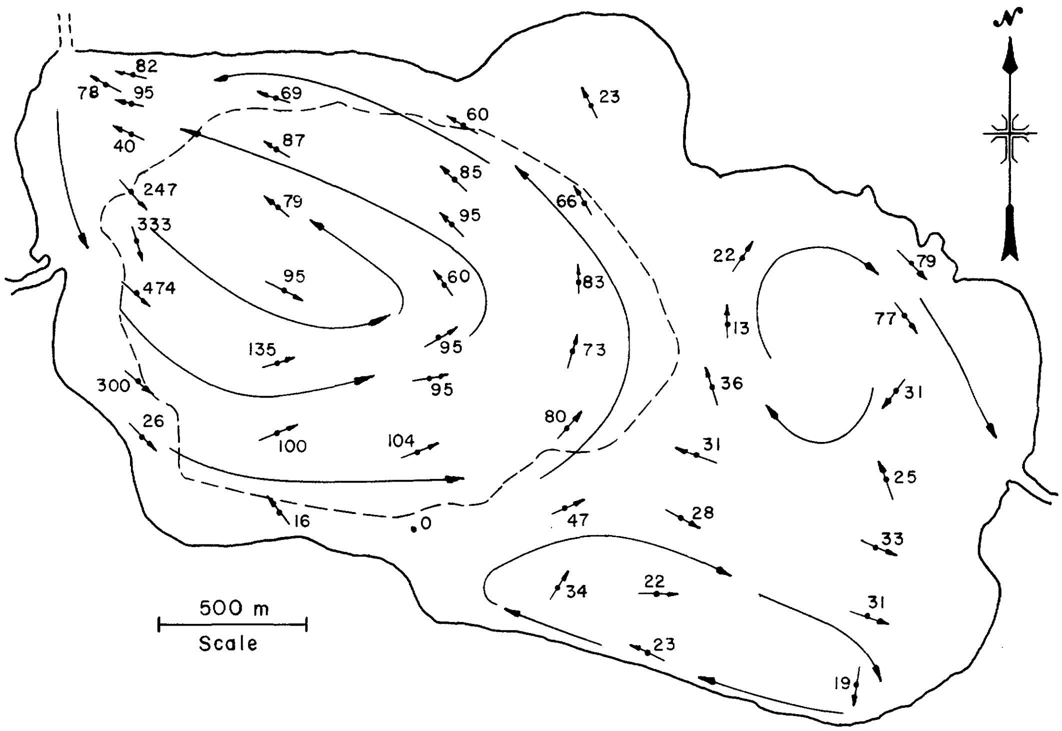

FIGURE 3-15.—*Under-ice current direction and velocity (cm/min) in Lower Twin Lakes, February 20, 1982.*

to 79 cm/min. This pattern contrasts that of February 1981 when numerous small gyres, with an average velocity of 30 cm/min, were observed. Operation of the powerplant circulated about two-thirds of the lake—including the deep basin. This circulation pattern continually brought planktonic organisms into the area near the front of the powerplant and increased their vulnerability to entrainment.

Summer.—Summer operation of the powerplant appears to influence surface currents only in the area near the tailrace channel. During powerplant operation in August 1984, surface currents moved generally from east to west—with predominantly easterly winds. The powerplant discharge did not appear to alter these wind-generated currents. It is suspected this may be due to the low temperature of the discharge that caused it to flow into the lower lake considerably below the surface.

* * * * *

Raft Monitoring of Selected Limnological and Meteorological Parameters

Chapter 4

S. D. Hiebert

Bureau of Reclamation
Applied Sciences Branch
Denver, Colorado

INTRODUCTION

Limnological and meteorological monitoring equipment mounted on rafts anchored at five stations on Twin Lakes system were used to obtain a near continuous record of selected water quality and weather parameters from the summer of 1980 through the fall of 1985. Temperature, pH, and DO (dissolved oxygen) were monitored at the 1-m depth at all stations. In addition, temperature, DO, pH, and ORP (oxidation-reduction potential) were monitored at the 13-m depth at the two stations centered in the upper and lower lakes. Meteorological parameters—including ambient light, air temperature, and average hourly windspeed—were monitored on two of the five instrument rafts.

The purpose of the monitoring/recording instrumentations was to detect diel changes in water quality at nominal depths and monitor short-term changes and long-term trends in water quality during Mt. Elbert Pumped-Storage Powerplant operations. Also, sensor reliability information were gathered. Preoperational data were collected in 1980-81.

Because of the large (≈800 000) number of data points collected from five stations over nearly 6 years, only data summarization (i.e., monthly, "core" period) is reported and discussed along with selected "snapshot" hourly results during shorter periods for comparison of preoperational and postoperational water quality parameters.

METHODS

The monitoring instrumentation was mounted on five rafts (fig. 4-1) that were moored at five research sampling stations (fig. 1-5) on Upper and Lower Twin Lakes and on Mt. Elbert Forebay. Each station's instruments were powered by a 12-volt dry cell battery that was recharged by a photovoltaic panel.

The limnological/meteorological monitoring equipment and software were designed and built for the study by Hydrolab Corporation. The monitoring instrumentation on each raft consisted of a surface data control unit that contained a microprocessor, memory, and digital display. The underwater data transmitters containing the water quality parameter sensors were connected to the surface data control unit. The temperature sensor on the underwater transmitter was a thermistor whose resistance decreased as temperature increased. The DO sensor on each underwater transmitter was a polarographic cell that produced a measurable current proportional to the amount of oxygen diffused through the cell's Teflon membrane. The pH was measured by a pH-sensitive glass electrode with a "wet" cell reference probe. The oxidation-reduction potential sensor consisted of a platinum electrode that developed a potential with respect to the reference cell according to the state of oxidation or reduction in the water. All these sensors were located at the base of the underwater transmitter.

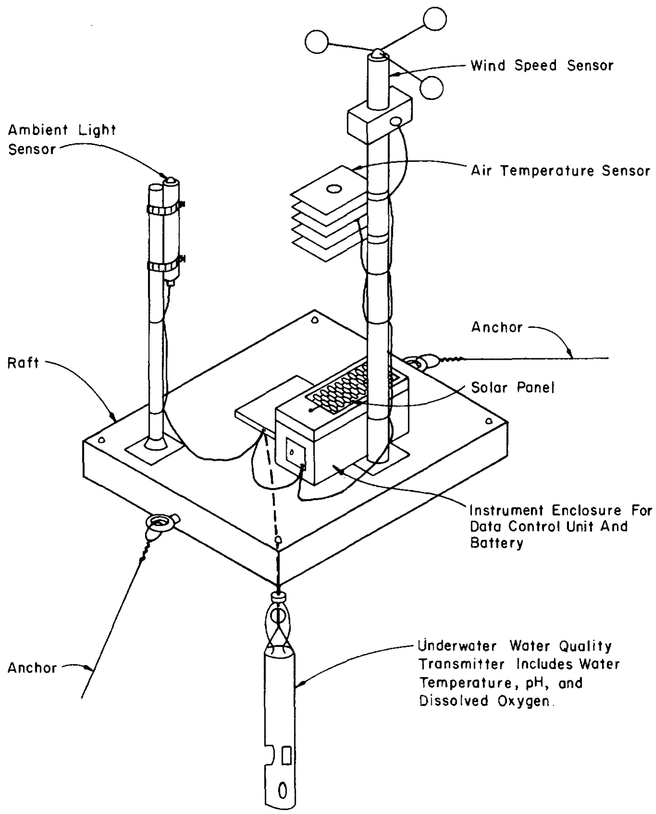

FIGURE 4-1.—*Schematic of typical raft and instrumentation used at Twin Lakes.*

At stations 3 and 5, there were raft-mounted light, windspeed, and air temperature sensors connected to the data control units. The light sensor consisted of a photo cell which produced a voltage according to the intensity of light striking the sensor. The wind sensor was a cup anemometer calibrated to record wind in miles per hour. Air temperature was measured with a thermistor similar to the underwater temperature thermistor. More detail on the monitoring equipment can be found in LaBounty and Hiebert (1981).

Equipment used in the retrieval of stored data included a portable Hewlett Packard Model 85 computer and memory interface module. The data from the control units were fed to the computer, and the underwater transmitters were recalibrated every 2 to 4 weeks—depending on the number of sensors and available memory. Final data reduction was performed with an IBM desktop computer and various data management software. Stations 2 and 4 had instruments that collected underwater data at the 1- and 13-m depths; stations 3 and 5 had instruments that monitored water quality data at the 1-m depth only. Frequent hourly and 2-hourly data collections were used to obtain a good estimate of the water quality at each station over the entire sampling period (Dandy and Moore, 1979).

It should be noted that the raft-mounted instruments were prototypes with limited field trials before use in this study. The prototype of the instrumentation, combined with extreme field conditions of temperatures below the components' operating ranges plus high humidities in the instruments' enclosures, produced some anomalies and suspect data. The suspect data was removed from the analyses—producing some data gaps. Sensor reliability and instrument performance are discussed later in this chapter.

RESULTS AND DISCUSSION

This section is divided into subsections on Mt. Elbert Pumped-Storage Powerplant operation, meteorological data, and station-by-station water quality pre- and postoperational data comparisons.

Mt. Elbert Pumped-Storage Powerplant Operation

Mt. Elbert Pumped-Storage Powerplant began operation in June 1981 with intermittent, short-duration tests of pumping and generation that continued into 1982. Generation and pumping operations, shown in the example on table 4-1, indicate time periods and water volumes. Similar information from additional time periods will be used later here when discussing visible effects of operation, however:

- The typical average operation schedule is nightly pumping from 2300 to 0600 hours the following day, when flow produces a volume of 1 472 777 to 2 454 629 m^3 (1194 to 1990 acre-ft).
- Daily generating from about 1000 to 1900 hours, when flow produces a volume of 2 466 964 to 3 638 771 m^3 (2000 to 2950 acre-ft).

The volume of water passed through the pump-turbine units varies with demand for generated electricity and available electricity to pump. In table 4-1, differentiation was not made between the units, while most generation was made with only one unit at a time. Volumes of water passed were measured in acre-feet over the total operation with most operating modes obtaining synchronous operation speed within 3 minutes of startup.

Meteorology

As part of the analysis of water quality parameters, meteorological data were collected so that any changes observed in the water quality parameters could be attributed to weather patterns, the operation of the pumped-storage powerplant, or a combination of both. Monthly meteorological data, primarily from station 3, are noted in table 4-2.

Table 4-1.—Mt. Elbert Pumped-Storage Powerplant operations—July 1984.

Day	Operating mode	Operating time, hour	Water passed, cubic meters	Volume, acre-ft
1	Generating	1700-2300	202 9078	1645
2	"	0700-1400	203 8946	1653
3	"	0900-1400		
	"	1800-2100	212 6523	1724
4	Pumping	0400-0700		
	"	2300-2400	107 6830	873
5	"	0000-0500	202 1677	1639
6	Generating	0900-1600	233 0047	1889
8	Pumping	2300-2400	18 3789	149
9	"	0000-0200	73 7622	598
	Generating	0900-1700	269 2691	2183
10	"	1000-1100	5 4273	44
11	"	1000-2100	349 4455	2833
12	"	0900-1200	55 8767	453
	Pumping	2300-2400	21 3392	173
13	"	0000-0600	124 3350	1008
	Generating	0900-1300	191 8065	1555
15	"	1800-2100	77 9561	632
16	Pumping	0200-0500	187 8593	1523
	Generating	1000-1800	236 8285	1920
17	Pumping	0300-0600	155 1720	1258
	Generating	1000-2300	365 9741	2967
18	Pumping	0300-0500	135 6830	1100
	Generating	2200-2300	30 5904	248
19	"	1000-1500	172 9342	1402
20	"	1800-2200	142 5905	1156
22	"	1000-1400	125 6918	1019
23	Pumping	0000-0400	157 2690	1275
	Generating	0800-1700	232 6347	1886
24	"	0800-0900	74 1323	601
25	"	0800-1700	272 1061	2206
27	"	1900-2200	101 0222	819
28	"	0900-1600	196 8637	1596
30	"	1700-2100	117 5508	953
31	Pumping	0100-0700		
	"	2300-2400	222 0268	1800
	Generating	0900-2200	366 7142	2973

Unfortunately, the meteorological data record has gaps because lightning and wind created static electric charges that disabled the sensors.

Warmest air temperatures were reached in July, with July 1981 having the warmest monthly average, 16.4 °C, of all years monitored. The summer daily air temperatures ranged from -4 °C (June 1983) to 25.6 °C (July 1981). The coldest month was usually December, with the coldest recorded air temperature in the tailrace channel (turbine afterbay) at -25.5 °C in December 1983.

September and November average windspeeds were generally lowest; winds in summer and winter averaged the highest speeds—usually greatest from 1000 to 1800 hours.

Table 4-2.— Average meteorological parameters from station 3—Lower Twin Lakes.

Year	Air temperature, °C	Light intensity	Wind m/s	Wind mi/h
1980				
Sept.	11.0	198	1.8	4
Oct.	7.9	175	2.2	5
Nov.	6.3	151	2.7	6
1981				
Apr.		200	2.7	6
May	9.3	212	2.7	6
June	13.5	246	2.7	6
July	16.4	278	2.7	6
Aug.	15.6		4.0	9
Sept.	10.1	155	0.9	2
Oct.	4.7	141	2.7	6
Nov.	1.1	134	0.9	2
1982				
Apr.	2.9	227	3.6	8
May	6.3	272	2.2	5
June	10.0	255	3.1	7
July	13.4	251	1.8	4
Aug.	13.4	216	4.9	11
Sept.	8.7	176		
Oct.	4.0	145		
Nov.	-4.9			
Dec.	-5.2			
1983				
Feb.	-3.2			
Mar.	-4.0			
Apr.	-2.5		4.5	10
May	3.5			
June	9.8			
July	14.2	260		
Aug.	14.0	228	4.1	11
Sept.	12.1	202		
Oct.	4.1	163		
Nov.	-2.1	163		
Dec.	-7.1			
1984				
Jan.	-7.5			
Feb.	-7.8	192	2.8	6.2
Mar.	-4.5	238	5.4	12
Apr.				
May	10.5	280	3.1	7
June	9.4	330	3.6	8
July	14.0	247	3.6	8
Aug.	13.0	207	3.1	7
Sept.	9.6	182	3.1	7
Oct.	2.0	152	4.9	11
Nov.	1.2	106		
1985				
May	10.7	218	5.4	12
June				
July	12.8	207		
Aug.	13.9	196		
Sept.	8.8	186		
Oct.	5.1	173		

Monthly average light intensity was usually highest in June, with the daily highs occurring at about 1300 hours mountain daylight time. There was up to a 70-percent reduction in summer light intensity because of cloudy conditions in the late afternoons as compared to clear days. Figure 4-2 shows an example of a 3-day diel cycle of light intensity, air temperature, and wind in May 1981.

Daily warming and cooling patterns appear similar between years by season, with no obvious unusual weather other than the 1981 average air temperatures being 1.2 °C warmer than the 1982-85 average air temperatures. Meteorological differences between Mt. Elbert Forebay and the tailrace channel were small, with Forebay air temperature ranging from 0.1 to 0.9 °C cooler, and higher average hourly winds in the summers of 1982-83. These differences are probably because Mt. Elbert Forebay is about 137 m (450 ft) higher in elevation and more exposed to the wind than the tailrace channel.

Water Quality

A "core" water quality data analysis period from June 1 to October 31 (each year) was used to standardize comparisons between stations 2 and 4 from 1981 through 1985. Other stations' graphed data are presented in non-"core" time periods

Station 1, at the eastern end of the lower lake, was monitored briefly in 1980 (Sept. 26 to Oct. 4) and summer through fall in 1981. Figure 4-3 shows hourly water temperatures and DO concentrations at the 1-m depths from July 9 through November 10, 1981. Maximum 1-m water temperature of 17.3 °C occurred August 6 at 0400 hours. Average water temperatures were 0.4 °C warmer in 1981 compared to the same sample period (late summer to fall) in 1980. Dissolved oxygen levels remained relatively stable, with a mean of 7.03 mg/L (standard deviation ± 0.26) from July 9 to November 10, 1981, and generally increased with decreasing water temperatures.

Temperatures fluctuated widely from July to early September (fig. 4-3) and then changed to a more consistent diel warming and cooling of the surface water. This change was probably caused by two factors: daily average windspeed—measured at station 3— dropped from 9 in August to 2.5 miles per hour in September (4 to 0.9 m/s); and the instrument raft was pushed into shallow water (8-m depth) in early September. Wind fetch from the westerly winds was greatest at station 1; consequently, raft movement on the waves and surface mixing would have been great enough to produce variable temperatures. The surface water temperature measurement became stable in September with the calmer weather. Station 1 was only monitored in late 1980 and 1981 because the stations nearest the powerplant were given priority in rotation of working instruments. Long-term monitoring at station 1 was eliminated during 1982-85.

At station 2, tables 4-3 and 4-4 show monthly average water temperature and DO for the "core" periods 1981-85. Maximum annual water temperatures at 1-m depth primarily occurred during August, with maximum temperatures ranging from 18.4 °C (Aug. 4, 1981, and Aug. 16, 1983) to 16.4 °C (Sept. 2, 1982). Diel temperature fluctuations were greater in summer (at both 1 and 13 m) than in the spring, before stratification, and in the fall after turnover. Figure 4-4 shows the 1984 hourly 1- and 13-m water temperatures, and DO with visible temperature stratification development and decline.

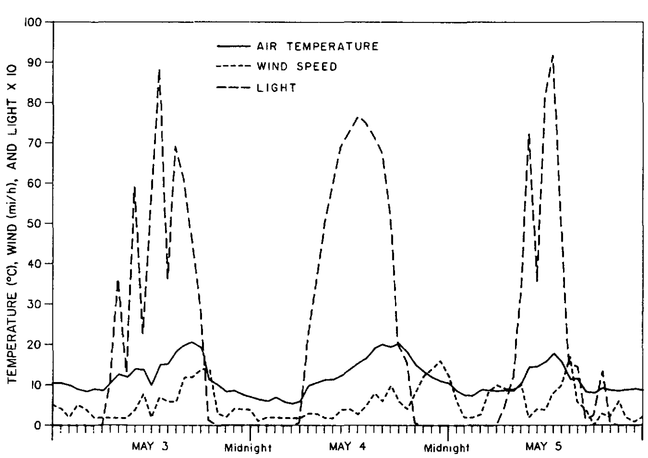

FIGURE 4-2.—*Air temperature, wind velocity, and ambient light in a 3-day "snapshot" from Lower Twin Lakes tailrace channel, station 3 (1981).*

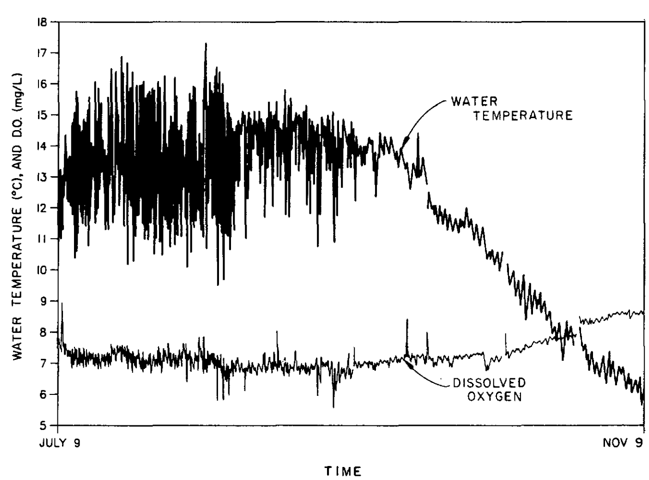

FIGURE 4-3.—*One-meter water temperature and dissolved oxygen concentration collected hourly from Lower Twin Lakes, station 1 (July 9 to Nov. 9, 1981).*

Table 4-3.—Average water temperatures at 1 and 13 m from station 2—Lower Twin Lakes.

	1981	1982	1983	1984	1985
		1 meter			
June	12.9	10.4	9.7	9.4	
July	16.4	13.8	12.5	15.1	14.7
Aug.	16.2	15.5	16.6	15.3	14.8
Sept.	14.0	12.0	13.2	13.1	11.3
Oct.	10.0	8.7	10.7	8.9	
		13 meters			
June	9.9	8.9	7.9	7.5	
July	11.3	7.7	8.9	9.4	
Aug.	10.4	11.1	9.3	10.4	
Sept.	11.4		11.5	11.1	
Oct.	10.5	8.8	10.4	8.7	

Table 4-4.—Average dissolved oxygen at 1 and 13 m from station 2—Lower Twin Lakes.

	1981	1982	1983	1984	1985
		1 meter			
June	7.7	8.7	9.4	8.8	7.9
July	6.7	7.9	8.2	7.8	7.6
Aug.	7.9	7.3	7.8	7.7	6.8
Sept.	7.3	7.4	8.1	7.4	6.8
Oct.	7.4	8.2	7.9	7.3	
		13 meters			
June	7.8	8.3	8.2	8.2	
July	6.6	6.7	7.7	7.8	
Aug.	6.6	6.6	7.1	6.9	
Sept.	6.4	7.5		6.6	
Oct.	7.7	7.8	7.0	6.9	

Figure 4-4 is typical of the "core" period from 1981 to 1985 at station 2. During most of the season, 1-m temperatures generally fluctuated more on a daily and weekly basis than at 13 m. The 13-m temperatures showed a steady seasonal rise and then a rapid cooling at turnover in the fall. Dissolved oxygen, at 1-m depth, ranged from a 5-season average August concentration of 7.2 mg/L to the May (1982-84) average of 8.6 mg/L. The DO at 13 m was lower by an average of 0.47 mg/L than the 1-m seasonal average and fluctuated more on a daily basis during the "core" period except during 1983. Seasonal DO concentrations were similar to those at 1 m in that they declined during the summer and increased in fall as the water cooled.

The 1-m "core" period average pH at station 2 ranged from 7.49 in 1982 to 6.81 in 1984. The pH at 13 m was an average 0.7 lower than the 1-m average over the 1981-85 period. The pH sensor reliability was suspect (see Instrument Performance sec.), as diel changes appeared to increase with duration of sensor immersion.

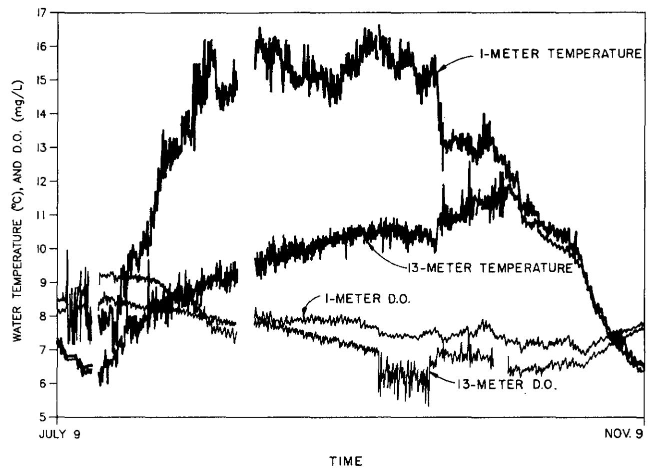

FIGURE 4-4.—*Typical seasonal 1- and 3-meter water temperature and dissolved oxygen concentration from Lower Twin Lakes, station 2 (1984).*

At the 13-m depth, ORP was similar over the seasons with "core" period averages ranging from 330 in 1982 to 392 in 1983. The ORP data at 13 m indicated an oxidizing condition at all times with a sensor "aging" trend of increasing ORP with time of immersion.

Pre- and Postoperational Data Comparison — Station 2

Water temperatures during operation averaged 1.7 °C lower at 1 m and 1.3 °C lower at 13 m than during the preoperational period (1981). The average postoperational summer maximum 1-m water temperatures were lower than preoperational temperatures by 1.8 °C. Similarly, air temperatures were warmer in 1981 (preoperational period) than 1982-85 (postoperational period); thus, the water temperature changes cannot be attributed entirely to powerplant operation.

Lower near-surface temperatures (during postoperation), while 13-m temperatures remain about the same, (between pre- and postoperational periods) are observations similar to those noted by Imboden (1979) and Potter et al. (1982). That is, pumped-storage operation causes an increase of vertical heat transport with a slight temperature increase in deeper water. In addition, the strength and duration of summer stratification in the lower lake is decreased by powerplant operation (Sartoris, 1989). Autumn water temperatures appeared to cool more rapidly during the postoperational period, with an average (Oct. 31, 1981) 13-m temperature of 7.9 °C and an average (1982-84) 13-m temperature of 7.4 °C. Onset of stratification in the spring was not recorded every spring, but extrapolation of data points from 1981, 1982, and 1984 indicate spring stratification, based on the last spring day of similar 1- and 13-m temperatures, was delayed 9 to 17 days from the preoperational year.

Fall turnover, based on the first date of 1- and 13-m water temperatures being equal, was similar between pre- and postoperational periods. These isothermal conditions at 1 and 13 m occurred on September 19, 1981; September 22, 1982; October 8, 1983; and September 26, 1984. Figure 4-5 shows hourly water temperatures at 1 and 13 m (from August 1 through 7 in 1981 and 1984), along with superimposed 1984 powerplant operations. Other than the 1984 lower 1-m water temperatures and similar 13-m temperatures, fluctuations attributable to powerplant operation are not apparent. Statistical tests could not be validly performed on the pre- and postoperational data because testing the single preoperational year against the 4 postoperational years would bias the results. Generally, DO concentrations followed a trend of being slightly higher during the postoperational period when water temperatures were lower. The pH levels varied from year to year, with no obvious outstanding differences between pre- and postoperational values. These fluctuations probably occurred more as a result of the natural system's variability from year to year.

Station 4 instruments were located on the upper lake and had identical equipment at the same depths as station 2. Monthly averages of water temperature and DO at 1 and 13 m are shown in tables 4-5 and 4-6. Daily fluctuations of parameters were similar to those observed at station 2, with smaller daily variations after fall turnover. Maximum water temperatures at the 1-m depth occurred usually during August and ranged from 17.8 °C (1981) to 16.0 °C (1982). Water temperatures at 13 m increased steadily to the seasonal high at fall turnover during all years, with the highest 13-m temperature (12.3 °C) in 1984.

As would be expected, oxygen levels varied inversely with water temperatures, with the lowest average "core" DO concentration of 6.4 mg/L occurring in August 1982. The pH levels fluctuated primarily because of sensor malfunction rather than lake conditions, with pH being higher at 1 m than 13 m by an average of 0.20 units in 1981-85. The pH at 1 m ranged from a minimum in 1983 of 6.58 to a maximum of 8.6 in 1982. At the 13-m depth, ORP was similar over the seasons with "core" period averages ranging from 333 in 1984 to 380 in 1981. Sensor aging also was noted for this parameter. The ORP data at 13 m indicated an oxidizing condition at all times with a trend similar to that observed at station 2 of increasing ORP with time of immersion.

Table 4-5.—Average water temperatures at 1 and 13 m from station 4—Upper Twin Lakes.

	1981	1982	1983	1984	1985
		1 meter			
June	8.8	8.0	7.2	9.3	11.7
July	15.1	11.1	10.3	13.7	13.9
Aug.	14.7	14.7		15.3	15.3
Sept.	12.4	12.5	13.5	12.7	12.7
Oct.	8.9	7.4	7.9	8.1	8.7
		13 meters			
June	8.6	7.0	5.7	6.9	
July	9.0	7.4	6.6	7.8	
Aug.	8.5	8.7		8.4	
Sept.	7.8	8.9	8.2	10.2	
Oct.	8.7	7.8	7.5	7.8	

Table 4-6.—Average dissolved oxygen at 1 and 13 m from station 4—Upper Twin Lakes.

	1981	1982	1983	1984	1985
		1 meter			
June	8.2	8.8	8.1	9.8	7.8
July	7.4	8.3	8.3	8.5	7.2
Aug.	7.2	7.3		6.0	6.0
Sept.	7.2	7.9	7.8	6.3	6.4
Oct.	7.7	8.1	7.8	6.9	6.5
		13 meters			
June	8.3	9.7	8.6		
July	7.8		8.3	8.0	
Aug.	7.2			8.0	
Sept.	7.9		8.0	7.0	
Oct.	7.2		8.1	7.1	

Pre- and Postoperational Data Comparison — Station 4

Lake levels were raised about 2.4 m during the postoperational period, which complicated the analyses. Rising lake levels inundated a large shallow area at the west end of the upper lake and probably affected the water quality parameters to some degree. Thus, observed changes in the water quality could have been caused by a combination of pump/generation, and inundation. Preoperational data were collected in 1981.

Preoperational water temperatures averaged 1.4 °C higher than postoperational temperatures at 1 m and 1 °C higher at 13 m. Water temperatures at 1 m and 13 m—to a lesser degree—fluctuated more widely on a daily basis in the spring period of postoperation than in the spring preoperational period. Figure 4-5 shows hourly water temperatures at 1 and 13 m for the same days in August 1981 and 1984, when the pumped-storage powerplant was in operation. Water temperatures were only slightly lower in 1984 during that period. The spring

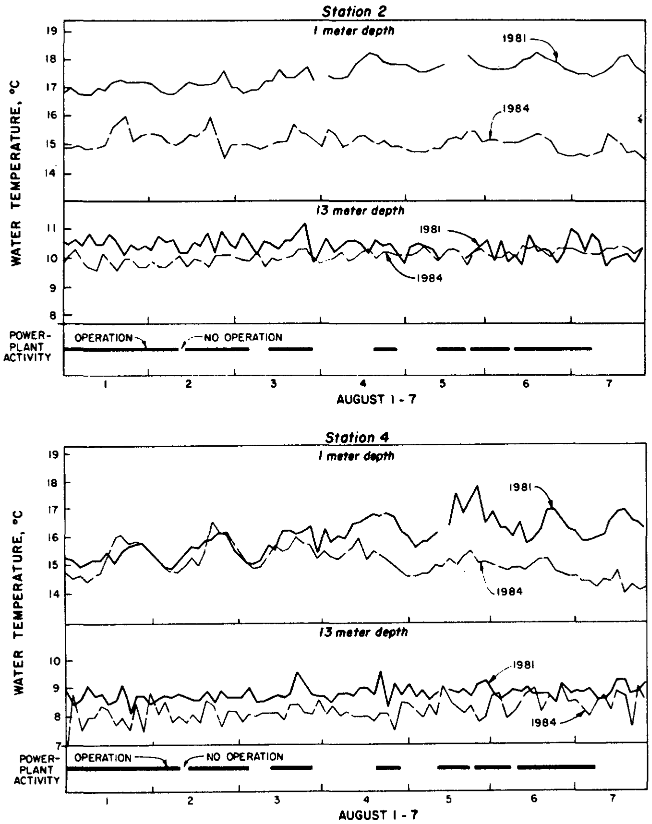

FIGURE 4-5.—*Water temperature at 1 and 13 m—pumped-storage powerplant operation (Aug. 1-7, 1981 and 1984).*

turnover was not monitored all years, but in 1982 and 1983 it was later than 1981 by 29 and 8 days, respectively. The fall turnover period, based on similar 1 and 13 m temperatures, was earlier in 1982 and 1983 than in 1981.

Dissolved oxygen levels appeared to drop more sharply in summer during the postoperational period than during the preoperational period. The pH levels fluctuated—probably due to sensor malfunction rather than environmental conditions—with no obvious changes during powerplant operation. The ORP did not appear different during postoperation than in preoperation.

Postoperational Effects — Station 3

On the lower lake, the raft at station 3 was located about 200 m in front of Mt. Elbert Pumped-Storage Powerplant. This station monitored water temperature, DO, and pH at 1 m, in addition to the meteorological parameters. Station 3 was located to obtain direct effect information from the pump or turbine flows and, consequently, had the most variable underwater measurements. The 1981 (preoperational) water quality conditions were similar to those at station 2, with "core" average water temperature of 12.9 °C, DO equal to 7.46 mg/L, and pH of 7.1.

Postoperational effects are illustrated in 2- and 9-day "snapshot" periods in 1984 (May 30-31, Aug. 22-31, Oct. 4-5), as shown on figures 4-6, 4-7, and 4-8. "Snapshots" from 1984 were used because powerplant operation was closest to normal on-line operation.

The changes observed at station 3 appear immediately after the powerplant begins daily operation, particularly in the generating mode, and more noticeably in the spring. Water temperatures dropped, often lower than the measured Mt. Elbert Forebay (Sta. 5) temperatures. The cooler water was probably imported from the nearby Forebay Inlet-Outlet Structure area. During the hours following startup for generation, water temperatures increased, reflecting mixing of the cooler Forebay water with the warmer lower lake water. Usually, water temperatures increased during the pumping mode, possibly as a result of pulling in the generally warmer epilimnetic water from the lower lake. During the summer and fall, the immediate effects were not as drastic. The greatest changes occurred after the powerplant had been idle for 2 or more days and the natural diel cycles had time to become re-established. Winter effects on the parameters were negligible during 1983 and 1984, with small temperature drops of 0.3 to 0.4 °C—some coinciding with generation and some not.

At station 3, dissolved oxygen levels did not follow any corresponding pattern to operation of the powerplant, possibly because the DO sensor response time was not as fast as the water temperature probe or because the DO levels were not affected passing through the powerplant. The pH at station 3 varied slightly (±0.2) during a pumping/generating cycle, which was less than during many periods when the plant was not operating.

Postoperational Effects — Station 5

Station 5 was located about 200 m north of the Mt. Elbert Forebay Inlet-Outlet Structure. It had instruments measuring water temperature, DO, and pH at 1-m depth as well as the meteorological parameters previously noted. The 1981 (preoperational) water quality conditions were similar to station 2, with "core" average water temperature of 13.1 °C, DO equal to 7.2 mg/L, and pH of 7.3. Average September water temperatures were warmer by 0.5 °C in 1985 than in preoperational 1981.

No discernible connections between fluctuations of any parameter and the pumping/generating operations of the powerplant were observed at station 5. Figures 4-9 and 4-10 present 1983 and 1985 "snapshots" of powerplant operation superimposed on the hourly water temperatures. These data, typical of the postoperational period, show no obvious disturbance of the diel trend. Any of the temperature variations during operation could have been caused by raising and lowering the water surface, which would have moved the temperature sensor up and down through the water column, or variations may reflect the temperature of the plume—from the Mt. Elbert Conduit—being drawn toward the penstock. This station had large data gaps (including the entire year of 1984) which make analysis difficult.

The DO changes were in line with seasonal temperature changes; fluctuations were caused by sensor malfunction. The pH was relatively stable during the periods when the pH sensor was considered reliable.

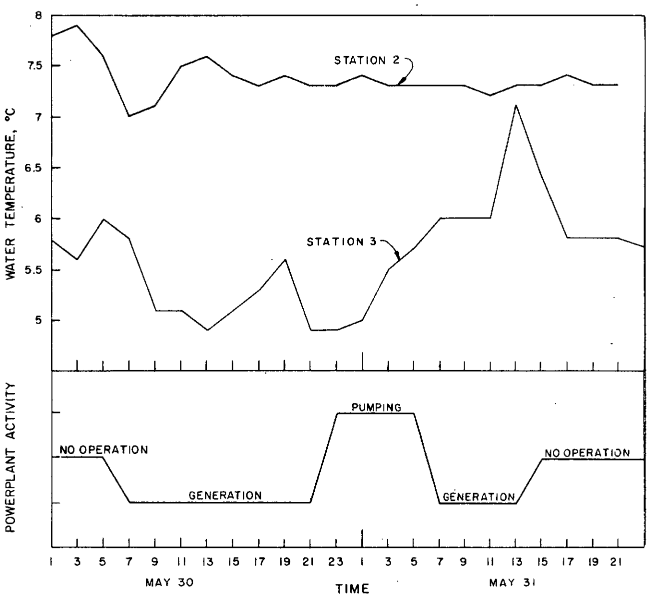

FIGURE 4-6.—*One-meter water temperature at Lower Twin Lakes, station 3 and station 2, and simultaneous powerplant operations (May 30-31, 1984).*

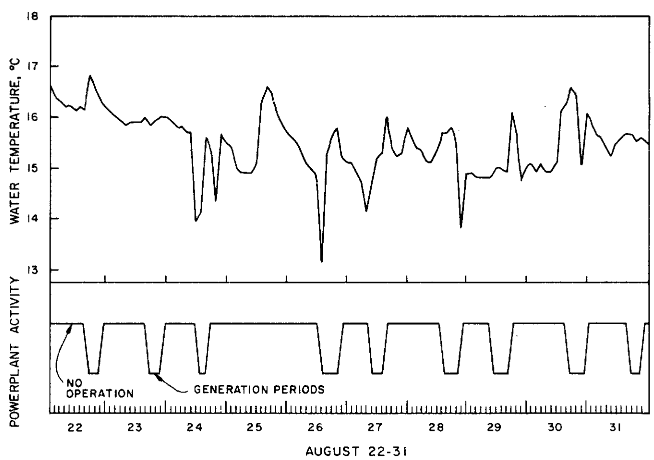

FIGURE 4-7.—*One-meter water temperature at Lower Twin Lakes, station 3, and simultaneous powerplant operations (Aug. 22-31, 1984).*

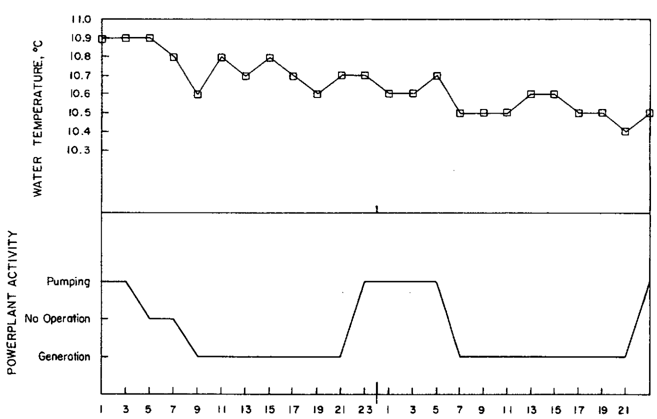

FIGURE 4-8.—*One-meter water temperature at Lower Twin Lakes, station 3, and powerplant operation (Oct. 4-5, 1984).*

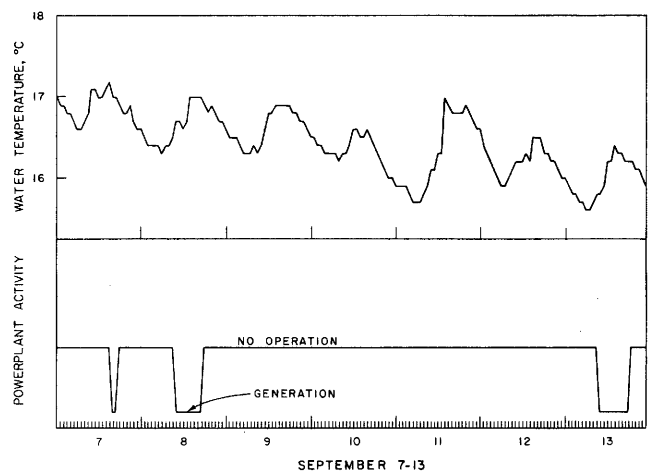

FIGURE 4-9.—*One-meter water temperature at Mt. Elbert Forebay, station 5, and simultaneous powerplant operations (Sept. 7-13, 1983).*

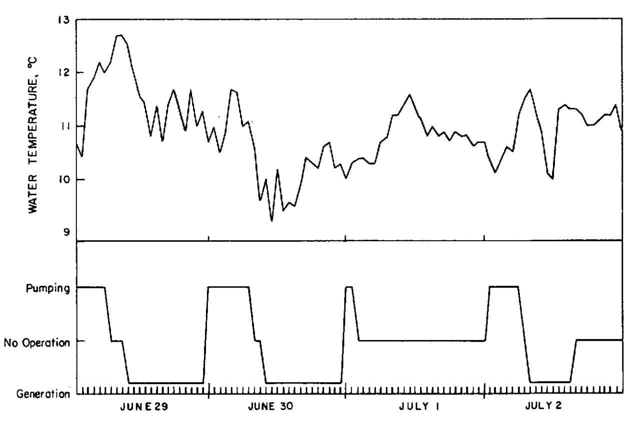

FIGURE 4-10.—*One-meter water temperature at Mt. Elbert Forebay, station 5, and powerplant operation (June 29 to July 2, 1985).*

Immediate Effects of Pumping and Generating at All Stations

As previously discussed, changes in water temperature were observed immediately at station 3, and were generally most obvious during generation. At stations 2 and 4, changes were not obvious immediately after startup of powerplant operation. Figure 4-11 presents 1-m water temperatures at stations 2, 3, and 5 over an operation cycle of generating and pumping. One-meter temperatures were lower at station 3 than at station 2. At station 2, the diel temperature cycle appeared to be unaffected.

In the later pumping cycle, as shown on figure 4-11, station 3 became warmer than the 1-m water at station 2 immediately after pumping began. Although not presented on figure 4-11, the 13-m temperatures at stations 2 and 1, and the 13-m temperatures at station 4 followed the same trend of no obvious effect. The DO and pH showed no obvious changes, although the DO levels inversely followed the temperatures to a small degree.

During regular powerplant operation, with one and two pump-turbines, water current measurements might help explain the lack of an immediately visible effect at stations 2 and 4 and also indicate the origin of the cooler/warmer water at station 3 during pumping and generating.

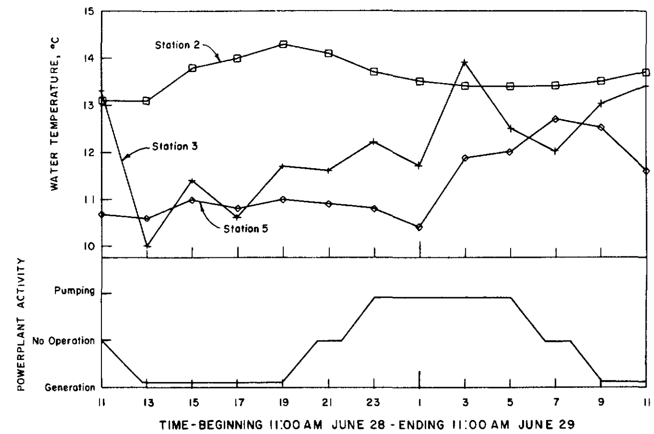

FIGURE 4-11.—*Comparative 24-hour water temperature at Lower Twin Lakes and Mt. Elbert Forebay and simultaneous powerplant operations (June 28-29, 1985).*

Instrument Performance

Incidental to diel cycles, long-term analysis, and pre- and postoperational differences, information was collected on the water quality sensor's performance. Throughout the study, the standard Hydrolab "wet" reference cell was used in all the underwater data transmitters to reference the pH (ions) and ORP readings. In 1984, Hydrolab Corporation and the U.S. Geological Survey—in testing acidity in mountain lakes—found that the wet cell reference was inaccurate in water of low conductance (<200 μS/cm). Twin Lakes conductance ranged from 35 to 99 microseimens per centimeter. Hydrolab Corporation felt the error was approximately ± 0.3 pH unit. Retrofitting to a Lazaran-type reference cell with high surface area for ionic communication in low conductance water was not possible; thus, monitoring continued as before.

Laboratory experiments on pH and reference probes showed a decline in pH observed after 3 to 5 days in distilled water. Applying the laboratory results to the field data suggested that pH readings could be suspect after the first week following recalibration. In addition, the continual increase in ORP over time between calibrations (fig. 4-12) could be a result of the inappropriate reference cell. The DO sensors drifted slightly over time and the sensors did not foul to any extent because of the relatively low productivity of the lakes. The 2- to 4-week period between calibrations seemed most reasonable, as the largest degree of sensor fouling occurred at stations 3 and 5 in 1985, when the time between calibrations was extended up to 7 weeks. During winter monitoring, the cold temperature resulted in battery component and electrical junction failures similar to those described by Walton (1982), whose instruments measured biological microclimates during the British Antarctic Survey.

One final note regarding prototypes: these instruments were limited in manufacture and not extensively field tested; thus, some operational problems resulted. Reliability increased, however, with operator familiarity and better documentation of the equipment's operation.

CONCLUSIONS

Conclusions are based on hourly, daily, monthly, and "core" period averages and trends. They could not be validly statistically tested.

1. Water temperatures at 1 and 13 m were warmer in the 1981 preoperational year than in the postoperational period (1982-85) at both stations 2 and 4. The DO inversely followed the temperature trends—averaging lower in 1981 than in 1982-85.

2. The pH and ORP appeared similar in both periods.

3. Powerplant operation effects at station 3 were immediately noticeable for temperature, then

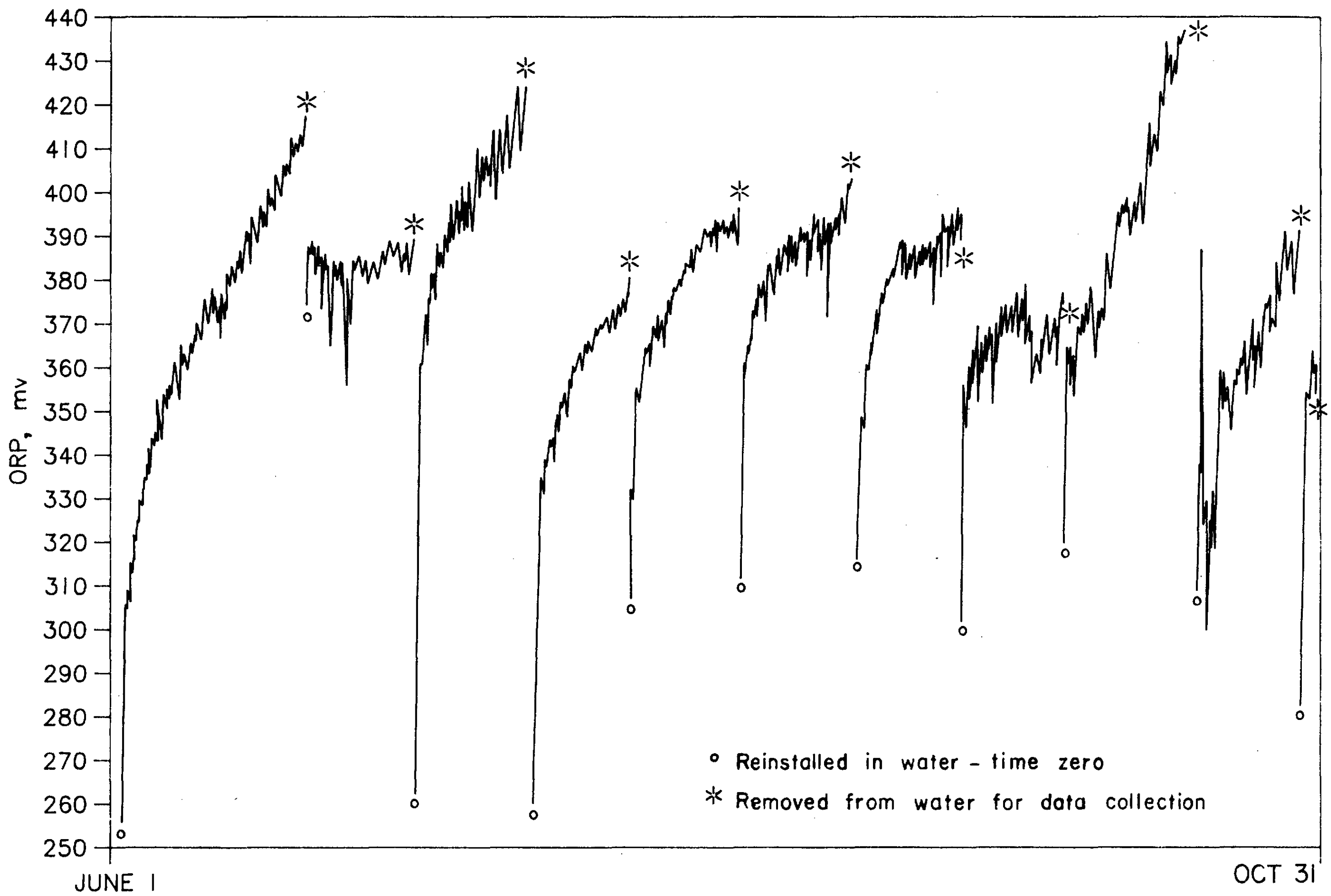

FIGURE 4-12.—*ORP variability between calibrations at Twin Lakes, station 2 (1981).*

stabilized at the temperature of water being passed through the powerplant. It appeared that the immediate, short-term effects were buffered by the volumes of the lower and upper lakes as well as Mt. Elbert Forebay. The DO and pH were not affected to the same degree as temperature.

4. Stratification periods were shorter during the postoperational period, with spring turnover occurring later and fall turnover occurring at nearly the same time.

5. Year-round trends were difficult to discern because of instrument and battery unreliability during winter monitoring. Winter operation broke the ice near the powerplant, and probably resulted in thinner ice and a shorter ice-cover season.

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Chapter 5
Water Chemistry Analyses

J. J. Sartoris

Bureau of Reclamation
Applied Sciences Branch
Denver, Colorado

INTRODUCTION

During the limnological surveys of the Twin Lakes system, water samples were collected routinely for laboratory chemical analyses. These analyses were of three general types: major ions, heavy metals, and nitrogen and phosphorus plant nutrients. Samples were usually collected from Lake Creek above and below Twin lakes, sampling stations 4 and 2 located in the middle of the upper and lower lakes, respectively, and sampling station 5 in Mt. Elbert Forebay (fig. 5-1).

This chapter discusses water chemistry data collected, from 1977 through September 1985, as standardization of analytical methods and sampling procedure during these years made the data more statistically comparable for analysis of trends and relationships among the various parameters. The 1971-76 data were sparse and scattered both spatially and temporally and are not discussed in this chapter; those interested in these data are referred to Sartoris et al. (1977).

The data presented here are further divided for discussion and analysis into two periods: the "pre-project" or *preoperational period*—1977 through 1981, and the "project" or *postoperational period*—1982 through September 1985. The various project features at Twin Lakes were built and brought into operation over a period of several years; thus, a specific cutoff date between preoperational and postoperational periods is somewhat arbitrary. January 1 was selected as the beginning of the postoperational period because 1982 was the first full year of operation for Mt. Elbert Pumped-Storage Powerplant. Testing of the first 100-MW unit began in June 1981; the unit went on-line in September 1981.

METHODS

Water chemistry samples from Twin Lakes and Mt. Elbert Forebay were collected with VanDorn or Kemmerer-type water samplers. In the lakes, samples were usually taken at 1-m depth, mid-depth of the water column, and within 1 m of the bottom. Mt. Elbert Forebay samples were taken at mid-depth only because of the relative shallowness of this impoundment and its high degree of mixing. Inflow and outflow samples were dipped directly from the stream.

Beginning in 1982, at stations 2 and 4, all samples for major ions and the 1-m and mid-depth samples for heavy metals were eliminated from the routine survey sampling procedure as a cost-saving measure. Thereafter, these samples were only included in the collection procedure during four surveys per year:

- At spring turnover,
- At fall turnover,
- At mid-winter (usually Jan. or Feb.), and
- At the peak of summer stratification (late July or early Aug.).

All water chemistry samples were collected in Nalgene bottles. The samples for nitrogen and phosphorus plant nutrient analysis were frozen soon after collection and kept frozen until they were analyzed in the laboratory. Samples for heavy metals were "spiked" with about 1 milliliter of concentrated nitric acid per 250-mL sample to reduce the pH to less than 2.0 to keep the metal salts in solution for analysis.

Laboratory analyses of all the water chemistry samples for Twin Lakes were provided by the Chemistry, Petrography, and Chemical Engineering Section at the Bureau of Reclamation's Research and

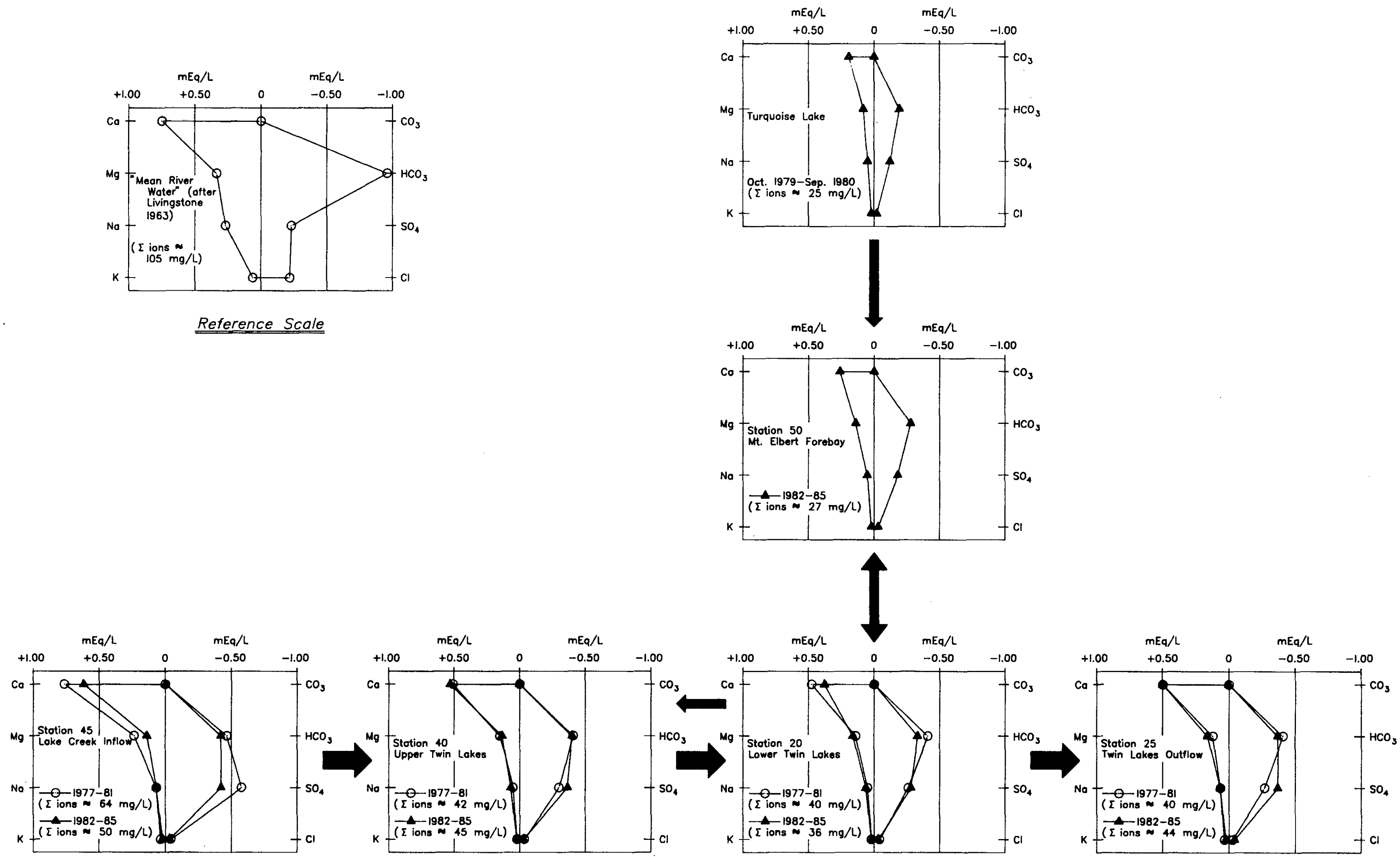

FIGURE 5-1.—*Ionic composition of the Twin Lakes system.*

Laboratory Services Division, Denver, Colorado. All analyses were performed according to the *National Handbook of Recommended Methods for Water-Data Acquisition* (USGS 1977), either in-house or contracted with other laboratories.

RESULTS AND DISCUSSION

Major Ions

These data included conductivity, or specific conductance, of each water sample, as well as the concentrations of TDS (total dissolved solids) and the major anions and cations. The sum of the concentrations of anions and cations was reported as salinity. The specific parameters are:

Cations: calcium (Ca^{+2}), magnesium (Mg^{+2}), sodium (Na^{+1}), and potassium (K^{+1});
Anions: carbonate (CO_3^{-2}), bicarbonate (HCO_3^{-1}), sulfate (SO_4^{-2}), and chloride (Cl^{-1});
Ionic indexes: conductivity (EC), total dissolved solids (TDS), and sum of ions (salinity).

Carbonate will not be considered further because it was rarely detected in the waters of the Twin Lakes system. Given the near-neutral pH range of Twin Lakes, the absence of CO_3^{-2} is not unexpected.

Mean preproject and project period values of the major ion concentrations and ionic indexes are listed in table 5-1. Table 5-2 lists corresponding means for Turquoise Lake, rivers in North America, and rivers worldwide for comparative purposes.

Generally, the waters of the Twin Lakes system may be characterized as a dilute solution of predominately calcium, bicarbonate, and sulfate. Sulfate is an especially important anion in the Lake Creek inflow to Twin Lakes. The relatively high proportion of sulfate in Lake Creek is probably explained by the presence of large deposits of jarosite, a hydrous sulfate of iron and potassium, at the headwaters of the South Fork of Lake Creek (Sartoris et al., 1977).

Preproject and project major ion mean concentrations from table 5-1 were converted into units

Table 5-1.—Ionic indexes and major ion concentrations in the Twin Lakes system.

Location	Lab. EC µS/cm	TDS mg/L	Cations, mg/L				Anions, mg/L			Sum* of ions mg/L
			Ca	Mg	Na	K	HCO_3	SO_4	Cl	
			Preoperational Means (1977-81)							
Lake Creek inflow	109	75	15.2	2.84	1.66	1.02	28.4	27.8	1.52	64.0
Upper Twin Lakes	74	54	10.2	1.85	1.18	0.96	25.1	14.2	1.20	42.0
Lower Twin Lakes	69	50	9.56	1.85	1.21	0.96	24.8	12.6	1.35	39.7
Twin Lakes outflow	70	52	10.0	1.48	1.28	1.00	25.0	13.2	0.73	39.9
			Postoperational Means (1982-85)							
Lake Creek inflow	69	53	12.4	1.78	1.60	0.67	25.4	20.0	1.17	50.1
Upper Twin Lakes	70	50	10.6	1.64	1.65	.75	24.7	17.0	0.88	44.7
Lower Twin Lakes	54	40	7.66	1.94	1.30	.62	20.0	13.7	0.87	35.9
Twin Lakes outflow	68	48	10.0	1.98	1.38	.64	22.8	17.7	1.07	44.0
Mt. Elbert Forebay	37	28	5.20	1.72	1.13	.60	17.3	8.55	1.20	26.9

*This value is also a mean and not the sum of the columns to the left.

Table 5-2.—Selected ionic data for comparison with the Twin Lakes system (table 5-1).

Location	Lab. EC µS/cm	TDS mg/L	Cations, mg/L				Anions, mg/L			Sum* of ions mg/L
			Ca	Mg	Na	K	HCO_3	SO_4	Cl	
Turquoise Lake, Colorado†	34	26	3.74	0.97	1.05	0.78	11.7	5.86	0.72	24.8
Mean composition of world rivers‡		120	15	4.1	6.3	2.3	58.4	11.2	7.8	10.5
Mean composition of North American rivers‡		142	21	5	9	1.4	68	20	8	132

* This value is also a mean and not the sum of the columns to the left. † Nesler (1981). ‡ Livingston (1963).

of milliequivalents per liter (mEq/L) and plotted as Stiff diagrams (fig. 5-1). The corresponding Stiff diagram for the mean ionic composition of rivers worldwide (Livingstone, 1963) is also included on this figure for scale.

During the postoperational period, two general trends are shown on figure 5-1. First, decreases in the ionic concentration of both the Lake Creek inflow and the lower lake are evident during the project period. These decreases were the only changes in ionic composition shown to be statistically significant by the Mann-Whitney U/Wilcoxon Rank Sum W Test. The decrease in Lake Creek salinity in the project period is significant beyond the 5-percent level (P=0.024), whereas that in Lower Twin Lakes is significant beyond the 1-percent level (P=0.004). At station 2, in the lower lake, dilution of the mean ionic concentration was a direct result of the project's import of extremely dilute water from Turquoise Lake by way of the Mt. Elbert Conduit and Forebay. Lake Creek inflow was more dilute from 1982 through 1985 because flows in the creek averaged much higher during this period than during the preproject years of 1977 through 1981 (see ch. 2).

The second apparent trend (fig. 5-1) is the greater similarity between the mean ionic composition of Lake Creek below the lakes and that of the upper lake during the postproject period. This similarity indicates a direct surface flow between the upper lake and the Twin Lakes outlet, with little dilution by the water imported from Turquoise Lake, which appears to plunge to the bottom of the lower lake during power generation. Water current measurements taken in lower Twin Lakes, in August 1980 and August 1984, support this interpretation (see ch. 3).

Heavy Metals

During this study, analyses were performed for five heavy metals:

- Iron, Fe
- Manganese, Mn
- Copper, Cu
- Zinc, Zn
- Lead, Pb

Cadmium, Cd, was added to the routine analysis list late in the preproject period; thus, comparisons of preproject and postoperational period concentrations of Cd suffer from unequal data sets.

A second major constraint on statistical analysis and interpretation of the heavy metal data is that analytical detection limits for the various metals changed several times and varied widely, as different chemists and laboratories worked on the samples from January 1977 through September 1985. It was necessary to assign reasonable and consistent values to those results reported as nondetectable, ND, at the various detection limits, DL, to be able to analyze statistically the entire data set. The following procedure was developed to handle this problem for each metal.[1]

1. All ND results at the lowest DL for a given metal were assigned a value of one-half the DL.

2. Then, the entire data set was scanned for all reported detectable and previously assigned values that were less than the next higher DL. These values were averaged, and the mean was assigned to all results reported as ND at this DL.

3. Step 2 was repeated for each successively higher DL until all the ND results had been assigned a non-zero value.

As a check on possible biases introduced by this method of assigning quantitative values to the ND results, correlations with time were calculated for both the DLs and the assigned values. Correlation did not exist in either case, nor was there any correlation between the various DLs and the assigned values aggregated over time.

The preceding discussion emphasizes the fact that chemical water quality data are often non-normally distributed. Non-normal data distributions are especially common in the case of trace substances, such as heavy metals, where a relatively large proportion of the concentrations are below DLs and a few individual concentrations are scattered toward the high end of the observed range. These types of data are analyzed best with nonparametric statistical procedures (Helsel 1987). In the following discussion of heavy metals in the Twin Lakes system, the distributional parameters used to describe the data are the maximum observed concentration, the median, and the quartiles. Differences between pre- and postoperational periods were evaluated with the Mann-Whitney U/Wilcoxon Rank Sum W Test.

Tables 5-3 through 5-5 show pre- and postoperational period distributional statistics on heavy metal concentrations in the Twin Lakes system. It should be noted that only bottom concentrations are listed for the upper and lower lakes because only bottom heavy metal samples were collected routinely at stations 2 and 4 throughout both periods. The number of samples available are thus

[1] J. W. Yahnke, personnel communication.

Table 5-3.—Iron and manganese concentrations in the Twin Lakes system.

Location	Oper-tional period	EPA criteria* μg/L	Concentrations, μg/L 25th Percentile	Median	75th Percentile	Maximum observed	No. of Samples
			Iron				
Lake Creek inflow	1977–81	†1000	238.8	325.0	477.5	3540	52
	1982–85	1000	270.8	344.0	419.0	2400	72
Bottom of upper lake (Sta. 4)	1977–81	1000	110.0	180.0	335.0	2080	58
	1982–85	1000	102.2	180.0	372.0	3600	70
Bottom of lower lake (Sta. 2)	1977–81	1000	40.0	80.0	120.0	860	57
	1982–85	1000	51.5	76.8	103.0	520	72
Mt. Elbert Forebay	1982–85	1000	51.5	76.8	103.0	520	72
			Mangnese				
Lake Creek inflow	1977–81	‡50	10.0	12.1	20.0	53	52
	1982–85	50	2.35	10.0	15.4	59	72
Bottom of upper lake (Sta. 4)	1977–81	50	20.0	30.0	52.5	2600	58
	1982–85	50	10.0	18.6	41.0	149	70
Bottom of lower lake (Sta. 2)	1977–81	50	10.0	16.0	27.5	320	57
	1982–85	50	5.65	15.0	23.8	345	72
Mt. Elbert Forebay	1982–85	50	0.14	5.1	7.0	28	72

* U.S. Environment Protection Agency (1986). † Freshwater aquatic life criterion.
‡ Domestic water supply criterion.

Table 5-4.—Copper and zinc concentrations in the Twin Lakes system.

Location	Oper-ational period	EPA criteria* μg/L	Concentrations, μg/L 25th Percentile	Median	75th Percentile	Maximum observed	No. of Samples
			Copper				
Lake Creek inflow	1977–81	6.5	7.00	11.0	18.0	40.0	52
	1982–85	5.2	5.42	11.4	18.0	31.7	72
Bottom of upper lake (Sta. 4)	1977–81	4.6	2.75	3.00	5.00	24.0	58
	1982–85	4.6	2.15	3.23	4.82	14.0	70
Bottom of lower lake (Sta. 2)	1977–81	4.4	2.00	2.35	4.00	13.0	57
	1982–85	3.9	0.92	2.70	3.23	11.0	72
Mt. Elbert Forebay	1982–85	3.0	1.28	3.23	5.18	39.1	72
			Zinc				
Lake Creek inflow	1977–81	58.4	3.20	5.50	12.0	195	52
	1982–85	46.9	0.36	4.00	8.75	36.0	72
Bottom of upper lake (Sta. 4)	1977–81	41.5	2.00	4.40	10.0	100	58
	1982–85	41.6	0.05	1.60	5.00	23.0	70
Bottom of lower lake (Sta. 2)	1977–81	39.8	2.50	4.40	9.00	50.0	57
	1982–85	35.1	0.05	1.60	4.75	25.0	72
Mt. Elbert Forebay	1982–85	27.2	1.60	3.14	7.00	26.0	72

* Freshwater aquatic life chronic toxicity criteria computed on the basis of hardness (U.S. Environmental Protection Agency, 1986).

more comparable to the number of inflow and Mt. Elbert Forebay samples than are the less frequently collected surface and mid-depth lake samples. Samples from the bottom of a lake are also more indicative of internal loading by hypolimnetic sediment release of reduced metals.

Iron and manganese (table 5-3) are the most abundant metals in the Twin Lakes system. From 1977 through 1985, iron was detected in 92 percent of all the samples collected, and manganese was detected in 71 percent of the samples. These metals are not classified as priority pollutants, in the sense of toxicity to aquatic life (EPA, 1986), but they are indicative of lake redox conditions, and iron plays an important role in the phosphorus budget of any aquatic system. In table 5-3, data show that iron is primarily imported into the lakes by the Lake Creek inflow, whereas manganese is mainly internally loaded by release from sediments during

Table 5-5.—Lead and cadmium concentrations in the Twin Lakes system.

Location	Oper-ational period	EPA criteria* µg/L	Concentrations, µg/L 75th Percentile	Maximum observed	No. of samples
		Lead			
Lake Creek inflow	1977-81	33.4	2.82	500	52
	1982-85	24.0	1.17	4.50	72
Bottom of upper	1977-81	20.0	1.35	11.0	58
lake (Sta. 4)	1982-85	20.0	1.17	6.00	70
Bottom of lower	1977-81	18.8	1.45	11.0	57
lake (Sta. 2)	1982-85	15.5	1.17	4.10	72
Mt. Elbert Forebay	1982-85	10.6	3.00	50.4	72
		Cadmium			
Lake Creek inflow	1977-81	1.78	0.10	0.10	6
	1982-85	1.33	0.06	6.00	57
Bottom of upper	1977-81	1.13	0.10	0.10	6
lake (Sta. 4)	1982-85	1.13	0.06	1.00	55
Bottom of lower	1977-81	1.07	0.07	0.20	6
lake (Sta. 2)	1982-85	0.90	0.06	5.00	57
Mt. Elbert Forebay	1982-85	0.64	0.06	0.57	72

* Freshwater aquatic life acute toxicity criteria computed on the basis of hardness (U.S. Environmental Protection Agency, 1986).

seasonal reducing episodes. Figures 5-2 and 5-3 further illustrate these trends. Higher median iron concentrations (fig. 5-2) at the bottom relative to the surface and mid-depth elevations in the two lakes, however, suggest internal loading is an important secondary source of iron in Upper Twin Lakes. The importance of internal cycling of manganese in both lakes is evident in the high bottom concentrations shown on figure 5-3.

Upper Twin Lakes has been particularly susceptible to hypolimnetic anoxia during the winter stratification season, and it is winter sediment releases that account for the prominent bottom peaks in both iron and manganese at station 4 (figs. 5-2, 5-3). The only statistically significant (P=0.029) pre- to postoperational change in the heavy metal concentrations in the system is the decrease in median manganese concentrations at the bottom of the upper lake (table 5-3, fig. 5-3). This change suggests the severity of winter anoxia in Upper Twin Lakes has been reduced since the preproject period.

Copper and zinc are the next most abundant heavy metals in Twin Lakes. In table 5-4, data indicate that concentrations of these metals in both lakes are largely the result of external loading from Lake Creek. Both copper and zinc are priority pollutants, and the EPA (U.S. Environmental Protection Agency) freshwater aquatic life chronic toxicity criteria for each are included in table 5-4 for comparison. Zinc does not appear to constitute a toxicity problem in the system; but copper could present some problems, in that concentrations in the Lake Creek inflow exceeded the chronic toxicity

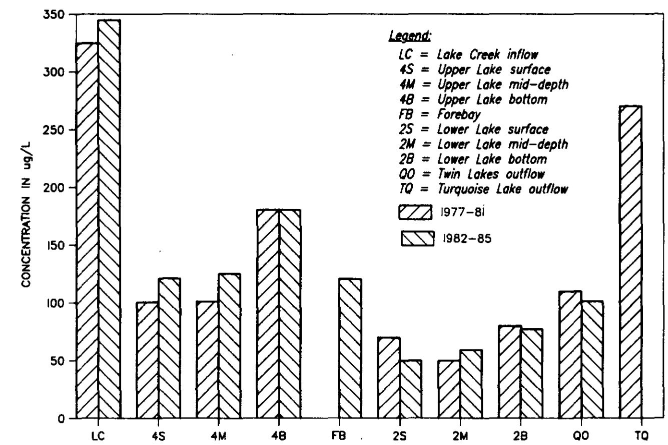

FIGURE 5-2.—*Median iron concentration.*

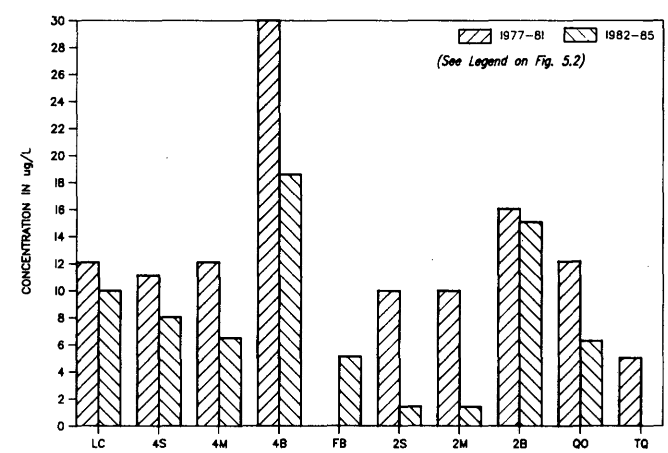

FIGURE 5-3.—*Median manganese concentration.*

criteria over 75 percent of the time during both the pre- and postoperational periods. Figure 5-4 shows the median copper data from table 5-4, with the addition of median concentrations at other points in the system for comparison. From this figure, it is clear Lake Creek is the main source of copper in the system, and that this situation has not changed significantly with the development of the pumped-storage project.

Lead and cadmium were rarely detected during both periods. Lead was detected in 42 percent of the samples collected from 1977 through 1985; cadmium was detected in 26 percent of the samples. Because these metals were so infrequently present in detectable amounts, table 5-5 includes only the 75th percentile and maximum observed concentrations; EPA acute toxicity criteria are listed for comparison. Occasionally, both metals have exceeded the acute toxicity criteria, but significant impact on the ecosystem has not been observed.

Lake Creek watershed lies within the Colorado Mineral Belt. Although mining activity in this watershed was never on the scale of the mines in the Leadville, Colorado, area—or the placers in the drainages to the north and south—large ore bodies have been naturally exposed in the basin of the South Fork of Lake Creek. This area is eroded from the intrusive igneous rock of the Tertiary Twin Lakes Batholith (Wilshire et al., 1965; Chronic, 1980). One of the minerals much in evidence is jarosite, which forms crusts on the slopes around Henschel Lake (fig. 1-3). Red Mountain (in the same area) is so named for the reddish color of its slopes which are the result of the oxidation of iron-bearing minerals. Sampling along the South Fork of Lake Creek in 1975-76 showed this tributary to be the major contributor of iron, manganese, copper, and zinc to Lake Creek (Sartoris et al., 1977).

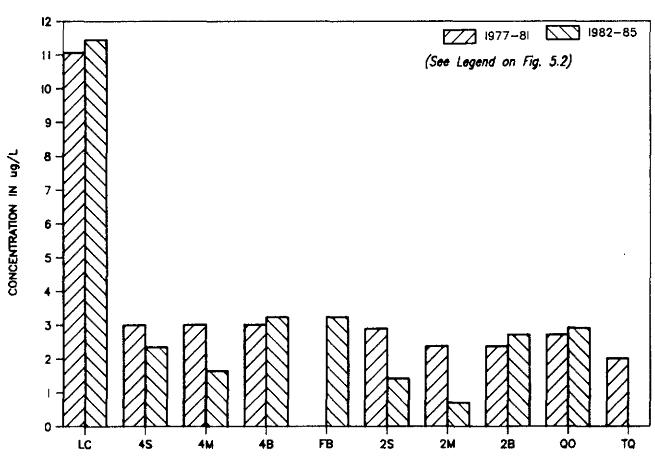

FIGURE 5-4.—*Median copper concentration.*

Since the end of the last Ice Age, sediments of Twin Lakes have accumulated heavy metals that were derived from the mineral deposits in the Lake Creek watershed by glacial action, erosion, and leaching, and were subsequently deposited in the bottom of the lakes. Under conditions of low dissolved oxygen, pH, and redox potential, heavy metals in the sediments are reduced to a soluble form and released into the overlying water. An extreme case of this release was observed and documented in April 1975, when the biota of Upper Twin Lakes was decimated by heavy metals released from the bottom sediments when the hypolimnion became anaerobic (Sartoris et al., 1977). This "winter kill" resulted from a combination of a long ice cover, with a thick blanket of snow that precluded oxygenation by photosynthesis, and the oxygen demand of the organic debris that has also accumulated in the bottom of this upstream sediment trap of the Twin Lakes system. The relatively small volume of the preproject upper lake and the lack of an effective inflow from the frozen watershed probably compounded the problem.

During the anaerobic period, manganese concentrations in the hypolimnion peaked at 2500 μg/L, and iron reached 1700 μg/L. The EPA freshwater aquatic life criterion for iron is 1000 μg/L (EPA, 1986). At turnover in mid-May, copper appeared in the bottom water at a concentration of 90 μg/L, and zinc was measured at 70 μg/L. The EPA freshwater aquatic life acute toxicity criteria for these metals (at the mean hardness value of Upper Twin Lakes at that time of 33.11 mg/L as $CaCO_3$) are 6.2 and 46 μg/L, respectively (EPA, 1986). A probable combination of heavy metals toxicity and phosphorus coprecipitation with the iron at turnover (Hasler and Einsele, 1948) successively caused high mortality among the benthos and plankton and then delayed recovery of primary production for about 2 years. The lower lake was much less affected, and was available to serve as a refugium for the fish and a possible source area for recolonization. Although nothing as serious has happened since, less extreme releases of iron and manganese from the sediments have occurred in both lakes during periods of thermal stratification and hypolimnetic oxygen depletion.

With the enlargement of the upper lake, in particular, and the water circulation induced by pumped-storage operation in winter, as well as summer, it appears less likely that a combination of events such as those that occurred in 1975 would happen again at Twin Lakes. In fact, the enlarged volume and increased circulation may explain the significant decrease in bottom manganese concentrations in the upper lake during the postoperational period. However, the possibility of a recurrence exists as long as the metals are present;

and the overall pre- to postoperational trend of increasing dilution and decreasing hardness of the water would make any large metal loading that much more serious. Mt. Elbert Forebay is especially vulnerable, with a mean hardness of only 20.07 mg/L as $CaCO_3$ (table 5-6).

The major changes brought about by the project relative to heavy metals are in the conditions surrounding their release and their relative toxicity to aquatic life. Enlarged lake volumes and increased water circulation should reduce the possibility of severe hypolimnetic anoxia and, thus, the possibility of such extreme metal releases as those that occurred in April 1975. On the other hand, the increased dilution and reduced hardness of the water (table 5-6), especially in Forebay, will reduce the capacity of the Twin Lakes system to buffer the toxic effects of any metals that may be dissolved in the water (EPA, 1986).

A final point to consider is the sharp reduction in diversions to Lake Creek through the Twin Lakes Tunnel, now that almost all diverted Western Slope (west of Continental Divide) water is introduced into Lower Twin Lakes through the Mt. Elbert Conduit, Forebay, and powerplant. The Twin Lakes Tunnel diversion may have had the effect of diluting the natural runoff of the Lake Creek watershed, in particular, the metal-laden contribution of the South Fork of Lake Creek.

Although not statistically significant, median concentrations of iron (fig. 5-2) and copper (fig. 5-4) in Lake Creek inflow to Twin Lakes increased from the pre- to postoperational periods, even though the mean TDS (table 5-1) and mean hardness (table 5-6) were reduced by the relatively high runoffs of 1982 and 1985. Campbell (in LaBounty et al., 1984) attributed the low productivity of Lake Creek above Twin Lakes to "... relatively high sustained concentrations of copper . . . [and] . . . occasional high concentrations of iron." Comparison of median copper and maximum iron concentrations in Lake Creek with EPA freshwater aquatic life criteria (tables 5-4, 5-3, respectively) supports her contention. Also, it suggests a combination of high copper concentrations and a reduced capacity to buffer their toxicity.

Table 5-6.—Mean hardness, as mg/L of $CaCO_3$, of waters in the Twin Lakes system.

	Preoperational period 1977-81	Postperational period 1982-85
Lake Creek inflow	49.54	38.24
Upper Twin Lakes	33.11	33.19
Lower Twin Lakes	31.51	27.12
Twin Lakes outflow	31.12	33.19
Mt. Elbert Forebay		20.07
Turquoise Lake outflow*	13.36	

* Adapted from Nesler (1981).

Nutrients

Nitrogen and phosphorus data were collected routinely from 1977 through 1985. Silica (SiO_2), an important diatom nutrient, was measured from November 1979 through November 1980; it was never found to be in limiting supply (LaBounty and Sartoris, 1981). The 1979-80 data are reported for characterization purposes (table 5-7).

Like the heavy metal data, the nitrogen and phosphorus analytical results also suffer from the problems of mixed qualitative (i.e., ND) and quantitative data and varying detection limits. The same procedure that was described in the Heavy Metals section was used here to assign reasonable nonzero values to the ND results for purposes of statistical analysis and summarization.

Nitrogen.—The nitrogen forms consistently analyzed for throughout the study were ammonia,[2] nitrite (NO_2^{-1}), and nitrate (NO_3^{-1}), all expressed as µg/L of elemental nitrogen. These three inorganic nitrogen forms are immediately available for uptake by the phytoplankton (Golterman, 1975). Total Kjeldahl nitrogen, TKN, which includes organic nitrogen and ammonia but not nitrite nor nitrate, was not consistently analyzed for during the postoperational period.

Although nitrogen is abundant in the atmosphere, atmospheric nitrogen is only made available to lake phytoplankton through fixation by prokaryotes (e.g., some bluegreens) or through rainstorms.

Table 5.7.—Mean silica concentration in the Twin Lakes system, 1979-80.

Location	Mean SiO_2 concentration mg/L	Number of samples 1979-80
Lake Creek inflow	5.84	15
Uppper Twin Lakes	5.25	14
Lower Twin Lakes	4.80	15
Twin Lakes outflow	5.03	11

[2] This form includes both the ammonium ion (NH_4^{+1}) and unionized ammonia (NH_3).

Although summer thundershowers are relatively common at Twin Lakes, their contribution to the lakes' nitrogen budget is unknown; nitrogen-fixing organisms were not identified in the plankton community during this study. Mineralization and recycling of organic detritus (i.e., dead organisms and excreta) appear to be the major sources of nitrogen in the Twin Lakes system. Briefly, this cycle may be summarized as (Cole, 1979): (1) nitrogenous organic material is converted to ammonia by both aerobic and anaerobic bacteria (ammonification), and (2) the ammonia is oxidized to nitrite and then nitrate by aerobic bacteria (nitrification).

As mentioned above, all three inorganic nitrogen forms are readily available for algal uptake. Golterman (1975), however, points out that ammonia and nitrate are the more important forms because nitrite is unstable—being oxidized rapidly to nitrate in an aerobic environment. Under anoxic or anaerobic conditions, ammonia tends to accumulate because ammonification can continue with or without free oxygen. Ammonia and nitrate are the most commonly detected inorganic nitrogen forms in the Twin Lakes system. Nitrite was detected in only a few (<10%) water samples during the entire study.

Nitrogen concentrations in the Twin Lakes system during both periods are summarized on figures 5-5 through 5-7. Total Kjeldahl nitrogen sampling was interrupted in the early postoperational period, but the available data are included here (fig. 5-5) as an indicator of organic loading in the system. Higher in-lake concentrations relative to inflow concentrations indicate this loading is largely internal to the lakes. The general increase in TKN concentrations in both lakes and the outflow between the pre- and postoperational periods was statistically significant ($P<0.050$). This change was most likely the result of flooding new terrain and its vegetation and organic matter.

Ammonia nitrogen is included in the TKN fraction along with organic nitrogen. In the Twin Lakes system (fig. 5-6), ammonia accounts for only about one-tenth of the TKN and this inorganic fraction is primarily contributed by internal rather than external loading. Changes in lake ammonia concentrations between the pre- and postoperational periods were not statistically significant. This lack of change in ammonia—even though TKN increased—suggests either: (1) the added organic material was resisting decomposition, or (2) the system was so efficient in ammonification and nitrification that no significant increase in ammonia concentrations could occur. System-wide trends in nitrate concentrations (fig. 5-7) indicate the second explanation is more likely correct. Even though external loading from Lake Creek was a major source of nitrate during both periods, internal loading was also important and this internal loading increased significantly in both lakes and the outflow during the postoperational period.

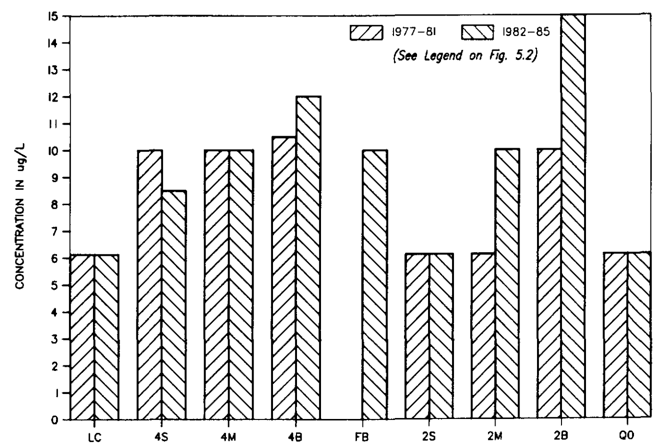

FIGURE 5-6.—*Median ammonia nitrogen concentration.*

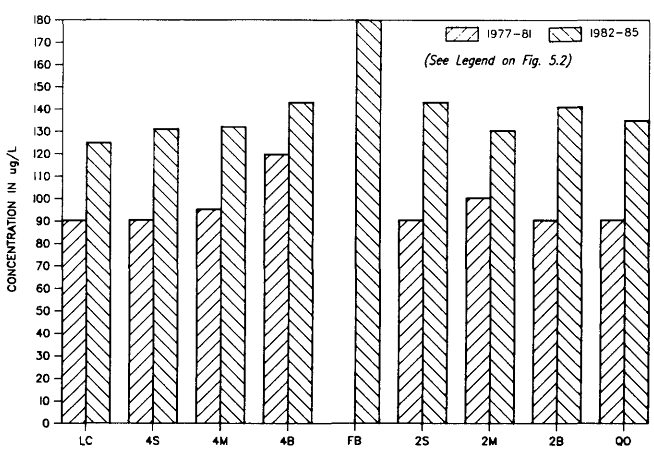

FIGURE 5-5.—*Median total Kjeldahl nitrogen concentration.*

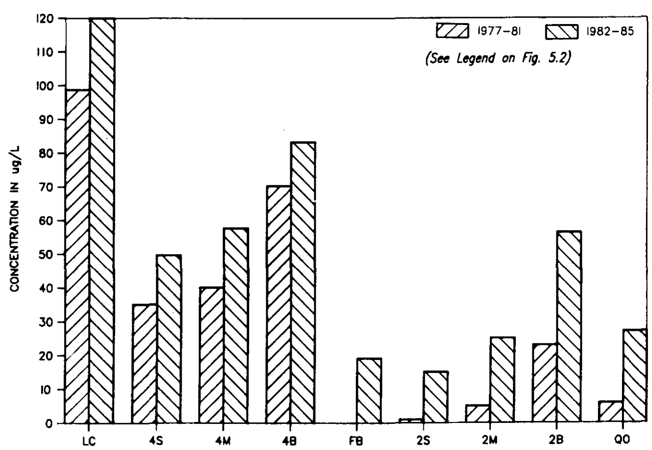

FIGURE 5-7.—*Median nitrate nitrogen concentration.*

The increase in the Lake Creek nitrate concentrations in the postoperational period (fig. 5-7), although not statistically significant, probably reflects the higher flows during 1982 and 1983. Nitrate is a relatively stable form of nitrogen, and a high runoff can carry an increased load of nitrate leached from the soil (Buckman and Brady, 1969; Cole, 1979). This same reasoning applies to possible direct leaching of nitrate from newly flooded lake littoral areas as well.

On average, nitrate is about five times as abundant as ammonia in the lakes and is the most important inorganic nitrogen form available to the phytoplankton. During both periods, nitrate concentrations tended to decrease in a downstream direction—from the inflow—through the upper and lower lakes to the outflow (fig. 5-7). This declining trend reflects phytoplankton uptake of nitrate in the lakes. Taylor et al. (1980) cited an ambient concentration of 300 μg/L of inorganic nitrogen "after the spring turnover" as the critical level for development of algal blooms, which are considered ". . . characteristic of mesotrophic and eutrophic water bodies." Figures 5-6 and 5-7 show that Twin Lakes fell well below this threshold throughout both periods.

Phosphorus.—The phosphorus forms analyzed during the study were total phosphorus and orthophosphate (PO_4^{-3}) phophorus, both expressed as μg/L of elemental phosphorus. Orthophosphate is the inorganic phosphorus form that is immediately available for algal uptake and growth (Cole, 1979). Total phosphorus, TP, however, includes various soluble and insoluble organic and inorganic phosphorus forms (USGS et al., 1977). Lambou et al. (1976) pointed out that, because of the ". . . high mobility and short turnover times of phosphorus . . . within the general phosphorus pool . . ." TP is often ". . . a good approximation of bioavailable phosphorus." This approximation must be applied to Twin Lakes because orthophosphate phosphorus was only rarely detected during either period.

Figure 5-8 summarizes the system-wide pre- and postoperational period TP concentrations at Twin Lakes. At no time did the median TP concentration in either lake equal or exceed the 10 μg/L ambient TP concentration cited by Taylor et al. (1980) as the cutoff point between oligotrophic and mesotrophic water bodies. All data collected during both periods indicate Twin Lakes are oligotrophic, phosphorus-limited lakes (Lambou et al., 1976; Taylor et al., 1980; LaBounty and Sartoris, 1981; LaBounty et al., 1984); however, there were statistically significant increases in TP concentrations during the postoperational period in the Lake Creek inflow and in both lakes. In Lake Creek, during

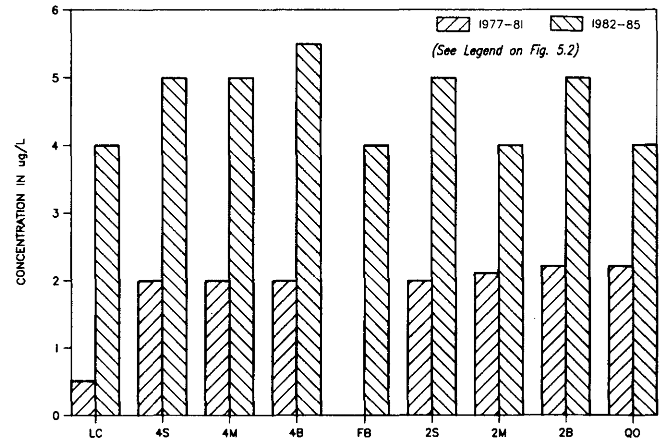

FIGURE 5-8.—*Median total phosphorus concentration.*

the 1982 through 1985 period, phosphorus increases were likely caused by the high runoffs of 1982 and 1983, as noted earlier in the discussion of nitrate. Because phosphorus often occurs in a particulate form—being adsorbed onto soil particles or chemically bound to iron (Buckman and Brady, 1969)—it is even more susceptible to increased loading with increased discharge than is nitrate.

Higher median concentrations of TP in the lakes compared with the Lake Creek inflow, during both periods (fig. 5-8), indicate internal cycling plays a major role in the phosphorus budget of Twin Lakes. Data also suggest at least some of the increased TP in the lakes, during the postoperational period, came from the newly flooded soils in the expanded littoral zone created by raising the lakes' water surface elevation.

SUMMARY

The raising of Twin Lakes' water surface elevation, during the postoperational period, appears to have resulted in an increased internal nitrogen loading in both lakes (figs. 5-5, 5-7). Evidence indicates TP concentrations in the lakes were at least partially increased by this same event; although a larger external loading by the higher Lake Creek inflows in 1982 and 1983 was definitely a significant (P<0.050) factor.

The phenomenon of increased internal nutrient loading from recently flooded soils in a new impoundment has been documented in the limnological literature (Grimard and Jones, 1982; Bayne et al., 1983). However, the high Lake Creek inflows into the unexpanded lakes in 1982 and 1983, followed by relatively normal inflows into the raised lakes in 1984 and 1985, made it difficult to separate the effects of external and internal nutrient loading.

CONCLUSIONS

Waters of the Twin Lakes system are extremely dilute and soft. The overall preoperational to postoperational period trend was for these waters to become even more dilute and still more soft because of dilution with Turquoise Lake water, by way of the Mt. Elbert Conduit and Forebay, and because of higher inflows from Lake Creek during 1982 and 1983. During both periods, calcium was the major cation, and the major anions were bicarbonate and sulfate.

Naturally exposed mineral deposits contribute large quantities of heavy metals to Twin Lakes by way of Lake Creek inflow. Iron, copper, and zinc are most concentrated in Lake Creek, although internal loading through sediment release is an important source of iron in the upper lake, and to a lesser extent in the lower lake. In Twin Lakes, manganese is mainly internally cycled from the sediments during periods of stratification and oxygen depletion. A major episode of winter anoxia, heavy metals releases, and decimation of plankton and benthos was observed in Upper Twin Lakes in 1975; less severe winter metal releases were not uncommon in the lakes during the preoperational period (1977 through 1981). Manganese releases from the bottom sediments of the upper lake decreased significantly during the postoperational period; possibly, this is because the combination of increased lake volumes and powerplant induced water circulation precluded extreme anoxia. Because of the reduced hardness of waters in the Twin Lakes system, any input of heavy metals is potentially more toxic. Lake Creek shows evidence of chronic copper toxicity.

Twin Lakes are oligotrophic; phosphorus-limited bodies of water that have remained so throughout both periods. Nevertheless, during the postoperational period, significant increases in nitrate and total phosphorus were observed in both lakes. These increases resulted from the inundation of large areas of previously dry land around the lakes and the relatively high inflows from Lake Creek. Because of the short time of observation (21 months) after lake levels were raised, it is unlikely the system had reached trophic equilibrium when this study ended. Therefore, it is impossible to predict, with any certainty, whether or not these elevated nutrient levels are anything more than a temporary upsurge and what, if any, their long-term effects may be.

* * * * *

Physical and Chemical Profiles

Chapter 6

J. LaBounty and J. Sartoris

Bureau of Reclamation
Applied Sciences Branch
Denver, Colorado

INTRODUCTION

Twin Lakes, a pair of dimictic natural lakes, have been altered to use the water resource downstream. Under natural conditions (see Juday, 1906) they develop thermal stratification in both summer and winter, as would be expected in a temperate climate. Stratification patterns are variable due to the "flow-through" nature of the lakes since importation from other watersheds began. Disruption of natural stratification patterns is more pronounced since the Mt. Elbert Pumped-Storage Powerplant began operation (LaBounty and Timblin, 1988). The lakes, however, remain thermally stratified in the summer, with stratification breaking down in early October.

The lakes freeze in December and remain frozen until late April or May. Both lakes remain at or near saturation in dissolved oxygen, except for some occasional depletion near the bottoms. Usually, the lakes are neutral to slightly basic in pH balance, but the balance may shift to slightly acidic in deeper areas during periods of stratification and low biological productivity. Twin Lakes are extremely dilute and poorly buffered, with conductivity almost always lower than 100 μS/cm.

Although Twin Lakes remain dimictic, dilute, and circum-neutral in pH, subtle changes in the physical-chemical profiles are notable between the pre- and postoperational periods.

METHODS

Data in this chapter were obtained during routine surveys at three permanent stations in the Twin Lakes system (fig. 1-5). Station 4 (in Upper Twin Lakes) and station 2 (in Lower Twin Lakes) were established during the preoperational period (1977 through 1981). Station 5 (in Mt. Elbert Forebay) was added during the postoperational period (1982 through 1985).

Physical and chemical data were measured in the field, at a series of set depth intervals, so as to delineate a surface-to-bottom profile of each parameter's variation with depth at a given station. The standard measuring depths were at the surface, 1-m, and 2-m intervals—thereafter to the bottom.

Light penetration and Secchi depth measurements differed from the above approach. Light intensity measurements were recorded at the standard depth intervals until light was not detectable, rather than all the way to the bottom. Secchi depth is a single measurement of the depth at which a black and white disk disappears from view. Secchi depth was only measured during the ice-free seasons of the year. All the other profile measurements were done year-round.

With Hydrolab Corporation electronic multi-parameter probes, measurements were taken of:

- Water temperature,
- Dissolved oxygen concentration, DO,
- Conductivity,
- Hydrogen ion concentration, pH, and
- Oxidation-reduction potential, ORP or Redox.

Three different Hydrolab models were used during the course of the study: a Surveyor Model

6D (1977 through Sept. 1978), a System 8000 (Oct. 1978 through Sept. 1984), and a Surveyor II (after Sept. 1984).

Secchi depths were measured with a standard 200-mm diameter Secchi disk. Light intensity profiles were measured with a Kahl Scientific Company limnophotometer, and light transmissivity profiles were measured with a Kahl transmissometer. All light transmissivity measurements were performed using a 0.5-m-light path length.

The data were analyzed using SAS Institute statistical software packages. Statistical significance of observed differences between stations and between study periods at the same station was tested using both the standard *t*-test and the nonparametric Wilcoxon Rank Sums Test.

RESULTS AND DISCUSSION

Temperature

Table 6-1 lists the average temperature for each lake, and the number of days each lake was stratified during the summer.

- Average temperature of the lower lake ranged from 4.9 (1978) to 8.6 °C (1977).
- Water in the lower lake was slightly cooler (6.2 to 7.0 °C) after powerplant operation began.
- The upper lake was consistently cooler (~0.7 °C) with an average temperature ranging from 5.2 (1978) to 7.0 °C (1977 and 1981).
- Average temperature of the upper lake was also cooler (6.1 to 5.5 °C) after the powerplant began operation.
- In the lower lake, the period of summer stratification (defined here as the period when a thermocline with at least a 1 °C drop in temperature per meter of depth exists) ranged from 15 days (1982) to 110 days (1977).
- In the upper lake, the summer stratification period ranged from 30 (1982) to 89 (1980) days.
- Although the average number of days that the lower lake (89) was stratified was greater than the upper lake (74) before powerplant operation, the trend reversed after operation (lower=40, upper=63).

Both lakes were stratified fewer days after powerplant operation began due to a change in inflow patterns. Transmountain diversions which had come by way of Lake Creek into the upper lake were replaced by diversions from Mt. Elbert Conduit from Turquoise Reservoir, through the powerplant, and into the lower lake.

Figures 6-1 and 6-2 show temperature isopleths for the upper lake drawn from data collected in 1977 through 1985. Figures 6-3 and 6-4 show the same information for the lower lake. The usual pattern in the upper lake is for summer stratification to develop by early July, peak by mid-August, and disappear in early October. During the summer, the difference in temperatures from surface to bottom is about 10 °C (~6 °C at the bottom and ~16 °C at the surface). The lower lake pattern is similar, with the differences being that maximum stratification is about 10 days sooner and the bottom and surface temperatures are about 2 °C higher at maximum stratification (bottom ~8 °C and surface ~18 °C).

Table 6-1—Average of selected physical-chemical parameters.

Year	Temperature, °C	Stratification, days	DO mg/L	pH	Conductivity, µS/cm
		Lower Twin Lakes			
Preoperational period					
1977	8.6	110	7.7	8.0	70
1978	4.9	73	8.2	7.9	73
1979	6.0	95	8.2	7.8	76
1980	7.4	109	7.8	7.8	75
1981	7.8	56	7.7	7.7	83
Postoperational period					
1982	6.6	15	7.8	7.6	65
1983	6.3	85	7.7	7.7	62
1984	5.8	32	7.5	7.5	61
1985	6.3	24	7.6	7.7	58
1977–81	7.0	89	7.9	7.5	72
1982–85	6.2	40	7.7	7.2	57
		Upper Twin Lakes			
Preoperational period					
1977	7.0	86	7.8	7.4	69
1978	5.2	64	8.3	7.4	76
1979	5.5	78	8.1	7.3	76
1980	6.2	89	8.3	7.3	78
1981	7.0	51	7.8	7.4	87
Postoperational period					
1982	5.5	30	8.0	7.4	76
1983	5.6	83	8.0	7.4	75
1984	5.1	85	7.7	7.3	77
1985	5.7	49	7.1	7.3	75
1977–81	6.1	74	8.1	7.4	77
1982–85	5.5	63	7.7	7.3	75

Figure 6-5 shows average water temperature isopleths from Upper and Lower Twin Lakes organized before and after powerplant operation began. Because these are averages, the thermocline does not show. In reality, a thermocline is present at the same time each year, but at varying depths. Figure 6-6 contains temperature profiles of both lakes from August 1903 (Juday, 1906), 1978, 1980, and 1984. Figure 6-7 contains typical summer temperature

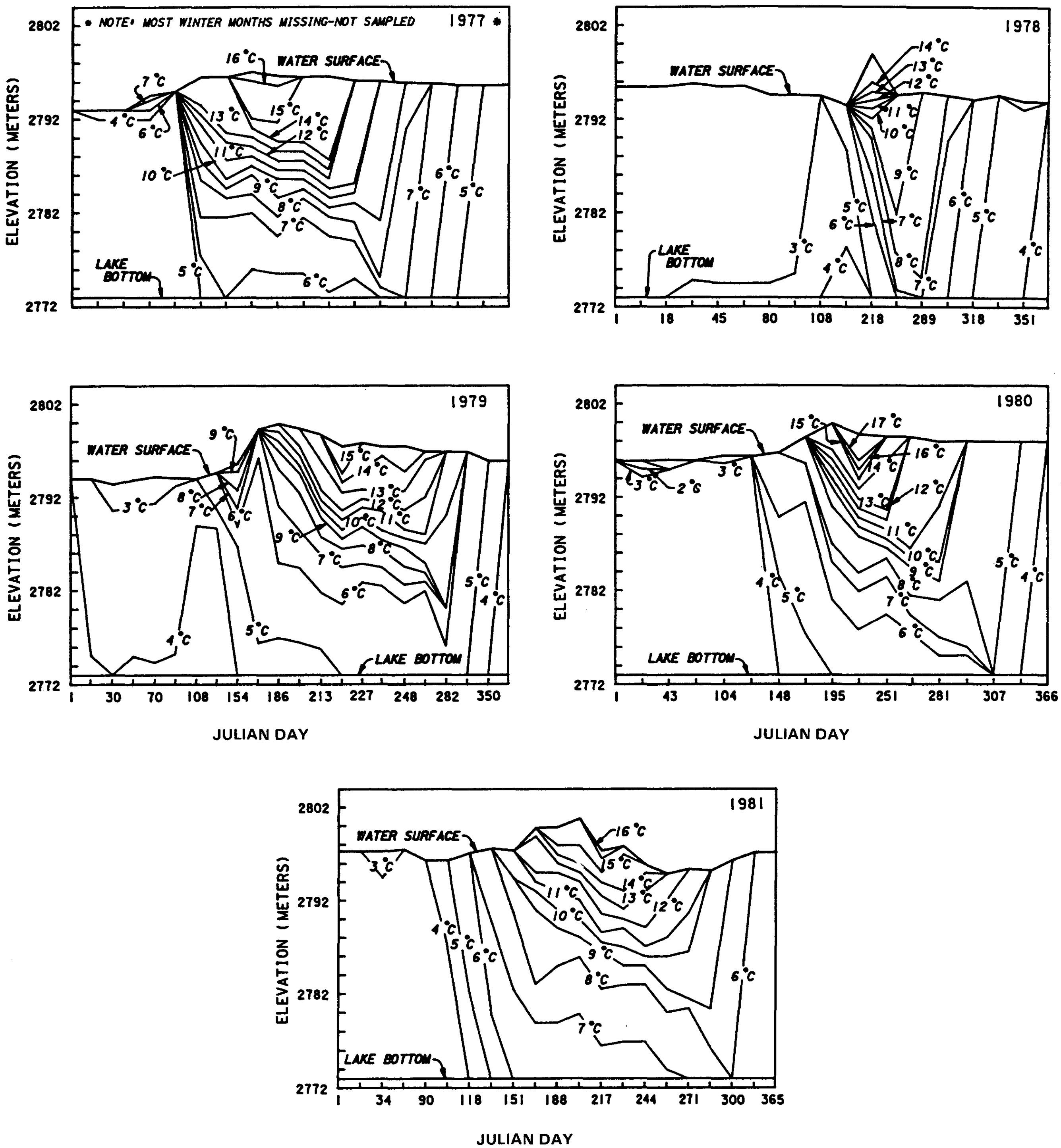

FIGURE 6-1.—*Temperature isopleths, Upper Twin Lakes, 1977–80.*

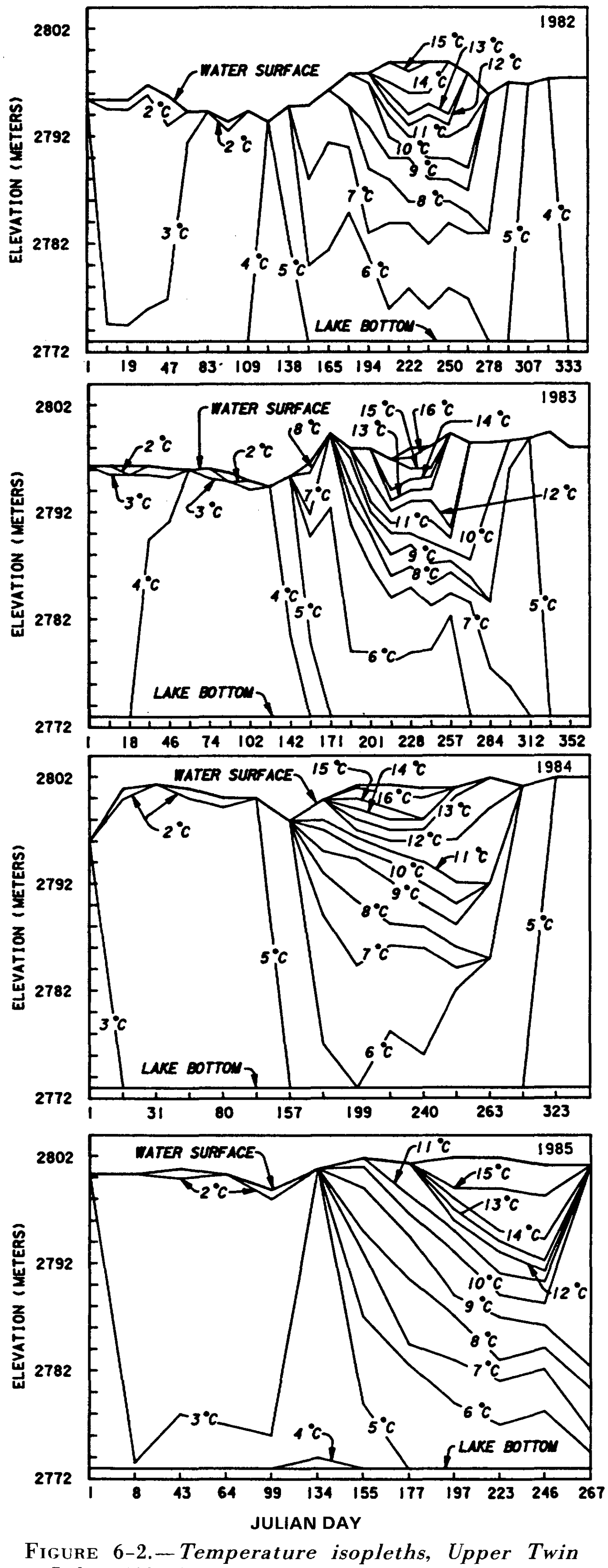

FIGURE 6-2.—*Temperature isopleths, Upper Twin Lakes, 1981-85.*

profiles before (1979, '80, and '81) and after (1982, '83, and '84) powerplant operation began. In each case, data are from August when stratification is strongest. It should be noted that in 1983 the powerplant did not operate during the entire ice-free season.

Twin Lakes are dimictic lakes; they are covered by a significant layer of ice in winter; they are warm to nearly 20 °C at the surface—in the middle of summer. The basic thermal regime, as measured by Juday in 1903, remains today. Twin Lakes still provide habitat for cold water species of fish such as the lake and rainbow trout. However, subtle differences exist in the thermal regime between the two lakes and over time.

As data on figure 6-7 indicate, and as discussed in LaBounty and Timblin (1988), Mt. Elbert Pumped-Storage Powerplant operations influence temperatures of Twin Lakes. Coincidently, the thermal characteristics of the lakes are influenced by inflow. Changes in the flow regime also have had subtle impacts on the thermal regime of Twin Lakes. The data analysis shows that both lakes are currently slightly cooler than before 1982; the upper lake remains consistently the coolest of the two lakes.

In addition to the slight disruption of thermal stratification, the average number of days whilst the lakes remain stratified have been reduced because of powerplant operations.

Temperature patterns are important to Twin Lakes ecology. Data for this study do not indicate any short-term effects that could lead us to state that the lakes are significantly different; however, the long-term influence of the subtle changes discussed above is yet unknown.

Dissolved Oxygen

Table 6-1 includes the average dissolved oxygen concentrations, mg/L, for Twin Lakes during the study. The average concentration of both lakes ranged from 7.7 to 8.1 mg/L throughout the duration of the study, with little difference between lakes or between years of pre- and postoperation.

Figures 6-8 through 6-11 show dissolved oxygen isopleths for both lakes for the years 1977-85 inclusive. Figures 6-8 and 6-9 show clear seasonal trends in Lower Twin Lakes with some oxygen depletion occurring both summer (maximum=late Aug.) and winter (maximum=Mar.). Figures 6-10 and 6-11 also show seasonal patterns in Upper Twin Lakes, but with less depletion during summer and greater depletion during winter. Figure 6-12

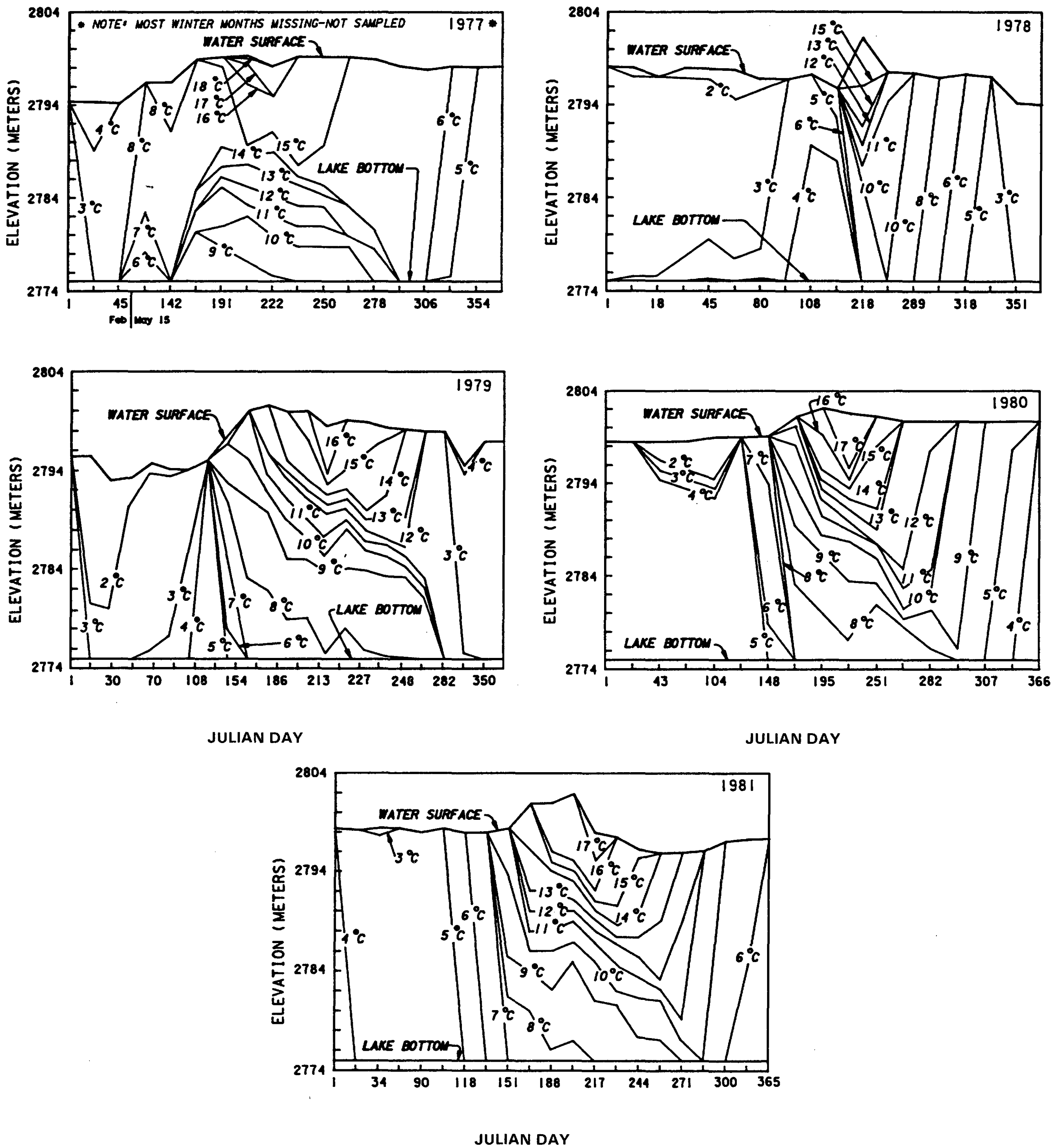

FIGURE 6-3.—*Temperature isopleths, Lower Twin Lakes, 1977-81.*

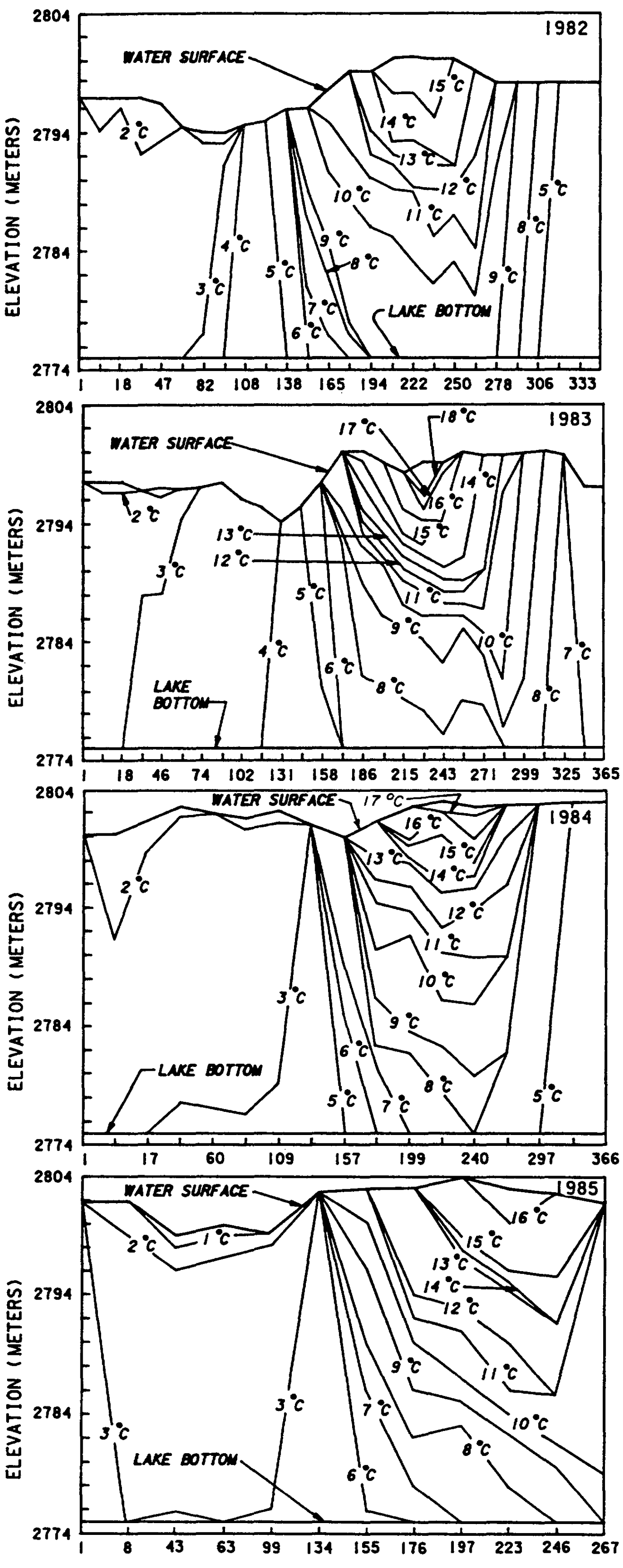

FIGURE 6-4.—*Temperature isopleths, Lower Twin Lakes, 1982-85.*

includes plots of average dissolved oxygen concentration isopleths from Twin Lakes organized before and after the powerplant began operation. In the lower lake, concentrations near the bottom are higher after operation whereas patterns in the upper lake are similar before and after operation.

Figure 6-13 shows typical dissolved oxygen profiles from each lake—winter and summer. These patterns typify all the years of the study, showing some oxygen depletion, but not to the degree of other lakes of similar size and depth that are more eutrophic. Instead, patterns are typical of clean oligotrophic lakes of similar size, shape, and depth.

Dissolved oxygen concentration is an important factor to the ecology of Twin Lakes. The lakes were near or at oxygen saturation almost year round during both study periods. However, in both periods during thermal stratification, some depletion of dissolved oxygen occurred in the deepest layers of the lakes. Statistically, little difference is distinguished, if any, before and after powerplant operation began. However, after resolving 14 years of data, while it was continually being collected, some subtle characteristics about this important parameter can be discerned. First, the powerplant seems to have caused a slight aeration in the deepest layers of the lakes during the times when stratification was greatest. This observation is supported by data shown on figures 6-8 through 6-11, where dissolved oxygen concentrations near the bottom (during the preoperational period) were at or below 1 mg/L. On one occasion, in 1974, oxygen was measured at 0 mg/L in the upper lake. During that winter, which was extremely harsh, the inflow over a long period of time was less. During other winters, inflow was great enough to cause some aeration as it flowed to the bottom. When powerplant operation began, the increased inflow of more dense (~4 °C) water further helped aerate the deep areas (figs. 6-8 to 6-11).

Lake trout habitat requirements include dissolved oxygen concentrations greater than 4 mg/L. In addition, *Mysis* shrimp, the food base for young lake trout, also require dissolved oxygen concentrations in excess of about 4 mg/L to survive. Because both of these animals inhabit the bottom layers of the lake, their habitat was greatly restricted during the portion of the year when oxygen depletion occurred. Thus, it would appear that the oxygenating effect of powerplant operation may be beneficial in expanding the habitat of these animals—both temporally and spatially.

Conductivity, pH, and Redox Potential

Clear seasonal stratification cycles were not evident for these three chemical parameters. However,

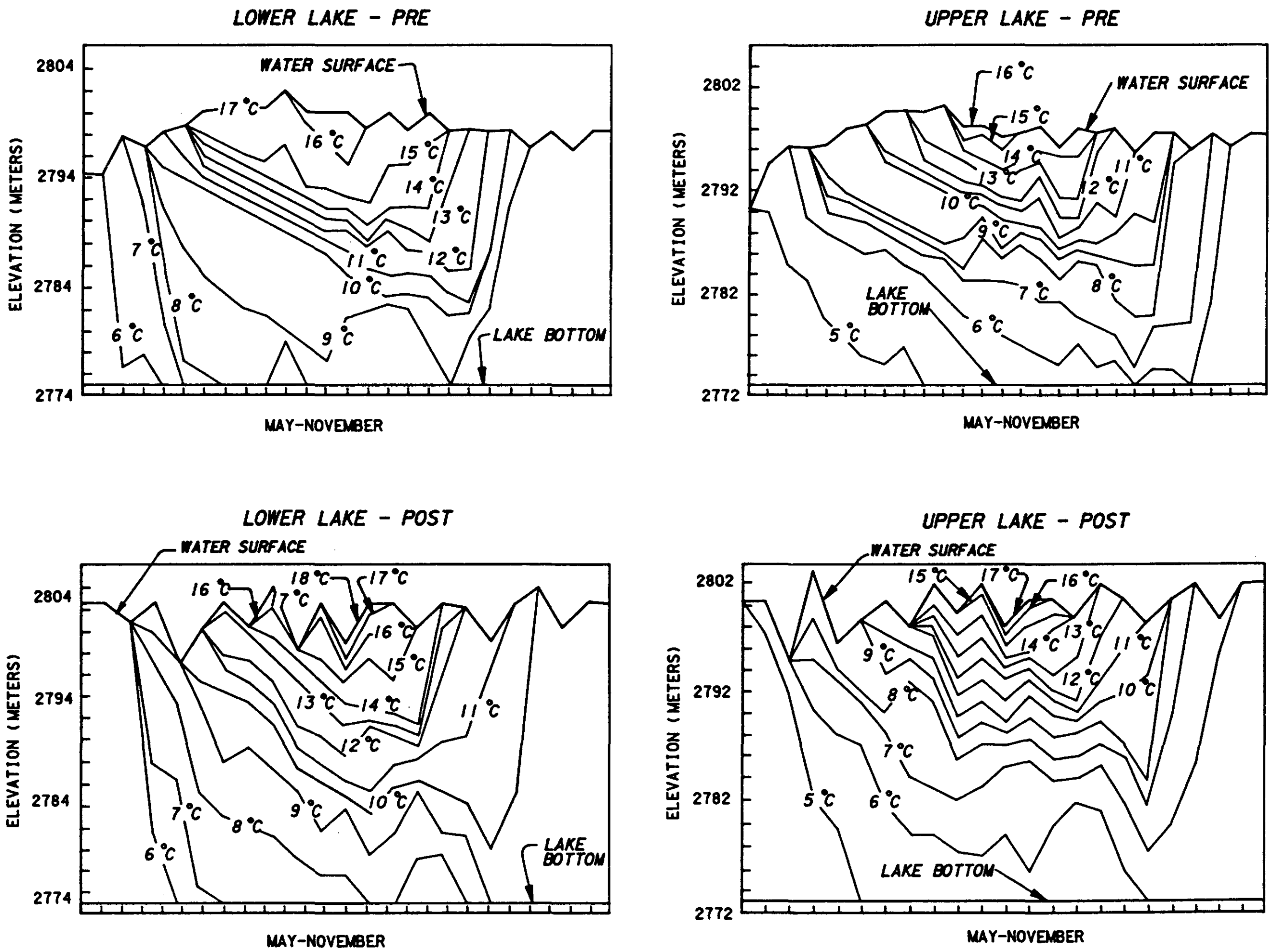

FIGURE 6-5.—*Average water temperature isopleths of Twin Lakes during pre- and postoperational periods.*

some statistically significant differences between stations and changes between study periods were observed in both the profile (surface-to-bottom) and bottom mean values of all three parameters.

Mean profile and bottom conductivity (specific conductance) values, during both study periods, are listed in table 6-2. Generally, these field conductivity values are in agreement with the laboratory values reported in chapter 5. However, table 6-2 values are statistically more reliable because they are based on a much larger sample. Field conductivity measurements are also not subject to the sample handling and storage problems that can affect laboratory results.

During the preoperational period, both the profile and bottom mean conductivities at station 4 (in Upper Twin Lakes) were significantly higher than those at station 2 in Lower Twin Lakes. During the postoperational period, profile and bottom mean conductivities at station 2 decreased significantly,

Table 6-2.—Mean conductivity, μS/cm, of the waters in the Twin Lakes system.

	Mean conductivity		
	Preoperational period 1977-81	Postoperational period 1982-85	Change
	Profile Mean		
Upper lake (Sta. 4)	77.34	75.46	NS‡
†(4-2)	**	**	
Lower lake (Sta. 2)	71.66	57.44	**
(2-5)	**	**	
Mt. Elbert Forebay (Sta. 5)		41.32	
	Bottom Mean		
Upper lake (Sta. 4)	80.88	78.82	NS
(4-2)	*	**	
Lower lake (Sta. 2)	74.34	59.26	**

† (4-2) = difference between stations 4 and 2.
‡ NS = not significant.
* Significant: $P<0.05$.
** Highly significant: $P<0.01$.

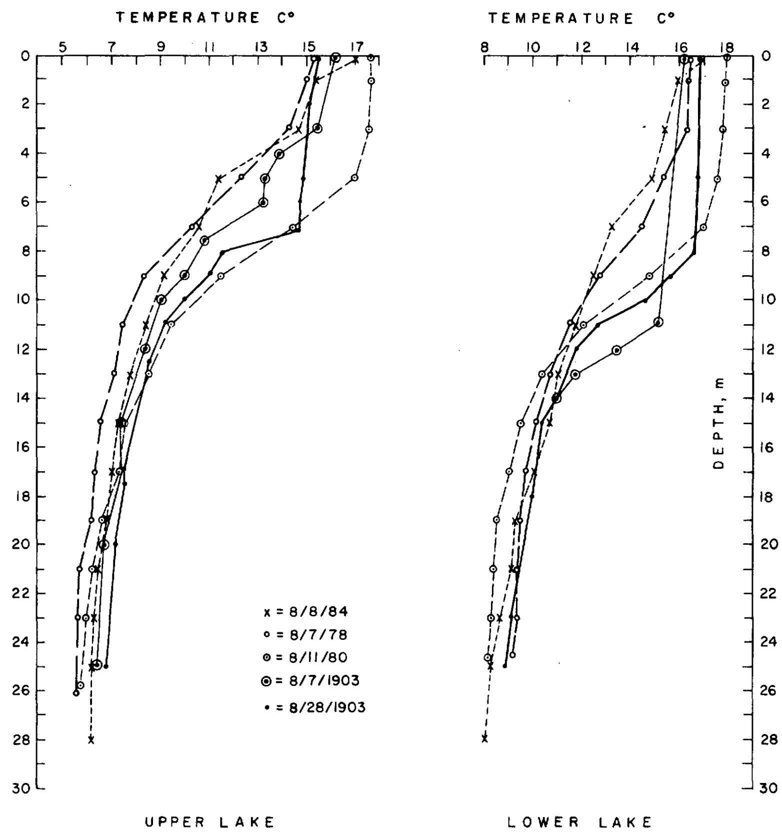

FIGURE 6-6.—*Temperature profiles of Twin Lakes.*

whereas those at station 4 were not significantly different from the preoperational period.

Station 5 (in Mt. Elbert Forebay) was always well-mixed from the surface to the bottom; thus, only the profile mean conductivity is reported here as an indicator of the dissolved mineral content of the water being imported into Twin Lakes from Turquoise Reservoir. The mean conductivity of this imported water was significantly less than that of Twin Lakes.

Profile and bottom mean pH values are in table 6-3. Bottom mean pH was not significantly different between stations 2 and 4, or between study periods at either station. The profile mean pH at station 4 also was essentially unchanged between study periods. However, at station 2, the profile mean pH, which was significantly higher than that at station 4 during the preoperational period, declined significantly during the postoperational period to a level not statistically different from that at station 4. At station 5, the profile mean pH was significantly lower than those in Twin Lakes.

Mean bottom redox (oxidation-reduction) potentials are listed in table 6-4. Although bottom redox potential may be useful as a semiquantitative indicator of the oxidation-reduction environment that mediates sediment-water chemical exchanges, redox measurements in the well-oxygenated upper water column are too unreliable to be meaningful (Brannon et al., 1978). Hence, only the bottom values were considered. The redox potentials listed in table 6-4 were adjusted to pH 7 to eliminate biases

Table 6-3.—Mean pH of waters in the Twin Lakes system.

	Preoperational period 1977-81	Postoperational period 1982-85	Change
	Profile Mean		
Upper lake (Sta. 4)	7.36	7.32	NS‡
†(4-2)	**	NS	
Lower lake (Sta. 2)	7.46	7.24	**
(2-5)		*	
Mt. Elbert Forebay (Sta. 5)		7.11	
	Bottom Mean		
Upper lake (Sta. 4)	7.00	7.01	NS
(4-2)	NS	NS	
Lower lake (Sta. 2)	7.04	6.95	NS

† (4-2) = difference between stations 4 and 2.
‡ NS = not significant.
* Significant: $P<0.05$.
** Highly significant: $P<0.01$.

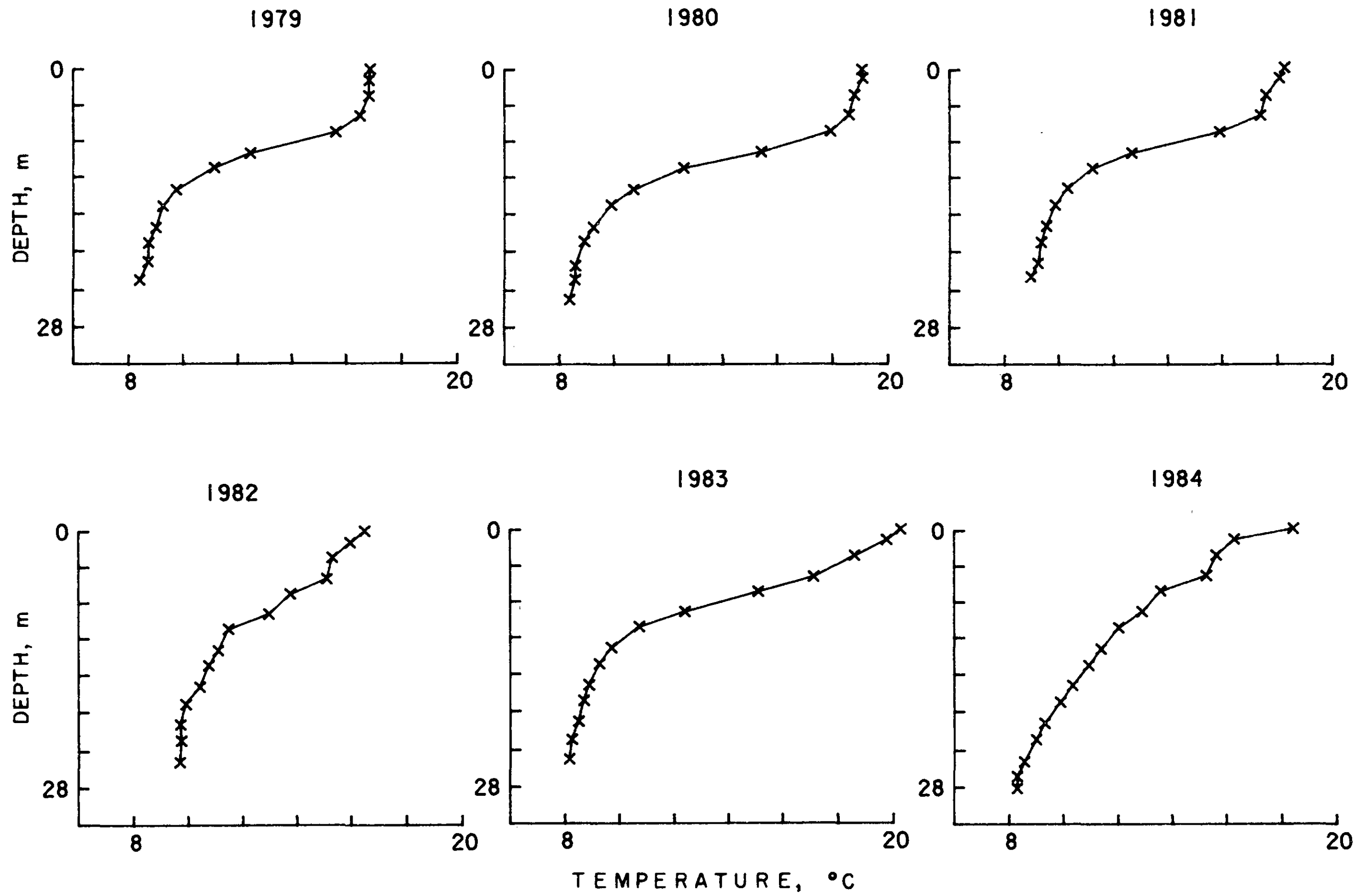

FIGURE 6-7.—*Typical summer temperature profiles. Mt. Elbert Pumped-Storage Powerplant operation for 3 years (1979, 1980, 1981) before and 3 years (1982, 1983, 1984) after pump-turbine operation began.*

introduced by pH variations; thus, the listed values are designated E_7. Redox potentials measured using the first Hydrolab probe, before October 1978, were found to be incompatible with values measured using the later model probes. The bottom redox potentials measured during 1977 and 1978 were therefore dropped from further consideration.

Mean bottom E_7 was not significantly different from station to station in either study period; but a significant decline in mean bottom E_7 occurred at both stations 2 and 4 during the postoperational period. At station 5, mean bottom E_7 was not statistically different from those in Twin Lakes (table 6-4).

During the preoperational period, Twin Lakes is characterized as dilute, poorly buffered, circumneutral lakes, on the basis of conductivity and pH. Upper Twin Lakes had a higher mean conductivity and a lower mean pH than did Lower Twin Lakes (tables 6-2, 6-3). The difference in conductivity between the two lakes was consistent with their relative positions in the system. Dissolved minerals from the Lake Creek watershed were first intercepted by the upper lake, which retained some before passing the rest on to the lower lake. The preoperational difference between the two lakes in

Table 6-4.—Mean bottom redox potential, E_7 (mV), of the waters in the Twin Lakes system.

	Preoperational period 1977-81	Postoperational period 1982-85	Change
	Mean Profile		
Upper lake (Sta. 4)	405	378	*
†(4-2)	NS‡	NS	
Lower lake (Sta. 2)	415	377	**
(2-5)		NS	
Mt. Elbert Forebay (Sta. 5)		390	

Note: 1977 and 1978 data not compatible.
† (4-2) = difference between stations 4 and 2.
‡ NS = not significant.
* Significant: $P<0.05$.
** Highly significant: $P<0.01$.

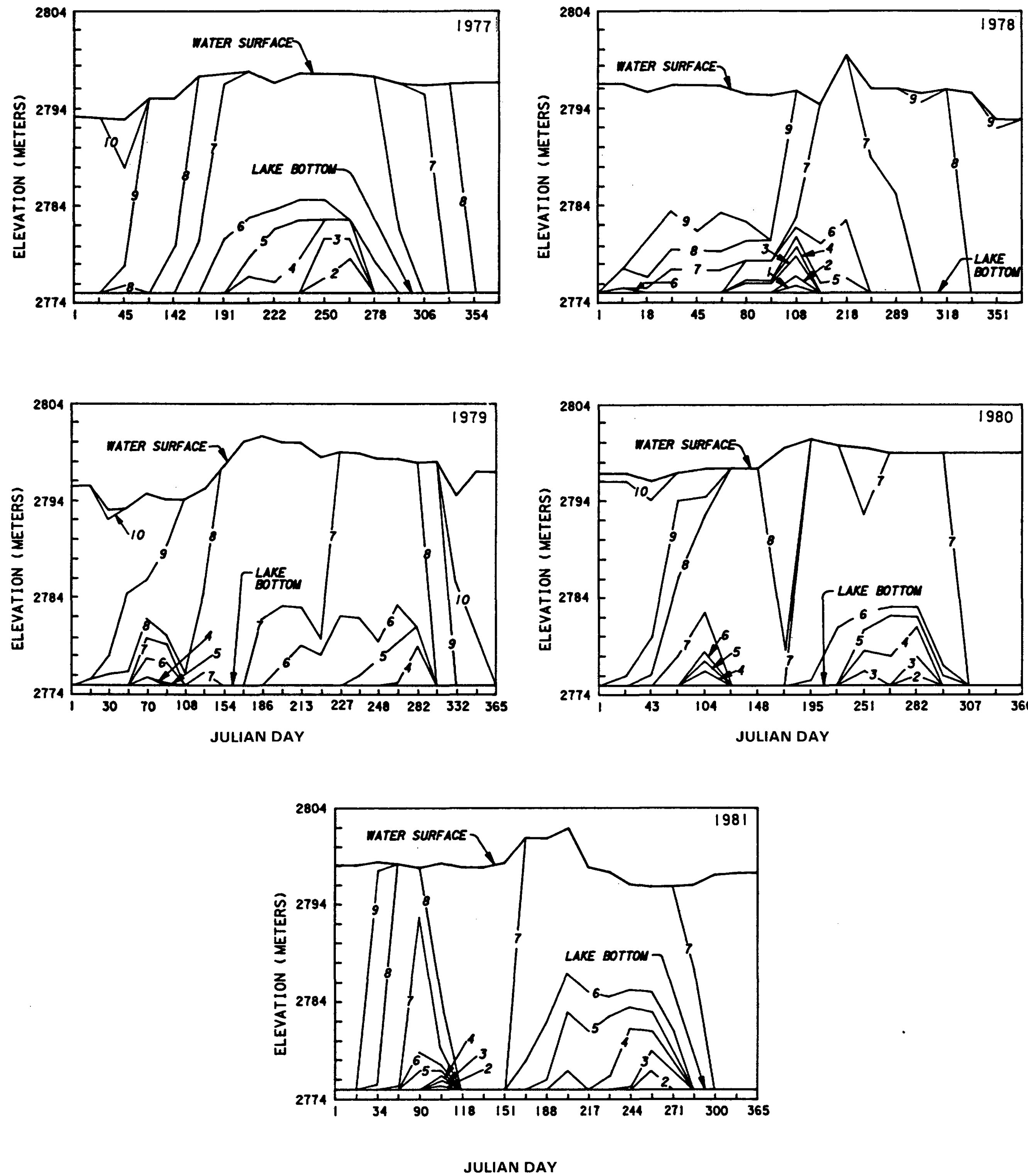

FIGURE 6–8.—*Dissolved oxygen isopleths, mg/L, Lower Twin Lakes, 1977–1981.*

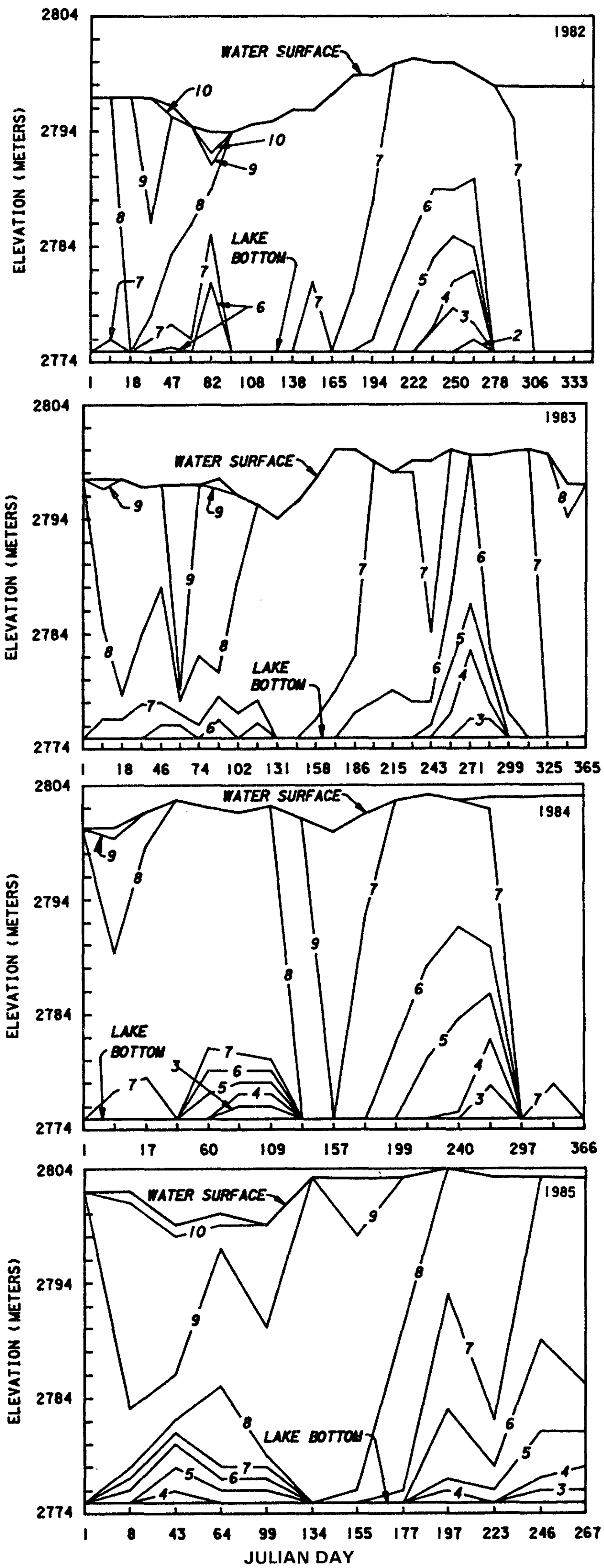

FIGURE 6-9.—*Dissolved oxygen isopleths, mg/L, Lower Twin Lakes, 1982-1985.*

mean pH was small but statistically significant. Lower Twin Lakes sustained higher average levels of primary production during this period than did the upper lake (see ch. 8), which seems to have had the effect of maintaining a slightly higher mean pH in the lower lake.

The mean conductivity and pH of Lower Twin Lakes were reduced *significantly* during the post-operational period by the importation of even more dilute and poorly buffered water from Turquoise Reservoir (Sta. 5 in tables 6-2, 6-3). Upper Twin Lakes was relatively unaffected by this imported water; its mean conductivity and pH continued to reflect the influence of Lake Creek.

Mean bottom E_7 values (table 6-4) declined significantly and almost uniformly, in both Upper and Lower Twin Lakes, during the postoperational period. Also, there was not significant difference between mean bottom E_7 values in Mt. Elbert Forebay and those in Twin Lakes. Redox potential is an indicator of state, rather than a driving variable. Brannon et al. (1978) called redox potential "a semiquantitative indicator of the intensity of reduction" and "a mixed potential generated by a variety of oxidation-reduction couples." In their laboratory studies and literature review, Brannon et al. (1978) found that dissolved organic matter can strongly influence the redox potential of a system; they observed that concentrations of volatile organic acids increased in their systems when the soils were first inundated. A similar increase in dissolved organics, from the recently flooded soils of Mt. Elbert Forebay and the expanded littoral areas of Twin Lakes, may have caused bottom redox potentials to decline during the operational period. In this case, it is possible that bottom redox potentials in the Twin Lakes system could eventually rise as these submerged soils become depleted in soluble organic matter.

Light

Light penetration.—Three indexes of light penetration in the Twin Lakes system are noted in table 6-5. The light extinction coefficients and euphotic depths were both calculated from measured light intensity profiles. A light extinction coefficient indicates the rate at which light passing through the surface of the water is absorbed, or attenuated—with depth. The coefficient varies inversely with water clarity. The euphotic depth is the depth at which light intensity is 1 percent of the surface light intensity. This depth represents the theoretical limit of photosynthesis; it is calculated using the light extinction coefficient. The Secchi depth is a subjective measurement of near-surface water clarity. Both Secchi depth and euphotic depth vary directly with water clarity.

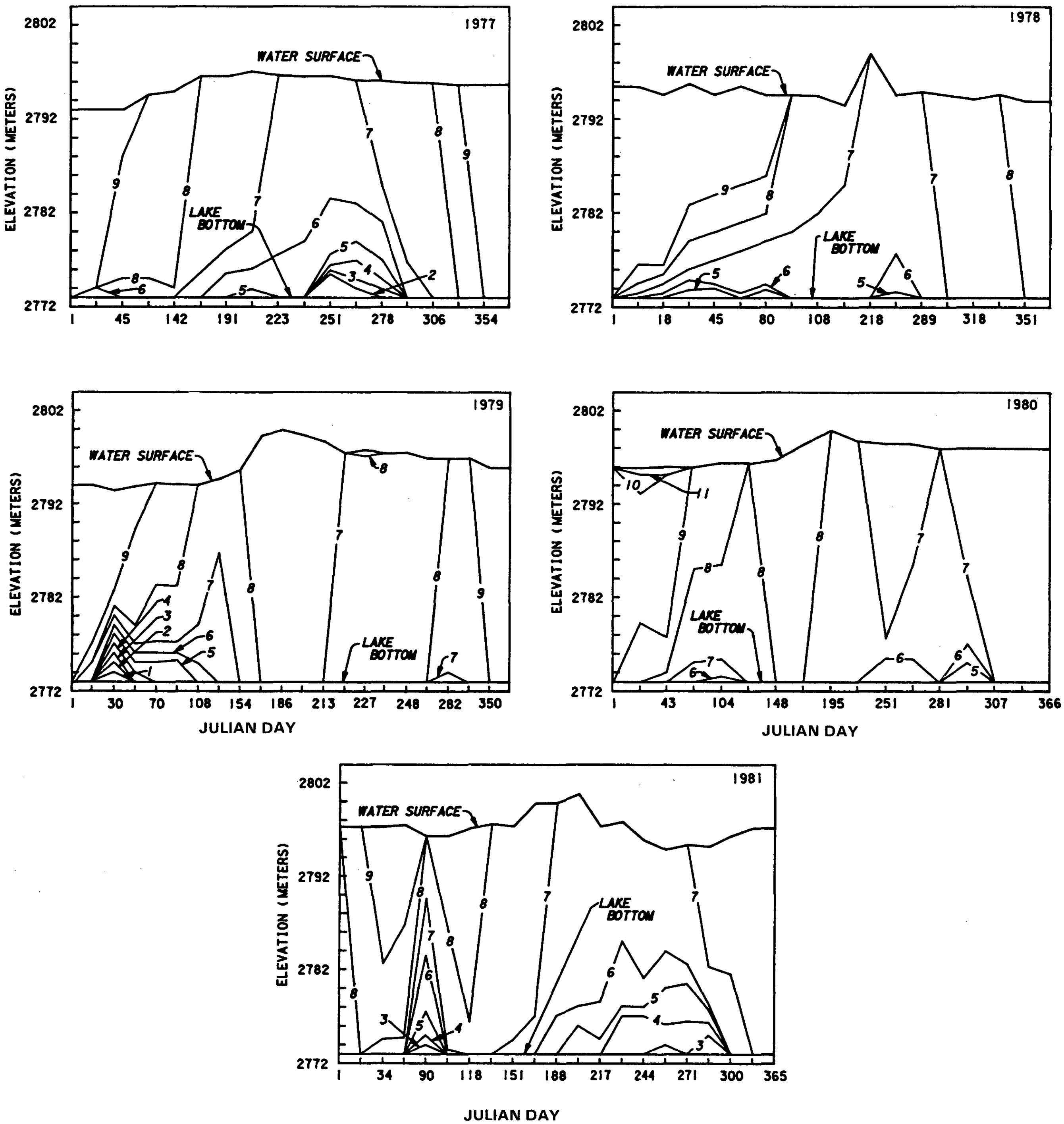

FIGURE 6-10.—*Dissolved oxygen isopleths, mg/L, Upper Twin Lakes, 1977-1981.*

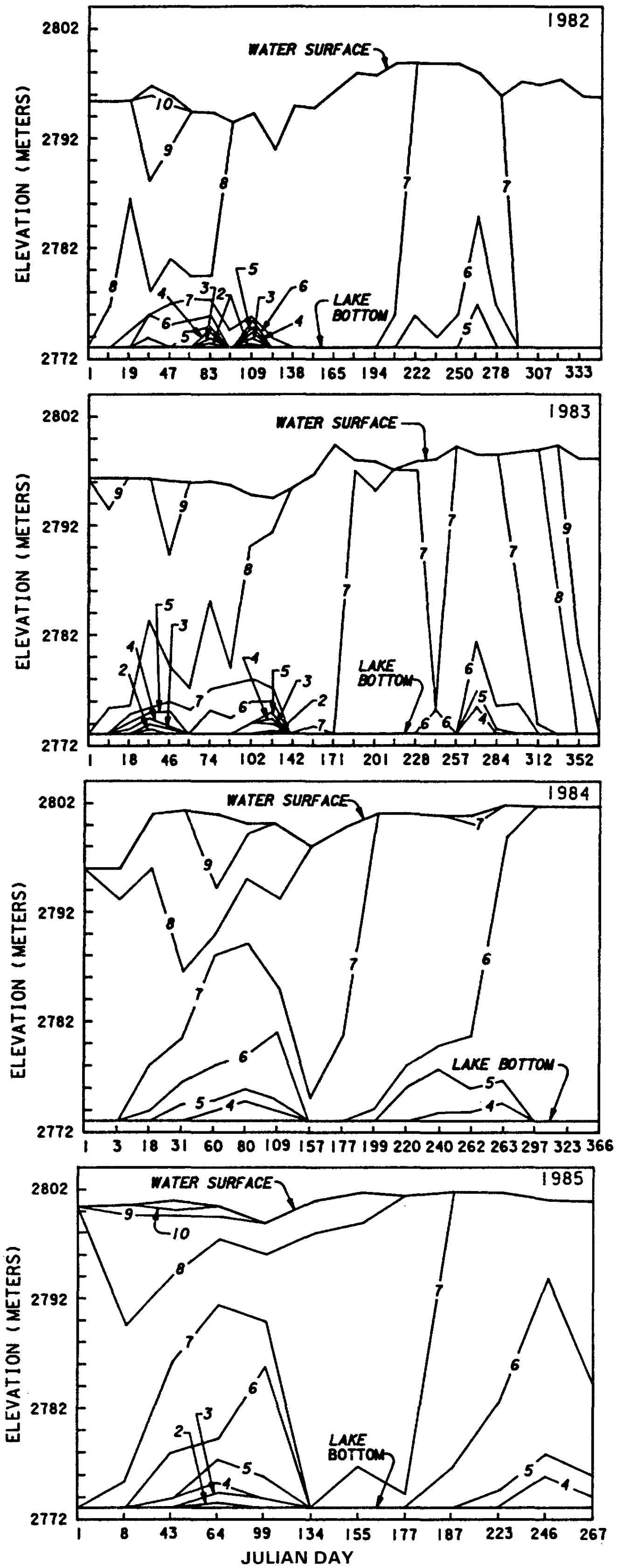

FIGURE 6-11.—*Dissolved oxygen isopleths, mg/L, Upper Twin Lakes, 1982-1985.*

At station 4, the mean preoperational light extinction coefficient was significantly greater than that at station 2. During the postoperational period, light extinction coefficients increased significantly at both stations, resulting in mean coefficients (Sta. 2 and 4) no longer statistically different. Thus, light extinction coefficients derived from the light intensity profiles indicated that the clarity of Twin Lakes was significantly reduced during the postoperational period (table 6-5).

Figure 6-14 shows the trends in mean annual light extinction coefficients in both of the lakes during the entire study period. Whereas the lower lake was significantly clearer than the upper lake (during the preoperational period) the significant increase in light extinction at station 4 and the highly significant increase at station 2 brought both lakes to a similar condition of reduced clarity during the postoperational period. Both lakes were affected to some degree by the increased inflow volumes (fig. 6-15) and rising lake elevations (fig. 6-16) which exposed new shorelines to erosion by water level fluctuations induced by powerplant pumping and generation operations (fig. 6-17). However, Lower Twin Lakes was most affected by the increased inflow and powerplant operations (figs. 6-15, 6-17). This is a direct contrast to the preoperational period, when all inflow had first entered the system through the upper lake— which served as a settling basin and "hydraulic shock absorber" for the lower lake.

The statistical results for euphotic depths were essentially the inverse of those for light extinction coefficients. During the preoperational period, the mean euphotic depth at station 2 was significantly greater than that at station 4; mean euphotic depths

Table 6-5.—Light penetration in the Twin Lakes system.

	Preoperational period 1977-81	Postoperational period 1982-85	Change
	Mean light extinction coefficient, m^{-1}		
Upper lake (Sta. 4)	0.50	0.56	*
†(4-2)	*	NS‡	
Lower lake (Sta. 2)	0.44	0.55	**
	Mean euphotic depth, m		
Upper lake (Sta. 4)	9.46	8.80	NS
†(4-2)	*	NS	
Lower lake (Sta. 2)	10.43	8.80	**
	Mean Secchi depth, m		
Upper lake (Sta. 4)	3.87	3.03	*
(4-2)	NS	*	
Lower lake (Sta. 2)	4.04	3.82	NS

† (4-2) = difference between stations 4 and 2.
‡ NS = not significant.
* Significant: $P<0.05$.
** Highly significant: $P<0.01$.

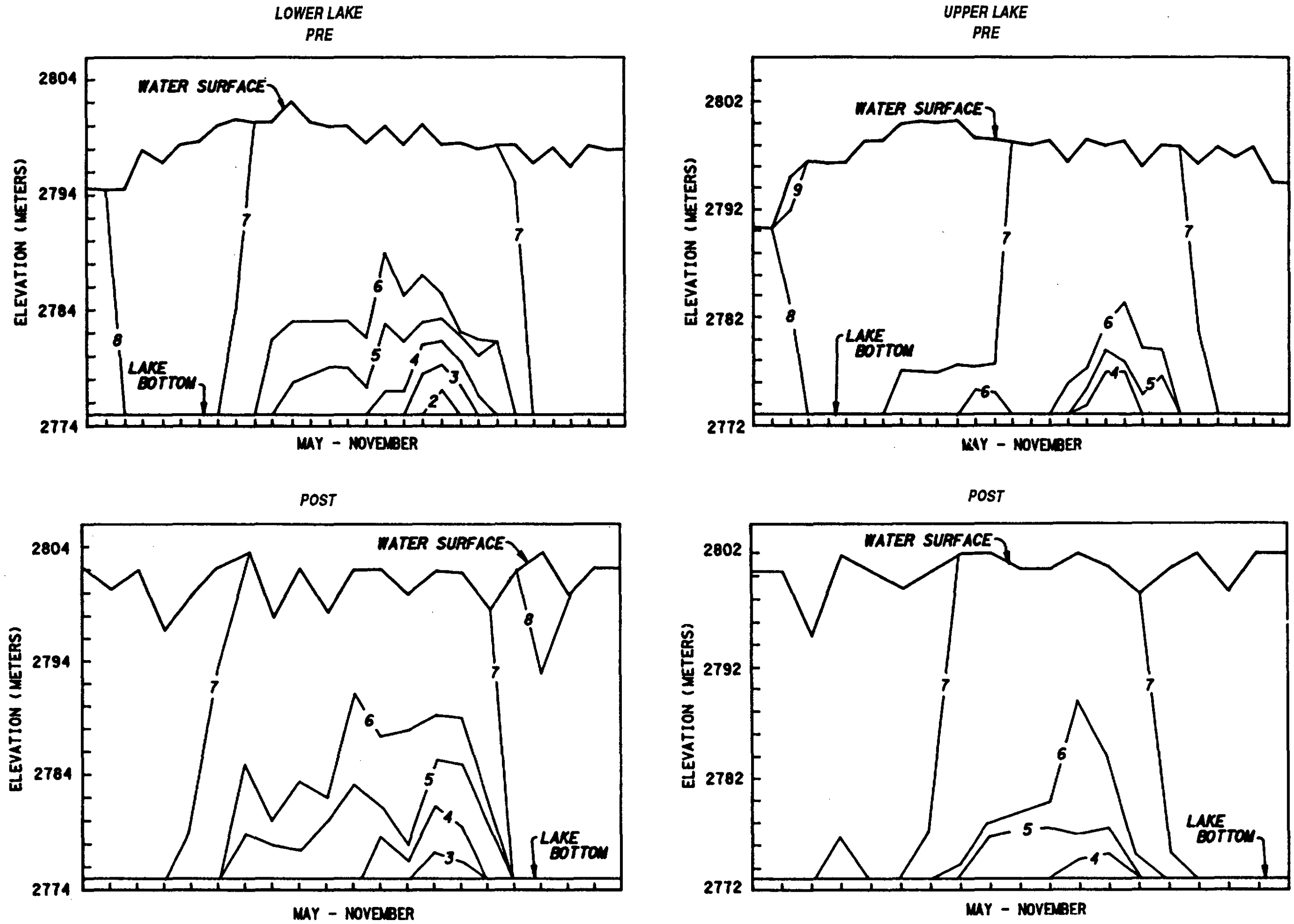

FIGURE 6-12.—*Average dissolved oxygen concentration, mg/L, isopleths in Twin Lakes of pre- and postoperational periods of Mt. Elbert Pumped-Storage Powerplant operation.*

also decreased to the same value at both stations. Only the decrease at station 2 was statistically significant; however, the finding of no significant change at station 4 probably resulted from the data transformation involved in calculating a depth from a coefficient.

Mean Secchi depth statistical results were not in complete agreement with the results derived from the light intensity profiles. Preoperational mean Secchi depths at stations 2 and 4 were not statistically different. During the postoperational period, mean Secchi depth at station 4 decreased significantly, whereas the decrease at station 2 was not significant. Thus, the mean Secchi depth at station 2 was significantly greater than that at station 4.

The Secchi disk results (table 6-5) do not contradict the light intensity profile results so much as they show a different aspect of the same situation. Because Secchi depth was only measured during the ice-free seasons of the year, the results are limited to "summer" conditions. Hence, it is evident why the mean Secchi depth in Upper Twin Lakes declined significantly during the postoperational period. Greater than average spring runoff in 1982 and 1983 caused high turbidity in the still unexpanded upper lake. During the summers of 1984-85, the upper lake had been raised which created a broad, shallow inflow area that spread and warmed the incoming Lake Creek waters. The result was that Lake Creek now entered Upper Twin Lakes as a near-surface inflow rather than as a plunging inflow as in the past. Sediment and debris, flushed out of the newly flooded area, was carried farther out into the deep part of the lake instead of being deposited along a near-shore plunge line.

Water clarity in Twin Lakes may recover somewhat as the new shoreline becomes established and the recently inundated areas are flushed of debris. However, the increased circulation in Lower Twin Lakes—induced by powerplant operations and the passing of diverted project waters—will probably maintain water clarity in this lake at less than preproject levels. Figure 6-18 shows Secchi disk data

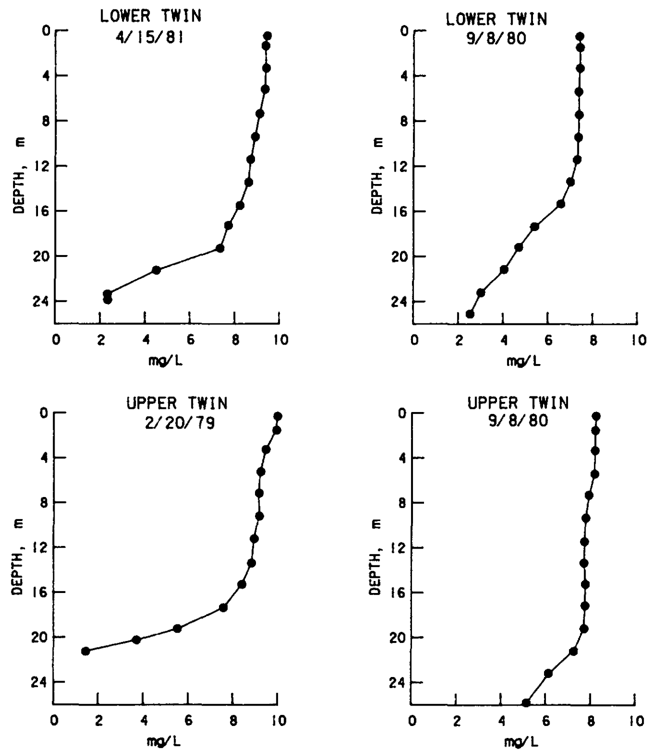

FIGURE 6-13.—*Dissolved oxygen profiles, on selected dates, in Twin Lakes.*

collected during July and August and how transparency has apparently decreased from the time (1903) when the lakes were studied by Juday (1906) to now (1986). The August Secchi depths dropped from 9 m in 1903 to 6 m in 1978 and 1980, and to less than 4 m in 1982, 1983, 1984, 1985, and 1986. Each of these decreases in transparency occurred after major changes in the hydraulic regime of the Twin Lakes system (see ch. 2).

Transmissivity.—Transmissivity is a measure of relative water clarity at a given depth. Figures 6-19 and 6-20 present selected transmissivity data from the upper and lower lakes, respectively. These profiles are representative for each of the four seasons before (1981) and after (1984) powerplant operation began. The thermocline, when present, is depicted as the shaded area. A typical pattern is evident within the water column of each lake, i.e:

- An initial increase in clarity near the surface,
- A drop in clarity near the bottom of the euphotic zone (and within the thermocline when it is located in the euphotic zone),
- Another increase in clarity below the euphotic zone, and
- Finally, a decrease in clarity near the bottom.

Generally, this pattern persisted after powerplant operation began, but in a modified sense; i.e., the definition of the transmissivity profiles was lessened due to a disruption of stratification.

There are no long-term averages for transmissivity as there are for the other limnological parameters. LaBounty et al. (1984) compared the 1981 and 1982 transmissivity data and concluded that an average 17-percent decrease in percent transmissivity existed in both lakes from January through May 1982—caused by powerplant operation. This conclusion is tempered because only 1981 data were used as the preoperational condition. Nonetheless, figure 6-21—presented to support the conclusion—shows the 1-year contrast in transmissivity, indicating the decrease in 1982 after powerplant operation began.

Analyses of data on figures 6-19 and 6-20 reveal important qualities of Twin Lakes. Many seasonal events are reflected in the transmissivity data. Several factors can cause percent transmittance to decrease, including: allochthonous sediment, zooplankton, phytoplankton, and resuspended material from bottom sediments. During winter and with ice cover, both lakes show decreased transmittance near the bottom, indicating the buildup of resuspended material from bottom sediments. Also, at times during winter, a decrease in transmittance exists just above the 10-m depth. This situation is especially true in the upper lake, due to either turbid inflowing water entering at this depth or a phytoplankton bloom under the ice.

During months when spring turnover occurs (April and May), the transmittance readings are isometrical. The profiles from early June through the end of September show some characteristic patterns each year. First—and perhaps most significantly to the lakes' food chain—within the thermocline, a phytoplankton bloom develops in both lakes. This bloom becomes well developed by late August. By mid-September, these algae are either dispersed or dead. Below the thermocline, or even within it if it is below the euphotic depth, densities of plankton are generally less—resulting in clearer water. Examination of plankton samples verifies this observation. The dominant species are either *Synedra* or *Dinobryon* (see ch. 8). During at least one survey, this dense phytoplanton layer was greater than 250 mm thick.

The decrease in transmittance near the bottom between early June and late September is due to either turbulence at the bottom (from the inflow of colder water) and ensuing resuspension of material from bottom sediments, or the accumulation of sinking debris from the upper water column. When fall turnover occurs each year (mid-Oct.), the nutrients that accumulate in this bottom area are dispersed throughout the water column and

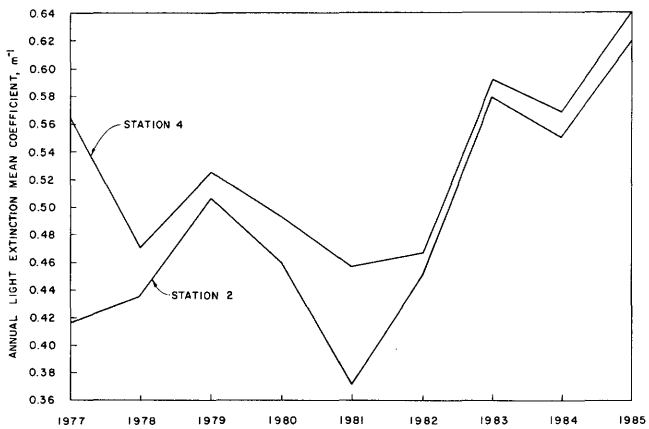

FIGURE 6-14.—*Mean annual light extinction coefficient.*

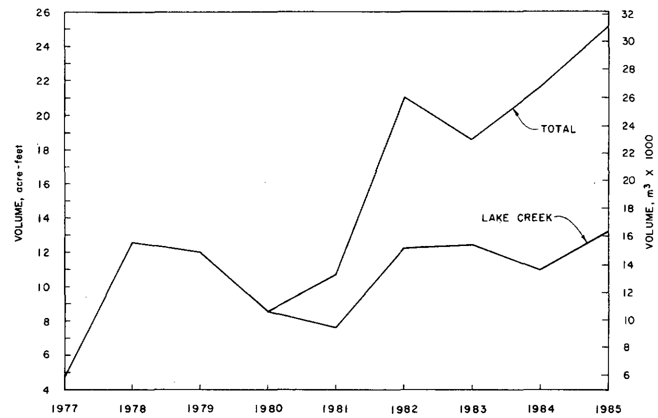

FIGURE 6-15.—*Mean annual inflow volume.*

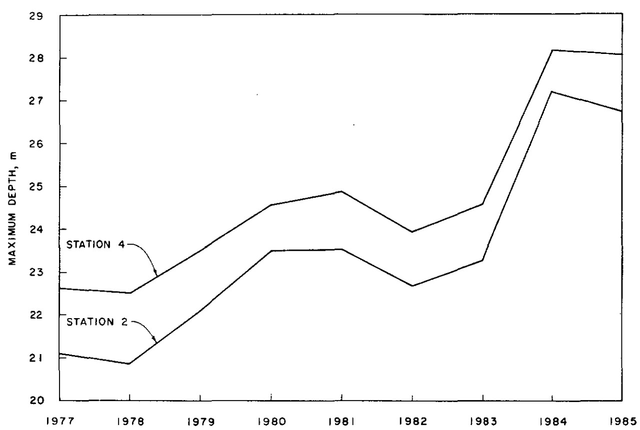

FIGURE 6-16.—*Maximum depth by year.*

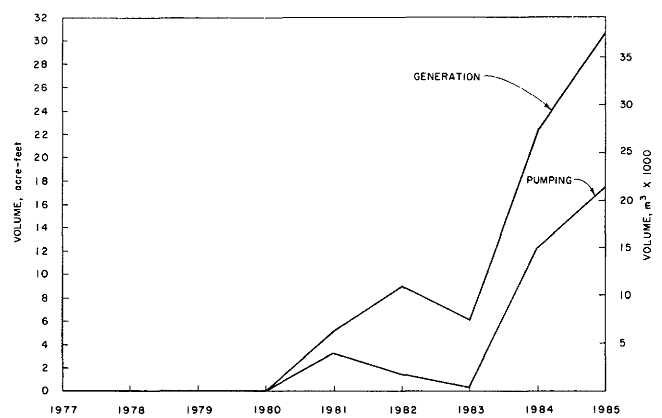

FIGURE 6-17.—*Mean annual pump-turbine operation.*

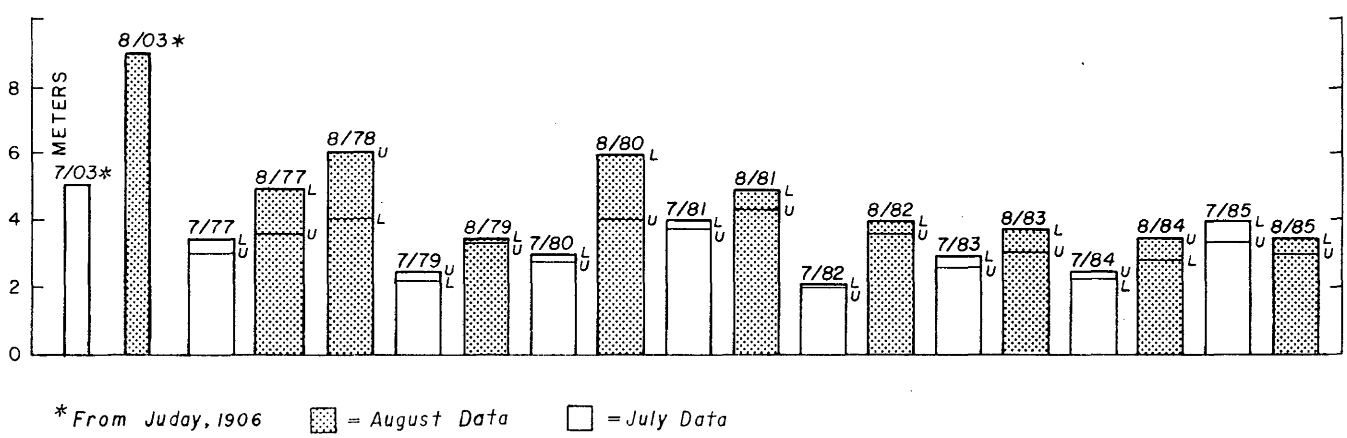

FIGURE 6-18.—*Secchi depths from July and August 1903 and 1977-86.*

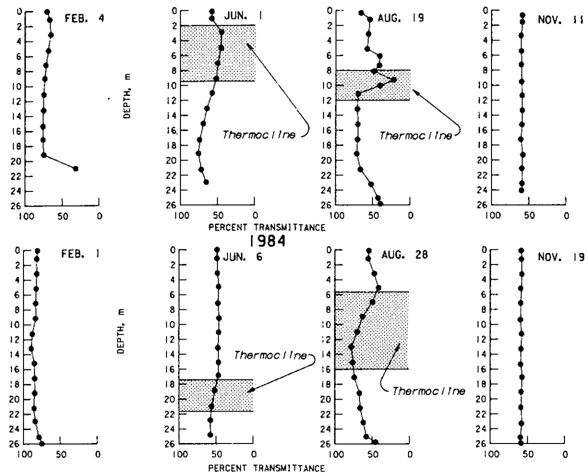

FIGURE 6-19.—*Light transmittance profiles in Upper Twin Lakes. 1981 (top)—1984 (bottom).*

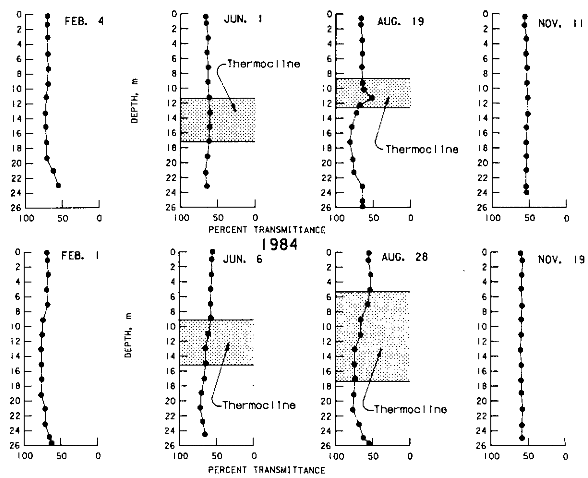

FIGURE 6-20.—*Light transmittance profiles in Lower Twin Lakes. 1981 (top)—1984 (bottom).*

are primarily responsible for the normal increase in primary productivity observed during October and November. Isothermal conditions are again reflected in the fall transmissivity profiles, which resemble those from spring, except that transmittance is 10 to 25 percent less because substantially

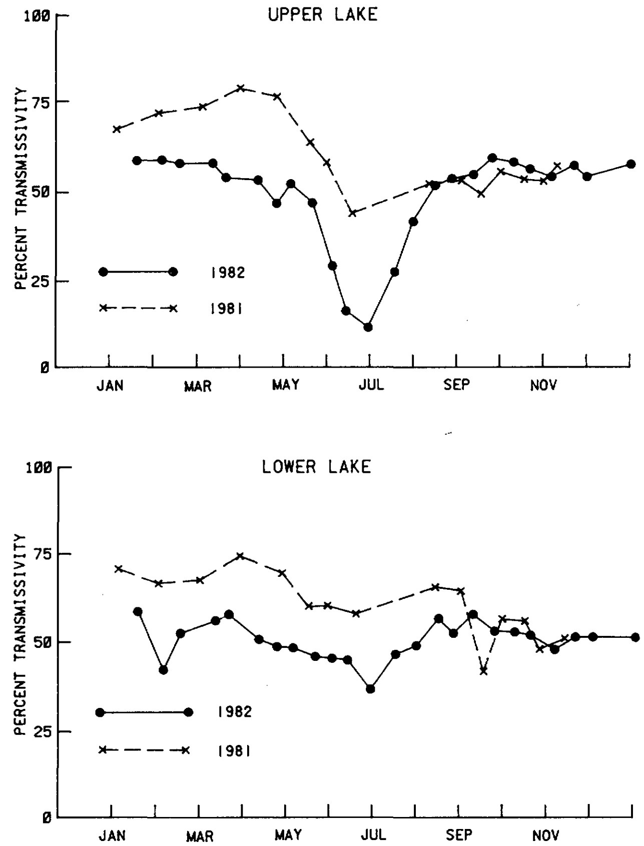

FIGURE 6-21.—*Average percent of light transmittance in Twin Lakes, 1981-82.*

more plankton and organic debris exist in both lakes during the fall than in the spring.

Overall, transmissivity of Twin Lakes is commonly between 50 and 80 percent. Figures 6-19 and 6-20 indicate that powerplant operation causes a greater dispersal of phytoplankton in the water column; i.e., the peaks of dense phytoplankton within the thermocline before powerplant operation began are not observed after powerplant operation began. As figure 6-7 shows, the integrity of the thermal stratification profile is disrupted to at least a limited degree, and this situation probably results in a more even dispersal of plankton.

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Sediments — Chapter 7

E. P. Bergersen

Colorado Cooperative Fish
and Wildlife Research Unit
Colorado State University
Fort Collins, Colorado

INTRODUCTION

Lake sediments provide a chronological record of the physical and biological indicators of past environmental conditions because they are the ultimate sink for many living and nonliving materials produced in the lake or by its watershed. By correlating a known event in the history of a lake with a position in the sediment profile, sedimentation rates may be estimated. With this information, it is possible to develop a history of past lake conditions and to perhaps gain some insight into future lake environments. Sediment composition and the biological history of recent sediments also provide baseline information against which any effects of powerplant operation on the sediments can be compared.

METHODS

Core Sampling

Twenty-four bottom samples (fig. 7-1; 7 from Upper Twin Lakes and 17 from Lower Twin Lakes) were collected in 1975 with a core freezer (Shapiro, 1958) to retain the stratigraphy of the uppermost sediment. Corer construction and collection techniques were described by Wunderlich (1974). These cores were examined in detail using atomic absorption analyses for profile changes in copper, lead, silver, and zinc. The cores also were subjected to semiqualitative analyses (spark source mass spectrometry) for 48 elements, examination of total carbon content, and measurements of particle size. A cluster of seven cores was taken from the lower lake in 1974 to determine sedimentation rate.

Cores were kept frozen until processed and then all were sliced at 5-mm intervals with a lapidary saw. Slices were rinsed with distilled water to remove surface contaminants, quartered, and air-dried or retained frozen until analyzed.

Analytical Design

Corresponding 5-mm segments were pooled from the seven core samples collected for sedimentation rate determination. One core from each lake was examined in detail for profile changes in copper, lead, silver, and zinc concentrations, and another single core from each lake was analyzed for carbon content. Four cores from each lake were quartered and then combined for the 48-element analysis:

- First quarter was the top 50 mm of the core (composite I),
- Second quarter was the 50- to 75-mm core section (composite II),
- Third quarter was the 75- to 100-mm section (composite III), and
- Fourth quarter was the 100- to 150-mm section (composite IV).

In one core (from each lake), the quarters were divided into 5-mm segments. Three of these segments from each quarter were combined, and the mixture was measured for sediment size. A small portion of these segments, as well as 5-mm segments from all remaining cores, was examined for changes in diatom community structure. The sediments were cleaned of organic matter by dichromate oxidation to improve diatom identification. The araphidineae to centric diatom ratio (Stockner, 1971) was calculated for every other segment of each core. Mean values were based on multiple counts from each segment.

Sediment Rate Estimation

Because visible layers of sediments were not evident in Twin Lakes, sedimentation rates were

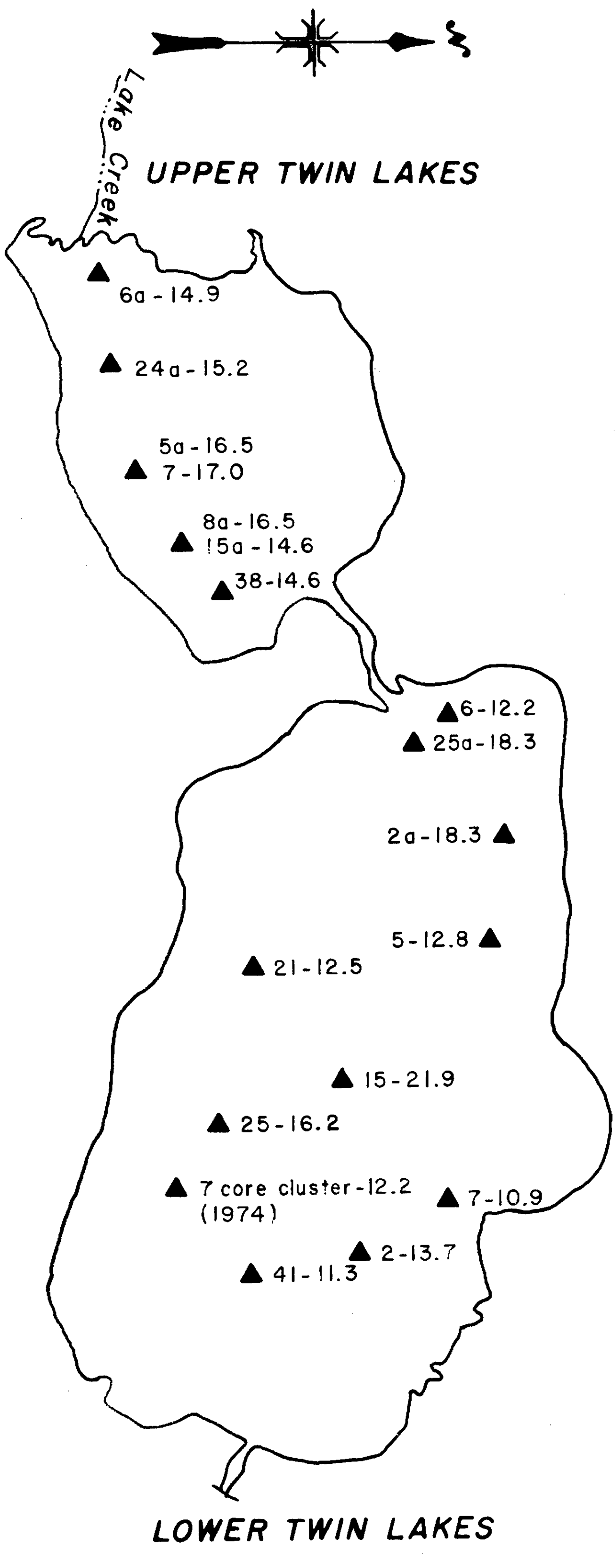


FIGURE 7-1.—*Core sample locations in Twin Lakes. Core identification number and depth (meters) where samples were collected in 1975—unless noted.*

estimated from analysis of the fallout radionuclide cesium 137 in sediments resulting from atmospheric nuclear testing (Ritchie et al., 1973). Personnel of Colorado State University's Radiation Biology Department measured the ^{137}Cs. The counts per minute of ^{137}Cs were corrected for background radiation and interference from ^{40}K, converted to picocuries, and expressed on a sample weight basis. Confidence limits were computed on the counting statistics.

To provide an additional check on estimated sedimentation rates based on ^{137}Cs counts, concentrations of copper, lead, silver, and zinc in the sediment profile were examined by atomic absorption for possible changes that might be related to some recorded event in the history of the lakes. An example of particular interest was the extensive mining and associated activities that occurred in the area 100 to 125 years ago that resulted in the possible increase in heavy metal deposition in the lakes. All atomic absorption analyses were conducted at the Colorado State University Analytical Chemistry Facility.

Total Carbon and Sediment Size

Total carbon content and measurement of sediment particle size from composites of selected core quarters were made at CSU's Analytical Chemistry Facility. Carbon was measured as evolution of CO_2 on combustion. A settling rate technique was used to estimate particle size fractions.

RESULTS AND DISCUSSION

Sediment Composition

Significant difference in the percentage weight fraction greater than 20 micrometers was not found between the lakes. However, the mean value from upper lake data (65%) was somewhat greater than that from the lower lake (61%), possibly reflecting coarser fluvial material entering from Lake Creek.

The mean value of total carbon content (% by weight) in the composited cores from the upper lake was significantly higher than in the lower lake (table 7-1). More recent sediments in the upper lake generally had higher carbon levels than the sediment in the lower lake. Individual cores had carbon contents similar to those of the combined cores, indicating low variability in carbon content of the segments from the pooled samples. Higher carbon levels in the upper lake and, in particular, of the upper levels in more recent sediment, suggest that the upper lake has (at least in recent times) served as a nutrient trap in the Twin Lakes system. The

Table 7-1.—Total carbon content of selected cores, percent by weight.

Combined quarter	Upper Twin Lakes Four composites	Upper Twin Lakes Core 6A	Lower Twin Lakes Four composites	Lower Twin Lakes Core 6A
I	2.6	2.9	2.0	1.5
II	2.5	2.4	1.6	1.9
III	2.0	2.0	1.6	1.6
IV	2.1	2.2	1.5	1.8

48-element survey of the combined core segments also supports this premise because the elements had higher concentrations more often in the upper lake than in the lower lake (table 7-2).

Of the four selected metals examined in detail (copper, lead, silver, and zinc), only copper showed a distinct change in its concentration profile. Increasing copper concentrations began at about the 100-mm depth in the upper lake and generally increased to the sediment surface. Below 100 mm, copper concentrations were 75 to 100 parts per million; but above 100 mm copper concentrations increased steadily from 100 to 250 parts per million near the sediment-water interface. Copper increased only slightly in the more recent sediments of the lower lake.

Zinc, and to some extent silver, showed a slight tendency toward higher values in the more recent sediments of the upper lake. Lead concentrations were fairly constant in all sampled sediments, but were much higher in the lower lake (table 7-3). This difference may be related to the anaerobic conditions observed in the upper lake during ice-off in 1975 (Sartoris et al., 1977). Thus, if lead becomes soluble under anaerobic conditions, it may be transported to the lower lake with spring overturn and runoff before it returns to the sediments.

Many mines in the Leadville-Twin Lakes vicinity contained silver and sulfide ores of copper, lead, and zinc (Ubbelohde et al., 1971). The distinct increase in copper in recent lake sediments and, to a lesser extent zinc and silver, could be attributed to the impact of past mining activity on the watershed. Based on estimated sedimentation rates for the lake, as described below, the increases in these metals, copper in particular, correspond quite well with the onset of mining in the area about 100 to 125 years ago.

Sedimentation Rate

The highest concentrations of ^{137}Cs in the sediment of the lower lake were in the 5- to 10-mm layer (fig. 7-2). This peak probably reflected the incorporation of both recorded peaks of ^{137}Cs that occurred in 1959 and 1963 (Ritchie et al., 1973). If that is true, the sedimentation rate was 7.5 mm in 12 years, as the ^{137}Cs deposition—using 1961 as the midpoint year plus 1-year lag time for incorporation into the sediment—or 0.63 mm/year with a possible range of 0.42 to 0.83 mm/year. High cesium levels were evident to a depth of at least 65 mm in the analyzed sample. The higher values in the more recent sediments were probably due to a number of factors including water movements near the bottom, activity of burrowing animals, the necessity of pooling core samples, and possibly the vertical migration of the ^{137}Cs in the sediment. Although these factors may not completely disrupt the stratigraphy of the sediments, they tend to blur it to some extent.

The sedimentation rate estimated for Twin Lakes was considerably higher than those obtained by Pennak (1963). Pennak used radio carbon methods for dating two lakes and two bogs in the Colorado Front Range (east of Continental Divide), but was lower than those reported for two alpine lakes in Rocky Mountain National Park, Colorado, and a plains reservoir near Fort Collins, Colorado (Wunderlich, 1974). The two lakes examined by Pennak were only 2.5 hectares and had no large inflows. Lake Creek, however, carries a considerable sediment load into Twin Lakes, so the differences in sedimentation rates are not unexpected. The alpine lakes in Rocky Mountain National Park were also small, with limited inflows; however, both had similar histories of cultural influences on their watershed and, since 1940, have been increasingly subjected to organic pollution from heavy use by park visitors (Wunderlich, 1974).

Assuming accelerated cultural activity associated with placer mining in the upper Arkansas River Basin began in 1859 (Ubbelohde, 1969), and a sedimentation rate of 0.63 mm/year, 49 to 96 mm of material have been deposited in Twin Lakes since that time. This establishes a reference point (albeit large) relative to other observed changes in the sediment profiles. The wide range of possible deposition since 1859 is primarily due to the apparent low rate of sedimentation in the system and, consequently, the lack of resolution in defining the cesium peaks.

Diatom Profiles

Stockner (1971) proposed the ratio of araphidineae to centrale (A/C) diatoms in lake sediment as an indicator of trophic condition at the time of deposition.

Occasionally, the A/C values from the lower lake cores only exceeded two and generally were less than

Table 7-2.—The 48-element survey of combined cores from Upper and Lower Twin Lakes. Most concentrations are in parts per million, but some are recorded as percent. Upper lake: cores 6A, 7A, 15A, and 24A. Lower lake: cores 2, 6, 15, and 41.

	Composite Number							
	I		II		III		IV	
Element	Upper	Lower	Upper	Lower	Upper	Lower	Upper	Lower
Uranium	7.0	7.0	7.0	2.3	12.0	3.0	7.0	3.0
Thorium	12.0	12.0	12.0	4.0	12.0	9.0	20.0	5.1
Bismuth	12.0	5.0	8.7	5.0	5.0	3.9	12.0	4.4
Lead	70.0	70.0	30.0	110	30.0	52.0	70.0	70.0
Thallium	0.8	0.8	0.4	0.4	0.8	0.8	1.2	0.4
Gold	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1
Platinum	<0.14	<0.14	<0.14	<0.14	<0.14	<0.14	<0.14	<0.14
Tungsten	6.9	3.8	3.4	4.6	4.6	2.3	4.6	2.0
Tantalum	0.9	0.9	1.5	0.9	1.5	1.5	1.8	0.9
Hafnium	6.0	8.0	6.7	6.0	12.0	8.0	20.0	12.0
Cerium	67.0	120	100	67.0	67.0	120	130	33.0
Lanthanum	41.0	55.0	55.0	27.0	46.0	55.0	91.0	18.0
Barium	1000	1000	620	570	1300	730	1300	730
Cesium	4.3	4.3	4.3	4.3	4.3	4.3	8.7	3.0
Iodine	29.0	6.3	15.0	2.9	4.9	4.4	5.5	2.9
Antimony	0.25	0.33	0.14	0.33	0.14	0.33	0.33	0.33
Tin	2.0	2.0	35.0	0.93	4.7	4.7	2.0	47.0
Cadmium	0.63	1.5	3.1	0.73	1.5	7.3	1.5	3.1
Silver	3.0	0.64	2.2	0.45	2.5	3.0	3.0	3.0
Palladium	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1
Molybdenum	6.0	12.0	6.0	6.0	6.0	6.0	6.0	6.0
Niobium	6.9	6.9	6.9	5.3	6.9	6.9	16.0	6.9
Zirconium	110	100	60.0	60.0	450	110	110	60.0
Strontium	140	380	140	140	140	260	210	210
Berkelium	110	110	110	11.0	110	94.0	210	94.0
Bromine	12.0	12.0	5.1	2.4	5.4	5.1	2.4	5.1
Selenium	1.8	0.93	0.7	0.47	0.47	0.47	0.35	0.47
Arsenic	1.2	3.1	0.92	4.6	<0.92	1.2	3.1	1.2
Germanium	0.6	1.4	0.6	0.28	0.6	1.2	2.8	1.2
Zinc	1200	330	330	330	210	120	260	120
Copper	560	330	330	250	330	180	250	180
Nickel	18.0	18.0	18.0	18.0	14.0	18.0	18.0	7.7
Cobalt	38.0	3.0	25.0	7.0	14.0	7.0	7.0	7.0
Iron	>1.0%	>1.0%	>1.0%	>1.0%	>1.0%	>1.0%	>1.0%	>1.0%
Manganese	1400	>0.5%	2800	2800	490	1400	1400	1400
Chromium	67.0	67.0	67.0	36.0	67.0	36.0	67.0	36.0
Vanadium	50.0	100	50.0	50.0	100	50.0	50.0	50.0
Titanium	2000	4300	3800	3000	4300	2000	3000	2000
Calcium	>0.5%	>0.5%	>0.5%	>0.5%	>0.5%	>0.5%	>0.5%	>0.5%
Potassium	>0.5%	>1.0%	>0.5%	>1.0%	>0.5%	>1.0%	>0.5%	>1.0%
Chlorine	74.0	74.0	42.0	74.0	21.0	21.0	42.0	42.0
Sulfur	560	560	560	870	320	560	870	560
Phosphorus	3700	>0.5%	1400	>0.5%	1400	>0.5%	3700	2900
Magnesium	>1.0%	>0.5%	>0.5%	>1.0%	>0.5%	>0.5%	>0.5%	>0.5%
Sodium	3200	>0.5%	1400	>0.5%	>0.5%	>0.5%	>0.5%	4800
Boron	15.0	10.0	10.0	10.0	10.0	10.0	5.0	10.0
Beryllium	4.0	8.0	1.7	4.0	4.0	1.7	8.0	1.7
Lithium	43.0	75.0	120	75.0	120	120	75.0	

Trophic level	*A/C ratio*
Oligotrophic	0 to 1.0
Mesotrophic	1.0 to 2.0
Eutrophic	> 2.0

one (fig. 7-3). In contrast, distinct increase in the ratio was found at about the 100- to 150-mm depth in midlake cores from the upper lake (figs. 7-4, 7-5, 7-6). Above 100 mm, A/C values ranged from about 2.5 to 5. This pattern suggests that a distinct change from an oligotrophic to eutrophic condition occurred at some time in the recent past in the upper lake but did not occur in the lower lake.

Based on the estimated sedimentation rates in the lakes and the known onset of cultural activity in the area, the increase in the A/C ratio in the upper lake was initially likely due to the mining activities in the area. The subsequent suspected change in the quality of water entering the system and the continued high A/C values were probably the result of mining and other human activities, such as water diversion and the development of the town of Twin Lakes. Because of this distinct increase in the ratio, the upper lake appears to have acted (and perhaps still acts) as a trap for nutrients or other materials entering the Twin Lakes system.

Table 7-3.-Copper, lead, silver, and zinc (mg/L v. depth) in cores from Twin Lakes. Upper lake: core 38. Lower lake: core 15.

Depth	Copper		Lead		Silver		Zinc	
mm	38	15	38	15	38	15	38	15
5	160	200	20	130	2.5	4.7	300	30
15	240				4.7		350	
20	250	170	25	140	8.0	3.4	300	250
30	200				10.		400	0
35	190	180	13	110	4.2	3.3	300	250
45	150				3.9		350	
50	240	190	35	120	4.2	3.5	300	300
60	150				3.4		600	0
70	180	160	35	120	3.0	3.0	230	250
80	150				3.3		300	
90	170	130	23	130	6.0	2.6	250	230
100	120	85		80	5.0	6.0	250	220
110	85	150	40	80	2.8	2.5	220	240
120		130		100		2.1		250
125	100	180	40	120	5.0	7.0	220	220
140	100	120	40	110	2.2	4.0	450	200
155	80	120	45	100	2.2	3.8	240	230
165		160		70		7.0		600
170	90	200	45	150	2.7	12	230	2
180	0	120	30	170	4.2	4.0	1500	220
190	100	110	40	110	3.7	2.4	230	200
200		95	90	40	3.4	2.5	350	1200
205			30		3.4		220	
210	110	110	120	70	3.2	2.5	300	200
215			80		8.0		300	
220		23	50	50	3.2	1.9	250	300
225		100		40		1.2		300
230	90		80		3.5		230	
240			20		3.5		300	
250	85		40		3.4		220	

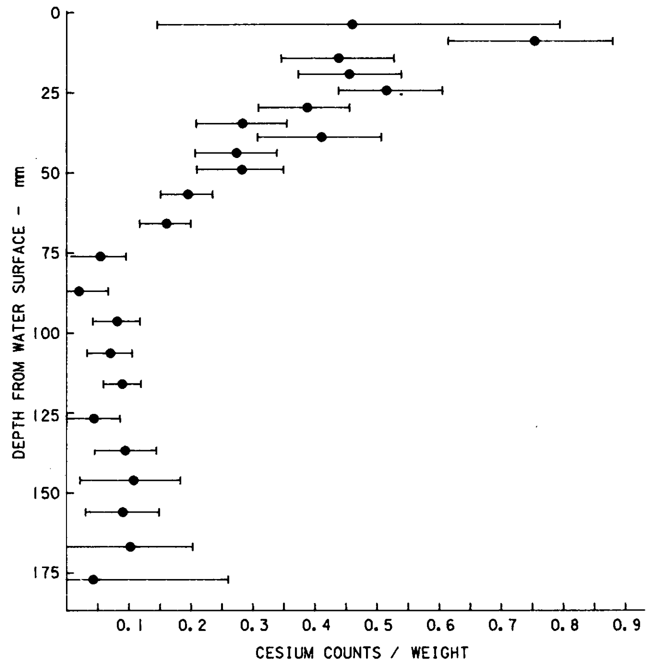


FIGURE 7-2.—*Depth v. Ce 137 in Lower Twin Lakes.*

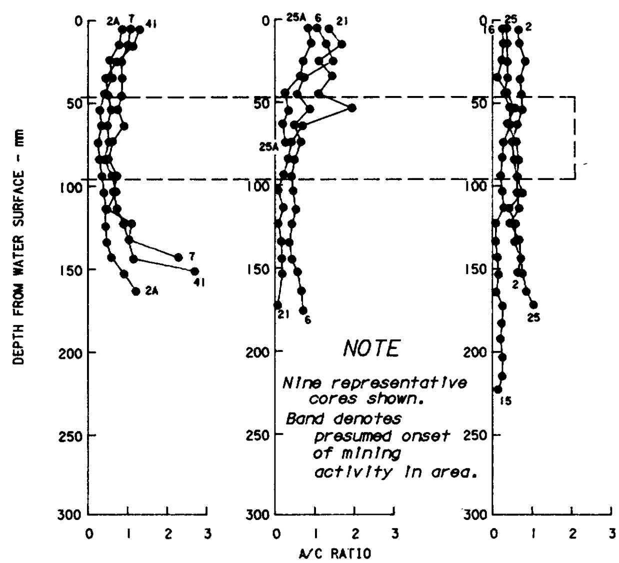


FIGURE 7-3.—*Depth v. ratio of araphidineae/centric in Lower Twin Lakes.*

Although high A/C values indicate eutrophication, primary production, and biomass of microinvertebrates and fish is less in the upper lake than the lower lake. However, the upper lake has an anaerobic hypolimnion during certain times of the year due largely to allochthonous production; an extensive swampy meadow once existed on the west shore of the lake (Juday, 1906) that appears to have contributed elevated amounts of organic matter to the lake. The suddenness and timing of the change in the A/C ratio suggests that the diatom community was responding not to a natural eutrophication process but to some other perturbation.

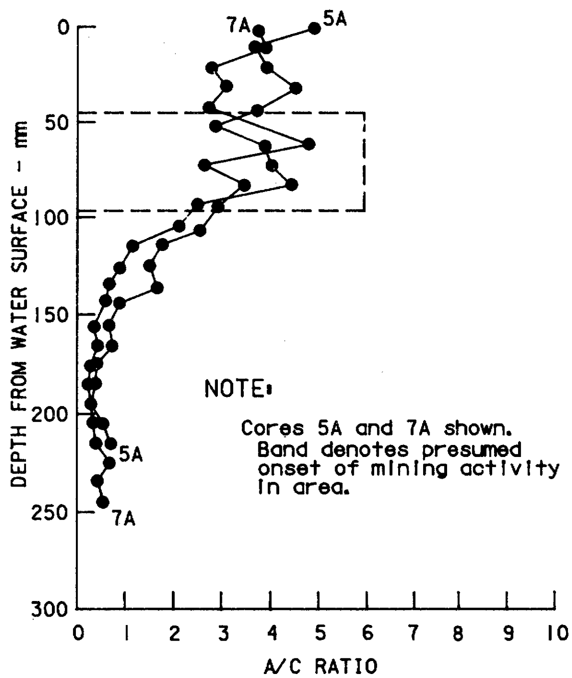


FIGURE 7-4.—*Depth v. ratio of araphidineae/centric in Upper Twin Lakes (cores 5A and 7A).*

Unusually high levels of copper have been found in the upper lake (LaBounty and Sartoris, 1976); and heavy metals brought into solution due to anaerobic conditions have been implicated in declines in the flora and fauna of the upper lake (LaBounty and Sartoris, 1976). The diatom groups probably respond differently to heavy metal stress as well as increased nutrient levels, which would affect the ratio between them. Differential responses to heavy metals and heavy metal pulses, which would be expected when occasional anaerobic conditions trigger high metal concentrations, could explain the erratic changes in the ratios noted in the upper 100 mm of the upper lake sediments.

The A/C values in two cores collected near the Lake Creek inlet to the upper lake were considerably higher at all depths than values from corresponding depths of all other cores. High ratios throughout the cores from this area indicate a higher sedimen-

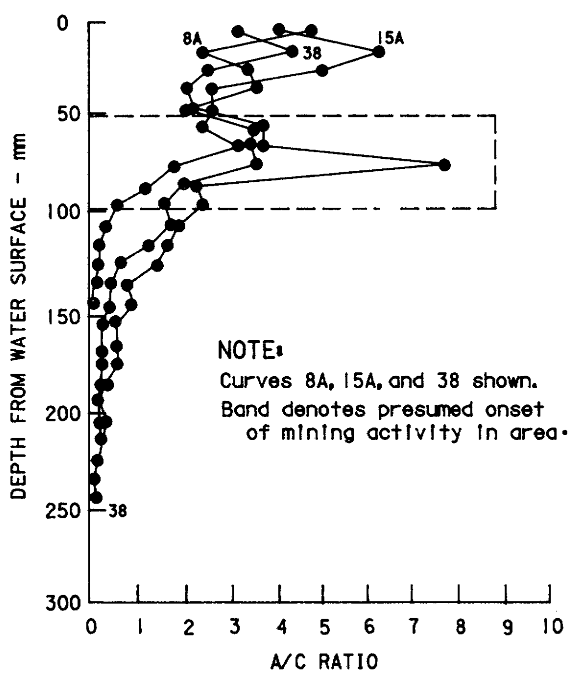


FIGURE 7-5.—*Depth v. ratio of araphidineae/centric in Upper Twin Lakes (cores 8A, 15A, and 38).*

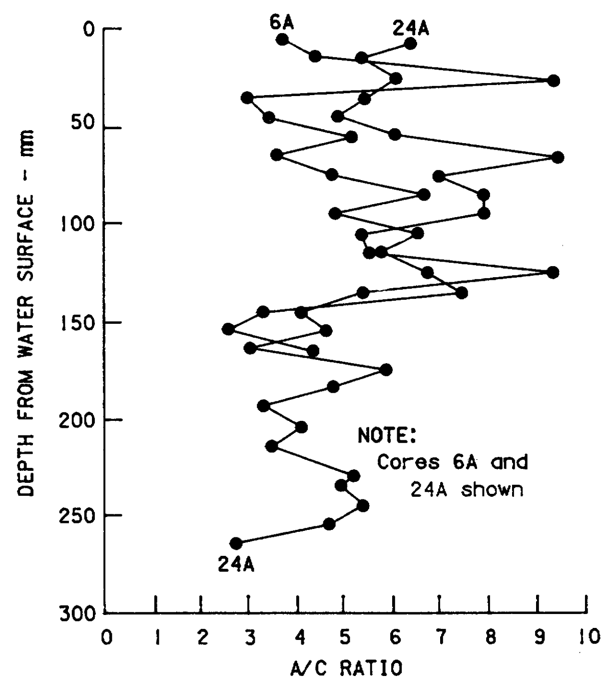


FIGURE 7-6.—*Depth v. ratio of araphidineae/centric in Upper Twin Lakes (cores 6A and 24A).*

tation rate in this area. Considering the depth to which high values extended, and comparing them with values of other profiles at similar depths in the upper lake, it appears that the sedimentation rate is at least 2.5 to 3 times greater near the inlet than elsewhere in the lake.

* * * * *

Primary Production Chapter 8

S. G. Campbell

Applied Sciences Branch
Bureau of Reclamation
Denver, Colorado

INTRODUCTION

Cole (1979) defined primary production as "stemming from the energy in organic substances that autotrophic organisms create from inorganic raw materials." Harris (1984) further explained that ". . . rates of productivity are commonly expressed in terms of chlorophyll or carbon . . ." and that ". . . population growth may also be measured by changes in cell number, changes in particulate carbon or other elements, or changes in chlorophyll." Primary production parameters measured at Twin Lakes included photosynthetic rate (carbon fixation rate), biomass of chlorophyll *a*, and the number and kind of algae within which the photosynthetic process occurred (phytoplankton density).

The ultimate goals for primary production estimates may seem simplistic; however, primary production estimates are generally used to classify ecosystems tropically. Also, primary production estimates can be used more practically as terms that define the relative water quality of an ecosystem, with oligotrophic and eutrophic systems generally being at the opposite ends of a scale of good and bad water quality, respectively. Other more practical applications for primary production estimates include:

- Determining fish stocking rates in managed ecosystems,
- Assessing sewage treatment requirements to meet point source pollution criteria, and
- Monitoring changes in previously undeveloped areas subject to increasing development.

Flynn (1988) commented that using the term "primary production" in aquatic ecology seemed undesirable because it fostered the view of aquatic ecosystems in discrete, artificial, trophic levels. The goal (established for primary production estimation at Twin Lakes) was quantification of the effects of operating a pumped-storage powerplant on the ecology of the lakes, of which, primary production was a major biotic component.

A working set of hypotheses about the effects of powerplant operation on both abiotic and biotic parameters that could affect primary production in Twin Lakes developed during the studies.

First, the glacial flour (the major sediment component in the lower lake) could be stirred by powerplant discharge and remain in suspension for a considerable time. Suspended sediments could significantly increase turbidity, particularly in the lower lake. In turn, increased turbidity could decrease the total volume of the euphotic zone, place a further limit on light penetration in the water column, and result in decreased chlorophyll *a* concentrations, carbon fixation rates, and possibly affect or influence both the magnitude and composition of the phytoplankton assemblage in Twin Lakes.

Second, water chemistry changes, particularly in the lower lake, might occur when a portion of the Fryingpan-Arkansas project, Western Slope (west of the Continental Divide—Rocky Mountains) water diversion passed through the Mt. Elbert Conduit directly into the lower lake (see also ch. 2). Previously, Western Slope diversions had been made into the upper lake through the Twin Lakes Tunnel which entered Lake Creek several miles west of the upper lake, passing first through the upper lake receiving basin, and then into the lower lake. Thus, the lower lake was buffered by the upper lake in years of high runoff and deprived of nutrient laden inflow in years of low runoff. Decreased nutrient concentrations in the lower lake could occur as a result of dilution from the relatively nutrient-poor water entering the lake through the Mt. Elbert Conduit. Increased nutrient concentrations in the upper lake were expected, when the lake levels were raised and the flat, marshy area (west of the lake) was inundated, that could result in increased algal biomass and possibly a change in species dominance in the phytoplankton assemblage.

Third, the temperature regime in the lakes might be affected by powerplant operation, perhaps resulting in temperature changes in the euphotic zone. A significant cooling of the euphotic zone might extend the period of diatom dominance of the phytoplankton assemblage at the lakes through summer months. A significant warming of the euphotic zone could increase algal biomass and allow facultative warm water algal species, such as blue-green algae, to become part of the phytoplankton assemblage and even dominate during the summer months. A possible breakdown in typical patterns of thermal stratification during the summer months might occur if generating and pumping activity affected the onset, duration, or breakdown of thermal stratification. Increased mixing of the euphotic zone could increase nutrient availability to the algal biomass and result in increased biomass or a different species dominating the phytoplankton assemblage.

The final expectation was any combination of the events that occurred at Twin Lakes, during the study period, such as: the raised water surface, the changed hydrologic regime, and the pumped-storage powerplant operation, could potentially affect primary production. However, determining which event resulted in a particular change could be difficult. The differences, if any, in primary production between pre- and postoperational periods might not be directly attributable to any single event; but could certainly be the result of cumulative changes in morphology, water surface elevation, hydrology, and powerplant operation.

METHODS

Chlorophyll *a*

Chlorophyll *a* samples were collected in the field according to methods reported by Strickland and Parsons (1972), and Holm-Hansen and Reimann (1978). Standard sampling depths for collection of chlorophyll *a* samples in both lakes were: 0.1, 1.0, 3.0, 5.0, 9.0, and 15.0 m. Water was collected with a vertical water bottle sampler at each depth interval and decanted into 1.9-L Nalgene containers. The sample containers were kept in an insulated box until they were filtered in the field laboratory at Twin Lakes. Replicate 750-mL aliquots from each depth were filtered through Whatman GF/C fiberglass filters using a Millipore Corp. manifold and vacuum pump apparatus at 238.4 kilopascals. The filters were suction-dried, folded in half (with sample surface inside) and placed in individually labeled manilla coin envelopes. The samples were frozen immediately and kept at -18 °C until processed.

Spectrophotometric processing of chlorophyll *a* samples was always performed within 3 weeks of collection and usually within 7 days. Extraction methods used for chlorophyll *a* samples followed procedures described by Holm-Hansen and Reimann (1978). A Beckman Model 24 spectrophotometer was used for analysis of chlorophyll *a* samples throughout most of both periods (1978–85). Chlorophyll *a* concentrations were calculated from analytical results using the trichromatic equations reported by Jeffery and Humphrey (1975).

Carbon Fixation Rate, C^{14}

The standardized C^{14} primary productivity test at Twin Lakes, Colorado, was performed using a modified technique reported by Steemann-Nielsen (1952) and completely described in LaBounty et al. (1985). In summary, water was collected from five depth intervals; 1, 3, 5, 9, and 15 m. Four replicate 300-mL aliquots from each depth interval were decanted into two clear and two black coated 300-mL BOD bottles (biological oxygen demand). Each of the clear (light) and one of the black coated (dark) replicates were inoculated with C^{14} tagged $NaHCO_3$ at a standard rate of 1 μCi per 100-mL water. The remaining dark bottle was used for field titration of bicarbonate alkalinity. Following a minimum of 4-hour field incubation, during which the bottles were resuspended in the lake at the original sampling intervals, the samples were pressure filtered through a membrane filter (0.8 μm pore size) at 275 to 420 kPa. Filters were desiccated and returned to the laboratory for further processing.

Final processing of filters consisted of placing each filter in a 20-mL capacity borosilicate glass vial and digesting with 1-mL quaternary ammonium hydroxide tissue solubilizer for 4 to 6 hours. Ten milliliters of xylene based fluor were added; samples were agitated briefly on each of 3 to 4 successive days allowing the chemiluminescence reaction to reduce to background levels. Then, the samples were placed in a Packard Tri-Carb 300 °C liquid scintillation counter where radioisotope activity was counted.

To determine the relative efficiency of the C^{14} tagged $NaHCO_3$ solution used to inoculate samples, five replicate 10-μL aliquots were obtained from each of the vials of solution used. The relative efficiency was used as the total ampoule count in the formula used to calculate carbon fixation rates reported in Vollenweider (1969).

Phytoplankton Density

Samples for plankton enumeration and identification were collected using a No. 20 (80 μm mesh apertures) closing net. Replicate hauls were taken at 5-m intervals to 20 m, then from 20 m to the bottom of the lakes. Samples were preserved with formalin to a 10-percent by volume concentration. Taxonomic identification and enumeration were performed by examining three replicate Palmer counting cell (0.1-mL capacity) subsamples from each sample using a binocular microscope with 70 to 100 × magnification. Before 1984, a Bausch & Lomb, Inc., stereo-zoom binocular microscope was used for routine plankton counting and identification; after 1984, a Zeiss model 16 binocular microscope equipped with both phase contrast and differential interference contrast modes was used.

Each type of algal organism was identified to genus and recorded as an individual in its occurring state; i.e. colonial forms were counted as one for each colony, and single-celled forms were counted as one for each cell. Colonial forms were converted to density of cells per liter by multiplying each colony count by an average number of cells per colony. This average was determined by counting the cells in 100 colonies of each colonial species and dividing by 100. An average *Asterionella* colony at Twin Lakes ranged from 2 to 16 cells with an average of 10 cells per colony. An average *Dinobryon* colony ranged from 4 to 22 cells with an average of 14 cells per colony. Phytoplankton density is reported here as a depth-weighted average concentration in cells per liter because sampling intervals collected in the field were not always consistent. The total length of the water column sampled during net hauls was factored into the calculation of phytoplankton density in this fashion.

Statistical Analysis

Twin Lakes data were analyzed using SPSS-X (the Statistical Package for the Social Sciences, release 2.0) on the Bureau of Reclamation mainframe computer (a CYBER 875) and the personal computer version of SAS (statistical analysis program, release 6.03). Both parametric and nonparametric statistical procedures were used to analyze Twin Lakes data. Helsel (1987) reported that water quality data were typically non-normally distributed; and when applied to non-normally distributed data, the power of parametric procedures was low and may even be erroneous. For this reason, the Kolmogorov-Smirnov goodness-of-fit test was applied to chlorophyll *a*, carbon fixation rate, and phytoplankton density data for Twin Lakes to determine data normality.

The null hypothesis for the test was that the data are normally distributed and a two-tailed probability of 0.025 or less was required to reject the null hypothesis. The null hypothesis was not rejected for either the carbon fixation rate or chlorophyll *a* data with probabilities greater than 0.025 in both lakes for these parameters throughout both periods. These two data sets followed a normal distribution pattern and parametric statistical procedures were used in the analysis. The probabilities for phytoplankton data in both lakes throughout both periods were less than 0.025 and the null hypothesis was rejected. Nonparametric statistical procedures were used to analyze phytoplankton data sets. In statistical analyses of combined data sets, phytoplankton density was $\log_{10}$ transformed and parametric procedures were used.

RESULTS AND DISCUSSION

Chlorophyll *a* concentrations and seasonal and water column distribution patterns at Twin Lakes for the preoperational period (1977-81) were reported by Campbell and LaBounty (1985). During the period 1977-81, seasonal maximum chlorophyll *a* values were most likely to occur in both lakes in the fall following breakdown of thermal stratification when isothermal conditions allowed complete mixing of the water column. Summer peaks in chlorophyll *a* were observed in both lakes, but never in both lakes simultaneously. The distribution of chlorophyll *a* in the water column was characterized by subsurface maxima occurring at or near the thermocline in the summer and, occasionally, just below the ice cover during the early winter if snow cover on the ice was absent or scant.

Yearly average chlorophyll *a* concentrations in Twin Lakes are summarized in table 8-1. The range of monthly average chlorophyll *a* concentrations (during the preoperational period) in the lower lake was 1.1 to 7.7 mg/m^3 (table 8-1). Typical seasonal trends in chlorophyll *a* concentration, including the pattern of spring and fall maximum chlorophyll *a* values and the tendency for a decline in chlorophyll *a* to occur just before the spring and fall mixing periods, were evident in the preoperational period (fig. 8-1). In contrast, the prespring and fall mixing period declines in chlorophyll *a* concentration—so evident in the preoperational period—are much less evident in the postoperational period, especially in the fall. The typical September decline in chlorophyll *a* concentration just before fall turnover was not observed in the upper lake during the post-

Table 8-1.—Monthly average chlorophyll *a* concentration, mg/m³, in Twin Lakes, Colorado.

Month	1977	1978	1979	1980	1981	1982	1983	1984	1985
Lower Twin Lakes									
Jan.	2.3	3.3	2.4	5.4	3.4	3.0	4.3	3.2	4.5
Feb.	2.6	2.0	2.2	2.9	4.8	2.4	4.4	3.1	3.1
Mar.		1.9	2.9	1.8	3.7	2.6	2.9	2.9	5.0
Apr.		1.8	2.3	1.8	2.2	2.8	2.6	2.3	2.0
May	1.1	1.2	3.5	2.1	2.7	3.5	2.0	2.0	3.0
June	2.1	3.7	4.2	2.5	2.9	2.9	3.2	4.6	4.5
July	2.4	3.9	4.9	2.6	2.9	2.8	3.6	3.9	3.2
Aug.	2.0	4.4	7.7	2.3	2.0	3.6	3.4	3.8	3.2
Sep.	2.2		2.1	3.0	2.4	3.6	2.9	5.0	3.4
Oct.	3.0	4.3	5.3	2.9	4.7	5.1	5.2	6.1	
Nov.	3.5	5.0	5.3	4.6	4.4	6.6	5.7	5.5	
Dec.	4.8	4.7	5.2				5.3		
Annual mean	2.6	3.3	4.0	2.9	3.3	3.5	3.8	3.9	3.5
	1977-81 mean = 3.2					1982-85 mean = 3.7			
Upper Twin Lakes									
Jan.	3.6	3.9	2.7	2.0	4.5	2.8	4.9	2.2	3.2
Feb.	6.0	5.7	1.3	1.1	5.6	1.8	3.9	1.6	2.2
Mar.	3.8	2.6	0.8	0.9	3.5	2.1	3.8	1.6	2.1
Apr.		2.3	0.4	0.8	1.8	2.3	3.4	1.4	1.3
May	2.2	2.7	1.3	1.4	2.3	2.1	2.3		2.4
June	2.6		0.6	1.3	2.0	1.1	1.4	1.8	2.4
July	7.2	1.1	0.5	1.0	3.8	0.7	0.4	2.4	1.7
Aug.	2.6	3.1	2.3	2.8	6.0	1.4	3.0	2.8	2.4
Sep.	1.8	1.3		6.0	4.5	2.1	4.5	3.8	3.5
Oct.	2.5	3.5	2.7	6.8	4.1	3.0	5.5	5.6	
Nov.	2.3	3.6	3.1	7.3	4.1	4.5	3.1	6.2	
Dec.	4.3	5.0	2.1				2.8		
Annual mean	3.5	3.4	1.6	2.9	3.8	2.2	3.3	2.9	2.4
	1977-81 mean = 3.0					1982-85 mean = 2.7			

operational period. Major summer or winter peaks in chlorophyll *a* were not observed in either lake during the postoperational period (fig. 8-1).

The range of monthly average chlorophyll *a* concentrations in the upper lake (during the preoperational period) was 0.4 to 7.3 mg/m³ (table 8-1). The seasonal trends in the upper lake, displayed on figure 8-1, result from spring runoff. The late May to early July period in each year shows a distinct decline in chlorophyll *a* concentration. Summer peaks in chlorophyll *a* concentration were observed twice during the preoperational period (1977 and 1981). Both years were low runoff years (see ch. 2). Summer peaks in chlorophyll *a* concentration in the upper lake were not observed in postoperational years.

Another facet of chlorophyll *a* biomass is the pattern of distribution within the water column. In the preoperational period, thermal stratification resulted in a horizontal accumulation of chlorophyll *a* biomass along the thermocline (Campbell, 1985). In the postoperational period, the duration of a thermocline in either lake was reduced significantly—particularly in the lower lake. Thermal stratification in the lower lake averaged 88 days during the preoperational period and 39 days in the postoperational period. The upper lake averaged 73 and 62 days of thermal stratification in the pre- and postoperational periods, respectively. Figure 8-2 shows typical patterns of depth distribution in a preoperational year (1981) when thermal stratification was documented in both lakes and a postoperational year (1982), which was the first year of powerplant operation. The difference in depth distribution was similar to the difference in seasonal maxima and minima in the lake. Chlorophyll *a* biomass was more evenly distributed through the water column at an overall average density that is similar between the pre- and postoperational periods.

Although chlorophyll *a* biomass average concentrations in the lakes remained similar in both the pre- and postoperational periods, the two lakes diverged from one another during the postoperational period and their relationship was no longer statistically significant. Chlorophyll *a* average concentrations generally increased in the lower lake and decreased in the upper lake, causing the divergence and lack of correlation between the two lakes during the postoperational period.

Another way to examine chlorophyll *a* concentration is to calculate the biomass in kilograms per euphotic volume. The euphotic volume is calculated from the depth of the 1-percent light level on a monthly basis. The results of this calculation are presented on figure 8-3. A general upward trend in chlorophyll *a* biomass is evident in both lakes. This trend may be partly explained by the new water surface elevation of the lakes that may have resulted in a "new reservoir" phenomenon, particularly, in the upper lake where a large marshy area west of the lake was inundated.

Depth-distribution trends in both lakes may have been altered by pumped-storage powerplant operations. Depth distribution of chlorophyll *a* biomass was altered to a more even distribution throughout the water column, possibly due to shorter duration of thermal stratification in the lakes following pumped-storage powerplant operation. The trend toward evenly depth distributed chlorophyll *a* was more evident in the upper lake than in the lower lake, but the fact that thermoclines often did not coincide with point sampling depths for chlorophyll *a* was noted.

Carbon Fixation Rates

The results of carbon fixation rate experiments at Twin Lakes for the preoperational period were reported by LaBounty et al. (1985). Carbon fixation rates were highest in spring and fall, with maxima that usually preceded chlorophyll *a* peaks by about 4 weeks. Carbon fixation rates were lower in winter and summer, although a small winter peak in productivity, like chlorophyll *a*, could occur in the lakes if there was scant snow cover on the ice. Mean carbon fixation rates in the upper lake were 261.11 and 208.22 $\mu gC/m^3/h$ for the pre- and postoperational periods, respectively (table 8-2). Generally, lower lake carbon fixation rates were higher than the upper lake with a preoperational period mean of 473.84 $\mu gC/m^3/h$ and a postoperational period mean of 350.71 $\mu gC/m^3/h$ (table 8-2). In both lakes, the preoperational period mean was greater than the postoperational period mean.

The maximum and minimum ranges for monthly carbon fixation rates were 1419 to 6 $\mu gC/m^3/h$ measured in the lower lake and 591 to 16 $\mu gC/m^3/h$ meas-

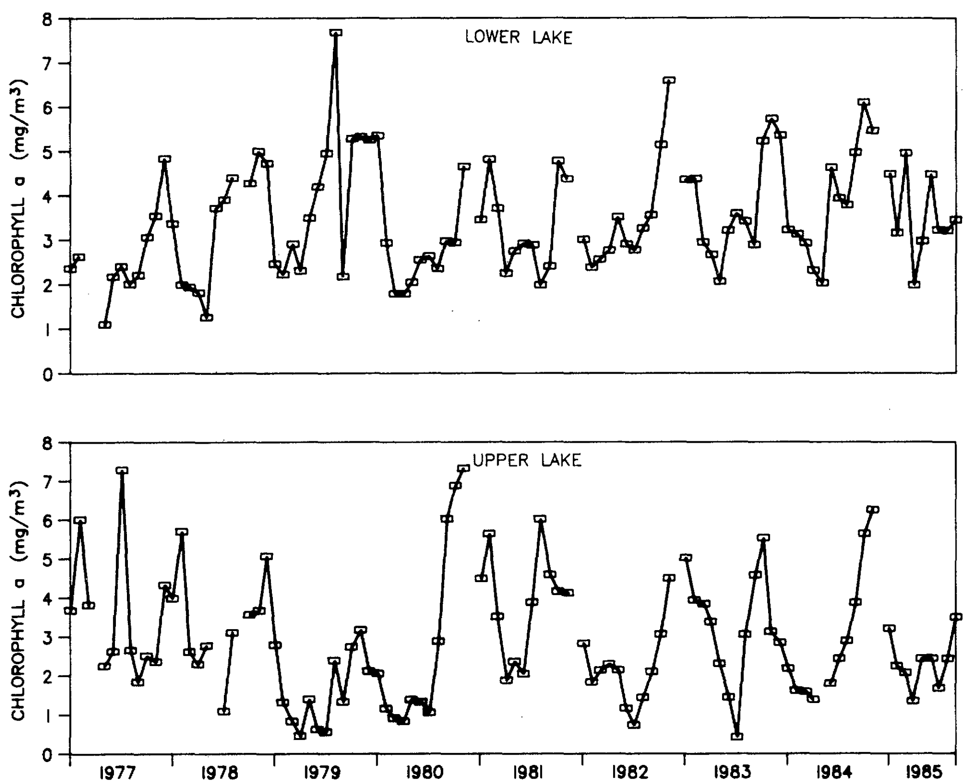


FIGURE 8-1.—*Monthly average chlorophyll* a *concentration in Twin Lakes, Colorado.*

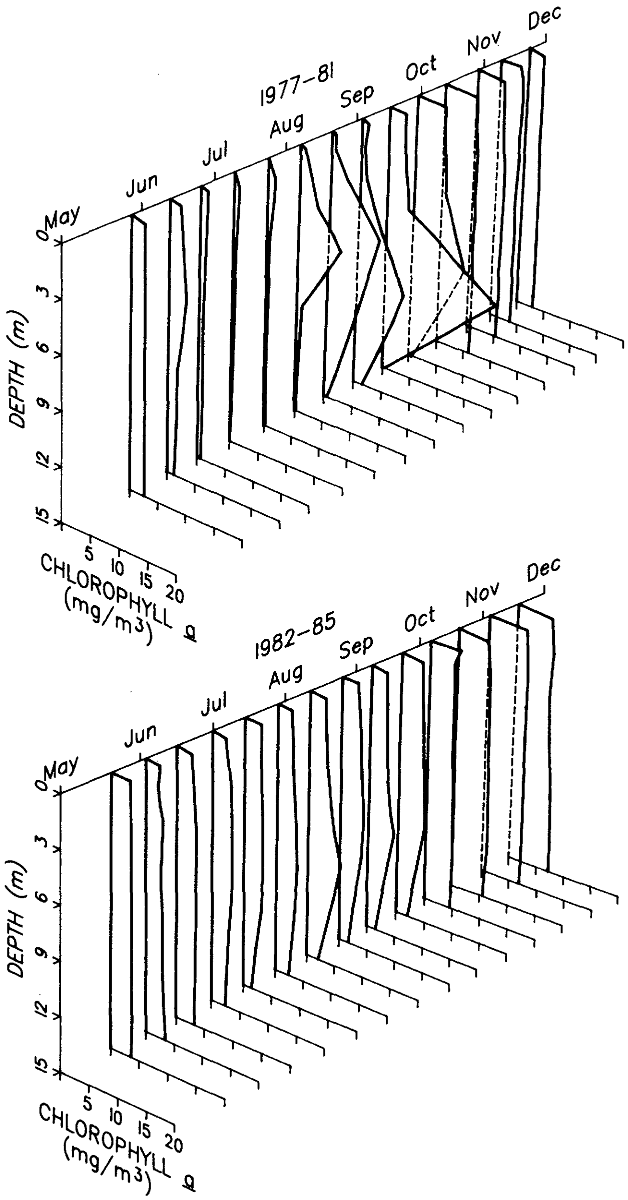


FIGURE 8-2.—*Representative depth profiles of chlorophyll a concentration for pre- and postoperational periods in Twin Lakes, Colorado.*

ured in the upper lake during the 1977-85 period (table 8-2). The dependency of carbon fixation rates on light penetration in both lakes was evident in the higher values during each May-November ice-free season (fig. 8-4). Once ice-cover was established on the lakes, the carbon fixation rates decreased sharply. This condition was more persistent in the upper lake as spring runoff and the associated turbidity of runoff water usually depressed carbon fixation rates until July or August.

During the preoperational period, carbon fixation rates in the lower lake were characterized by a bimodal distribution, with highest rates occurring just after spring and fall turnover. The mean maximum monthly carbon fixation rate in the lower lake during the preoperational period was 1418.83 $\mu gC/m^3/h$ and the mean minimum monthly carbon fixation rate was 52.77 $\mu gC/m^3/h$ (table 8-2).

The postoperational period in the lower lake was also characterized by spring and fall peaks in carbon fixation rate; however, beginning in 1983, the magnitude of carbon fixation rate peaks were reduced by approximately 45 percent from the mean spring maxima observed from 1977 to 1982 (fig. 8-4). During the last 3 years of the postoperational period (1983-85), the lower lake was buffered from runoff by the upper lake and with its larger volume—less susceptible to flushing during runoff. Therefore, the spring and fall bimodal distribution maxima in the lower lake were not as evident in these postoperational period years. The mean maximum monthly carbon fixation rate in the lower lake during the postoperational period was 847.6 $\mu gC/m^3/h$, and the mean minimum monthly carbon fixation rate was 6.16 $\mu gC/m^3/h$ (table 8-2). The yearly means reflect the pre- versus postoperational period declines in the lower lake carbon fixation rates (table 8-2).

Throughout both periods, the upper lake exhibited a skewed normal distribution (fig. 8-4). The mean maximum monthly carbon fixation rate in the upper lake during the preoperational period was 591 $\mu gC/m^3/h$ and the mean minimum monthly carbon fixation rate was 16 $\mu gC/m^3/h$ (table 8-2). The mean maximum carbon fixation rated in the upper lake was 503 $\mu gC/m^3/h$ and the mean minimum monthly carbon fixation rate was 25 $\mu gC/m^3/h$ (table 8-2). Discernable differences in carbon fixation rate seasonal patterns between periods can not be determined from the data displayed on figure 8-4.

During the preoperational period, the lakes differed in magnitude of carbon fixation rates (fig. 8-4). During the postoperational period, the lakes were more similar in magnitude mainly because carbon fixation rates declined drastically in the lower lake beginning in 1982 (fig. 8-4). The reasons for this decline may be related to changes in inflow pattern to the lakes, the greater impact of powerplant generation on the lower lake, and the fact that initial powerplant operation occurred before the lake surface elevation increased.

Phytoplankton

Phytoplankton density, species composition, and seasonality of phytoplankton at Twin Lakes were reported in annual progress reports (LaBounty et al., 1980, 1984; LaBounty and Sartoris, 1981, 1982).

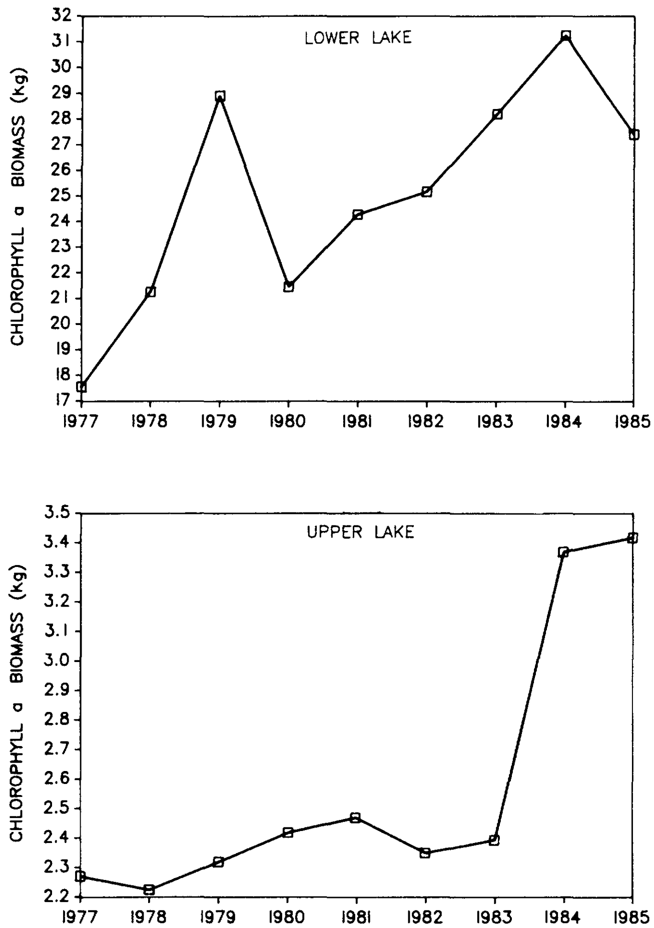


FIGURE 8-3.—*Annual mean chlorophyll* a *biomass in the entire euphotic zone in Twin Lakes, Colorado.*

Typically, the phytoplankton population at Twin Lakes is characterized by a spring/diatom, fall/chrysophycean species succession, with greatest abundance occurring in the fall—following turnover. Summer maximum phytoplankton densities were observed in the lower lake in 1979 and 1984 and in the upper lake in 1977 and 1981, but the lakes did not experience simultaneous summer maxima. A list of phytoplankton, by genus, observed at Twin Lakes is summarized in appendix A. The dominant algal species were the diatoms:

- *Asterionella formosa* Hassall,
- *Synedra amphicephala* Kuetz,
- *S. delicatissima* W. Sm.,
- *S. radians* Kuetz,
- S. *ulna* var. *danica* (Kuetz), and
- Chrysophycean algae *Dinobryon cylindricum* Imhof.

Reynolds (1984) categorizes the dominant phytoplankton species *Asterionella*, *Synedra*, and *Dinobryon* as representative of mesotrophic freshwater ecosystems and the vernal/diatom, summer/chrysophycean or B-->E species succession represents well-mixed, nutrient-rich conditions followed by stratification and nutrient depletion. However, Twin Lakes was classified as oligotrophic based on a range of production values. LaBounty et al. (1985) classified Twin Lakes in the ultraoligotrophic to oligotrophic category based on ranges of less than 50 mgC/m^2/d for ultraoligotrophic to 50 to 300 mgC/m^2/d for oligotrophic systems reported by Likens (1975). LaBounty et al. (1984) classified Twin Lakes in the oligotrophic category based on ranges of 0.3 to 3.0 mg/m^3 chlorophyll *a* reported by Likens (1975). Dominant phytoplankton in Twin Lakes were representative of oligotrophic systems according to the classification scheme developed by Likens (1975). Regardless of differences among authors attempting to classify and categorize ecosystems, Twin Lakes is a cold, high altitude lake surrounded by mountains and limited by severe winter conditions and a short growing season with relatively low production values.

Mean annual cell densities of total depth-weighted phytoplankton are summarized in table 8-3. Pre- and postoperational period mean total phytoplankton densities were similar in the lower lake, 24 063 and 25 227 cells per L, respectively. However, if 1984 peak phytoplankton densities were removed from the mean, a significant difference between pre- and postoperational periods would result (table 8-3, fig. 8-5). By comparison, the upper lake showed a significant increase in total phytoplankton densities during the postoperational period, with values of 9358 and 11 549 cells per liter, respectively. Phytoplankton means, for 2 years of the postoperational period of record, were near the previous maximum yearly mean observed in the upper lake during the drought year 1981 (table 8-3, fig. 8-5). This trend may have continued in 1985, but the study period terminated at the end of September before the fall maxima occurred.

Generally, the lower lake has higher mean densities of total phytoplankton than the upper lake (table 8-3). However, in all years since 1981, except 1984, the lakes have had similar yearly means. In 1984, the lower lake experienced a tremendous peak in total phytoplankton density (fig. 8-5). Both lakes experienced a marked decrease in phytoplankton density in 1982, the first year of powerplant operation.

The maximum monthly mean total phytoplankton density in the lower lake was 402 257 and the minimum was 52 cells/L (table 8-3). The maximum monthly mean total phytoplankton density in the upper lake was 92 149 and the minimum was 22 cells/L (table 8-3). Minimum phytoplankton den-

Table 8-2.—Monthly average carbon fixation rate, μgC/m³/h, in Twin Lakes, Colorado.

Month	1977	1978	1979	1980	1981	1982	1983	1984	1985
					Lower Twin Lakes				
Jan.		125	108	96	358	255	275	101	69
Feb.	191	94	53	147	286	235	89	66	120
Mar.		168	432	173	86	228	50	37	6
Apr.		313	382	142	305	84	59	149	128
May		587	1419	567	879	764	237		668
June	345		995	762	554	739	375	423	441
July	334		641	774	373	625	246	372	612
Aug.	460	447	305	392	340	522	209	304	461
Sept.	542		190	565	590	848	391	352	777
Oct.	796	1328	644	841	730	822	503	579	
Nov.	478	593	989	979	638	608	495	318	
Dec.	202	81	346				88		
Annual mean	419	415	542	494	467	521	251	270	365
	1977-81 mean = 474				1982-85 mean = 351				
					Upper Twin Lakes				
Jan.		167	76	16	261	291	139	60	88
Feb.	347	104	47	48	305	156	123	67	84
Mar.		194	47	27	79	236	109	49	33
Apr.		305	68	39	193	139	72		154
May		326	43	176	591	412	166		401
June	499		68	102	215	263	107	179	297
July	567		240	331	399	65	25	211	296
Aug.	459	249	330	186	435	338	106	341	453
Sept.	332		104	322	364	339	254	347	503
Oct.	577	364	469	377	427	249	238	434	
Nov.	536	329	366	365	448	181	197	277	
Dec.	185	102	110				57		
Annual mean	438	227	164	181	338	243	133	218	257
	1977-81 mean = 261				1982-85 mean = 208				

sities tended to occur in the late spring, when ice and snow cover was usually heaviest and in early summer, as a result of increased flushing and turbidity conditions associated with runoff (fig. 8-5). Either lake could experience a spring peak in phytoplankton density when the phytoplankton assemblage was dominated by diatoms. Summer and fall peaks in phytoplankton density were usually dominated by chrysophycean algal species such as *Dinobryon.*

Monthly trends in phytoplankton density in the lower lake tend to show bimodal or even trimodal distribution peaks during the year. Spring peaks in phytoplankton density occurred in 1979, 1981, and 1984 (fig. 8-5). Summer peaks occurred in 1978, '79, '80, '81, and '84. Peaks in phytoplankton density, following fall mixing periods, were observed throughout the study. Monthly trends in phytoplankton density in the upper lake show greatest abundance occurring in the fall. The phytoplankton assemblage in the upper lake, during the fall maxima, was dominated by the chrysophycean algae *Dinobryon* sp. The smaller upper lake basin was rapidly flushed during spring runoff and resulted in lower phytoplankton densities (fig. 8-5). The exception to this general trend in phytoplankton densities in the upper lake occurred in 1981. As noted in chapter 2, 1981 was a low inflow year. Hydraulic retention time and duration of stratification were longer in 1981, resulting in maximal peaks in phytoplankton densities in the upper lake.

The distribution of phytoplankton among the dominant algal species is shown on figure 8-6. The data displayed in this figure show the spring/diatom fall/chrysophycean species succession that was typical of both lakes during the preoperational period. This species succession was more marked in the lower lake because spring runoff tended to suppress the development of a spring phytoplankton population in the upper lake in most years. The change in postoperational phytoplankton assemblages was clearly evident. A fall shift to a chrysophycean dominated phytoplankton assemblage in the preoperational period changed to a nearly

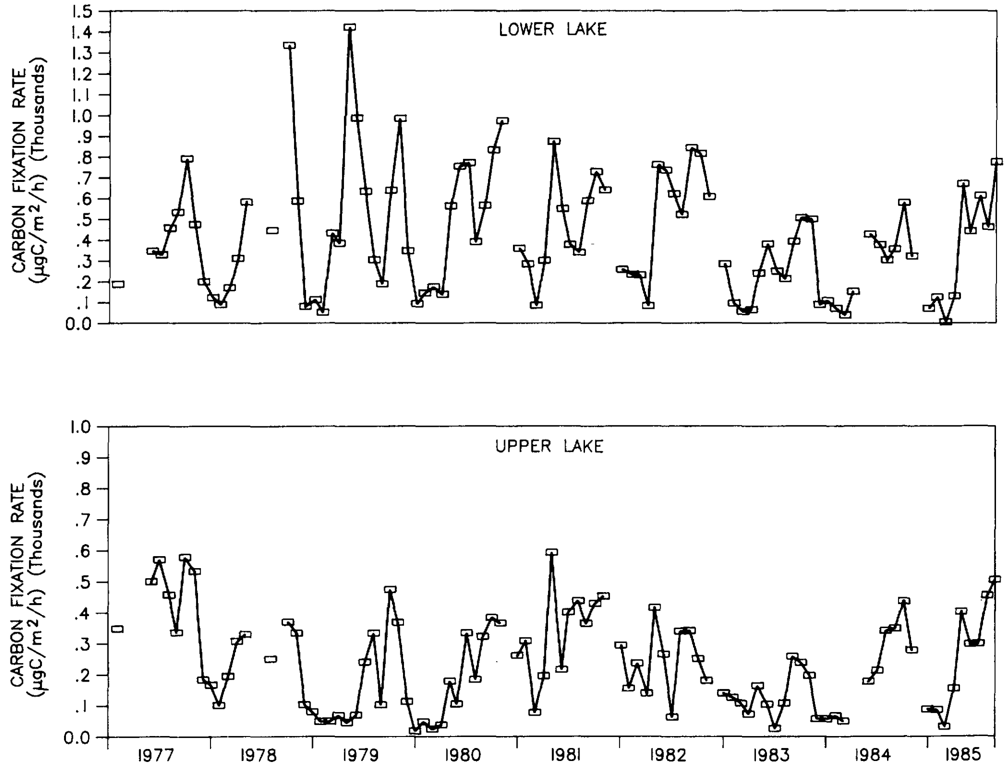


FIGURE 8-4.—*Monthly average carbon fixation rates in Twin Lakes, Colorado.*

constant diatom dominated assemblage during the postoperational period. In the upper lake, the fall shift from diatoms to chrysophycean algae was replaced by an assemblage dominated by chrysophyceans during the postoperational period.

Reynolds (1984) reported that changes from spring diatom assemblages to mid-summer chrysophycean assemblages were directly related to recurrent fluctuations in physical stability and relative mixing. Induced mixing of the euphotic zone—resulting from powerplant discharge into the lower lake and the change in the Lake Creek inflow pattern—with inflow entering and spreading out into a wide, shallow, delta area in the upper lake probably caused the change in species succession observed at Twin Lakes in the postoperational period.

The phytoplankton assemblage also experienced a fairly significant change between pre- and postoperational periods. Several different species of algae new to the Twin Lakes ecosystem were observed throughout both periods. Figure 8-7 compares two species of *Dinobryon* collected in 1984 from the lower lake shortly before *Dinobryon* sp. disappeared from the assemblage almost completely. One species, identified as *Dinobryon bavaricum* Imhof (Ahlstrom, 1937) was imported from Turquoise Lake to Mt. Elbert Forebay and then through the powerplant into Lower Twin Lakes. The normally occurring spring/diatom fall/chrysophycean algal species succession was disrupted when the powerplant began regular operations in 1984. As long as the powerplant continues to cause induced mixing of the euphotic zone in the lower lake, continuous diatom dominance of the phytoplankton assemblage may prevail.

The Twin Lakes phytoplankton assemblage includes a considerable subnet-size (<80 µm) component that was quantified beginning in 1983 (Campbell, 1987). Results of size fractionation experiments indicate that 80 to 90 percent of the chlorophyll *a* containing cells are in size classes smaller than the apertures of the closing net used routinely to collect plankton samples at Twin Lakes. The presence of this subnet-size component quantified in both chlorophyll *a* and carbon fixation estimation procedures, was not included in Twin Lakes phytoplankton density data. Therefore, a certain element of uncertainty exists in interpretation of phytoplankton population changes between pre- and postoperational periods.

Phytoplankton density seemed to remain at about the same level between pre- and postoperational periods in the lower lake. In 3 of the 4 postoperational years of record (1982, '83, and '85), mean phyto-

Table 8-3.—Monthly average depth-weighted total phytoplankton density, cells/L, in Twin Lakes, Colorado.

Month	1977	1978	1979	1980	1981	1982	1983	1984	1985
				Lower Twin Lakes					
Jan.	17 021	8 770	9 218	21 235	11 272	4 812	4 319	23 032	7 507
Feb.	41 236	8 149	7 938	392	12 987	1 727	3 121	14 784	1 009
Mar.		7 187	17 873	87	43 434	1 663	2 530	27 844	96
Apr.		6 013	34 246	52	92 414	2 713	1 913	8 594	211
May	1 832	1 705	66 392	1 087	8 209	14 726	6 684	11 839	9 064
June			41 224	1 773	7 139	9 059	10 886	19 557	10 635
July	3 756		85 239	8 230	4 247	2 323	12 329	402 267	33 193
Aug.	2 645	29 317	53 132	82 706	2 568	1 312	3 378	91 364	21 653
Sept.	9 253		21 790	15 171	22 376	7 490	11 426	82 336	10 520
Oct.	20 183	54 481	12 331	14 040	73 772	9 692	51 154	13 053	
Nov.	41 687	73 478	52 611	16 207	23 691	15 979	56 578	33 207	
Dec.	25 804	29 356	27 437				49 337		
Annual mean	18 157	24 273	35 785	14 634	27 464	6 500	17 805	66 171	10 432
		1977-81 mean = 24 063			1982-85 mean = 25 227				
				Upper Twin Lakes					
Jan.	6 233	7 125	10 288	723	7 470	1 079	4 500	14 541	5 513
Feb.	8 832	4 944	1 862	22	11 866	1 080	1 095	10 018	3 127
Mar.		6 642	271	30	23 163	1 563	2 153	6 307	408
Apr.	1 362	5 428	528	63	26 596	1 221	2 397	2 348	317
May	2 734	3 054	791	82	18 523	1 755	2 006		5 586
June			405	107	14 566	444	1 382	5 421	5 376
July	6 585		884	1 446	28 876	605	156	33 386	13 552
Aug.	4 103	2 428	7 737	1 046	16 719	87	228	47 349	19 670
Sept.	562		16 134	4 060	38 084	1 931	12 586	53 472	10 505
Oct.	6 929	11 987	27 059	5 967	19 163	3 797	71 280	13 358	
Nov.	17 509	25 813	24 090	6 471	12 125	8 572	92 149	7 484	
Dec.	10 362	29 330	5 456				22 432		
Annual mean	6 521	10 750	7 959	1 820	19 741	2 012	17 697	19 368	7 117
		1977-81 mean = 9 358			1982-85 mean = 11 549				

plankton density was equal to or less than the lowest annual mean density observed during the preoperational period. In July 1984, a tremendous *Dinobryon* sp. bloom occurred in the lower lake. In the 0 to 5 m depth stratum, *Dinobryon* sp. concentration was 1754310 cells per liter. The water surface elevation of the lakes was increased in the late fall of 1983 and the upsurge in algal populations may have resulted from increased nutrient input to the lakes as shorelines were inundated. If the bloom period in 1984 is eliminated from the postoperational mean, phytoplankton densities in the lower lake would be considerably less than the preoperational period mean density.

Phytoplankton density seemed to increase in the upper lake in 1984 and 1985 following the inundation of the delta area. The increased volume of water in the lakes—located in a broad shallow delta area in the upper lake—that created a more favorable habitat for production of phytoplankton may have been responsible. This shallow delta may have initially functioned as a nutrient source. The original marshy area was covered with willows (*Salix* spp.) and other water tolerant shrubs and many low-lying areas had intermittent standing water pools during spring runoff when water levels were generally high. When the water surface of the lakes was raised in late 1983, many acres of vegetation and leaf litter were inundated. In August 1984, the delta area was sampled in several locations. High plankton populations, high pH values, considerably warmer water temperatures, and evidence of gas bubbles issuing from rotting organic debris on the bottom were observed. Under these circumstances, it is likely that water soluble organics were readily leached from the vegetation and were immediately cycled through the primary producers prevalent in the area. Thus, a natural cycle of decomposition in the delta area would gradually result in increased organic loading to the main basin of the upper lake.

The apparent decrease in phytoplankton density observed in 1985 (fig. 8-5) may have been a function of a partial year of data where the fall maximum phytoplankton density was absent from the data base.

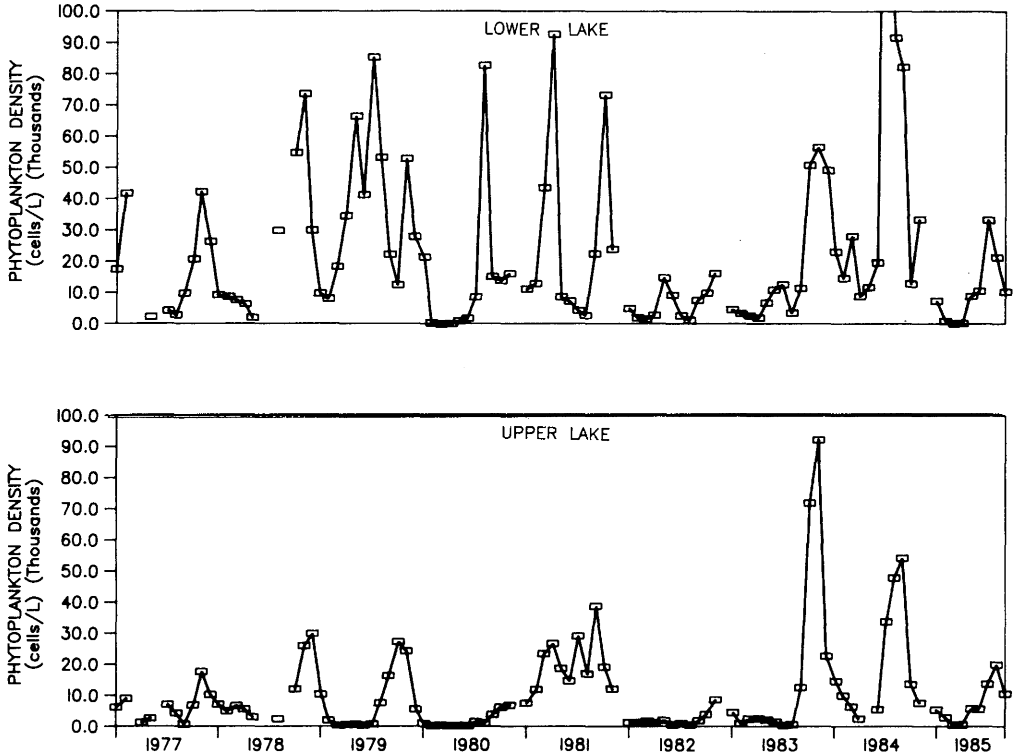


FIGURE 8-5.—*Monthly average depth-weighted phytoplankton densities in Twin Lakes, Colorado.*

Statistical Analysis Results

The complete Twin Lakes data set for statistical analysis consisted of 68 variables; however, that data set was reduced to 35 variables having less than or equal to 33 percent missing values. Table 8-4 summarizes the abiotic and biotic variables used for statistical analyses of primary production. The data set also was limited to observed values for any variable and no compensation for detection limit changes throughout both periods (as discussed in ch. 5) was made. Representative variables from each category of limnological data, except trace metals, were included in the reduced data set (table 8-4). Of the trace metals, iron was the only variable which met the missing values (≤33%) criteria. Iron concentration did not show any significant relationship to primary production in any of the analyses and was eliminated from the reduced data set. The null hypothesis for all statistical analyses was that pre- and postoperational periods were the same. The significance level to reject the null hypothesis was 0.05.

Primary production, a biotic component of an ecosystem, is influenced or determined by abiotic components. Straskraba (1980) stated that ". . . quantitative interpretation of biotic interrelations in freshwater ecosystems, . . . depends on an understanding of the changes in the physical matrix in which they occur." Brylinsky (1980) also recognized the importance of abiotic and biotic interrelationships and reported a general scheme for statistical analyses of productivity in 93 lakes and reservoirs. The abiotic and biotic variables used for statistical analyses were categorized as driving, site, and state variables.

Driving variables were those which included factors having an effect on a system, but which were not, in turn, affected by the system. Driving variable examples are:

- visible incident radiation
- day length range
- air temperature
- precipitation

Site variables were those specific for a site and did not vary with a site. Site variable examples are:

- latitude
- altitude
- mean depth
- maximum depth
- surface area
- volume
- drainage area

State variables were those affected by the system and usually those requiring prediction. State

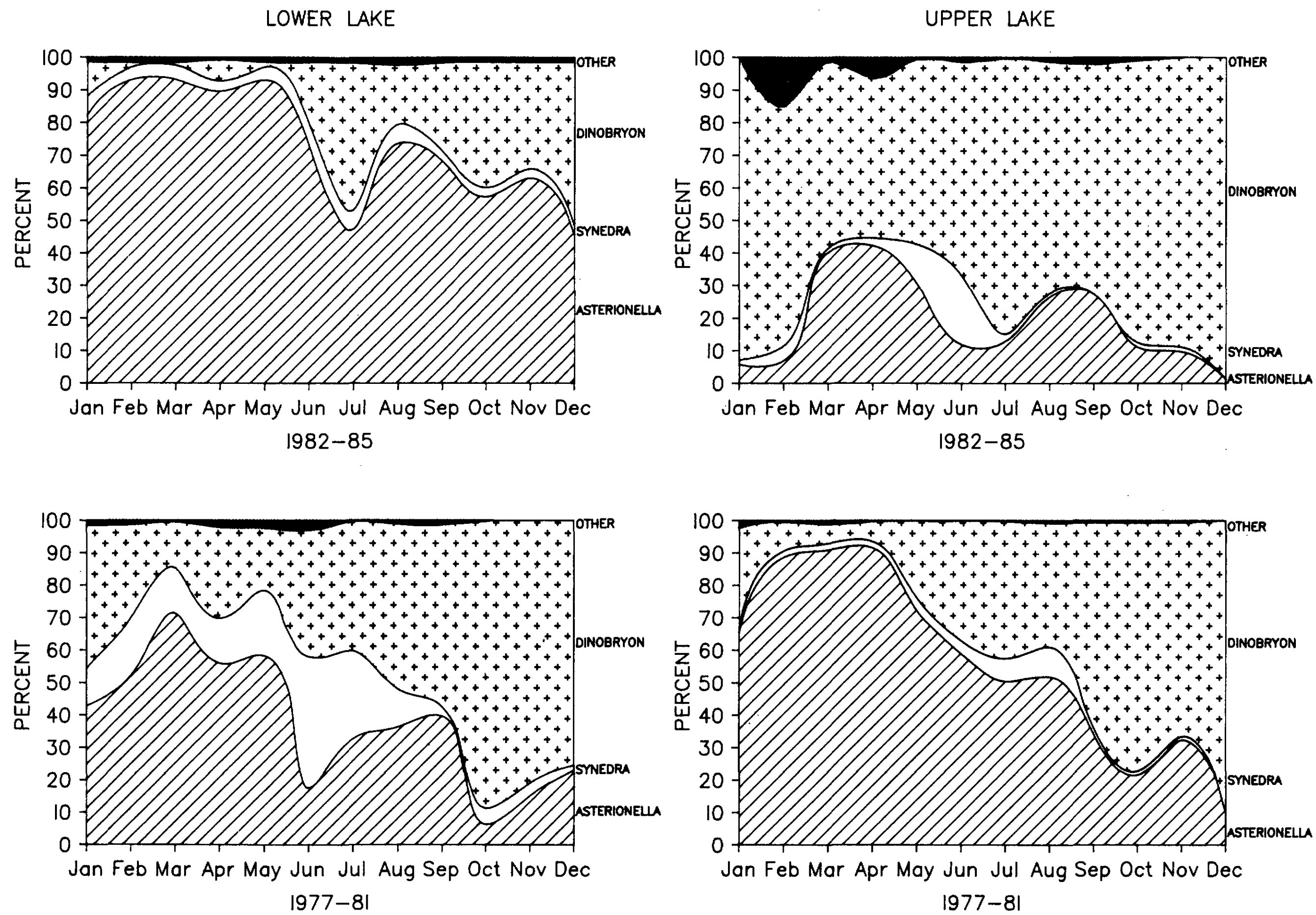


FIGURE 8-6.—*Phytoplankton assemblage percent population distribution contrasting composites of pre- and postoperational periods in Twin Lakes, Colorado.*

variable examples are:

- pH
- hydraulic retention time
- stratification duration
- water temperature
- euphotic depth
- conductivity
- bicarbonate alkalinity
- nitrate nitrogen
- dissolved oxygen
- chlorophyll *a*
- phytoplankton biomass
- zooplankton biomass
- photosynthetic efficiency

Brylinsky's method of categorizing a large number of variables into driving, site, and state variables was modified and applied to the Twin Lakes data set. Because Brylinsky's analysis used mean values from a single growing season for many lakes and reservoirs, and the Twin Lakes data set compared a single ecosystem through time, some components such as:

- latitude
- altitude
- day-length range
- visible incident radiation
- precipitation

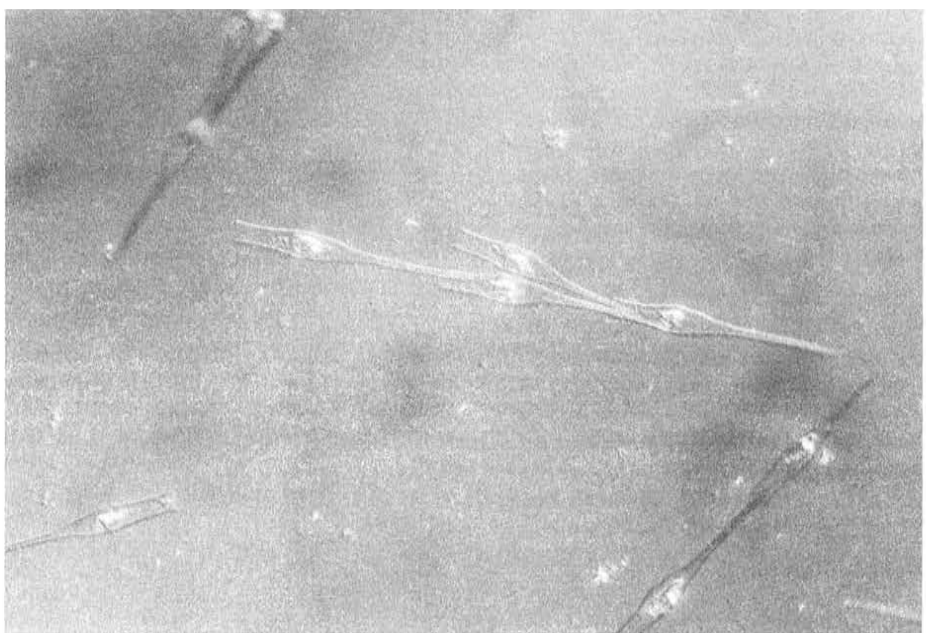
either were not available or had no application for the Twin Lakes statistical analyses.

Twin Lakes' site variables included the surface area and its volume—mentioned by Brylinsky—and inflow sources rather than drainage area, specific drainage area, or the ratio of drainage area to volume (table 8-4). Twin Lakes state variables were further subdivided into:

- physical
- chemical
- biological characteristics

Mean values for pre- and postoperational periods for each variable in the Twin Lakes data set are

Dinobryon bavaricum Imhof



Dinobryon cylindricum Imhof

FIGURE 8-7.—*Photomicrographs × 400 (magnification) of two species of* Dinobryon *collected in Twin Lakes, Colorado, 1984.*

summarized in table 8-4. Based on Wilcoxon Rank Sum Test, nearly all of the variables in the site parameters category, except discharge volumes through the Twin Lakes Tunnel and Lake Creek, were significantly greater ($P=\leq 0.05$) in the postoperational period in both lakes.

In the physical characteristics category, stratification duration was significantly less ($P=\leq 0.05$) in the postoperational period in both lakes, and light extinction coefficient was significantly greater ($P=\leq 0.05$). Retention time and euphotic depth in the lower lake were significantly less ($P=\leq 0.001$) in the postoperational period.

In the chemical characteristics category, average bicarbonate alkalinity and bottom E_7 were significantly less ($P=\leq 0.05$) in the postoperational period

Table 8-4.—Summary of abiotic and biotic parameters used for statistical analyses of primary production in Twin Lakes, Colorado.

	Lower Twin Lakes		Upper Twin Lakes	
	1977-81	1982-85	1977-81	1982-85
Site Parameters				
Lake surface area, hectares	1589.3	†1699.8 >	499.8	678.7 >
Lake volume, $m^3 \times 10^6$	89.3	104.4 >	31.7	39.8 >
Twin Lakes Tunnel volume, m^3/m	43.0	44.4	*	
Lake Creek volume, m^3/m	86.9	129.5		
Mt. Elbert Conduit volume, m^3/m	9.0	138.0 >		
Powerplant generation volume, m^3/m	14.5	229.1 >		
Powerplant pumping volume, m^3/m	9.1	102.0 >		
Total inflow volume, m^3/m	138.9	311.9 >		
Total outflow volume, m^3/m	124.6	287.8 >		
Physical Characteristics				
Average temperature, °C	7.0	6.2	6.1	5.5
Maximum temperature, °C	9.1	8.5	8.7	8.2
Minimum temperature, °C	5.1	4.5	4.2	3.7
Retention time, days	229.0	‡131.0 <	88.0	86.0
Stratification duration, days	88.6	39.0 <	73.6	61.8 <
Light extinction coefficient, μ^{-1}	0.44	0.55 >	0.50	0.55 >
Euphotic depth, meters	10.4	8.8 <	9.5	8.8
Chemical Characteristics				
Average dissolved oxygen, mg/L	7.9	7.5	8.1	7.7
Maximum dissolved oxygen, mg/L	9.0	9.1	9.1	9.1
Minimum dissolved oxygen, mg/L	5.4	5.4	5.5	5.1
Average pH	7.5	7.3 <	7.4	7.3
Maximum pH	7.8	7.6 <	7.7	7.6
Minimum pH	7.0	6.9	6.9	6.9
Average conductivity, $\mu S/cm^2$	71.7	57.4 <	77.3	75.5
Maximum conductivity, $\mu S/cm^2$	75.6	61.6 <	87.0	84.6
Minimum conductivity, $\mu S/cm^2$	63.4	53.6 <	70.1	66.4
Bottom E_7, mV	472.5	376.6 <	463.7	378.2 <
Bicarbonate alkalinity, mg/L	20.1	15.9 <	20.3	18.8 <
Nitrate nitrogen, mg/L	35.9	77.5 >	66.7	119.8 >
Biological Characteristics				
Average chlorophyll *a*, mg/m^3	3.2	3.7 >	3.0	2.7
Average carbon fixation rate, $\mu gC/m^3/h$	473.8	350.7	262.1	208.2
Ratio, μgC/mg chlorophyll *a*	0.13	0.09 <	0.09	0.08
Depth-weighted zooplankton, No./L	33.4	20.7 <	14.8	13.6
Log_{10} phytoplankton	3.5	3.8	3.1	3.3
Depth-weighted phytoplankton, cells/L	24 507.4	25 742.2	9332.7	11 719.8
Total benthos, No./m^2	2509.8	2447.2	2486.8	3149.9

* Flow volumes are the same for both lakes.

† > Significantly greater ($P \leq 0.05$) in postoperational period based on Wilcoxon Rank Sum Test.

‡ < Significantly less ($P \leq 0.05$) in postoperational period based on Wilcoxon Rank Sum Test.

in both lakes and concentration of nitrate-nitrogen was significantly greater ($P=\leq0.05$). Conducutivity and pH were significantly less ($P=\leq0.05$) in the lower lake in the postoperational period.

In the biological category, average chlorophyll *a* was significantly greater (P=0.0317) in the lower lake, and depth-weighted zooplankton density was significantly less (P=0.003) in the postoperational period. Although not statistically significant at the 0.05 probability level, carbon fixation rates in both lakes showed a generally decreasing trend throughout the postoperational period and the ratio of carbon per unit chlorophyll *a* in the lower lake was significantly less (P=0.0039) in the postoperational period.

Pre- and postoperational period comparisons between the two lakes were also made using the Wilcoxon Rank Sum Test. Although the two lakes were essentially two separate basins in series in the preoperational period, we expected them to become more alike (less separate) in the postoperational period. However, the lakes retained some distinct differences and the results of the Wilcoxon Rank Sum Test provide support for this observation. Site parameters were not included in this analysis because the physical size of the lakes was always different and flow data are the same for both lakes.

In the physical characteristics category, hydraulic retention time in the lower lake was significantly greater (P=0.0156) than in the upper lake during the preoperational period. However, retention time was not significantly different (P=0.7962) in the two lakes in the postoperational period. Stratification duration in the lower lake was significantly greater (P=0.0001) than in the upper lake during the preoperational period and significantly less (P=0.0010) than in the upper lake during the postoperational period. Light extinction coefficients in the lower lake were significantly less (P=0.0367) than in the upper lake during the preoperational period. However, light extinction coefficients were not significantly different (P=0.6860) in the lakes during the postoperational period. Euphotic depth in the lower lake was significantly greater (P=0.0045) than in the upper lake during the preoperational period. However, euphotic depth was not significantly different (P=0.8022) in the two lakes during the postoperational period.

In the chemical characteristics category, average and maximum monthly pH values in the lower lake were significantly greater (P=0.0052 and 0.0136, respectively) than in the upper lake during the preoperational period, but were not significantly different (P=0.1962 and 0.8432, respectively) in the two lakes during the postoperational period. Average bicarbonate alkalinity was not significantly different (P=0.4534) in the lakes during the preoperational period, but was significantly less (P=0.0001) in the lower lake than in the upper lake during the postoperational period.

In the biological characteristics category, average chlorophyll *a* concentration was not significantly different (P=0.2560) in the lakes during the preoperational period, but was significantly greater (P=0.0002) in the lower lake than in the upper lake during the postoperational period. The ratio of carbon per unit chlorophyll *a* in the lower lake was significantly greater (P=0.0003) than in the upper lake during the preoperational period. However, in the postoperational period, the ratio of carbon per unit chlorophyll *a* in the two lakes was not significantly different (P=0.5941).

Table 8-5 summarizes results of Pearson correlation analyses on the Twin Lakes data set. Both seasonal and operational effects were evident in correlations among abiotic and biotic variables. The lakes' operation pattern was discussed in detail in chapter 2. Generally, the lakes' surface area and volume increased following runoff and remained maximal through the growing season. The lakes were drawn down in late winter and early spring and this pattern followed the seasonal trends observed at the lakes in primary production parameters. The effect of annual runoff was a decrease in primary production parameters in response to increased flushing and turbidity during peak runoff. Seasonal effects were determined using the Kruskal-Wallis one-way ANOVA test to quantify seasonality in primary production parameters. Chlorophyll *a*, carbon fixation rate, and phytoplankton density were all skewed toward the fall season throughout the study period in both lakes ($\chi^2 = \leq0.05$). Graphically, all biotic variables show this same skewed distribution pattern (figs. 8-1, 8-4, 8-5) and relationships among the biotic parameters were common. Correlations among primary production parameters and flow variables, lake surface area, lake volume, and other parameters such as temperature, DO, pH, as well as biological variables—which also exhibit a strong seasonal trend—were all common throughout both periods.

Lake surface area and lake volume in the site parameters category showed strong positive correlations to primary production variables during the postoperational period, particularly in the upper lake (table 8-5). For the upper lake, both lake surface area and volume increased significantly during the postoperational period. As noted in chapter 5, a "new-reservoir" phenomenon resulting from increased nutrient input following inundation of new flooded shorelines may have occurred in the

Table 8-5.—Summary of correlation analyses of abiotic and biotic variables related to primary production in Twin Lakes, Colorado.

Preoperational period (1977-81)	Lower Twin Lakes			Upper Twin Lakes		
	Chlorophyll *a*	Carbon fixation rate	Log_{10} phytoplankton	Chlorophyll *a*	Carbon fixation rate	Log_{10} phytoplankton
Site Parameters						
Lake surface area, hectares	+					
Lake volume, $m^3 \times 10^6$	+					
Twin Lakes Tunnel volume, m^3/m		+ +		- -		
Lake Creek volume, m^3/m		+ +		- -		-
Mt. Elbert Conduit volume, m^3/m						
Powerplant generation volume, m^3/m						
Powerplant pumping volume, m^3/m						
Total inflow volume, m^3/m		+ +		- -		
Total outflow volume, m^3/m		+ +		- -		
Physical Characteristics						
Average temperature, °C		+ +			+ +	
Maximum temperature, °C		+			+ +	
Minimum temperature, °C		+ +			+ +	+ +
Retention time, days		- -			-	
Stratification duration, days						
Light extinction coefficient, μ^{-1}						
Euphotic depth, meters						
Chemical Characteristics						
Average dissolved oxygen, mg/L		-			- -	
Maximum dissolved oxygen, mg/L		- -			- -	
Minimum dissolved oxygen, mg/L	+ +					-
Average pH					+	
Maximum pH				+ +	+ +	+
Minimum pH		+ +				
Average conductivity, $\mu S/cm^2$						
Maximum conductivity, $\mu S/cm^2$					- -	
Minimum conductivity, $\mu S/cm^2$						
Bottom E_7, mV					+	
Bicarbonate alkalinity, mg/L	-					
Nitrate nitrogen, mg/L						
Biological Characteristics						
Average chlorophyll *a*, mg/m^3			++		+ +	+ +
Average carbon fixation rate, $\mu gC/m^3/h$			+	+ +		+ +
Ratio, μgC/mg chlorophyll *a*						
Depth-weighted zooplankton, No./L				+ +	+ +	+ +
Log_{10} phytoplankton	+ +	+		+ +	+ +	
Depth-weighted phytoplankton, cells/L	+ +	+		+	+ +	
Total benthos, No./m^2	+ +					

* Significance levels: 0.05 = + + or - -, 0.01 = + or -

Table 8-5.—Summary of correlation analyses of abiotic and biotic variables related to primary production in Twin Lakes, Colorado — Continued.

Preoperational period (1982–85)	Lower Twin Lakes Chlorophyll *a*	Lower Twin Lakes Carbon fixation rate	Lower Twin Lakes Log_{10} phytoplankton	Upper Twin Lakes Chlorophyll *a*	Upper Twin Lakes Carbon fixation rate	Upper Twin Lakes Log_{10} phytoplankton
Site Parameters						
Lake surface area, hectares			++		++	++
Lake volume, $m^3 \times 10^6$			++		++	++
Twin Lakes Tunnel volume				--		-
Lake Creek volume, m^3/m		+		--		
Mt. Elbert Conduit volume, m^3/m						
Powerplant generation volume, m^3/m					++	
Powerplant pumping volume, m^3/m		+			++	+
Total inflow volume, m^3/m				--		
Total outflow volume, m^3/m				--		
Physical Characteristics						
Average temperature, °C		++	++		++	
Maximum temperature, °C		++	++		++	
Minimum temperature, °C		++	++		++	
Retention time, days					--	
Stratification duration, days		-	+	+		++
Light extinction coefficient, μ^{-1}	+	--			-	
Euphotic depth, meters	-	++				
Chemical Characteristics						
Average dissolved oxygen, mg/L		--	-			
Maximum dissolved oxygen, mg/L		--	--		--	-
Minimum dissolved oxygen, mg/L						
Average pH	+	++				
Maximum pH		++				
Minimum pH	+					
Average conductivity, $\mu S/cm^2$						
Maximum conductivity, $\mu S/cm^2$						
Minimum conductivity, $\mu S/cm^2$			+			
Bottom E_7, mV						
Bicarbonate alkalinity, mg/L		++			+	
Nitrate nitrogen, mg/L						
Biological Characteristics						
Average chlorophyll *a*, mg/m^3			++			++
Average carbon fixation rate, $\mu gC/m^3/h$						
Ratio, μgC/mg chlorophyll *a*						
Depth-weighted zooplankton, No./L			++	++		++
Log_{10} phytoplankton	++			++		
Depth-weighted phytoplankton, cells/L				+		++
Total benthos, No./m^2						++

* Significance levels: 0.05 = + + or – –, 0.01 = + or –

upper lake. The increased surface area and volume of the upper lake, by reducing the effect of runoff, may have aided primary production. The range for hydraulic retention time in the upper lake following raising of the lakes (during runoff) was 23 to 29 days which translates to an exchange rate of 0.97 to 1.3 times compared to an exchange rate range of 1.25 to 2.73 times in the preoperational period. While average chlorophyll *a* seemed slightly less at the deep station on the upper lake in the post-operational period (table 8-4), total chlorophyll *a* biomass actually exhibited a substantial increase in both lakes following raising of the lakes in 1984 (fig. 8-3).

Both lakes showed small increases in mean annual phytoplankton densities in the postoperational period. The increased surface area and volume of the lakes could have resulted in increased phyto-plankton density in both lakes. However, an examination of yearly mean depth-weighted phyto-plankton densities in the lower lake during the postoperational period indicates that an increase occurred in just 1 year, 1984, when densities averaged 66 171 cells/L (table 8-3). For the remaining 3 years, the yearly mean phytoplankton densities ranged from 6500 to 18 805 cells/L, which were among the lowest densities observed during both periods (table 8-3). The increased phytoplankton densities observed in 1984 immediately following raising of the lakes might have been a temporary phenomenon which masked an actual decline in the net-size phytoplankton assemblage in the lower lake during the postoperational period.

Flow variables in the site parameters category were generally positively correlated to primary production in the lower lake and negatively correlated to primary production in the upper lake (table 8-5). LaBounty et al. (1985) reported a positive correlation between primary production and Lake Creek inflow (r = 0.91) in the lower lake during the preoperational period. The positive relationship between primary production and flow variables may be explained by the dependency upon inflow as a nutrient source in the lower lake. During the pre-operational period, the lower lake was dependent upon runoff passing through the upper lake for nutrient input. In low inflow years, production in the lower lake was limited when nutrients were retained in the upper lake basin (Campbell and LaBounty, 1985; LaBounty et al., 1985). In contrast, primary production variables in the upper lake exhibited a strong negative relationship to inflow variables in the preoperational period—possibly indicating the effects of flushing during runoff. In the preoperational period, all inflow was routed by way of Lake Creek into the upper lake which significantly reduced the hydraulic retention time of the lake.

Based on hydrologic records, retention time in the upper lake during June ranged from 11 to 24 days in preoperational period. The entire volume of the upper lake was replaced from 1.25 to 2.73 times during maximum runoff. LaBounty et al. (1985) reported a negative correlation between primary production and Lake Creek inflow in the upper lake (r = -0.58) during the preoperational period. Chlorophyll *a* and phytoplankton densities were still negatively correlated with inflow in the upper lake during the postoperational period; however; carbon fixation showed a strong positive relationship to powerplant operation (table 8-5). A clear cause-effect explanation for this correlation can not be found, however; when generation and pumping volumes are plotted against carbon fixation rates, the data follow the same distribution trends.

Of the physical characteristics category variables, temperature had a strong influence on production in the lakes and was most clearly evident in the carbon fixation primary production parameter (table 8-5). Rhee and Gotham (1981) found that temperature is interactively important to carbon fixation rate in culture. Harris (1980) stated that the photosynthetic capacity of natural populations is clearly a function of temperature and listed numerous authors reporting a correlation between temperature and specific growth rates of algae in situ. Reynolds (1984) reported that specific maximal growth, like most physiological processes, respond to temperature changes provided that other factors continue to be saturating. Temperature variables and carbon fixation rates were always significantly positively correlated in both lakes throughout both periods.

Carbon fixation rates were negatively correlated with retention time in both lakes during the preoperational period and in the upper lake during the postoperational period (table 8-5). Retention time was least during the growing season and greatest in the winter (under ice cover) when primary production was generally minimal. The absence of correlation with retention time in the lower lake during the postoperational period can be explained by the change in hydrology in the lower lake resulting from powerplant operation. In the preoperational period, retention time decreased during run-off and gradually increased until the following spring. After the powerplant began operating, retention time in the lower lake increased at a much slower rate through December. In some years (e.g., 1982), retention time remained at about the same level throughout the year (fig. 8-8).

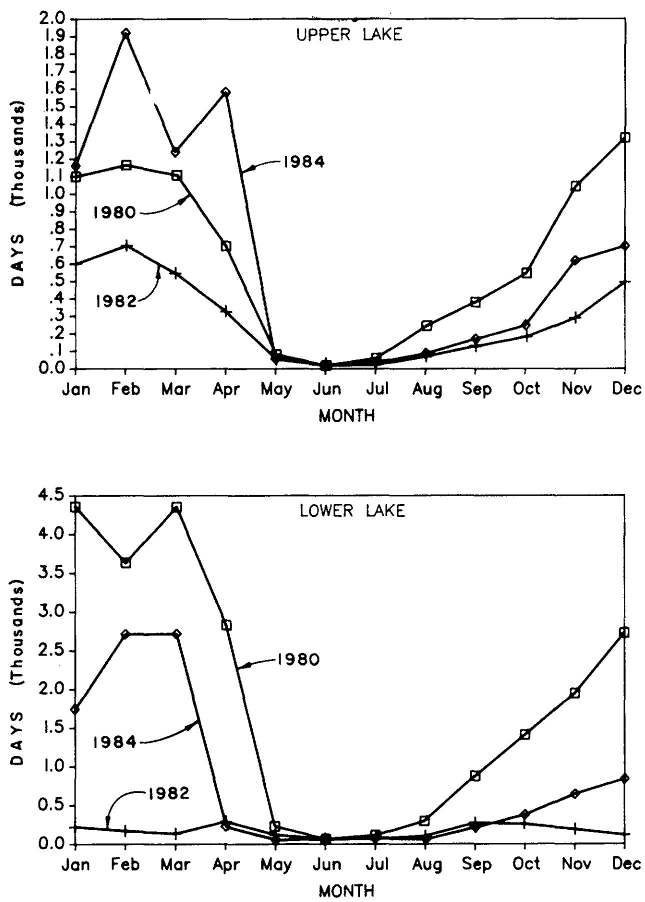


FIGURE 8-8.—*Average hydraulic retention time during representative pre- and postoperational years in Twin Lakes, Colorado.*

Duration of stratification was both positively and negatively correlated with primary production parameters in the lakes (table 8-5). Phytoplankton density in the upper lake during the preoperational period exhibited a strong negative correlation with duration of stratification. Nutrient cycling in the upper lake seemed dependent upon allocthonous input from inflows. During the preoperational period, cooler incoming flows plunged over the lip of the bowl-shaped basin to enter the hypolimnion of the upper lake. Nutrients in the epilimnion were rapidly depleted and as the duration of stratification increased, phytoplankton populations decreased in response to nutrient limitation. In the postoperational period, duration of stratification was generally positively correlated with primary production parameters in both lakes (table 8-5). In the upper lake, the inflow pattern was altered because incoming flows spread out, warmed in the shallow delta area, and entered the epilimnion of the main basin. Nutrients were more readily available in the epilimnion of the upper lake throughout the growing season in the postoperational period and phytoplankton densities generally increased as the duration of stratification increased.

Phytoplankton populations in the lower lake also exhibited a positive correlation with stratification duration in the postoperational period. The duration of stratification in the lower lake decreased significantly ($P=\leq 0.05$), from 88.6 to 39 days (table 8-4) as a direct result of powerplant operation. However, nutrient cycling in the lower lake seemed dependent on autochthonous cycling within the epilimnion during stratification periods. In the postoperational period, any increase in duration of stratification in the lower lake allowed phytoplanktonic organisms to exploit environmental conditions when the water column was not experiencing rapid perturbations. Haffner, et al. (1979) and Harris, et al. (1979) have shown that rapid and continued perturbations in the mixing regime of a lake may reduce the efficiency with which phytoplankton can exploit available nutrient resources. Under such conditions, primary production parameters are all suppressed. Therefore, increasing phytoplankton densities, as duration of stratification increased, may be explained by the relative stability in the water column that allowed an increase in biomass to occur.

The negative relationship exhibited between carbon fixation rate and stratification duration in the lower lake during the postoperational period may be related to the effect of powerplant operation on water transparency. As the powerplant generated, a turbulent plume of discharge water entrained bottom sediments, decreasing light penetration. Light is the major energy source driving the productivity of aquatic ecosystems (Wetzel, 1975; Cole, 1979; Goldman and Horne, 1983). Because the carbon fixation process is light dependent, increased mixing, which results in less light penetration, would decrease carbon fixation.

An increased light extinction coefficient was observed seasonally in both lakes, i.e., spring runoff and ice-covered periods. However, higher light extinction coefficients were generally observed in the lower lake throughout the postoperational period. As previously mentioned, light extinction coefficient was one of the abiotic variables significantly greater in the lakes during the postoperational period. Generally, primary production parameters in both lakes were negatively correlated with light extinction coefficient and positively correlated with euphotic depth in both lakes during the postoperational period (table 8-5).

In the chemical characteristics category, primary production parameters generally exhibited a strong negative relationship to dissolved oxygen variables (table 8-5). Because dissolved oxygen and temperature exhibit a strong inversely proportional relationship, and temperature is strongly correlated with primary production, a corresponding inversely proportional relationship between dissolved oxygen and primary production—which is an artifact of the relationship between dissolved oxygen and temperature—might result. Therefore, the inversely proportional relationship between dissolved oxygen and carbon fixation rate might have been an effect of a relationship with a common variable, temperature.

Primary production exhibited a strong positive relationship to pH variables in both lakes throughout both periods (table 8-5). Photosynthesis can result in increasing pH in an ecosystem (Goldman and Horne, 1983); thus, increasing pH—which resulted from primary production—was generally observed in the lakes throughout both periods. Average bicarbonate alkalinity generally exhibited a strong positive relationship to primary production variables in both lakes during the postoperational period. Bicarbonate alkalinity was titrated for determination of available carbon during C^{14} tests at the lakes. Bicarbonate alkalinity decreased significantly in the postoperational period in the lower lake and may have indicated carbon limitation; however, since both variables decreased significantly between periods, the correlation between the two may have resulted from similar distribution patterns in the postoperational period. The strong positive correlation between carbon fixation rate and bicarbonate alkalinity in the upper lake (throughout the study period) may have indicated periodic or seasonal carbon limitation because the upper lake basin was flushed repeatedly during runoff and bicarbonate alkalinity decreased by dilution from increased flows.

In the biological characteristics category, primary production variables showed strong positive correlations with other biotic variables. Some caution must be exercised in interpreting these results as all biological parameters exhibited skewed distribution patterns. The similarity in distributions may be enough to cause autocorrelations among biological variables. Chlorophyll and carbon fixation rates were not generally correlated (table 8-5). Changes in carbon fixation rate precede changes in chlorophyll *a* at Twin Lakes. LaBounty et al. (1985) reported that the lag between carbon fixation rate and chlorophyll *a* at Twin Lakes was approximately 4 weeks, but that response time seemed to vary inversely with temperature. Chlorophyll *a* is an integrative variable, reflecting environmental conditions over a period of a week or so before its measurement (Harris, 1980). Carbon fixation rate is an instantaneous measurement reflecting current environmental conditions which may or may not result in a change in biomass at some future time. This inherent difference, an integrative variable and an instantaneous one, may be much like comparing apples and oranges; and the time lag between fixation and biomass may complicate the relationship between the two variables, masking any statistical correlation.

Phytoplankton density in the lower lake during the preoperational period was significantly correlated with the biotic variables average chlorophyll *a* and average carbon fixation rate (table 8-5). Although these variables are measured components of primary production and a relationship among these variables could be expected, any interpretation must be qualified by certain inherent factors in the data. Chlorophyll *a* and carbon fixation rates were estimated from autotrophic organisms in size classes equal to or greater than 1.0 μm. Phytoplankton densities were estimated from autotrophic organisms in size classes equal to or greater than 80 μm. Investigations of size-class distributions for autotrophic organisms at Twin Lakes indicated that 80 to 90 percent of chlorophyll *a* concentration was contributed by size classes less than 80 μm (Campbell, 1987). Because phytoplankton density does not include those autotrophic organisms in the size classes contributing to most of the production in the lakes, the relationship among biotic variables may be the result of similar, skewed distribution patterns.

The results of maximum R^2 improvement regression analysis on primary production variables is summarized in table 8-6. The percent of total variance explained by the model ranged from 66.2 to 95.3 percent. The criteria established for the model was a maximum of 12 variables used with greater than 65 percent of the total variance explained as the desired result. The criteria was always met during the analysis. This parametric statistical analysis procedure was used to sift through all the variables in the Twin Lakes data set to determine those few variables which account for the largest variation in primary production.

Site parameters, which included lake surface area, volume, inflow and outflow variables, accounted for relatively little of the variability in primary production in the lower lake throughout both periods (table 8-6). The exception was the phytoplankton component of primary production in the postoperational period to which site parameters contributed 22.1 percent. In the upper lake, site parameters contributed little to the total variation in primary production variables in the preoperational period, but contributed most of the variation to all primary production variables in the postoperational period. The change in lake morphology

Table 8-6.—Results of maximum R^2 improvement regression analyses of abiotic and biotic variables related to primary production at Twin Lakes, Colorado.

Lower Twin Lakes	Preoperational (1977–81)			Postoperational (1982–85)		
	Chlorophyll *a*	Carbon fixation rate	Log_{10} phytoplankton	Chlorophyll *a*	Carbon fixation rate	Log_{10} phytoplankton
Site Parameters						
Lake surface area, hectares	12.8		3.1			18.4
Lake volume, $m^3 \times 10^6$	0.1					1.0
Twin Lakes Tunnel volume, m^3/m						
Lake Creek volume, m^3/m						
Mt. Elbert Conduit volume, m^3/m						1.6
Powerplant generation volume, m^3/m						
Powerplant pumping volume, m^3/m						
Total inflow volume, m^3/m				8.8		1.1
Total outflow volume, m^3/m						
Category Sum:	12.9	0.0	3.1	8.8	0.0	22.1
Physical Characteristics						
Average temperature, °C		24.7			20.1	
Maximum temperature, °C					33.4	
Minimum temperature, °C			30.6			
Retention time, days	8.8	2.3				
Stratification duration, days	0.1	7.1			10.6	
Light extinction coefficient μ^{-1}	0.2	3.2		12.2	3.8	
Euphotic depth, meters			4.4	0.7		
Category Sum:	9.1	67.9	4.4	12.9	67.9	0.0
Chemical Characteristics						
Average dissolved oxygen, mg/L		0.8		10.4	0.8	0.7
Maximum dissolved oxygen, mg/L						23.9
Minimum dissolved oxygen, mg/L					4.8	
Average pH				13.9		
Maximum pH		8.0		19.3	1.5	
Minimum pH	0.1	6.5	4.1		0.9	
Average conductivity, $\mu S/cm^2$			1.3		0.3	
Maximum conductivity, $\mu S/cm^2$	11.4		10.4		1.6	
Minimum conductivity, $\mu S/cm^2$						
Bottom E_7, mV	0.1	0.2	0.5		6.8	
Bicarbonate alkalinity, mg/L		0.2	11.1		7.3	
Nitrate nitrogen, mg/L	2.0			2.7		0.5
Category Sum:	13.6	15.7	27.4	46.3	24.0	25.1
Biological Characteristics						
Average chlorophyll *a*, mg/m^3		1.3	22.3			
Average carbon fixation rate, $\mu gC/m^3/h$	4.6		3.9	2.6		3.0
Ratio, μgC/mg chlorophyll *a*						
Depth-weighted zooplankton, No./L	2.0		5.1			34.6
Log_{10} phytoplankton						
Depth-weighted phytoplankton, cells/L				7.7		
Total benthos, No./m^2	33.0	9.1		8.2		5.5
Category Sum:	39.6	10.4	31.3	18.5	0.0	43.1
Percent of total variance:	75.2	94.0	66.2	86.5	91.9	90.3

Table 8-6.—Results of maximum R^2 improvement regression analyses of abiotic and biotic variables related to primary production at Twin Lakes, Colorado — Continued.

Upper Twin Lakes	Preoperational (1977-81)			Postoperational (1982-85)		
	Chlorophyll *a*	Carbon fixation rate	Log_{10} phytoplankton	Chlorophyll *a*	Carbon fixation rate	Log_{10} phytoplankton
Site Parameters						
Lake surface area, hectares		0.8		3.4		29.6
Lake volume, $m^3 \times 10^6$				6.6		
Twin Lakes Tunnel volume, m^3/m				16.9	0.8	12.6
Lake Creek volume, m^3/m			11.3	4.5	1.0	0.8
Mt. Elbert Conduit volume, m^3/m				9.2		0.1
Powerplant generation volume, m^3/m						
Powerplant pumping volume, m^3/m				37.3		
Total inflow volume, m^3/m				9.5	0.6	0.9
Total outflow volume, m^3/m	11.5	0.9				
Category Sum:	11.5	1.5	11.3	50.1	39.7	44.0
Physical Characteristics						
Average temperature, °C	11.0		10.1		16.7	
Maximum temperature, °C	0.6					
Minimum temperature, °C	14.1	60.4	27.1			
Retention time, days				6.6		
Stratification duration, days	1.6		12.8			4.8
Light extinction coefficient, μ^{-1}		0.2	6.5			
Euphotic depth, meters	11.1	0.8	2.3	7.6	12.3	
Category Sum:	38.4	61.4	58.8	14.2	29.0	4.8
Chemical Characteristics						
Average dissolved oxygen, mg/L			0.7		0.7	
Maximum dissolved oxygen, mg/L		2.5			2.3	
Minimum dissolved oxygen, mg/L		0.1		7.3		
Average pH			0.3			
Maximum pH	14.6		3.9		4.5	0.3
Minimum pH		1.4	1.4			
Average conductivity, $\mu S/cm^2$	0.1					9.0
Maximum conductivity, $\mu S/cm^2$	2.6			2.4	9.8	11.4
Minimum conductivity, $\mu S/cm^2$				2.9		
Bottom E_7, mV		7.1			1.2	3.2
Bicarbonate alkalinity, mg/L	1.6	3.9	1.9		7.9	
Nitrate nitrogen, mg/L						
Category Sum:	18.9	15.0	8.2	12.6	26.6	23.9
Biological Characteristics						
Average chlorophyll *a*, mg/m^3		2.9				
Average carbon fixation rate, $\mu gC/m^3/h$	1.6		3.0			3.5
Ratio, μgC/mg chlorophyll *a*						
Depth-weighted zooplankton, No./L	8.2					15.2
Log_{10} phytoplankton		2.7			14.4	
Depth-weighted phytoplankton, cells/L						
Total benthos, No./m^2						
Category Sum:	9.8	5.6	3.0	14.4	0.0	18.7
Percent of total variance	78.6	83.5	81.3	91.3	95.3	91.4

Note: m^3/m = cubic meter per month.

and inflow to the lakes (see ch. 2) was clearly reflected in this shift between periods and most clearly evident in the upper lake where the changes were greatest.

The changes in inflow to the lakes occurred in a stepwise fashion. First, the powerplant began testing one unit in September 1981, and Mt. Elbert Forebay was drained for installing a polyvinyl chloride liner to prevent leakage. The powerplant operated extensively in 1982, but mostly in generation mode because mechanical problems prevented "pump-back" operation for much of the early postoperational period. The maximum volume of the lakes remained the same in all preoperational years; however, in 1982, the total lake volume was low following a major drought year, 1981.

In the second step, a bipartite inflow pattern to the lakes was begun when Western Slope diversions and/or exchanges began to enter both the upper and lower lakes. Western Slope diversions began to enter the lower lake through the Mt. Elbert Conduit in 1981; however, diversion—through the Twin Lakes Tunnel—entering the upper lake continued during 1982 and 1983. Both lakes received Western Slope diversions during 1982-83; these years are considered "transitional," showing both pre- and postoperational period characteristics. In addition, the powerplant did not operate from March to September 1983 because of mechanical difficulties. The lakes quickly returned to the preoperational pattern of thermal stratification during the summer months in 1983, but the inflow pattern was considerably altered from preoperational conditions.

Finally, beginning in 1984, the configuration of the lakes behind the Twin Lakes Dam was established. Both pump-turbines in the Mt. Elbert Pumped-Storage Powerplant were in place and operational. As a result, the mean water depth in the lakes increased 2.5 meters; Western Slope water diversions impacted only the lower lake, and both generating and pump-back activities occurred on a more regular basis.

As noted in chapter 6, before raising of the water surface elevation of the lakes, Lake Creek inflow—which consisted of both natural inflow and Western Slope imports by way of the Twin Lakes Tunnel—entered the upper lake at a cooler temperature than the surface waters of the lake. The cooler water plunged over the lip of the bowl-shaped basin of the lake and entered at it's own temperature stratum. This plunging inflow made the bottom water mass cooler and strengthened thermal stratification in the upper lake during the summer months. Following raising of the lakes, Lake Creek inflow entered a broad, shallow, delta area west of the main upper lake basin and spread out, warmed quickly and entered the upper lake as a near surface overflow. Direct entry of inflow water into the epilimnion of the upper lake may have introduced nutrients directly into the euphotic zone resulting in the generally increasing carbon fixation rates and phytoplankton densities observed throughout the summer months during the postoperational period.

Bipartite inflow to the lakes may have benefitted the upper lake by reducing the impact of generally increased flow conditions throughout the postoperational period. The upper lake, during the preoperational period, functioned as a reregulating reservoir and a settling basin for entrained sediment during annual runoff. With the change in inflow and morphology of the upper lake, during the postoperational period, the impact of runoff was distributed over a greater surface area and was not as great in magnitude as in high runoff years during the preoperational period.

The effect of the altered inflow regime on the lower lake was directly related to the passage of water through the powerplant. The step-wise nature of changes in the lakes made the impact of powerplant operation greatest in 1982 before the total volume of the lakes was increased in late 1983.

Physical characteristics, which included temperature, water column stability, and light variables, were particularly important to carbon fixation rates in both the pre- and postoperational periods in the lower lake (table 8-6). Temperature contributed 67.9 percent of the total variability in the carbon fixation rate variable throughout both periods in the lower lake. Physical characteristics made an important contribution to primary production variables in the upper lake, during the preoperational period, but not in the postoperational period. The effect of changes in lake morphology and inflow variables on the upper lake was so significant, all other contributions to the total variability in primary production during the postoperational period were diminished.

Water column stability indicators, i.e., hydraulic retention time and stratification duration, were important contributors to variability in primary production in both lakes (table 8-6). The retention time (time necessary for complete exchange of lake volume) in the lakes decreased significantly between periods. As discussed in chapter 2, the mean retention time decreased from about 1 year to less than 1/2 year on an annual basis. The greatest change in retention time occurred in the winter months.

Figure 8-8 illustrates average monthly retention times in the lakes:

- During 1980, a representative preoperational period year;
- 1982, a transitional year; and
- 1984, a true postoperational period year.

The relatively normal inflow year, 1980, ranked 27th in the 38 year record of flow (ch. 2). An above normal inflow year, 1982, ranked 3d in the 38 year record of flow. Also an above normal inflow year, 1984, ranked 6th in the 38 year record of flow.

The retention times during 1980 in both lakes are fairly representative of the pattern present in the lakes during the entire preoperational period (fig 8-8). The retention time was two orders of magnitude greater in the winter than in the summer; the difference between the retention times seemed to be a function of the relative size of the lakes (lower lake is three times larger than the upper). In the upper lake, during the 1982 winter, retention time was an order of magnitude greater than in any other previous year. All water systems throughout the Western States were full. In an attempt to provide additional storage in the Colorado River Basin before 1983 runoff, Western Slope diversions were passed through Twin lakes from both the Twin Lakes Tunnel (by way of Lake Creek) and from Turquoise Lake (through the Mt. Elbert Conduit) to Pueblo Reservoir during the winter months in both 1982 and 1983. In the upper lake, the retention times for January, February, and December were much shorter.

For the lower lake, the retention times in all months of 1982 reflect the sustained and relatively more rapid flushing of the lake as diversion water was passed through the lake in an inflow-equal-outflow process (fig. 8-8). The relative size of the lakes no longer was part of the retention time pattern, but strictly a measure of the amount of flow entering and leaving the lakes regardless of relative size. In 1984, the winter months were more similar to preoperational conditions, but the lower lake was still more subject to flushing than in preoperational period years. Both lakes showed an increase in the time period for maximum flushing compared to preoperational years. In preoperational years, flushing was confined to the June-July months in any year. In postoperational years, flushing began with ice-off in both lakes, and continued through September. Although mean total inflow to the lakes remained the same, during the post-operational period; inflow and flushing rates were more sustained than during the preoperational period.

The effect of increased flushing and decreased retention time on primary production parameters was three-fold:

1. Increased flushing favored diatom dominance of the algal assemblage as diatoms compete more successfully during periods when the water column is well mixed and the normally occurring species succession was disrupted.

2. Phytoplankton densities declined in the lower lake during the postoperational period, except (as previously discussed) in 1984 immediately following raising of the water surface elevation of the lakes. Algal species at Twin Lakes require approximately 3 to 5 weeks to enter the exponential growth phase at 12 °C (Campbell, unpublished). During the postoperational period, the retention times decreased in both lakes to that range, of 3 to 5 weeks in June and July, which meant that the algae were being flushed from the lake before they could reproduce in numbers sufficient for algal biomass to increase.

3. Some factor—possibly increased turbidity—in the lower lake was affecting light penetration. Carbon fixation rates are completely dependent upon light as a basis for the reaction by which carbon fixation occurs. Less light penetration equals less carbon fixation. Finally, less carbon fixation could have created a situation where algal cells were stressed. Under those conditions, chlorophyll *a* can seem to increase when other parameters are decreasing. Carbon fixation may have occurred—not at a level necessary for growth and reproduction—but at a level for maintenance only.

Variables in the chemical characteristics category contributed more toward the total variability in primary production in both lakes in the postoperational period (table 8-6). In the lower lake, conductivity and bicarbonate alkalinity both decreased significantly between periods. The decrease in both variables illustrated a dilution effect, and—in the case of bicarbonate alkalinity—may have indicated that carbon limitation was occurring at times when imports through the Mt. Elbert Conduit were high. In the upper lake, dissolved oxygen and bottom E_7 contributed more toward the total variability in primary production variables in the postoperation period. The shift observed between periods may have indicated a more favorable winter environment in the upper lake. A higher survival rate in over-wintering populations of both algae and zooplankton yields a larger population base for initial spring production in any ecosystem.

A relatively small contribution to the total variability in primary production was made by variables in the biological characteristics category in both lakes throughout both periods (table 8-6). The relationship among the biotic components in the lakes was affected by a similar, skewed distribution pattern and the relative size of the contribution, in most cases, to primary production tends to support that hypothesis.

Table 8-7 summarizes maximum R^2 improvement regression analysis by limiting the number of variables contributing to the total variability in a dependent variable to those contributing equal to or greater than 10 percent each and the minimum number which contribute equal to or greater than 50 percent. These criteria were met for all dependent variables except phytoplankton density in the lower lake during the preoperational period.

In the lower lake, during the preoperational period, the variables: total benthos number, lake surface area, maximum conductivity, and retention time explained 66 percent of the total variance in the dependent variable chlorophyll *a* (table 8-7). The relationship between these variables and chlorophyll *a* was proportional for benthos and retention time and inversely proportional for lake surface area and conductivity. In the postoperational period, the variables: maximum and average pH, light extinction coefficient, maximum dissolved oxygen, and total inflow explained 64.6 percent of the total variance in chlorophyll *a* (table 8-7). The relationship between these variables and chlorophyll *a* was proportional for average pH and light extinction coefficient, and inversely proportional for maximum pH, maximum dissolved oxygen, and total inflow.

For the dependent variable average carbon fixation rate in the lower lake, during the preoperational period, the variables minimum and average temperature and total benthos number contributed 64.4 percent of the total variance (table 8-7) The relationship between these variables and average carbon fixation rate was proportional for minimum temperature and inversely proportional for average temperature and total benthos number. In the postoperational period, the variables minimum and maximum temperature, and stratification duration explained 64.1 percent of the total variance in the dependent variable carbon fixation rate (table 8-7). The relationship between these variables and carbon fixation rate was proportional for minimum temperature and inversely proportional for maximum temperature and stratification duration.

Total variance in the dependent phytoplankton density variable in the lower lake, during the preoperational period, was poorly explained by the maximum R^2 regression model (table 8-7). The variables that met the equal to or greater than 10-percent contribution to total variance criteria were average chlorophyll *a*, average bicarbonate alkalinity, and average conductivity. These variables explained 43.8 percent of the total variance in log-transformed phytoplankton density. The relationship between phytoplankton and average chlorophyll *a* was proportional.

During the postoperational period in the lower lake, the total variance explained by the maximum R^2 regression model was much greater (table 8-7). The variables depth-weighted zooplankton, maximum dissolved oxygen, and lake surface area explained 76.9 percent of the total variance in the dependent phytoplankton density variable. The relationship between phytoplankton density and zooplankton was proportional and inversely proportional between maximum dissolved oxygen and lake surface area.

In the lower lake, the variables in the maximum R^2 regression analysis of dependent variables chlorophyll *a* and phytoplankton density were completely different between pre- and postoperational periods (table 8-7). Some of the variables that were significantly correlated with chlorophyll *a* or phytoplankton either were not present in the R^2 model or were not among the variables that contributed equal to or greater than 10 percent of the total variation. It is possible some sort of interaction among independent variables had a masking effect in the model. For example, because saturation of dissolved oxygen is a temperature dependent function, variables that were significantly correlated with temperature were often correlated with dissolved oxygen. If the model fit was better with dissolved oxygen than with temperature, then temperature might be dropped from the model. The relationship between a primary production variable and temperature is a reasonable one while the relationship between dissolved oxygen concentration might be less reasonable. Temperature was significantly correlated to carbon fixation rate in the lower lake (throughout both periods) and also accounted for most of the variance in this dependent variable in the R^2 model.

In the upper lake, during the preoperational period, the variables maximum pH, minimum and average temperatures, total outflow, and euphotic depth explained 62.3 percent of the total variation in the dependent variable chlorophyll *a* (table 8-7). The relationship between chlorophyll *a* and both maximum pH and minimum temperature was proportional and was inversely proportional with total outflow, euphotic depth, and average temper-

Table 8-7.—Summary of maximum R^2 improvement regression analysis on primary production at Twin Lakes, Colorado.

Preoperational period, 1977-81		Postoperational period, 1982-85	
Lower Lake			
Dependent variable: Average chlorophyll *a* concentration, mg/m^3			
Total variance explained	= 0.7482%	Total variance explained	= 0.8671%
Total benthos number	= .330	Maximum pH	= .193
Lake surface area	= .128	Average pH	= .139
Maximum conductivity	= .114	Light extinction coefficient	= .122
Retention time	= .088	Maximum dissolved oxygen	= .104
		Total inflow volume	= .088
Dependent variable: Average carbon fixation rate, μgC/m^3/h			
Total variance explained	= 0.9378%	Total variance explained	= 0.9190%
Minimum temperature	= .306	Minimum temperature	= .334
Average temperature	= .247	Maximum temperature	= .201
Total benthos number	= .091	Stratification duration	= .106
Dependent variable: Log_{10} Depth-weighted phytoplankton, cells/L			
Total variance explained	= 0.6606%	Total variance explained	= 0.9033%
Average chlorophyll *a*	= .223	Depth-weighted zooplankton	= .346
Average bicarbonate	= .111	Maximum dissolved oxygen	= .239
Average conductivity	= .104	Lake surface area	= .184
Upper Lake			
Dependent variable: Average chlorophyll *a* concentration, mg/m^3			
Total variance explained	= 0.7584%	Total variance explained	= 0.9108%
Maximum pH	= .146	Twin Lakes Tunnel volume	= .169
Minimum temperature	= .141	Log_{10} d.w. phytoplankton	= .144
Total outflow	= .115	Total inflow	= .095
Euphotic depth	= .111	Mt. Elbert Conduit volume	= .092
Average temperature	= .110		
Dependent variable: Average carbon fixation rate, μgC/m^3/h			
Total variance explained	= 0.8354%	Total variance explained	= 0.9537%
Minimum temperature	= .604	Powerplant pumping volume	= .373
		Average temperature	= .167
		Euphotic depth	= .123
		Maximum conductivity	= .098
Dependent variable: Log_{10} Depth-weighted phytoplankton, cells/L			
Total variance explained	= 0.8143%	Total variance explained	= 0.9141%
Minimum temperature	= .271	Lake surface area	= .296
Stratification duration	= .128	Depth-weighted zooplankton	= .152
Lake Creek volume	= .113	Twin Lakes Tunnel volume	= .126
Average temperature	= .101	Maximum conductivity	= .114
		Average conductivity	= .090

[1] Minimum number of variables contributing to total variance criteria: should contribute approximately 0.10 of total variation; variables listed should total 0.50 of total variation.

ature. During the postoperational period, chlorophyll *a* variation in the upper lake was explained by the flow variables Twin Lakes Tunnel, total inflow, and Mt. Elbert Conduit, and the phytoplankton density. These four variables explained 50 percent of the total variance in chlorophyll *a* concentration in the upper lake during the postoperational period. The relationship between chlorophyll *a* and both phytoplankton and total inflow was proportional and was inversely proportional to inflow from Twin Lakes Tunnel and the Mt. Elbert Conduit.

Sixty percent of the variation in dependent variable carbon fixation rate in the upper lake, during the preoperational period, was explained by minimum temperature (table 8-7). The relationship between these two variables was proportional. During the postoperational period, 76.1 percent of the total variance in carbon fixation rate was explained by the variables powerplant pumping volume, average temperature, euphotic depth, and maximum conductivity. The relationship between carbon fixation rate and all these variables was proportional.

Minimum and average temperature, stratification duration, and Lake Creek inflow variables explained 61.3 percent of the total variation in phytoplankton in the upper lake during the preoperational period (table 8-7). The relationship between phytoplankton and minimum temperature was proportional and was inversely proportional to stratification duration, Lake Creek inflow, and average temperature. In the postoperational period, 77.8 percent of the total variance in phytoplankton was explained by the variables lake surface area, depth-weighted zooplankton, inflow from the Twin Lakes Tunnel, and maximum and average conductivity. The relationship between these variables and phytoplankton density was proportional except maximum conductivity, which was inversely proportional to phytoplankton in the upper lake during the postoperational period.

As in the lower lake, the variables contributing equal to or greater than 10 percent of the total variance in dependent variables chlorophyll *a* and phytoplankton were different in the pre- and postoperational periods (table 8-7). Temperature was an important contributing variable to carbon fixation rate throughout both periods.

Of the variables that contributed equal to or greater than 10 percent of the variability in primary production in Twin Lakes (throughout both periods) temperature, lake surface area, some flow variable (e.g., total inflow, Lake Creek inflow, etc.), and conductivity were significant in all statistical analyses results. A principal components analysis on the Twin Lakes data set was performed.

- Temperature was the single most important variable in both lakes throughout both periods.
- Dissolved oxygen was the next variable; however, its relation to temperature is so strong that it probably was a component of the temperature variable.
- Lake surface area and volume were important variables in the principal components analysis throughout both periods in each lake.
- Flow variables were important in each lake throughout both periods, but the upper lake also showed a strong relationship to total outflow while the lower lake did not.
- Conductivity was the final variable important in principal components analysis in each lake throughout both periods.

Lake surface area, flow variables, and conductivity, were significantly different between the periods and, as expected, the change in these variables had a definite impact on primary production in Twin Lakes in the postoperational period.

The strong positive correlation between temperature and primary production parameters, the contribution to total variance in the maximum R^2 regression analysis, and the results of principal components analysis all indicate that temperature was the single most important driving variable in each lake throughout both periods.

Significant change in annual mean temperatures was not observed between periods in the lakes, however; a 1 or 2 degree decrease in mean summer temperatures in both lakes remains a possibility and the effect on primary production could be profound.

CONCLUSIONS

A brief summary of the set of expectations discussed in the introductory section of this chapter is listed here for reference:

- Powerplant operation might stir bottom sediments and increase turbidity.
- Diversions from the Western Slope of the Continental Divide might change water chemistry in the lakes.
- Thermal regime of the lakes might be affected by powerplant operation.
- Changes in primary production parameters likely could be related to the lakes' increase in elevations, and to changes in hydrologic regime.

Most of these expectations were confirmed. However, some of the expected changes may have been mitigated by the physical and hydrologic changes in the lakes which occurred during both periods. Therefore, the quantifiable effects of Mt. Elbert Pumped-Storage Powerplant operation on primary production at Twin Lakes are limited. The relative lack of postoperational period data restricts quantifiable conclusions.

The postoperational period includes 2 years of transitional data:

- 1982, wherein the powerplant had considerable testing upon the first unit before increasing the lakes' elevation, and
- 1983, during which the powerplant was nonfunctional throughout most of the growing season.

Because the lakes' elevation was increased in late 1983, only 1984 and 1985 (through Sept.) represent the postoperational data base.

Any interpretation of changes resulting from pumped-storage powerplant operation must be made with caution because the postoperational data base is so limited. Long-term changes could not reasonably be manifested in 2 years, especially if the powerplant operates intermittently or if additional changes in the hydrology of the lakes are implemented. Follow-up studies of Twin Lakes' limnology are certainly necessary before any definitive conclusions about pre- versus postoperational conditions can be made.

In the lower lake, chlorophyll *a* concentration was significantly greater in the postoperational period. Chlorophyll *a* concentrations in two lakes were not significantly different from one another in the preoperational period, but were significantly different in the postoperational period. Depth-weighted phytoplankton densities were significantly greater in the upper lake in the postoperational period. A phytoplankton bloom occurred in both lakes shortly after the water surface elevation of the lakes was increased in 1983. Phytoplankton densities reached maximum observed values in both lakes which increased the mean for the postoperational period in both lakes. When this year is removed from the data set, phytoplankton densities are significantly less in the lower lake in the postoperational period. The net-size phytoplankton species succession from a spring/diatom to a fall/chrysophycean algal assemblage was disrupted during the postoperational period. Diatom species dominated the net-size phytoplankton assemblage in the lower lake beginning in 1984 and intensifying in 1985. Induced mixing of the euphotic zone by powerplant operation may have created more favorable environmental conditions for diatoms that are dependent on mixing to maintain relative position in the euphotic zone. Chrysophycean species dominated the upper lake phytoplankton assemblage almost completely beginning in 1984 after the lakes' elevation was increased.

Carbon fixation rate in pre- and postoperational period comparisons decreased in both lakes, although the decrease was not statistically significant. The decline in mean annual carbon fixation rates was a sustained process that persisted in both lakes throughout the postoperational period. The decline in carbon fixation rates seemed to be directly related to the decrease in light penetration in the lower lake following powerplant operation. Carbon fixation processes are limited by and dependent upon light as the driving force for the reaction during which the carbon is fixed intracellularly. Decreased light penetration in the lower lake may have resulted from suspension of bottom sediments once the powerplant began operating. Decreased light penetration in the upper lake may have been a temporary phenomenon resulting from entrained sediments originating in the new inundated delta area. Inflow to the upper lake spread out over a much larger area in the postoperational period, and entered the lakes as a near-surface, turbid overflow rather than the cold, plunging inflow observed in the preoperational period.

The seasonality of primary production at Twin Lakes did not change during the postoperational period. All primary production parameters were significantly skewed toward greatest magnitude following fall turnover in both lakes and in both the pre- and postoperational periods.

The greatest change in the lakes was from a single inflow to a dual inflow. Primary production in the upper lake might benefit from this change on a long-term basis as autochthonous nutrient—cycling in the warmer, shallower delta area—increases nutrient loading in the main basin of the upper lake. Primary production in the lower lake might benefit from increased nutrient loading resulting from entrained organic debris from powerplant generation; however, the long term effects of decreased light penetration and more rapid flushing of the lake may offset increased nutrient loading.

During the preoperational period, the lakes seemed to fluctuate dynamically around a set of mean circumstances that were determined by site and physical variables, i.e., the natural variation in weather, annual precipitation and spring runoff cycles, the severity of conditions during March and April of each ice-covered season, temperature, the availability of light in the water column, and water column stability. During the postoperational period, the lakes still responded to the same gross (overall) hydrologic cycles, but the lakes' total volume was larger; the total flow increased, but was bipartite, with a portion of the total entering the upper lake and a portion entering the lower lake. The effects of runoff and increased flushing on the lakes shifted from the upper lake to the lower lake in the postoperational period. The import of water from Turquoise Lake affected the chemical characteristics of the lower lake—particularly, conductivity and bicarbonate alkalinity. The change in lake morphology affected the thermal regime in the upper lake, while powerplant operation affected the thermal regime in the lower lake. Neither lake exhibited the same degree of thermal stratification in the postoperational period. Essentially, the lakes changed from two lake basins in series to a lake and reservoir basin in series. Although the change in mean temperatures was not statistically significant, a degree temperature decrease (-1 °C) during the summer growing season may have significant adverse impacts on primary production in lower lake in the future.

Twin Lakes ecosystem was vulnerable to severe winter conditions in the preoperational period. A winter kill phenomenon was documented in the lakes during the winter of 1975 and the biota of the upper lake did not completely recover for several years. Powerplant operation may decrease the likelihood of another winter kill in the lakes; however, if the powerplant does not operate in the winter and a winter kill occurs, the effects may be more severe than in the past. Conceivably, in the future, a winter kill could eradicate the lake trout fishery in Twin Lakes.

* * * * *

Appendix A.—Algal species identified by three different taxonomists in samples collected at Twin Lakes, Colorado.

	CHLOROPHYTA	
Cosmarium bioculatum var *depressum*[1]	*Arthrodesmus ralfsii*[2]	*Ankistrodesmus* sp.[3]
Cosmarium impressulum	*Cosmarium bioculatum*	*Arthrodesmus* sp.
Dictyosphaerium pulchellum	*Dictyosphaerium pulchellum*	*Desmidium* sp.
Eudorina elegans	*Oocystis borgei*	*Dictyosphaerium pulchellum*
Mougeotia sp.	*Scenedesmus incrassatulus*	*Mougeotia parvula*
Oocystis borgei	*Scenedesmus bijuga*	*Mougeotia* sp.
Sphaerocystis schroeteri	*Sphaerocyctis schroeteri*	*Sphaerocystis schroeteri*
Staurastrum longiradiatum	*Spirogyra* sp.	*Spirogyra* sp.
Tetraspora lacustrus		
	CHRYSOPHYTA	
Dinobryon cylindricum	*Chrysosphaerella longispina*	*Characiopsis* sp.
Mallomonas elongata	*Dinobryon bavaricum*	*Dinobryon bavaricum*
Mallomonas pseudocoronata	*Dinobryon cylindricum*	*Dinobryon cylindricum*
	Mallomonas caudata	*Dinobryon sertularia*
	Mallomonas pseudocoronata	*Mallomonas* sp.
		Phaeothamnion confervicola
	BACILLARIOPHYTA	
Asterionella formosa	*Asterionella formosa*	*Achnanthes flexella*
Cyclotella bodanica	*Cymbella minuta*	*Achnanthes linearis*
Cyclotella comta	*Cymbella ventricosa*	*Anomoeoneis vitrea*
Cyclotella meneghiniona	*Fragilaria crotonensis*	*Asterionella formosa*
Cymbella affinis	*Gyrosigma scalproides*	*Caloneis ventricosa* var *truncatula*
Cymbella cuspidata	*Melosira italica*	*Cyclotella comta*
Cymbella ventricosa	*Melosira varians*	*Cyclotella meneghiniona*
Diatoma hiemale var *mesodon*	*Navicula canalis*	*Cyclotella ocellata*
Eunotia sp.	*Nitzschia fonticola*	*Cymbella minuta*
Fragilaria crotonensis	*Rhopolodia gibba* var *ventricosa*	*Diatoma tenue*
Gomphonema sp.	*Stephanodiscus astrea*	*Fragilaria* sp.
Hannaea arcus	*Stephanodiscus hantzschii*	*Frustulia rhomboides* var *amphipleuriodes*
Melosira distans var *alpigena*	*Surirella augustata*	*Hannaea arcus*
Melosira islandica subsp. *helvitica*	*Surirella ovata*	*Melosira granulata*
Navicula rhynocephala	*Synedra amphicephala*	*Melosira granulata* var *angustissima*
Pinnularia abaujennis var linearis	*Synedra ulna*	*Melosira italica*
Pinnularia major	*Tabellaria fenestrata*	*Navicula radiosa* var *tenella*
Pinnularia mesolepta	*Tabellaria flocculosa*	*Nitzschia linearis*
Stauroneis phoenocenteron	*Tetracyclus lacustris*	*Nitzschia* sp.
Surirella sp.		*Pinnularia nodosa*
Synedra amphicephala		*Pinnularia obscura*
Synedra delicatissima		*Rhopolodia gibba*
Synedra radians		*Stephanodiscus alpinus*
Synedra ulna var *danica*		*Stephanodiscus hantzschii*
Tabellaria flocculosa		*Surirella angustata*
		Surirella sp.
		Synedra delicatissima
		Synedra radians
		Synedra ulna
		Tabellaria fenestrata
		Tabellaria flocculosa
	CYANOPHYTA	
Anabaena flos-aquae	*Oscillatoria* sp.	*Oscillatoria agardhii*
Oscillatoria amphibia		
Oscillatoria tenuis		
	PYRROPHYTA	
Ceratium hirundinella		*Glenodinium* sp.
Glenodinium gymnodinium		
Piridinium inconspicuum		
Priodinium villei		

[1] Identification by Claire Norton, Colorado State University, Fort Collins, Colorado

[2] Identification by Sharon Campbell, Bureau of Reclamation, Denver, Colorado

[3] Identification by Chadwick & Associates, 5721 South Spotswood Street, Littleton, Colorado

Benthos — Chapter 9

R. A. Roline

Bureau of Reclamation
Applied Sciences Branch
Denver, Colorado 80225

INTRODUCTION

Benthos is the community of bottom-dwelling organisms in a lake. Benthic animals are diverse and represent many phyla. The benthic organisms referred to in this report are considered macrobenthos and are easily visible to the eye. The range for density and diversity of benthic organisms may be numerically quite broad. These organisms are often thought to be representative of the total aquatic environment over an extended period of time. According to citations compiled by LaBounty and Sartoris (1976), benthic fauna have been commonly used to describe the nature of lakes. Twin Lakes has a benthic fauna similar to that of mesotrophic lakes, although in other aspects they resemble oligotrophic lakes.

According to Brinkhurst (1974), benthos is that assemblage of animals living in or on the sediments and dependent upon the decomposition cycle for most if not all of its basic food supply. Benthic organisms are important in the mixing and exchange of substances within the bottom sediments as well as between the sediments and the overlying water (LaBounty and Sartoris, 1976; Krezoski et al., 1978; Graneli, 1979). They are also important in food webs by facilitating decomposition of organic material and providing food for numerous fish species (Griest, 1977; Krieger, 1980).

In research on lake trout *Salvelinus namaycush* (Walbaum), Griest (1977) found that diptera larvae occurred in at least 40 percent of the stomachs examined of fish between 127 and 965 mm in length; however, the volume only accounted for about 6 percent of the total stomach contents. In contrast, Krieger (1980) found that benthic fauna in Twin Lakes was an important component in the diet of both white (*Catostomus commersoni*) and longnose (*Catostomus catostomus*) suckers. Well over 60 percent of the gut content of these species of various ages consisted of benthic organisms.

This chapter discusses changes in the composition, abundance, and mass of the benthic fauna in Twin Lakes between the pre- and postoperational periods of the Mt. Elbert Pumped-Storage Powerplant and construction of the new Twin Lakes Dam.

COMPOSITION OF BENTHOS

In both Upper and Lower Twin Lakes, a narrow littoral zone is present and a layer of fine silt is found across the bottom below depths of 4 m (LaBounty and Sartoris, 1976). This littoral zone changed following project completion; presently, a relatively large marsh area at the upper end of the upper lake has been inundated. Few types of benthic organisms have been found in the main basins of the lakes; however, benthos associated with the littoral zone has not been studied in detail. The benthos found in the main basin of both the lower lake, Sta. 2, and the upper lake, Sta. 4, consisted nearly exclusively of organisms from the families Sphaeriidae (fig. 9-1) and Chironomidae (figs. 9-2, 9-3), and from the class Oligochaeta (fig. 9-4). The taxonomic composition of these benthic organisms is as follows:

Mollusca
  Bivalvia
    Pelecypoda
      Sphaeriidae
        Pisidium casertanum (Poli)
        Pisidium pauperculum sterki

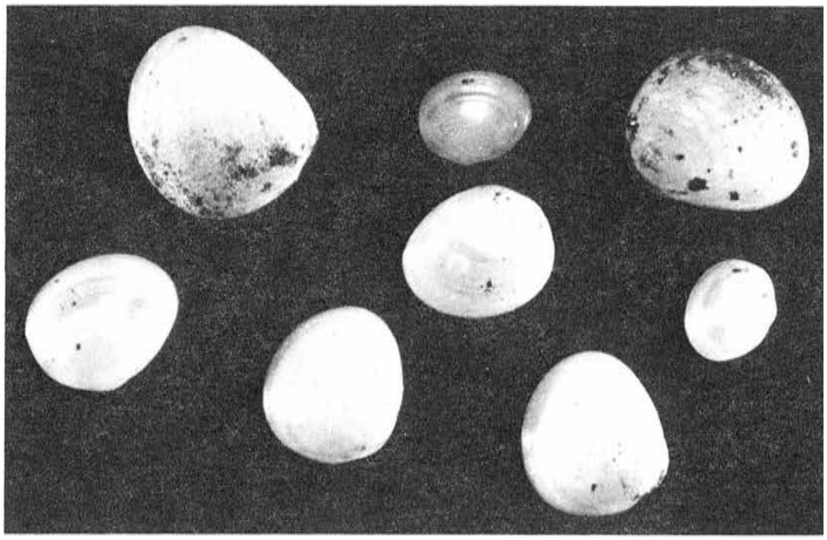


FIGURE 9-1.—Sphaeriidae (*fingernail clams), Pisidium spp.*

FIGURE 9-2.—*Chironomid larvae.*

Arthropoda
  Insecta
    Diptera
      Chironomidae
        Chironomus spp.
        Phaenopsectra spp.
        Dicrotendipes modestus

Annelida
  Oligochaeta

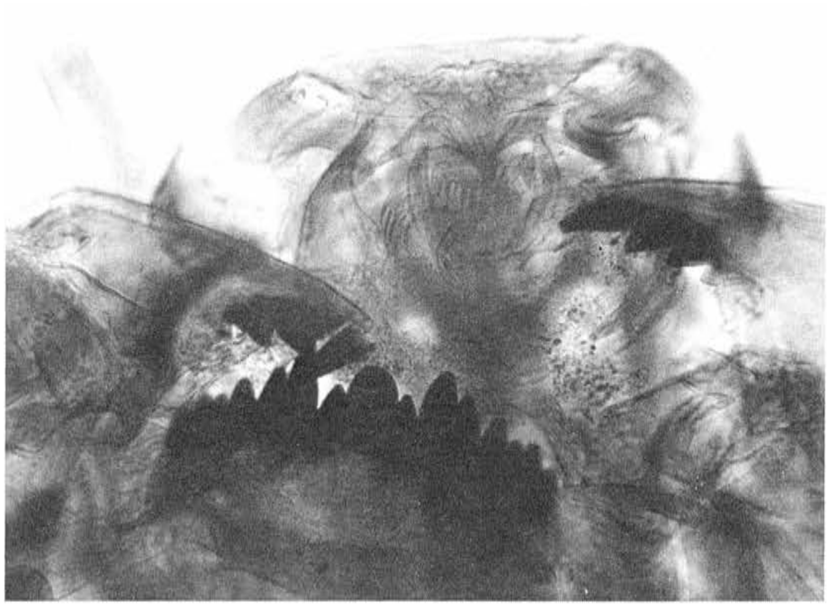
The taxonomy of benthos in Twin Lakes has not been examined fully; however, main groups of organisms can be discussed with regard to some changes in water quality, and construction and operation of Mt. Elbert Pumped-Storage Powerplant and Twin Lakes Dam.

METHODS

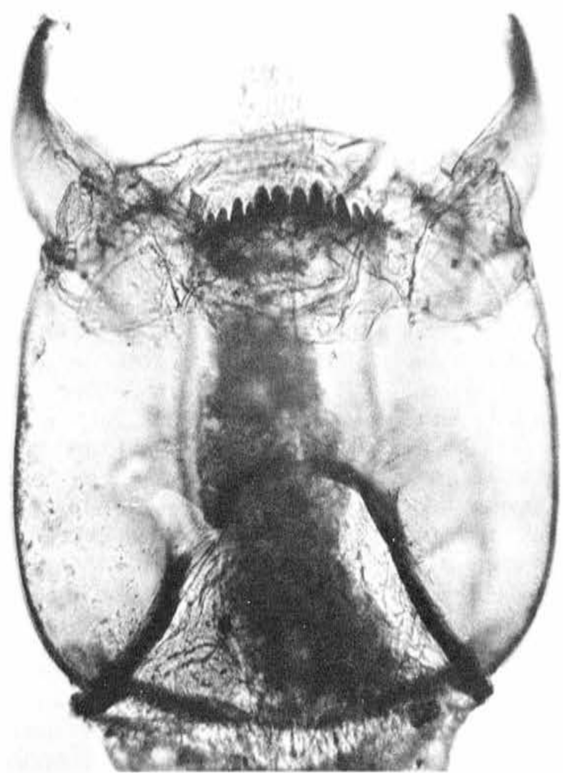
Throughout both periods, benthic samples were collected with the Peterson, Ponar, and Ekman dredges. Powers and Robertson (1965), Hudson (1970), and Rinne (1973) have compared the efficiency of different dredges and found the Ekman dredge to be more reliable in collecting sediment similar to that found in Twin Lakes. However, LaBounty and Sartoris (1976) found that other dredges proved equally efficient when they were set on the bottom rather than dropped.

Benthic samples were collected from mid-basin in both lakes (Sta. 2 and 4) monthly from June 1974 through April 1975 and twice monthly from June through November 1975. Samples were collected at least once a month from December 1975 through September 1985 at both stations 2 and 4. Data from 1977 through 1981 were used to characterize preoperational conditions, and data from 1982 through September 1985 were used to describe postoperational conditions of the benthic community. On one occasion, June 14, 1975, sample transects were made across both lakes to determine the relationship between depth and abundance of benthic fauna.

Triplicate or duplicate samples were collected on all sampling dates, and the samples were washed and presieved through a 600-μm U.S.A. Standard sieve (No. 30) and preserved in a 10-percent (approximately) formalin solution for counting and laboratory analyses. Specimens were identified using keys developed by Mason (1973) and Pennak (1978). Mass was measured using techniques described by Weber (1973).



a. *Chironomus* sp.



b. *Phaenopsectra* sp.

c. *Dicrotendipes modestus*.

FIGURE 9-3.—*Labial plates of three species of chironomid larvae.*

FIGURE 9-4.—*Oligochaete.*

RESULTS AND DISCUSSION

Comparisons between benthos populations and mass measurements between the pre- and postoperational periods were made with data collected from midlake on each of the lakes. These lakes were quite uniformly bowl-shaped, and the benthos collections represented the major groups of benthic macrofauna found throughout the main lakes.

In the collection of benthic samples on transects across each of the lakes, LaBounty and Sartoris (1976) found that differences in density and types of benthic fauna reflected, in part, dissimilarity in bottom substrate in various areas of the lakes. Where sediment became coarser, the density of benthos declined. The chironomid larvae occurred in greatest numbers in the fine sediment (fig. 9-5) found throughout the main basins while the density of *Pisidium* increased in the shallower depths. These benthic samples collected on a transect indicated that "patchiness" of organism density does occur.

During the winter of 1974-75, the hypolimnion of the upper lake became anaerobic, which resulted in the release of toxic heavy metals from the sediment. This release of heavy metals caused a great reduction in benthos abundance in 1975. Since that time, anaerobic conditions have not been reached in the hypolimnion of either lake.

Summary results from 5 years of preoperational and from 4 years of postoperational benthic data are shown in table 9-1. The Mann-Whitney U-Wilcoxon Rank Sum W and the Kolmogorov-Smirnov Goodness-of-Fit tests were used to evaluate differences between benthos composition, density, and mass between lakes and between pre- and postoperation of the project.

During the preoperational period, the densities of chironomids, oligochaetes, and clams (*Pisidium*) were significantly different ($P<0.05$) between lakes. The upper lake had a greater density of oligochaetes while the lower lake had a greater density of both clams and chironomids. This relationship continued to hold true following project operation; however, statistically significant ($P<0.05$) changes also occurred within each lake. In the upper lake, the density of chironomids declined as oligochaete density and total mass of chironomids plus oligochaetes increased. In the lower lake, the density of clams increased as chironomid density decreased. The mean density of clams in the upper lake increased and the density of oligochaetes in the lower lake decreased following project operation, but it was not significant ($P>0.05$).

In the upper lake, strong correlations existed between total density of the benthos ($P<0.01$) and nitrate concentration and water depth. Data representing the specific types of organisms showed that both clam and oligochaete densities were highly correlated ($P<0.05$) with nitrate concentration and water depth; however, chironomid densities were not significantly correlated with these parameters ($P=0.36$). The high correlation between the total density of the benthos with changes in the physical and chemical environment reflected the flooding of the marsh area at the upstream end of the upper lake. The lake surface area increased by 32 percent or 65 hectares. This increase was caused by a rise in water level and resulted in an influx of organic material that provided food for the benthic organisms. Mean nitrate concentrations in the upper lake between the pre- and postoperational periods rose from 66.7 to 119.8 μg/L.

In the lower lake, the density of the chironomids and nitrate concentrations and water surface area were negatively correlated ($P<0.01$); however, clam density was not correlated with either parameter ($P>0.05$) and oligochaete density only with water surface area ($P=0.01$). This finding indicated that the general increase in water depth and water surface area has not had as substantial an impact on the benthic fauna of the lower lake as it has had on the upper lake. In supporting this conclusion, the mean nitrate concentration in the lower lake (reflecting the contribution of organic material) increased from 35.9 to 77.5 μg/L, and the increase in lake surface area was only 0.4 percent or 3 hectares.

The data show a general increase in the combined chironomid plus oligochaete density and the associated mass in the upper lake with a decrease in the combined chironomid plus oligochaete density in the lower lake following project operation. Mean total mass of these organisms in the lower lake increased—indicating the average weight of individual organisms was greater.

This study has shown that there has not been any substantial overall impact on the benthos caused

FIGURE 9-5.—*Sediment mud sample collected at station 2.*

by project operation. However, the operation could conceivably impact chironomid densities, particularly in the lower lake, if the hydraulic retention time for water in the lake is reduced during the period of chironomid pupation and emergence. The reduced retention time may be both a result of normal pumped-storage powerplant operation as well as increased flushing rate which may impact mature chironomids in the lake. Fraley et al. (1989) stated that dam operation can result in direct washout of plankton in the water column as well as reductions in the standing crop of benthic organisms and insects on the water surface. LaBounty and Sartoris (1976) observed that a hatch occurred in June and concluded that the chironomid larvae of Twin Lakes are univoltine. As a general comparison, the retention time for water in the lower lake was 67 days in June 1980 before project operation and 53 days in June 1984 following full project operation. The exchange of water between Mt. Elbert Forebay and the lakes from operation would be considerably more frequent. Future demands for water and power will impact these retention times or flushing rates to a large extent.

Chironomid densities increased in the lower lake by 1985 (table 9-1) following full project operation, which could indicate that flushing is not a severe problem and the necessary food base for bottom-feeding fishes therefore should continue to thrive. Additional information is needed to support this apparent trend.

Water-level elevation in both lakes, but particularly the upper lake, is important in controlling the input of allocthonous material into the aquatic ecosystem. If the water level was lowered in the upper portion of the upper lake so as to allow drying or freezing of the sediment, many of the benthic organisms would be lost from the impacted area (Kaster and Jacobi, 1978). This influx of organic material from the recently inundated areas is similar to what would be expected from the "new reservoir phenomenon" in which the productivity of a newly filled reservoir increases rapidly to a relatively high level before reaching a lower level of equilibrium.

It seems likely that an increase in mixing and circulation caused by powerplant operation would have had a positive impact on the dissolved oxygen in the bottom waters of both lakes, particularly during the ice-on period. There has not been a significant ($P>0.01$) overall change in dissolved oxygen in either lake, as the annual means have risen slightly in the lower lake (5.8 to 5.9 mg/L) and have been reduced in the upper lake (6.0 to 5.5 mg/L). The mean dissolved oxygen concentration near the bottom of each lake—before and after project

Table 9-1.—Mean annual abundance and dry mass of benthos collected—pre- and postoperational periods.

Year	*Pisidium* (clams) No./m²	Chironomid larvae No./m²	Oligochaetes No./m²	Chironomids and oligochaetes No./m²	Chironomids and oligochaetes kg/ha
			Upper Twin Lakes		
Preproject operational period					
1977	29	nd[1]	nd	2333	0.20
1978	14	757	1868	2625	0.14
1979	nd	997	1456	2453	0.17
1980	61	1128	1641	2769	1.13
1981	36	691	1702	2393	1.69
$\overline{\chi}$	35	893	1667	2515	0.67
Postproject operational period					
1982	125	260	1541	1801	0.84
1983	34	670	1736	2406	0.49
1984	323	976	3520	4496	1.54
1985	211	578	2372	2950	1.22
$\overline{\chi}$	173	621	2292	2913	1.02
			Lower Twin Lakes		
Preproject operational period					
1977	511	nd	nd	1357	0.35
1978	275	795	449	1244	0.15
1979	189	3000	688	3688	0.33
1980	538	1784	485	2269	2.62
1981	698	1670	538	2208	1.26
$\overline{\chi}$	442	1812	540	2153	0.94
Postproject operational period					
1982	689	759	535	1294	0.98
1983	1234	810	364	1174	1.09
1984	1197	943	325	1268	1.25
1985	666	2225	223	2448	1.93
$\overline{\chi}$	947	1184	364	1546	1.31

[1] nd denotes no data. $\overline{\chi}$ denotes mean.

operation—shows this to a greater degree during the ice-on winter months. For example, values in the lower lake increased from 4.7 to 5.4 mg/L while values in the upper lake decreased from 4.3 to 3.6 mg/L. The lowest value that occurred in the upper lake following project operation was 0.85 mg/L. This reduction in dissolved oxygen concentration in the upper lake is probably due to the input of organic material from the inundated marsh area creating a greater oxygen demand in the sediments. It appears much less mixing of oxygenated water occurs in the upper lake than in the lower lake; however, development of anaerobic conditions in the upper lake should be precluded by normal powerplant operation during the winter months by both reducing the ice cover in addition to some increase in circulation.

Many environmental factors affect the type and abundance of benthic fauna. According to LaBounty and Sartoris (1976), if the trophic status of Twin Lakes was to be identified using benthic fauna only, the upper lake would be classified oligotrophic and the lower lake mesotrophic. This classification, of course, is an oversimplification of most aquatic ecosystems due to a variety of limnological factors. Brinkhurst (1974) cited numerous classification schemes for lakes based on physical, chemical, and biological data, including benthic fauna. Although it has not been successfully demonstrated that these classifications can be used universally; Valle's (1927) lake typology classification—using benthic organisms—indicates the lower lake is slightly more productive than the upper lake, and the general classification of these lakes by LaBounty and Sartoris (1976) is reasonable.

* * * * *

Zooplankton

Chapter 10

Davine M. Lieberman

Bureau of Reclamation
Applied Sciences Branch
Denver, Colorado

INTRODUCTION

Zooplankton play an important role in a lake's ecosystem. Zooplankton are secondary producers that transfer energy from the lower trophic levels—the primary producers—to the consumers. A simple aquatic system depends on the zooplankton community, an integral part of the food web, as nourishment for other zooplankton and fish. The food web interaction is influenced by both predation and availability of food, as well as the physical and chemical components of the lake. Twin Lakes are oligotrophic, montane lakes. The food web is fragile and uncomplicated, and it may be altered by a sudden change in water quality. For example, in April 1975, when heavy metals were released from the bottom sediments by anaerobic conditions in the hypolimnion of the upper lake, both benthos and plankton suffered high mortality (Sartoris et al., 1977). A great diversity does not exist in the biota of Twin Lakes; therefore, the organisms present must withstand or adapt to various environmental changes such as spring runoff from snowmelt, introduction of exotic species, water quality perturbations, and the effects of operating a pumped-storage hydroelectric powerplant.

Prior to these studies, little work was performed on the zooplankton of Twin Lakes. Juday (1906) studied the zooplankton of Twin Lakes from 1902 to 1903, and found a more diverse littoral and limnetic zooplankton community than has been documented in recent years. Notably, Juday collected 10 species of cladocerans that were primarily littoral forms and have since disappeared from the lakes. Cladocerans are only infrequently collected now and rarely in open waters. Fifty-two years after Juday's study, Reed and Olive (1958) studied the effects of altitude on the distribution of small crustaceans in Colorado; their report included a species list for Twin Lakes. This species list is now outdated and incomplete. The common zooplankton from the limnetic waters of both the lower and upper lakes have been collected and identified since 1977; a total of 11 species was found. The paucity in the number of species is typical of the limnetic communities of small-to-medium-size lakes in Colorado (Pennak, 1957).

This chapter describes the zooplankton composition, seasonal trends, densities of Upper and Lower Twin Lakes, and the changes that occurred in zooplankton population between the Mt. Elbert Pumped-Storage Powerplant preoperational (1977–81) and postoperational (1982–85) periods. Little information is available on pumped-storage effects on zooplankton populations in the literature.

METHODS

Zooplankton samples were collected regularly from 1977 through 1985 at two stations in Twin Lakes: one located in the upper and one located in the lower lake. A No. 20 (0.076-mm mesh opening) Birge-style closing net and bucket was used for vertical haul collections. Two replicate samples were collected from the surface to 1 m above the bottom at 5-m intervals. Eight samples were collected from each lake during each field survey. Samples were transferred from the plankton bucket to 125-mL polyethylene plastic bottles and preserved with sufficient formalin to yield a 2 to 4 percent solution.

Zooplankton were identified to species or lowest possible taxa and enumerated. Three 1-mL subsamples were pipetted from each replicate onto individual Sedgwick-Rafter counting chambers. The entire slide was counted according to the procedure described by Armitage (1961). The three subsample counts were averaged for each replicate and depth interval; these data were totaled to represent the entire water column. Copepods were divided into either a nauplii- or copepodid-adult group. Only the copepodid-adult group was separated further into genera and species. Immature stages were grouped together with the adult forms when counting the rotifers and cladocerans.

The zooplankton concentration, n, is the number of zooplankton per liter of water, and is calculated from the equation (Welch, 1948):

$$n = \frac{c\,(a \times 1000)}{L} \tag{7-1}$$

where:

a = average number of zooplankters in all counts in counting units of 1-mL capacity,
c = volume of original concentrate, mL, and
L = volume of original water filtered or vertically hauled.

The volume of original water hauled was calculated from the equation (Edmonson and Winberg, 1971):

$$L = \pi r^2 h \tag{7-2}$$

where:

r = radius of the net, cm, and
h = length of haul, m.

The microscopes used for enumeration and identification were: Leitz-Wetzler Ortholux compound light microscope, Zeiss-light microscope with a differential interference contrast lighting system, and Bausch & Lomb, Inc., binocular microscope.

The taxonomic keys used in the identification of zooplankton included: Needham and Needham (1962), Ward and Whipple (1966), Pennak (1978), Dumont and Green (1980), and Dodson (1981).

Statistical methods used to analyze zooplankton data were done by student's t-test and analysis of variance (ANOVA) with Tukey grouping.

RESULTS AND DISCUSSION

The zooplankton species collected from Twin Lakes, which can be described as cosmopolitan, included three species of cladocera, three species of copepods, and five species of rotifers. The lack of species diversity reflects the simplicity of the food web in the lakes.

Copepod nauplii and copepodid-adult copepods were usually the dominant zooplankters for the duration of this study. The three cladoceran species were not commonly collected and generally they comprised less than 1 percent of the total zooplankton population (LaBounty and Sartoris, 1982). At the turn of the century, Juday (1906) found that cladocera comprised from 20 to 45 percent of the adult zooplankton population. Juday's work (1906) included the littoral zooplankton population, which is often more diverse than the limnetic population. Juday also collected and identified:

- 10 species of cladocera
- 5 species of copepods
- 6 species of rotifers

for a total of 21 species of zooplankton, which is in contrast to the small number of species now present in the limnetic waters of Twin Lakes (table 10-1).

Apparently, changes in the species diversity and composition have occurred since the turn of the century. Only three cladoceran species were collected during these studies: *Daphnia pulex*, *Daphnia rosea*, and *Bosmina longirostris*. The probable cause for this change in cladoceran composition was the introduction in 1958 of opossum shrimp, *Mysis relicta*; thereafter, the cladoceran population in Twin Lakes was almost decimated.

- *Bosmina longirostris* was collected from both the upper and lower lakes, and Mt. Elbert Forebay.
- *Daphnia pulex* and *Daphnia rosea* were not found in the upper lake and were only rarely collected in the lower lake.
- *Daphnia* spp. were commonly collected in Mt. Elbert Forebay prior to mid-1981, before Mt. Elbert Pumped-Storage Powerplant began "pump-back" operation (Bergersen and Maiolie, 1981).

Before mid-1981, water diverted from Turquoise Reservoir into Mt. Elbert Forebay was used only for power generation, with no return of water from Lower Twin Lakes to Forebay. When initial testing began on the first 100-MW pump-turbine unit in both pumping and turbine modes, water pumped from the lower lake introduced *Mysis relicta* into Mt. Elbert Forebay. Thereafter, predation by *Mysis relicta* reduced the *Daphnia* spp. population in Forebay and, as a result, the *Mysis* sp. rapidly increased (Maiolie, 1987). Goldman et al. (1978) described a similar situation that occurred at Lake Tahoe, California, where declines in *Daphnia* abundance were reported following the introduction of *Mysis relicta.*

The seasonal trend for total zooplankton densities in the lower lake usually showed a late spring/early summer maximum and secondary fall peak. This general trend remained fairly constant during both

Table 10-1.—Comparison of zooplankton species collected from Upper and Lower Twin Lakes, Colorado.

1902-03 (Juday, 1906)		1977-85
	ROTATORIA	
Anurea aculeata Ehrenberg		*Asplanchna* sp.
Anurea cochlearis Gosse		*Brachionus* sp.
Asplanchna sp.		*Kellicottia longispina* Ohlstrom
Notholca longispina Kellicott		*Keratella cochlearis Bry* de St. Vincent
Polyarthra sp.		*Polyarthra vulgaris*
Triarthra longiseta Ehrenberg		
	COPEPODA	
Cyclops albidus Jurine		*Diacyclops bicuspidatus thomasi* Forbes
Cyclops pulchellus Koch		*Diaptomus connexus* Light
Cyclops serrulatus Fischer		*Diaptomus judayi* Marsh
Cyclops viridis americanus Marsh		
Diaptomus judayi Marsh		
	CLADOCERA	
Alona affinis O. F. Muller		*Bosmina longirostris* (Mueller)
Alona guttata Sars		*Daphnia pulex* Leydig
Camptocercus rectirostris biserratus Schoedler		*Daphnia rosea* Sars
Chydorus sphaericus O. F. Muller		
Daphnia hyalina Burckhardt		
Drepanothrix dentata Euren		
Eurycercus lamellatus O. F. Muller		
Graptoleberis testudinaria Fischer		
Latona setifera O. F. Muller		
Pleuroxus procurvatus Birge		

the preoperational and postoperational periods (figs. 10-1, 10-2). In the lower lake, copepods were present in collections throughout the year.

Copepod nauplii densities peaked in the late spring, following spring turnover of the lower lake (figs. 10-3a, 10-4a). Both *Diaptomus connexus* and *Diaptomus judayi* peaked in the fall during preoperational years and then shifted toward spring/summer peaks during the postoperational period (figs. 10-3b, 10-4b). Densities of the dominant copepod *Diacyclops bicuspidatus thomasi* displayed three to four peaks per year, generally in the winter, summer, and fall (figs. 10-3b, 10-4b).

The cladocerans that were collected in the lower lake were few; these populations were dominated by *Bosmina longirostris*. *Daphnia pulex* and *Daphnia rosea* were rarely collected. It is difficult to determine when the cladocerans reached a peak because of the limited number of organisms found in the samples (figs. 10-3c, 10-4c). However, more cladocerans seemed to occur in late summer and early fall seasons.

Rotifer densities in the lower lake generally exhibited bimodal peaks in the early summer and fall (figs. 10-3d, 10-4d). Rotifer populations were dominated by *Keratella cochlearis* during the summer and fall months and *Kellicottia longispina* during the winter months. *Polyarthra vulgaris*

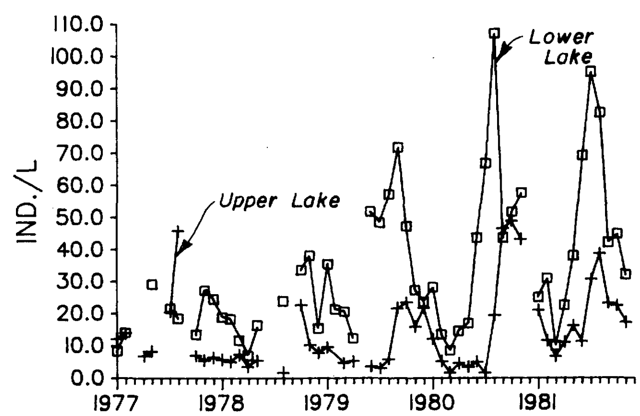


FIGURE 10-1.—*Comparison of monthly zooplankton densities in the lakes for the preoperational period.*

occurred all year. *Brachionus* sp. were most abundant during the fall months, and *Asplanchna* sp. occurred sporadically in the summer and fall.

The greatest total zooplankton densities in the upper lake usually occurred in late summer or early fall, lagging behind the late spring/early summer peak in zooplankton densities in the lower lake. Copepod nauplii were most abundant during the summer and fall months (figs. 10-5a, 10-6a). In the upper lake, *Diacyclops* sp. and *Diaptomus* spp. densities often followed similar seasonal cycles, with

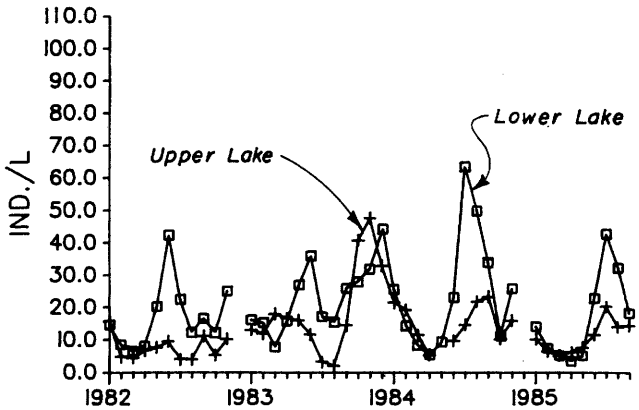


FIGURE 10-2.—*Comparison of monthly zooplankton densities in the lakes for the postoperational period.*

one species occasionally peaking before or after the other. Major peaks occurred two to three times per year, usually during the winter, spring, and fall months (figs. 10-5b, 10-6b). Cladoceran densities peaked sporadically, and populations were dominated by *Bosmina longirostris* in the upper lake (figs. 10-5c, 10-6c). During 1984, the upper lake's water level was raised and the volume of water was increased. Cladoceran populations increased as a result of inundating nutrient-rich areas of the lake. Rotifer densities exhibited bi- and trimodal peaks and were most abundant in the spring, summer, and fall (figs. 10-5d, 10-6d).

Generally, Twin Lakes underwent two turnover periods: one in May, after the ice-off period, and one in October. During the preoperational period, the upper and lower lakes were influenced greatly by the volume of Lake Creek inflow into the upper lake. High flows flushed the nutrients and plankton from the upper and lower lakes; consequently, zooplankton densities were lower than those that occurred during low runoff years. Usually, flushing of the lakes began in June and lasted into July, depending on the volume of water going through the lakes. A change in the inflow pattern to the lakes occurred in 1981 with the additional inflow of water from the Fryingpan-Arkansas diversions entering the lower lake through the Mt. Elbert Conduit. Diversions from the Western Slope (of the Continental Divide—Rocky Mountains), by way of the Twin Lakes Tunnel, continued to enter the upper lake through 1983, after which the upper lake no longer received these diversions.

During the same time period, the powerplant began operation (1982), although it was confined almost exclusively to the generation mode. During 1984, pumping operation began on a more or less routine basis. To further complicate matters, the upper and lower lakes increased in volume by 140

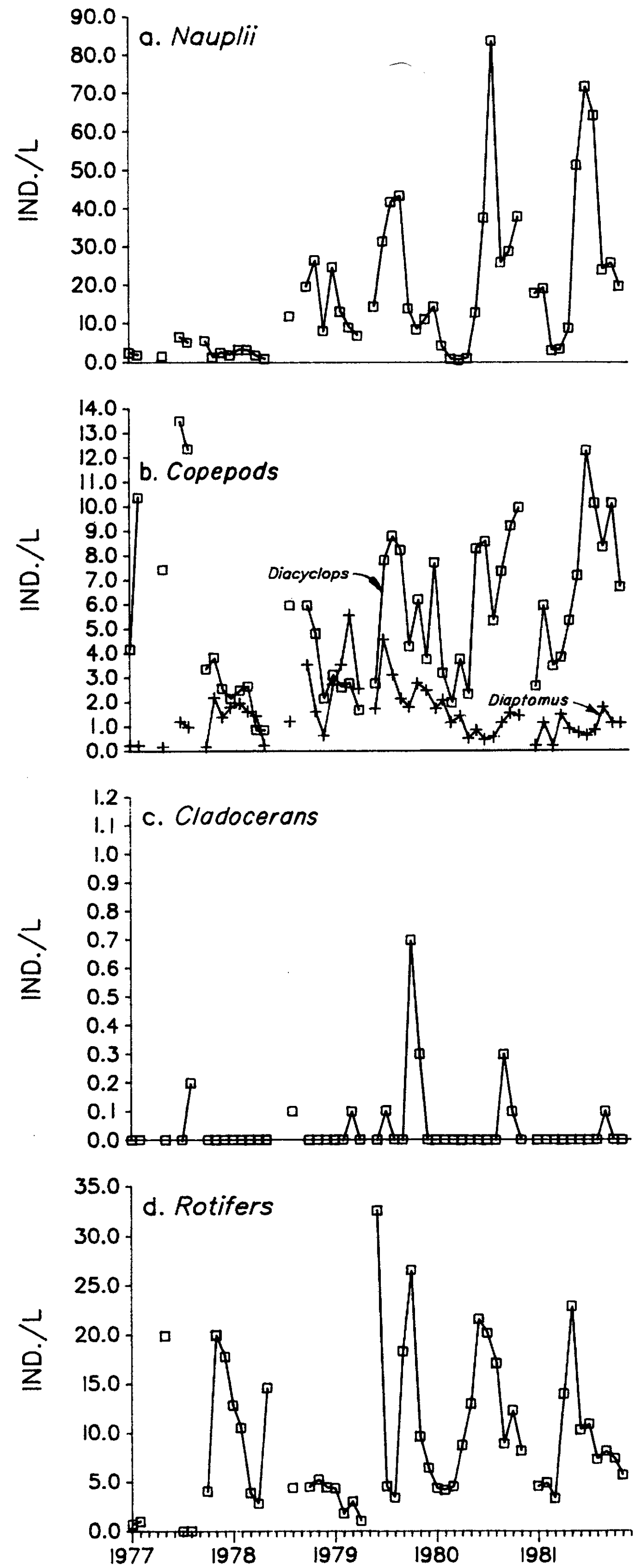


FIGURE 10-3.—*Nauplii, copepod, cladoceran, and rotifer densities in Lower Twin Lakes for the preoperational period.*

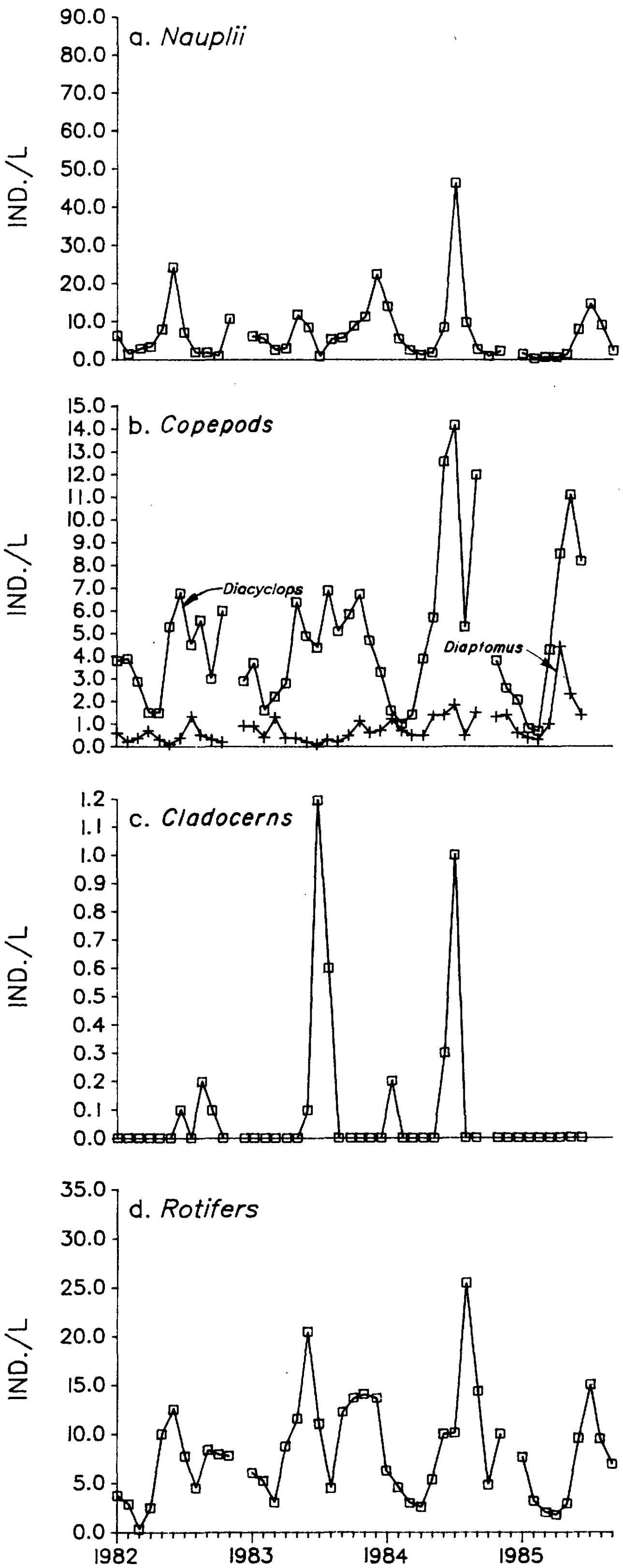


FIGURE 10-4.—*Nauplii, copepod, cladoceran, and rotifer densities in Lower Twin Lakes for the postoperational period.*

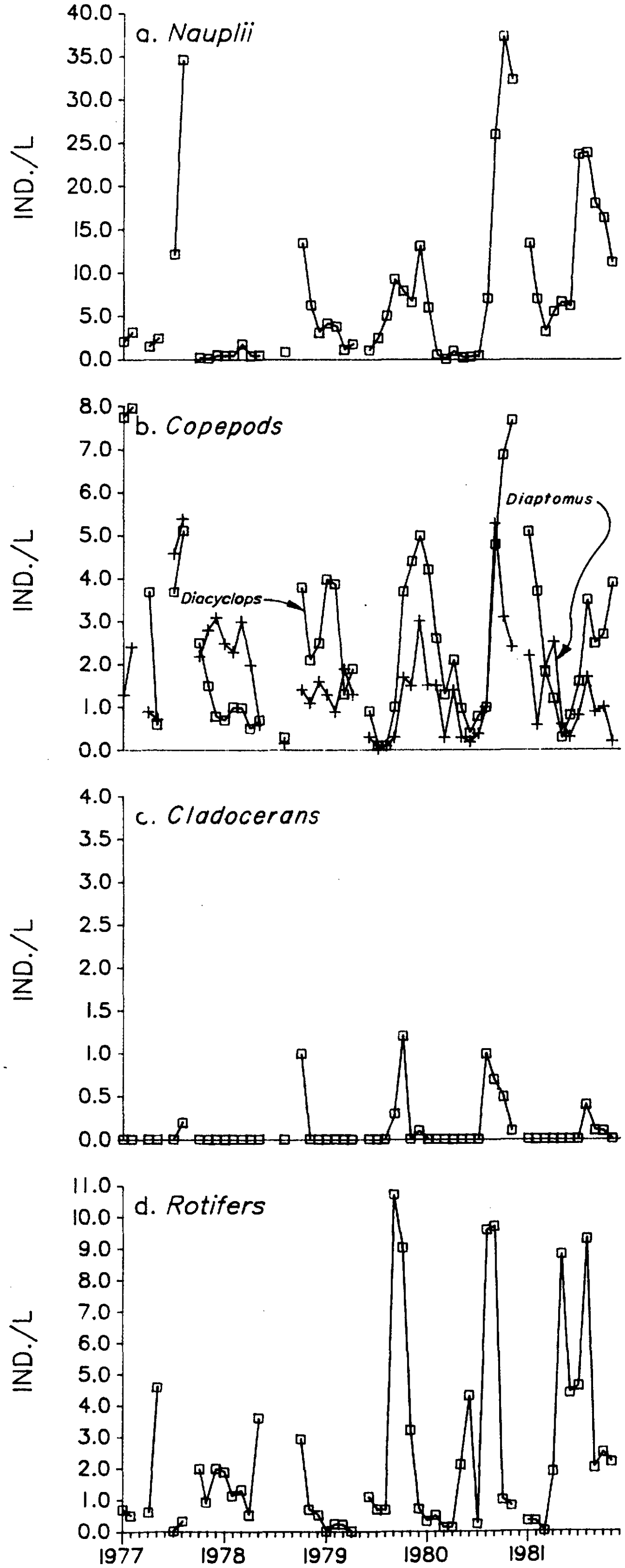


FIGURE 10-5.—*Nauplii, copepod, cladoceran, and rotifer densities in Upper Twin Lakes for the preoperational period.*

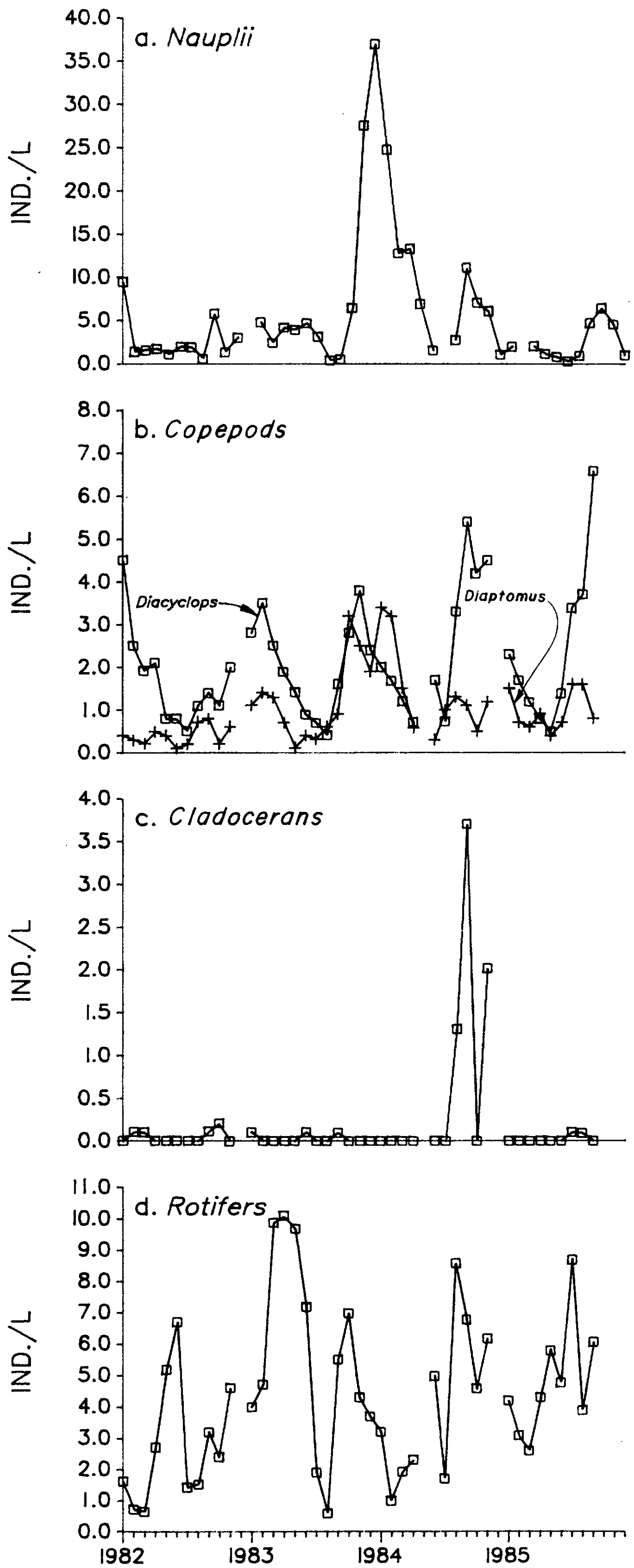


FIGURE 10-6.—*Nauplii, copepod, cladoceran, and rotifer densities in Upper Twin Lakes for the postoperational period.*

and 1 percent of the original volume, respectively. Thus, not only are the effects of powerplant operation to be considered, but changes in flow patterns and the raising of the lakes have to be taken into account.

For the lower and upper lakes, mean zooplankton densities were 36.1 and 15.3 individuals (ind.) per liter during preoperational sampling, and 21.0 and 13.2 ind./L during postoperational sampling. The zooplankton densities of the lower lake were significantly greater than the upper lake ($P<0.005$) in both the pre- and postoperational periods—due in part to flushing of the upper lake by spring runoff. Generally, both lakes' monthly hydraulic retention times decreased beginning in May (ch. 8, table 8-7). During the months of June 1977-85, the mean hydraulic retention time in the upper lake was less than 30 days. The mean hydraulic retention times for the lower lake, during the preoperational period in June, were:

- 175 days in 1977
- 62 days in 1978
- 56 days in 1979
- 67 days in 1980
- 105 days in 1981

The mean hydraulic retention times for the lower lake, during the postoperational period, in June generally were lower than the 1977-81 period:

- 46 days in 1982
- 65 days in 1983
- 53 days in 1984
- 60 days in 1985

The decrease in the mean hydraulic retention time, in turn, caused a decrease in zooplankton densities beginning in 1982. In the upper lake, rapid flushing caused a lag in seasonal development and lower summer zooplankton densities relative to the lower lake. Other investigators have reported that a mean flushing rate of less than 18 days may adversely affect zooplankton development and production (Hayward and Van Den Avyle, 1986). When the hydraulic retention time in the upper lake was less than 14 days during the month of June (i.e., 1978, '79, '80, '82, and '83), total zooplankton densities peaked in September or October; whereas when the retention time reached 23 to 30 days (i.e., 1977, '81, '84, and '85), zooplankton densities peaked earlier—during July or August.

Residence times of between 50 and 250 days have been found to be sufficient to allow the establishment of plankton populations that reflected the productive potential as well as effects of species interactions in the reservoir (Hayward and Van Den

Avyle, 1986). Thus, the short hydraulic retention time in the upper lake accounted for the significant difference in zooplankton densities between the two lakes during both the pre- and postoperational periods. Lower Twin Lakes benefitted from the spring flushing of the upper lake—trapping the nutrients which supported a June to August zooplankton bloom in all years.

Hydraulic retention time for the lower lake decreased somewhat between the pre- to postoperational periods, because of the additional inflow to the lower lake from the Fryingpan-Arkansas diversions. There was a significant decrease in total zooplankton densities from a mean of 36.1 ind./L, during the preoperational period, to a mean of 21.0 ind./L during the postoperational period. This decrease may have been caused, in part, by increased flushing. The hydraulic retention time calculated for the lower lake did not account for the volume of water conveyed back and forth from Mt. Elbert Forebay during pumped-storage operation. Maiolie (1987) estimated that a total of 1.37×10^9 cubic meters of water could be pumped per year, with the annual exchange volume being 12 times the volume of Lower Twin Lakes, or a volume equal to the lower lake being exchanged every 30 days during peak powerplant operation.

Unfortunately, zooplankton entrainment data for Twin Lakes are not available to indicate whether or not zooplankton densities were adversely affected by pumping operation. Some data from the literature suggest that zooplankton densities may even increase with "pump-back" operation (Potter and Meyer, 1982). Pumping operation caused a disruption in the thermal stratification, particularly in the lower lake. Other investigators have encountered similar circumstances (Buikema and Loeffelman, 1979). During the preoperational period, the duration of thermal stratification was reduced from 88 days to 39 days in the postoperational period. Reduction of thermal stratification may have potentially provided increased area in the water column for zooplankton habitat. However, both the mean water temperature and the euphotic depth decreased, thus limiting the proportion of the water column habitable by zooplankton. The mean water temperature in the lower lake decreased from 7.0 to 6.2 °C and the maximum temperature decreased from 9.1 to 8.5 °C, mainly from cooler inflow entering the lower lake and cooler air temperatures that occurred during the postoperational period.

Water temperatures often have been reported to be a controlling factor in the seasonal variation of zooplankton densities (Hodgkiss and Chan, 1976; Sharma and Pant, 1984; Stemberger and Gilbert, 1985). The euphotic depth in the lower lake significantly decreased ($P<0.005$) between the pre- (10.4 m) and postoperational periods (8.8 m). This decrease was probably caused by powerplant operation and change in the flow patterns. Increased vertical mixing in a reservoir often decreases the light intensity to which plankton are exposed (Sommer, 1985). Total seasonal zooplankton densities were highly correlated ($P<0.001$) with mean water temperatures during both the pre- and postoperational periods. Generally, as the temperature of the lake increased, total zooplankton densities increased.

Greatest annual zooplankton densities occurred in the lower lake during the preoperational period (i.e., 1979, 1980, 1981) (table 10-2). Mean densities reached a high of 45.0 ind./L in 1981 for the preoperational sampling and 24.7 ind./L in 1984 for the postoperational period sampling. Mean annual zooplankton density during 1982 was 17.1 ind./L—the lowest density reported for all sampling years except for 1985, which was not a complete sampling year. A combination of factors caused zooplankton densities to be reduced drastically in 1982:

- The hydraulic residence time reduced from an annual mean of 874 days in 1981 to 150 days in 1982.
- The maximum water temperature decreased from 9.95 °C in 1981 to 8.47 °C in 1982.
- The summer stratification in 1982 lasted only 15 days—as compared to 56 days in 1981 and 85 days in 1983.
- The total inflow doubled from 1981 to 1982; the volume of water going out of the lakes increased by 78 percent.
- The powerplant began operation in the pumping mode.

In 1983, mean annual zooplankton densities increased by 36 percent, probably because the powerplant did not operate from March through October and, as a result, the lower lake was stratified for 85 days. During the summer months, from June through August 1977-81, the total percent of *Diacyclops bicuspidatus thomasi* was distributed in the water column of the lower lake as follows:

- 50 percent from 0 to 5 m
- 30 percent from 5 to 10 m
- 14 percent from 10 to 15 m
- 6 percent from 15 to 20 m

There was not obvious migration of *Diacyclops* sp. to hypolimnetic waters during the summer months from 1982-85—even with the disruption in the thermocline period and the thermal regime. The greatest mean annual density of 6.98 ind./L occurred in 1981—a low runoff year. Densities of *Diacyclops*

Table 10-2.—Annual mean zooplankton densities, individuals per liter (ind./L) of all groups, in Lower and Upper Twin Lakes, Colorado.

Year	Copepod nauplii	*Diacyclops* sp.	*Diaptomus* spp.	Rotifers	Cladocerans	Total Zooplankton
			Lower Twin Lakes			
Preoperation period						
1977	7.31	7.20	0.81	3.14	0.12	18.58
1978	7.05	3.12	1.54	8.67	.02	20.39
1979	10.15	4.56	3.01	18.87	.10	36.68
1980	11.20	6.21	1.25	22.59	.03	41.29
1981	9.04	6.98	0.89	28.09	.01	45.01
Postoperation period						
1982	6.25	4.05	0.46	6.32	.03	17.12
1983	10.40	4.46	0.55	7.76	.15	23.32
1984	8.82	5.97	0.97	8.76	.14	24.66
1985	6.56	4.68	1.46	4.21	.01	16.93
			Upper Twin Lakes			
Preoperation period						
1977	1.28	3.45	2.75	6.32	0.39	14.66
1978	1.41	1.40	1.64	3.06	.12	7.63
1979	2.27	2.30	1.10	4.88	.13	10.67
1980	2.61	2.97	1.59	10.16	.20	17.53
1981	3.31	2.45	1.15	12.37	.07	19.33
Postoperation period						
1982	2.79	1.72	0.40	2.72	.04	17.67
1983	5.72	2.06	1.20	10.00	.03	19.01
1984	4.12	2.52	1.41	6.51	.70	15.27
1985	4.84	2.41	0.98	2.46	.03	10.71

bicuspidatus thomasi decreased significantly, from 5.86 to 4.76 ind./L between the pre- and postoperational periods (table 10-3), demonstrating that because of light and temperature limitations, the species were not able to use a greater proportion of the water column, as had been reported in other studies (Potter and Meyer, 1982). The vertical distribution of *Diaptomus* spp. appeared to shift to the lower depths during the postoperational period. During the summer preoperational period, the total percent of *Diaptomus* spp. was distributed in the water column as follows:

- 49 percent from 0 to 5 m
- 37 percent from 5 to 10 m
- 14 percent from 10 to 15 m

During the postoperational period, the total percent of *Diaptomus* spp. was distributed in the water column as follows:

- 28 percent from 0 to 5 m
- 49 percent from 5 to 10 m
- 20 percent from 10 to 15 m
- 3 percent from 15 to 20 m

Although *Diaptomus* spp. migrated to greater depths during the postoperational period, densities did not increase, rather they decreased significantly between the two periods—from 1.61 to 0.72 ind./L ($P<0.001$). *Diaptomus* spp. densities drastically decreased in 1982 and 1983, but seemed to rebound in 1984 and 1985 (table 10-2). The greatest percentage of rotifers was collected from the 0- to 5-m depth interval and depth distribution was similar throughout both periods. During the preoperational period, rotifer densities also decreased significantly from 19.06 to 6.99 ind./L during the postoperational period ($P<0.001$) (table 10-3). The significant decline in zooplankton densities between the pre- and postoperational periods in the lower lake indicates a combination of factors played a role in this reduction.

The upper lake, as noted previously, had significantly lower zooplankton densities than the lower lake during both sampling periods. Total zooplankton densities decreased from 15.31 to 13.15 ind./L between the pre- and postoperational periods (table 10-3). Total zooplankton densities, during the preoperational period, were correlated positively with mean water temperatures ($P<0.001$). During the postoperational period, however, total zooplankton densities did not correlate with water temperature. Copepod nauplii densities increased significantly from 2.33 to 4.45 ind./L ($P<0.001$); whereas the copepodid-adult densities decreased from 4.08 to 3.16 ind./L, which was not significant. Rotifer

Table 10-3.—Mean zooplankton densities, individuals per liter (ind./L), and standard deviation(std. dev.) for preoperational (1977-81) and postoperational (1982-85) periods in Upper and Lower Twin Lakes, Colorado.

Zooplankton group	Lower Twin Lakes 1977-81 Mean	Lower Twin Lakes 1977-81 Std. dev.	Lower Twin Lakes 1982-85 Mean	Lower Twin Lakes 1982-85 Std. dev.	Upper Twin Lakes 1977-81 Mean	Upper Twin Lakes 1977-81 Std. dev.	Upper Twin Lakes 1982-85 Mean	Upper Twin Lakes 1982-85 Std. dev.
Copepod nauplii	9.49	8.71	8.46	5.91	2.33	3.08	† 4.45	2.95
Diacyclops sp.	5.86	3.82	‡ 4.76	3.20	2.42	1.99	2.17	0.36
Diaptomus spp.	1.61	1.12	† 0.72	0.72	1.66	1.58	0.99	0.43
Rotifers	19.06	20.37	† 6.99	7.84	8.72	10.91	†5.34	8.40
Cladocerans	0.03	0.03	0.08	0.08	0.18	0.18	0.20	0.20
Total Zooplankton	36.05	24.73	† 21.01	13.38	15.31	14.21	13.15	10.75

† *t* -test shows significant difference between 1979 and 1981, and 1982-85 where $P<0.0001$.
‡ Significant difference where $P<0.05$.

densities decreased significantly from 8.72 to 5.34 ind./L. The mean annual zooplankton density reported for 1982 was 7.67 ind./L—a decrease of 60 percent from 1981. In 1982, mean densities of *Diacyclops* sp. were reduced by 30 percent, *Diaptomus* spp. by 65 percent, and rotifers by 78 percent relative to 1981. The decline in the upper lake zooplankton population in 1982 can be attributed to the increase in the flushing rate during the winter months of 1982, which continued on through the summer and fall.

The duration of stratification in the upper lake in 1982 lasted 30 days as compared to 51 days in 1981 and to 83 days in 1983. In 1983, between March and September, electrical generation did not occur. The mean annual zooplankton density increased from 1982 to 1983 by 2.5 times. This increase also was observed for the opossum shrimp, *Mysis relicta*, which increased in density from 1982 to 1983 by three times (Maiolie, 1987). The greater stratification period during the growing season contributed to a regain of population densities. Maiolie (1987) found that the lack of powerplant operation in 1983 may have strongly influenced the increase in the mysid densities in the upper lake, even though water was withdrawn from the lower lake through flow changes caused by project operation. Even so, zooplankton densities followed a similar trend as the opossum shrimp densities in the upper lake; it can only be speculated that the decrease in flushing rate and the lack of summer powerplant operation had a direct effect on increased zooplankton densities in 1983.

In 1984 the level of the lakes was raised, a 130-hectare marshy area (west of the lakes) was inundated, and the upper lake's volume increased by 140 percent of the original volume. At the same time, mean zooplankton density decreased from 19.0 (1983) to 15.3 ind./L (1984). This decrease in density was due to dilution because total zooplankton numbers in the upper lake actually increased from 0.636×10^{12} (1983) to 0.744×10^{12} (1984). Total zooplankton numbers were greater during 1984 than during any other sampling year.

The increased zooplankton numbers can be attributed to rapid colonization of the newly inundated land, with its abundance of plant nutrients and organic debris. During this period, an increase in ammonia nitrate nitrogen and total phosphorus concentrations occurred in the upper lake (ch. 5). The influx of nutrients often occurs in newly impounded reservoirs—causing an initial phase of high biological productivity (Goldman and Kimmel, 1978; Potter and Meyer, 1982). As habitat expansion ceases and readily available internal supplies of nutrients and food decline, the biological productivity also decreases until an equilibrium phase is finally reached (Baxter, 1977). Mean total zooplankton numbers did decrease during 1985 to 0.569×10^{12} but this may have reflected the shorter sampling period rather than actual zooplankton numbers.

CONCLUSIONS

The significant decrease in *Diacyclops bicuspidatus thomasi*, *Diaptomus* spp., rotifers, and total zooplankton densities that occurred in the lower lake between the pre- and postoperational periods was caused by a combination of factors. During the postoperational period, increased inflow to the lower lake—from Western Slope diversions—plus powerplant operation (in the pumping mode) caused a reduction in duration of summer thermal stratification and decreased the euphotic zone significantly that, in turn, decreased the availability of habitat for the zooplankton population.

Hydraulic retention time greatly influenced the upper lake; it was the major factor controlling the zooplankton population, accounting for the significant differences in densities between the upper and lower lake in both the pre- and postoperational periods. Total zooplankton densities in the upper lake decreased between the pre- and postoperational periods, from raising of the lakes, which caused an increase in water volume and, in turn, diluted the zooplankton. Total zooplankton numbers actually increased in the upper lake, beginning in 1984, because of increased habitat availability and increased primary productivity.

* * * * *

Mysids

Chapter 11

E. P. Bergersen, R. E. Gregg, M. A. Maiolie, with K. Hamilton

Colorado Cooperative Fish and Wildlife Research Unit
Colorado State University
Fort Collins, Colorado

INTRODUCTION

Mysis relicta, commonly called opossum shrimp, is a large North American zooplankter native to cold, deep, oligotrophic lakes in regions that were once glaciated. They are mainly benthic dwellers feeding on organic materials on the surface of the bottom mud. However, during vertical migrations they feed on smaller zooplankton. In its native habitat, *Mysis* is eaten by benthic and deepwater fish such as coregonids, char, and sculpins.

In the belief that *Mysis* could become a major food source for cold water fish in lakes with limited forage and that they could cause an increase in fish growth, mysids were stocked in lakes of western North America. Several hundred mysids from Clear Lake, Minnesota, were stocked into Twin Lakes October 5, 1957. After about 10 years, the *Mysis* population in both the upper and lower lakes became abundant enough to appear in plankton samples and lake trout (*Salvelinus namaycush*) stomachs. By 1970, the population was so abundant that it became the source of stocking programs in Colorado and several other states.

Because *Mysis* had become an important part of the diet of lake trout, the possibility of impacts on the *Mysis* population due to entrainment—during Mt. Elbert Pumped-Storage Powerplant operation—concerns ensued over the lake trout fishery. Consequently, studies began in 1974 to describe the natural history of *Mysis* in Twin Lakes, and in 1979 to determine impacts of entrainment on this population. Gregg (1976) investigated vertical migration patterns, life history, and distribution patterns as well as the effects of turbulence and turbidity on the shrimp. Entrainment studies conducted by Maiolie (1987) are discussed in chapter 14.

METHODS

Natural History

Vertical migration.—The vertical distribution of *Mysis* was examined by taking four simultaneous samples: one at the surface, two at intermediate depths, and one near the bottom. In 1974 and 1975, sampling was accomplished by attaching three nets and a benthic trawl to a single cable. Nylon nets, tapered from a 500-mm diameter mouth to a 125-mm diameter cod-end, were constructed of 0.76-mm mesh.

A benthic trawl was constructed of aluminum tubing, and the sampling net was sewn to a rectangular aluminum frame. A round, aluminum, deflection skid was attached across the front of the trawl to prevent it from digging into the soft sediments. A strip of heavy canvas was attached from the trawl frame backwards to prevent the netting from snagging or tearing on debris. Flowmeters were suspended in the mouth of the nets with elastic bands. The sampling boat was equipped with a winch, metering wheel, and inclinometer. The depth at which each net would sample could be determined from the cable angle and the length of cable released. A preliminary haul would determine the approximate cable angle for a measured boat speed. Using this angle and the depth indicated by fathometer, the nets could then be attached so that the trawl was on the bottom, the first net was at two-thirds of total depth, the second net was one-third total depth, and the third net was slightly beneath the water surface.

The nets and trawl were pulled at a measured speed for 10 minutes. Samples were taken before and after sunset, every 2 to 3 hours during the night before sunrise, and after sunrise. These samples were taken once during September, October, and November 1974, and June, July, and August 1975. The entire sample from each net was preserved independently in 10 percent formalin and returned to the laboratory for analysis.

Winter samples were taken during one 24-hour period in each month when sufficient ice covered the lake. During freeze-up in December and spring thaw in May, unsafe ice conditions made sampling impossible. Net hauls were made across a single transect laid beneath the ice (Gregg, 1976). Representative diagrams of vertical movements are provided in this study, but a complete set of diagrams is in Gregg (1976).

Life history.—Because individual net samples collected from 1 to over 18 000 mysids, subsampling was necessary to handle the large number and volume of samples. Small samples were analyzed entirely, whereas large samples were subsampled with a modified Waters subsampler (Waters, 1969). Following subsampling, mysid size and sex data were recorded for each separate net sample in 1974 and 1975. Size and sex data were obtained from mysids collected in November 1983 and November 1985.

Mysids were measured from the tip of the rostrum to the tip of the telson—excluding setae. Size was recorded in 1-mm categories: mysids from 10.00 to 10.99 mm were recorded as 10 mm. Mysids were classified into five categories according to sex characteristics:

- Juvenile,
- Immature male,
- Immature female,
- Mature male, and
- Mature female.

External characteristics of mysids, which were used to distinguish these categories, were described in Gregg (1976). Female mysids in mature breeding condition and the stage of development of the brood were recorded. Berrill (1969) gave a detailed account of the development of the embryos of *Mysis relicta*. In the Twin Lakes study, four stages of brood development were noted:

- Eggs present,
- Embryos present and elongated,
- Embryos with appendages present, and
- Larvae present.

Larger females that had previously released the brood were also noted. Population structure, recruitment periods, life cycle, and growth were determined by analyzing monthly size-frequency distributions. The portion of each size-class represented by different sex and maturity classes was calculated to give a size and sex frequency analysis at the surface, two intermediate depths, and the bottom.

Population distribution.—The purpose of this portion of the study was to determine the distribution of the *Mysis* population and to detect spatial and temporal changes in mysid density. These characteristics of the population were analyzed using a monthly series of benthic trawls taken from Upper and Lower Twin Lakes in 1974 and 1975.

Traverse lines extending the length of both the upper and lower lakes were established and divided into equal sections. During each monthly sampling period, one sample was collected from a randonly chosen length of each section of the traverse lines. In addition, three samples were collected at random from areas greater than 15 m deep in Lower Twin Lakes, along with one sample from an area of depth greater than 15 m in the Upper Twin Lakes. The samples were collected with a large benthic trawl having a mouth width of 1.5 m. The trawl was similar to and operated in the same manner as the trawl described in the vertical migration work. Observations of *Mysis* distribution were made during a series of dives by the Bureau of Reclamation's Lower Missouri Regional Dive Team during June 1974, August 1974, and June 1975 to substantiate findings.

Abundance.—A phototrawl, consisting of a bottom sled trawl supporting an underwater camera, was developed to aid in documenting changes of shrimp abundance in Upper and Lower Twin Lakes and of Mt. Elbert Forebay (fig. 11-1). The trawl was 1.5 m long, 1.3 m wide, 0.7 m high, and was constructed of 19-mm diameter aluminum tubing. Skids, 150 mm wide by 1150 mm long, flanked each side of the trawl and prevented it from sinking into the bottom. A removable net, frame, and scoop could be attached to the center of the phototrawl—allowing shrimp samples to be collected. The net was 1.6 m long, with 1-mm bar mesh, and had a mouth 454 mm wide by 288 mm high. Its cod-end terminated in a removable bucket. When towed along the lake bottom, the scoop preceded the net mouth and "pushed" the shrimp off the bottom into the net.

Supports were welded to the front of the trawl to hold a Canon model F1 camera having a 35-mm lens and autowinder—all in a waterproof housing. The camera was directed downward and capable to photograph 0.083 square meter of the bottom (0.90 ft^2). The camera was set at 1/60 second at *f*-13.5,

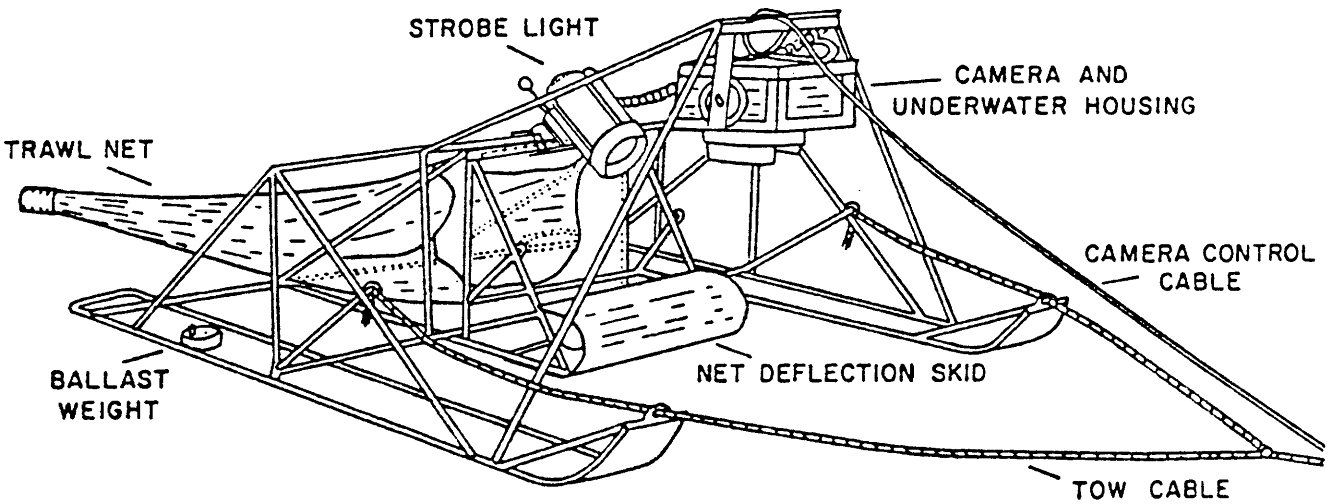


FIGURE 11-1.—*Phototrawl with camera equipment and removable net-scoop assembly attached.*

with the film plane 560 mm off the bottom, and used 100 ASA color print film. A Farallon Oceanic model 2001 strobe was mounted behind the camera housing. A remote shutter cable 75 m long activated the camera and enabled an entire roll of film to be exposed without servicing the camera.

A General Oceanics, Inc., model 2030 digital flowmeter was used to measure the boat speed and the distance trawled. Chart-recording depth-finders were used to record water depth and to ensure all trawling was conducted at depths greater than 15 m. An outboard powered boat equipped with a 75 m length of 2.4-mm diameter steel cable towed the phototrawl at 0.6 to 0.8 m/s. All trawls were conducted between 0800 and 1600 hours when the majority of shrimp were on the lake bottom, as found in the 1974-75 studies (Gregg, 1976). Additional tests in which a net was towed horizontally at depths as close as 100 mm above bottom reconfirmed that shrimp were lying close to the bottom during midday. During 1981 and 1982, trawls lasted 7 min 12 s, when 36 frames were exposed at 12-s intervals. Four trawls were run in the upper lake, five in the lower lake, and four in Mt. Elbert Forebay. Between 1983 and 1985, trawls lasted 3 min 36 s, with 18 frames exposed at 12-s intervals. Twice the number of trawls were made during these years; 8 were conducted in the upper lake, 10 in the lower lake, and 8 in Forebay. Position and direction of each trawl were chosen randomly, but they were stratified by lake area. All surveys were conducted after fall lake turnover when bottom oxygen was replenished.

Because photographic studies by Bergersen and Maiolie (1981) and net trawling by Gregg (1976) showed that mysids in Twin Lakes were clumped on the lake bottom, normality could not be assumed for the raw data. Log_e transformations (Elliott, 1977) were applied to the data:

$$y = \log_e (x + 1)$$

where x equals number of shrimp per photograph.

Statistical analyses were then applied to the transformed data y. Confidence intervals and differences between lakes were determined by one-way analysis of variance, with nested classifications for photographs, trawls, areas, and lakes (Steele and Torrie, 1960). Results of statistical analysis were converted to the number of shrimp per photograph by the reverse of the above equation (x = antilog y - 1, where y equals the transformed data). Then, number of shrimp per photograph was converted to shrimp per square meter by multiplying by 12.077. Geometric mean densities were calculated for each lake by $\log_e$ transformation instead of arithmetic mean, which is easily distorted by the skewed nature of the distribution. Total number of shrimp in each lake was estimated by multiplying the geometric mean density of shrimp by each lake's area:

- Upper Twin Lakes: 2 480 000 m^2,
- Lower Twin Lakes: 7 102 000 m^2, and
- Mt. Elbert Forebay: 1 092 000 m^2.

Differences were considered significant at the 5-percent level or less.

Because of the different sampling techniques used to gather abundance information since 1974, only data from 1980 to 1986—when the phototrawl was used—are compared in the analyses. Shrimp density estimates in 1974-75 were made with a large benthic trawl, but in 1977 and 1979 estimates were made with a smaller trawl. An estimated 58 percent of the sample was lost from the large trawl because the mesh size was too large to prevent juvenile escapement (Nesler, 1981). In 1977 and 1979, samples collected resulted in underestimated shrimp densities because the floating polyethylene tow rope frequently lifted the trawl off the bottom. The amount of time the trawl was off the bottom increased directly with depth and trawling speed (Bergersen and Maiolie, 1981).

Effects of turbulence and turbidity.—Twin Lakes mysids that had been acclimated to laboratory aquaria for several months were subjected to turbulence and turbidity in varying degrees over a period of 8 days to determine the increase in mortality. The trials were conducted as a factorial experiment using three levels of turbidity (0, 735 to 779, 1470 to 1558 American Public Health Association turbidity units) and four levels of turbulence ($K = 0, 1.9, 3.0, 5.2$). Experimental chambers—12, each containing 20 mysids—were used, each having one unique combination of the two factors. Chambers were 7-l, 150-mm diameter plastic cylinders. The cylinders stood upright in a fiberglass Min-O-Cool trough maintained at a constant temperature of 6 °C. Each cylinder was equipped with a drain at the bottom and a removable screen matching the inside diameter of the tube. The chamber having zero turbulence and zero turbidity served as the control for each trial. The turbulence was created by individual diaphragm aerators which were controlled by a timing mechanism. The turbulence began at 2400 hour and continued for 8 hours. After a 16-hour rest period, the cycle was repeated. The sediments causing turbidity began to settle out during the rest period and were resuspended during the turbulent period. The greater the turbulence, the more thoroughly the sediments remained in suspension. At a designated time on the 2nd, 4th, 6th, and 8th days, *Mysis* in each cell were examined for survival.

The turbidity was created with sediments collected from near the powerplant tailrace channel. Portions of these sediments were put into suspension by stirring with a magnetic device for 6 hours. After allowing 1 hour for the sediments to settle, supernatant was siphoned without disturbing the heavier sediments that had settled out. The supernatant contained a high concentration of fine particles that remained in suspension. This concentrated solution was used as the high turbidity level. The middle level was produced by diluting the concentrated solution one-to-one with well water. The zero turbidity level was well water. Turbidity was measured with a Hellige turbidity meter.

Turbulence was measured indirectly. The reoxygenation or diffusion constant K, was directly proportional to the amount of turbulence in a system and the amount of airflow; thus, a linear relationship between airflow (in cubic centimeters per minute) and K was established. The diffusion constant was used as a measure of turbulence and determined by the relationship between it and airflow, which was known (Gregg, 1976).

The data were analyzed with a four-way analysis of variance and a split-plot design to group between-cell variation due to main-factor interaction and replicate effects separately from within-cell variation due to repeated measure of the time factor. Turbulence and turbidity made up most of the main-effect variation. Part of the between-cell variation was due to replication effects as well as interactions with replications considered to be part of the experimental error (Winer, 1962).

A simple laboratory experiment was conducted to determine mysid reaction to turbulence. A fiberglass Min-O-Cool trough was maintained at 6 °C. The trough was marked lengthwise into one-quarter sections (cells). An aeration device or submersible pump was placed in the end section to create turbulence. The mysids were allowed 48 hours to acclimate and distribute themselves in the trough without turbulence; then, counts were made of the number in each cell. This nonturbulent condition served as the control and was repeated three times. The turbulence then was started. In each case, *Mysis* were left in the trough for 48 hours and then the number in each cell counted.

RESULTS

Vertical Migrations

Vertical migrations were found to occur during all open-water months, but the intensity of migrations varied between months. Migrations were greatest in June, July, and August, then decreased in September. Vertical migration increased in October, and by November the extent of migration within the population was only slightly less than that observed in the summer.

In June, juvenile mysids were found in the water column from the bottom to 9 m below the water surface by 1700 hours; adults did not begin to move off the bottom until about 2030 (fig. 11-2). Adults and juveniles were found throughout the water column at 2230. The numbers of mysids of all age groups off the bottom peaked at 0100. By 0500, 92 percent of the sample had been taken in the bottom trawl. The 2230 surface sample was dominated by

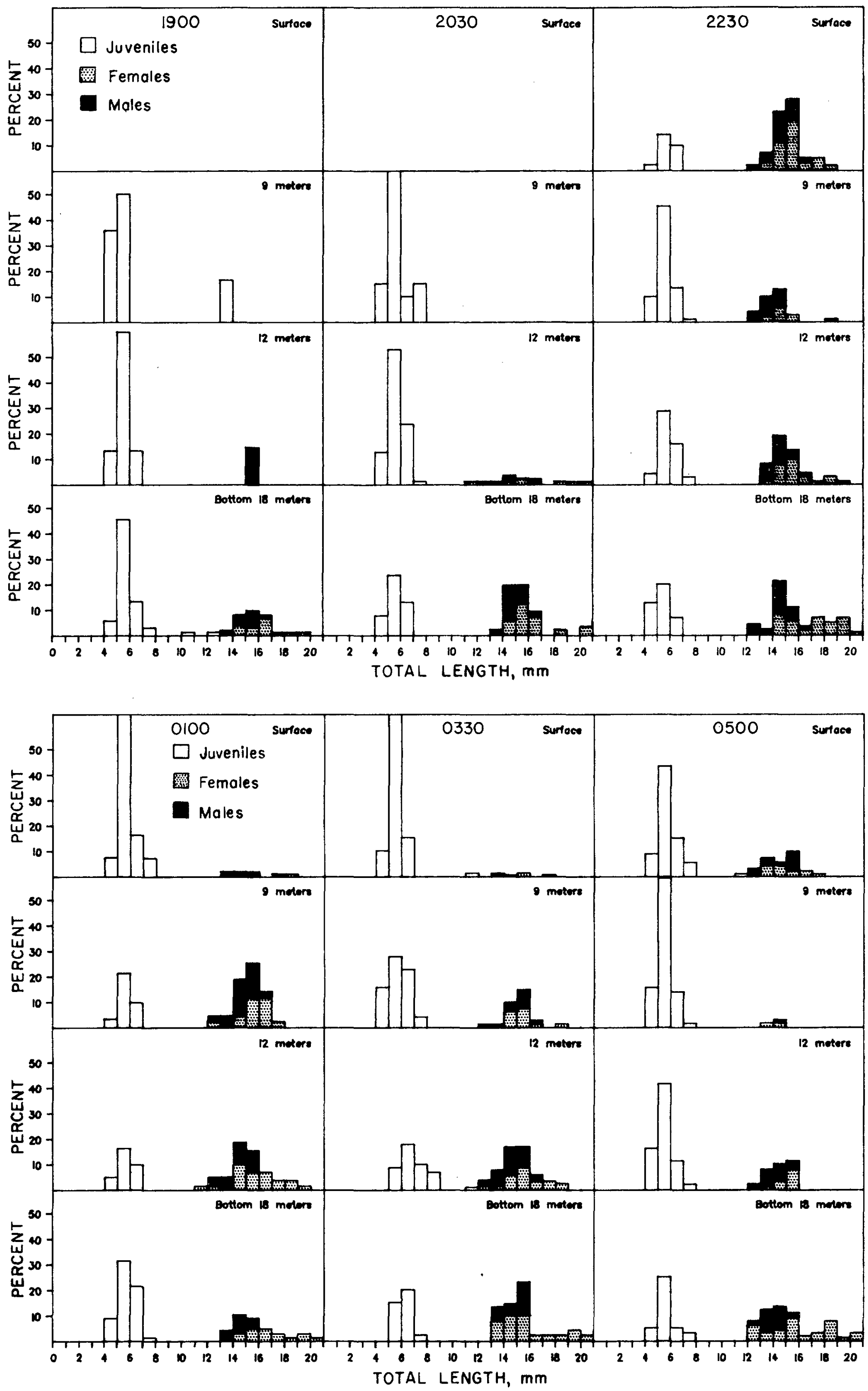


FIGURE 11-2.—*Size frequencies of* Mysis relicta *captured in individual net samples at four depths during the night of June 7-8, 1975.*

adults in contrast to surface samples taken at 0100 and 0300 hours, which were dominated by juveniles.

During July, longer days and later sunsets delayed the mysid vertical migration until about 2100. The migration began as it did in June; juvenile mysids reached the surface waters, remained there throughout the night, and were last to descend at dawn. Small numbers of migrating adults reached surface waters, but the majority of the mature mysids remained below the thermocline at 4.5 to 6 m. Migrations involved a large percentage of the population, and the average numbers of shrimp migrating were similar to those for June. A unique observation was made at 2400 hours in July 1975; the night was dark and the lake surface was calm. Mysids were observed swimming or resting at the lake surface; some actually contacted the surface film. When a beam of light was directed at a mysid, it would remain motionless for several seconds and then dart toward the bottom. Mysids used an abdominal reflex reaction to vanish from sight with a previously unobserved speed.

Lower Twin Lakes was strongly stratified in August 1975 and dissolved oxygen was beginning to be depleted in the hypolimnion. August vertical migrations began at 2030 hours. Again, large numbers of juveniles migrated first; adults did not reach the surface. Mysids were more numerous below the thermocline than above it. Only a small number of juveniles occupied the surface waters. The number of mysids reaching the surface was significantly smaller than in previous months. Mysids were not sighted on the lake surface even though the night was dark and the surface calm. Throughout the August sampling night, the number of captured mysids increased progressively with the depth of the sample. In all samples, the greatest percentage of shrimp occupied the benthic zone.

Vertical migration during September 1975 was less than at any other sampling time (fig. 11-3). The majority of the population remained on the bottom throughout the night. The greatest portion of the migration occurred immediately following sunset. Apparently, mysids entered the epilimnion early and retreated to depths of 6 m or more. Water temperature decreased at a constant rate from surface to bottom. A bottom temperature of 10.6 °C was near the yearly high for that depth. Bottom DO (dissolved oxygen) was becoming depleted, especially in the center of the lake where it was 2 mg/L; DO in the sample area did not drop low enough to affect *Mysis*. Size and sex composition of the surface population were similar to that of the bottom population. At dawn, juvenile mysids were the last to return to the bottom.

The lake had undergone fall overturn prior to the October sampling. The lake was isothermal at 8.0 °C and DO was greater than 7 mg/L at all depths. *Mysis* migrations increased during October. The early rapid rise to the surface noted in September was not evident; mysids were not taken at the surface before 2230 hours. At that time, mysids arrived at the surface and remained there until dawn. The density of mysids was greater in the water column and less on the bottom than in September. Mysids were found in greater numbers at 12 and 6 m depths than at the surface. Sex and size distributions were relatively constant, with a notable exception occurring in surface catches. Mature females dominated these samples, but many mature males were also present. These samples were collected during breeding season and may indicate a breeding swarm or an increased movement of mature mysids during the period. Mysids remained in the water column throughout the night and retreated at sunrise.

Water temperature during November was isothermal at 4 °C and DO was near saturation at all depths. Vertical migrations began immediately following sunset (fig. 11-4). The number of shrimp in the water column increased as the night progressed.

Only four samples were collected because of inclement weather. The greatest number of mysids taken from the water column was captured just before sunrise. Size and sex composition of the sampled population did not show any notable differences with time or depth. Most samples included representatives of juveniles, males, and females. During fall 1974, mysids were more abundant at all depths in the water column than they were during any other month of the year.

The results show light and temperature are important factors controlling mysid migrations, and agree with other observations (Beeton, 1960; Taraguchi, 1969; Brownell, 1970). Migrations coincided with changing light intensities, beginning progressively later as day length increased. Because thermal gradients modify the intensity of the response by adult mysids to illumination, adults dominated surface samples only in October when the lake became isothermal. Earlier in the year, mature mysids avoided warm surface waters, except for brief appearances in June and September, while juveniles migrated actively into warm surface waters.

Life History

The life cycle of *Mysis relicta* in Twin Lakes was described by Gregg (1976) by examining size-frequency data. Young-of-the-year mysids appeared

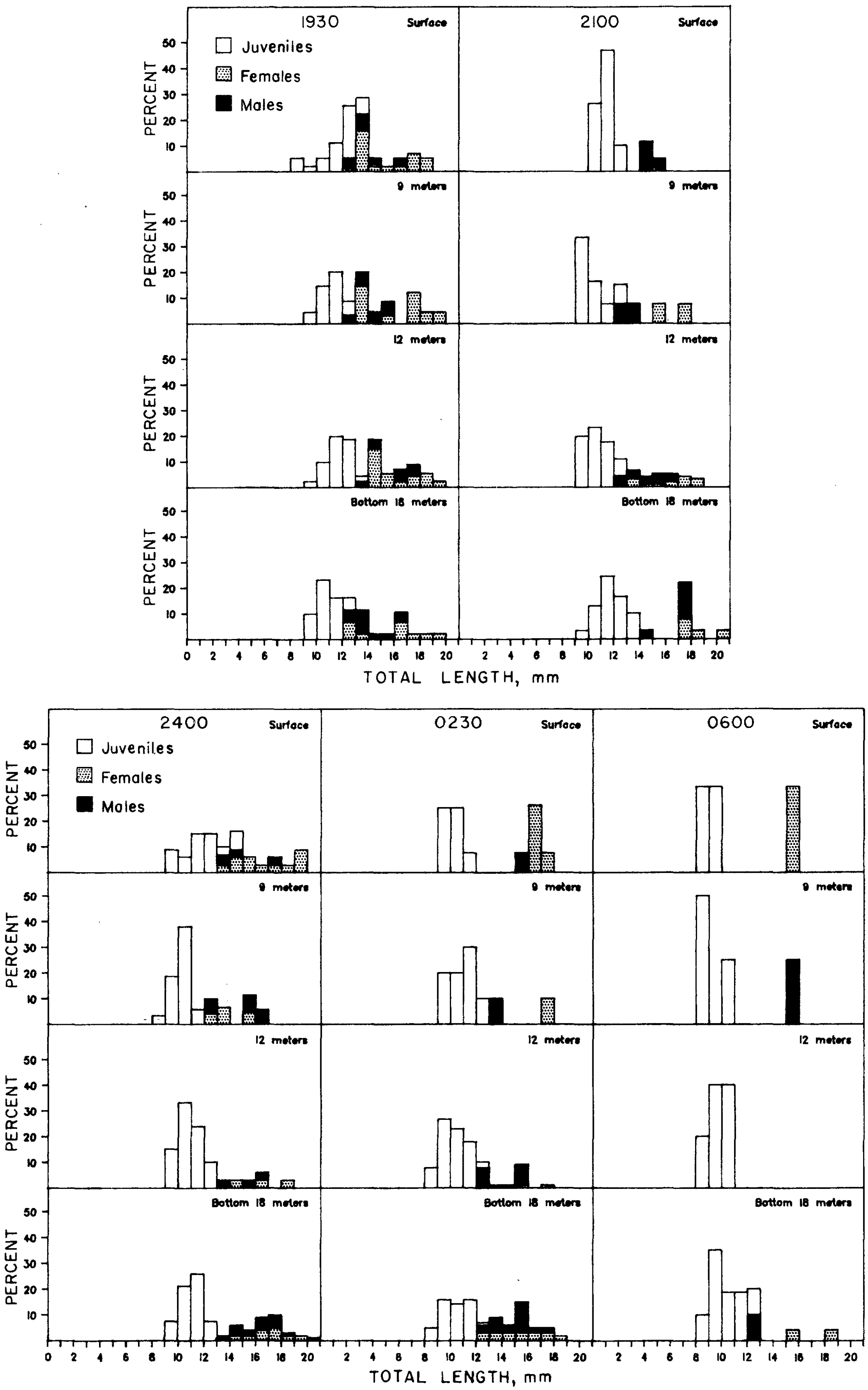


FIGURE 11-3.—*Size frequencies of* Mysis relicta *captured in individual net samples at four depths during the night of September 28-29, 1974.*

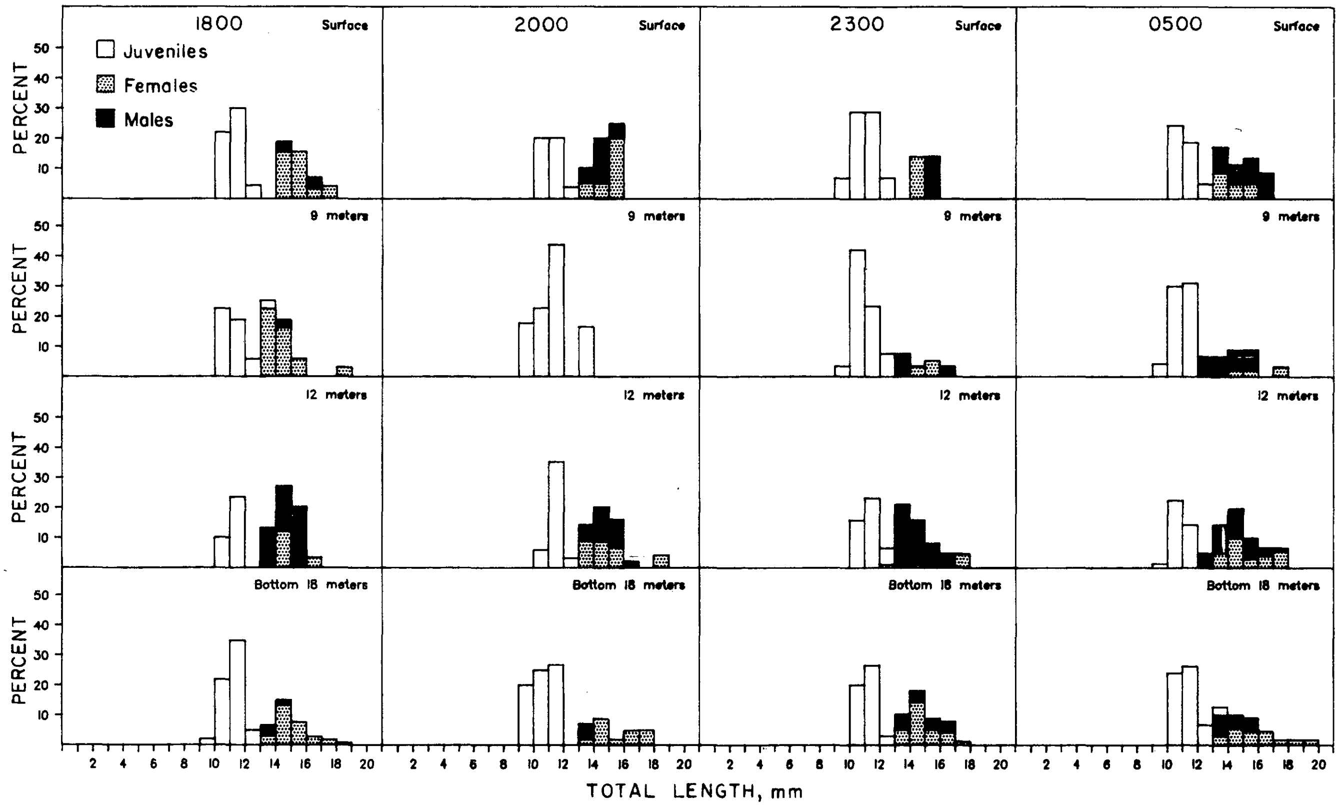


FIGURE 11-4.—*Size frequencies of* Mysis relicta *captured in individual net samples at four depths during the night of November 25-26, 1974.*

in April and were 4 mm long. By October, they were 10 mm long and sexually identifiable. Juveniles continued to grow during the winter and by January they were recognizable as immatures at a maximum length of 12 mm (fig. 11-5).

Following the second summer of growth, breeding characteristics became evident during October and November. Brood pouches developed in females and mating and subsequent release of eggs into the marsupium took place from November to January. Females 12 to 22 mm long were gravid during January, February, and March. Embryos in various stages of development were found in the marsupium during April, when the earliest of the young were released. Although males die after breeding, females may survive into the third year. The presence of small numbers of these large gravid females in the fall suggests that some females survived to breed a second time. These 2.5 year-old females reached a maximum size of 22 mm. It is not known if these females successfully produced a second brood.

The growth rates and maximum sizes attained by Twin Lakes mysids indicate a relatively slow-growing population comprised of small individuals. Growth to 14 mm took 14 months and the average growth

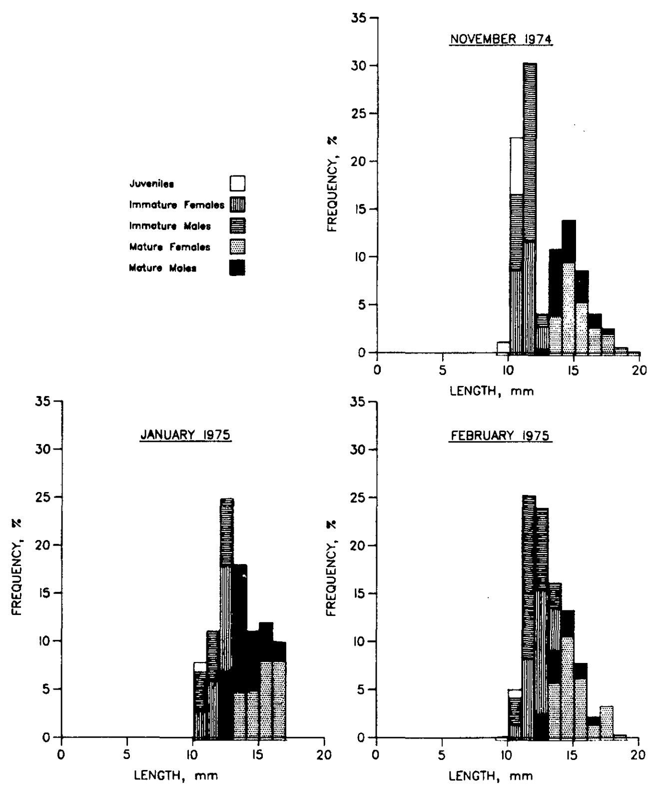


FIGURE 11-5.—*Length-frequency of* Mysis relicta *sampled in Twin Lakes.*

rate was 0.71 mm/month. Individuals larger than 20 mm were rare. Furst (1965) compared growth of 15 different populations and concluded that a slow growth rate coincided with a 2-year life cycle. The scarcity of food in oligotrophic lakes prevents mysids from accumulating sufficient oil reserves for breeding until the second year. Because breeding begins only at temperatures below 8 °C and is most active at 3 to 4 °C (Gosho 1975), mysids reaching reproductive capacity waited until fall to breed.

A trend in the November population toward a smaller proportion of juvenile shrimp in lower Twin Lakes from 1983 to 1985 was evident in age structure data (fig. 11-6). In 1983, juveniles and immatures comprised 64 percent of the population, which is similar to 1975 data. One-half as many juveniles made up the 1984 population, and by 1985 they made up only 24 percent of the population. In 1986, however, the population rebounded; 83 percent of the population was juvenile or immature. A population comprised of low numbers of juveniles is characteristic of a declining population or one that is just maintaining itself. Because juveniles tend to remain in the water column longer than adults, they may be more vulnerable to entrainment and age-specific mortality (see ch. 14). The proportion of juveniles in the November population of Upper Twin Lakes varied from year to year, but it was generally high. Juveniles made up 73, 24, 40, and 77 percent of the Upper Twin Lakes population in 1983, '84, '85, and '86, respectively (fig. 11-7). Juveniles and immatures made up 43, 49, 31, and 84 percent of Mt. Elbert Forebay populations in 1983, '84, '85, and '86, respectively (fig. 11-8).

Population Distribution

The divers consistently observed that mysids were ubiquitous in the Twin Lakes system. Mysids were seen anywhere from a depth of 1.5 m in bright sunlight to the complete darkness of the lake bottom. They were found in all types of substrate: rock, rubble, sand, glacial flour, and vegetation zones. Large concentrations of *Mysis* were observed during daylight on a rocky, sandy shelf at a depth of 2.4 m. Mysid densities were variable and estimates ranged from 10 to 50 per square meter to a maximum of 500 to 1000 per square meter. Mysids were not seen in the water column during the day, but were always within 0.2 m of the bottom. Mysids were not observed in the outlet stream.

The distribution of benthic mysids within a small area was compared with the distribution of mysids over the whole lake. Grid and radial replicates represented samples taken within a small area in August 1975. The radial replicates consisted of 10 samples taken in two similar small areas. The entire lake sample consisted of the 14 lower lake samples taken during the August 1975 sampling period. All data were transformed, and variance comparisons were made with F-tests. Because significant differences in variance were not found between the radial or grid and whole lake samples, *Mysis* density is apparently no more homogeneous in a small area than the whole lake. This suggests that the clumped distribution of *Mysis* is present even in a small area.

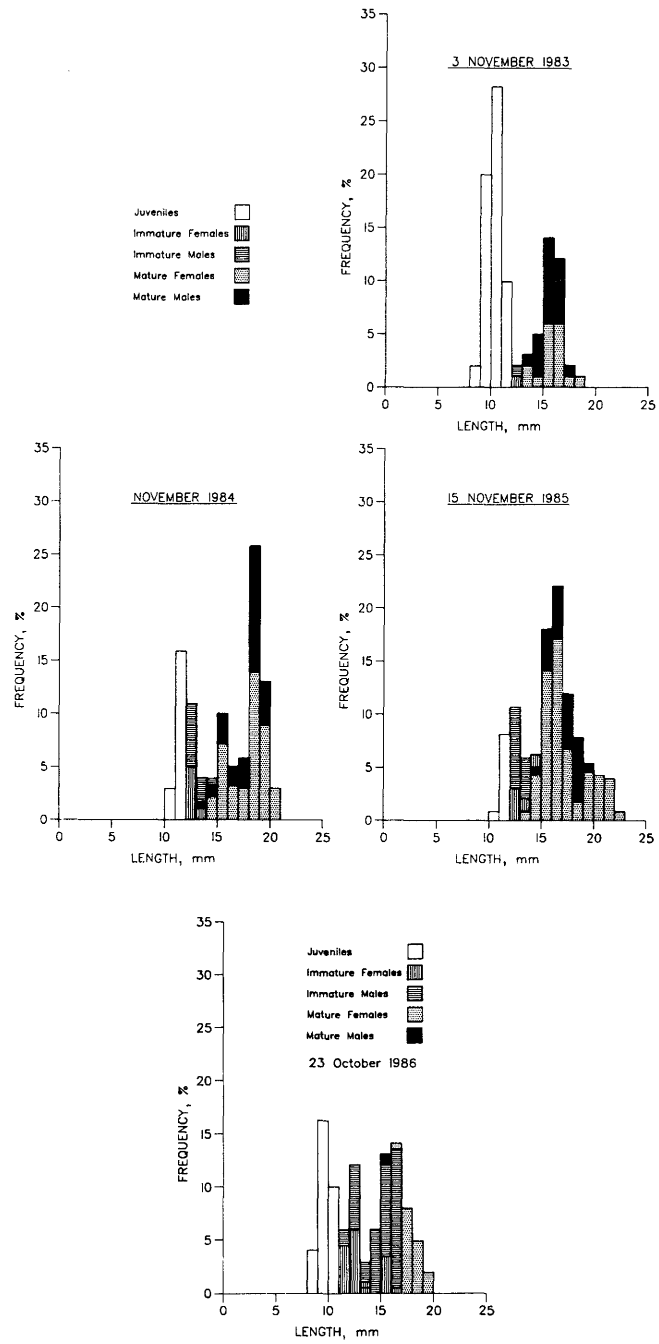


FIGURE 11-6.—*Length-frequency of* Mysis relicta *sampled in Lower Twin Lakes.*

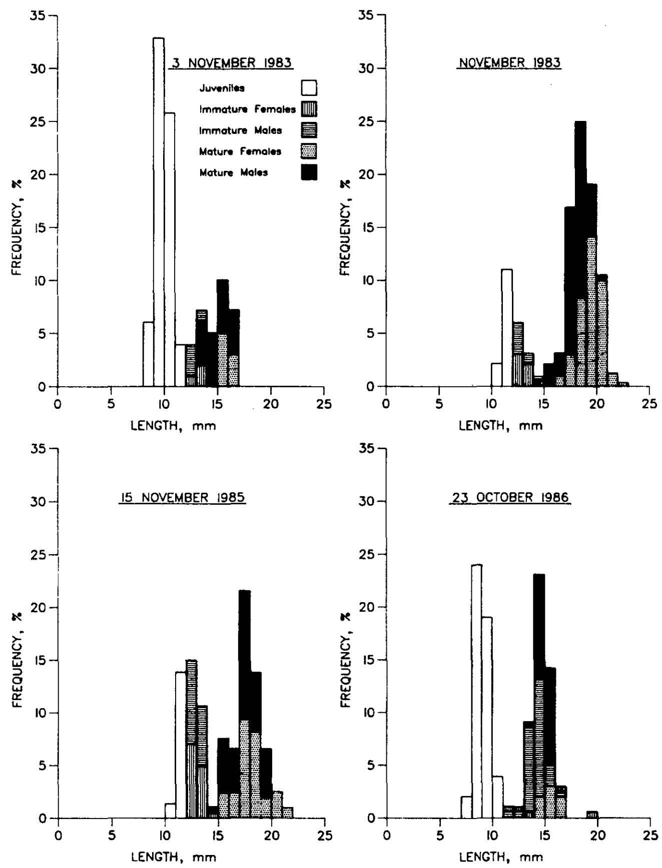


FIGURE 11-7.—*Length-frequency of* Mysis relicta *sampled in Upper Twin Lakes.*

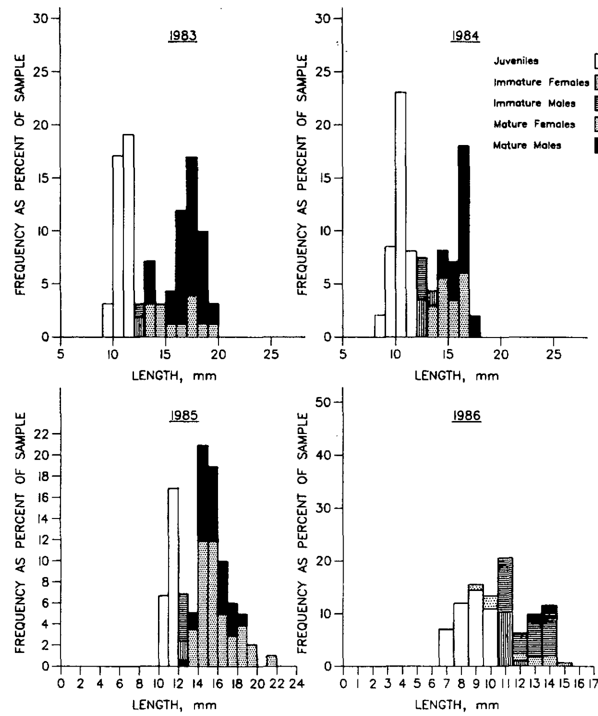


FIGURE 11-8.—*Length-frequency of* Mysis relicta *sampled in Mt. Elbert Forebay in October and November 1983-86.*

Abundance

Lower Twin Lakes.—Shrimp density in the lower lake was first estimated at 112 shrimp per square meter during 1981 (fig. 11-9). The population decreased significantly during 1982 to 54 shrimp per square meter, but increased 2.2 times during 1983, with densities reaching 121 shrimp per square meter. The 1983 density was significantly higher than the 1982 density but was not different from that estimated in 1981. The shrimp density remained high during 1984 (110 shrimp per m^2), but then dropped significantly to 70 shrimp per square meter during 1985. This is a decline in total estimated number of 271 million, from 781 in 1984 to 510 million shrimp in 1985. Samples from 1986 phototrawls indicated that the population was no longer declining; whole lake estimates increased by 292 million over the fall 1985 estimate to 802 million. However, the density of 105 shrimp per square meter, was still markedly less than the densities of Mt. Elbert Forebay or of the upper lake.

Shrimp density in the vicinity of the powerplant was not found to be significantly different from other areas of the lower lake during 4 out of the 5 years of this study. Only during 1984 were lower densities recorded in this area. During other years, shrimp density was near or exceeded the mean density for the lower lake.

A significantly greater mean density of all samples was found in 1974 (230 per m^2), which may have been due to the loss of organisms in the upper lake as a result of toxic levels of heavy metals. In late April and May 1975, DO concentration of the bottom 3 m of the hypolimnion became depleted, causing a reducing environment. Heavy metals in Lake Creek from early mining activities entering the lake had been trapped in the lake sediment, but were released during the reducing conditions. Concentrations were sufficient to be deleterious to primary production, which then would have affected other parts of the biological system. Chironomid and copepod numbers were also greatly reduced in the upper lake in 1975 (LaBounty and Sartoris, 1976).

When phototrawling began in 1981, the shrimp density was estimated to be 112 shrimp per square meter (fig. 11-9). As in the lower lake, the density was found to be significantly lower in 1982 (32 shrimp per m^2), but it rebounded in 1983. The shrimp population continued to increase and by 1985, 195 shrimp per square meter were estimated

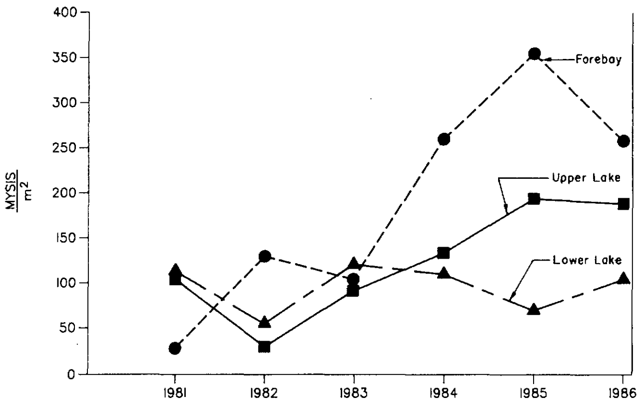


FIGURE 11-9.—Mysis relicta *densities in Upper and Lower Twin Lakes and Mt. Elbert Forebay, fall 1981-86.*

to exist in the upper lake. Population stabilization may be inferred from the 1986 data in which shrimp density was estimated to be 189 shrimp per square meter.

The 1982 decline in the shrimp populations of both lakes is associated with two events. The spring runoff from Lake Creek in 1982 was 69 percent greater than the previous year; and 63 650 m^3 (51.6 acre-ft) of water flowed through the pump-turbine units from May through September. This water flow was sufficient to cause currents to run back through and erode the channel connecting the upper and lower lakes, and to create a gyre in the upper lake. The hydraulic residence time of the upper lake declined from 1.2 to 0.4 years, and thermal stratification was weak. Other biological measurements such as zooplankton numbers and primary productivity were also markedly reduced in both lakes. In 1983, high spring inflows were also observed, but electrical generation did not occur during the growing season. Mysid and other zooplanktonic populations regained densities similar to or greater than those observed in 1981. High flushing rates, increased water movements, and increased turbidity appear to have seriously affected the planktonic communities.

In October 1983, the new dam was completed and the lake level increased 3 m by January 1984. The most heavily vegetated inundated area was a 130-hectare section west of the upper lake. This area contained an estimated 4000 metric tons of vegetation composed mostly of willows (*Salix* spp.), mixed grasses, and alders (*Alnus* sp.) (Bergersen and Maiolie, 1981), which provided an influx of nutrients. In 1983, higher densities of mysids were recorded in the upper lake than the lower lake for the first time, which may have been a response to the effects of inundation and the absence of pumping.

Pumping and generation began in 1984; in 1984 an average 27.5×10^6 m^3 (22.3 acre-ft) per month was released and an average 37.7×10^6 m^3 (30.6 acre-ft) per month was released during generation in 1985. Limnological measurements indicated that increased turbidity was associated with significant declines in primary production in the lower lake. By 1985, zooplankton densities of the lower lake had decreased by nearly one-half, and the *Mysis* population specifically experienced a significant 18 percent decline. Because a constricted channel no longer existed to direct currents created by generation into the upper lake after the lake was enlarged, the upper lake's biological system appeared to be relatively unaffected; although, currents were sufficient to maintain open water along its shoreline during the winter.

Mt. Elbert Forebay.—No shrimp existed in Forebay after it was drained during 1979. It has had water in it continuously since then, which has allowed the shrimp population to become established. The estimated density was 32 shrimp per square meter in 1981, and it increased significantly to 133 shrimp per square meter during 1982 (fig. 11-9). During 1983, the calculated Forebay shrimp density was 103 shrimp per square meter. Shrimp density again increased significantly to 262 shrimp per square meter during 1984, and to 355 shrimp per square meter during 1985. The 1985 increase was not significantly different from 1984. By 1985, the total estimated mysid population of Forebay was 379 million. Growth of Forebay mysid population occurred despite the large amount of water withdrawn between August 1981 and October 1983; the entire Forebay volume was exchanged 20 times during that period.

The nutrient concentrations and primary production in Forebay have traditionally been substantially lower than those of Twin Lakes, yet the mysid population has been thriving. It is suspected that an organic hash produced by entrainment of fish and other organisms, especially mysids, during "pump-back" cycles is sustaining this population. Additionally, the increased frequency of pump-back cycles moved shrimp from the lower lake to Forebay, which may have kept shrimp density above some lower carrying capacity. Shrimp density in pump-back cycles averaged nearly 10 shrimp per cubic meter of water discharged into Forebay. Thus, over 60 times more shrimp were added to Forebay than were flushed out, assuming equal volumes of water (see ch. 14). As the number of pump-back cycles increased, Mt. Elbert Forebay mysid population expanded.

Turbidity and Turbulence

Turbidity had no significant effect on mortality, but turbulence significantly increased mortality.

However, the interaction of turbidity did not appear to influence mortality significantly. In fact, at the lowest level of turbulence, the higher turbidity actually improved survival. Differences of 3 °C between replicates may explain some of the significant variability. Higher temperature appeared to increase mortality.

A *chi-square* test indicated that mysids were equally distributed among the four cells when turbulence was not present. However, under turbulent conditions, the number of mysids increased with distance from the source of the turbulence.

* * * * *

Fish Populations Chapter 12

T. Nesler and L. Finnell (retired);
Colorado Division of Wildlife
Fort Collins, Colorado; and

E. P. Bergersen, L. Walch, J. Griest,
and D. Krieger, with K. Hamilton
Colorado Cooperative Fish and
Wildlife Research Unit
Colorado State University
Fort Collins, Colorado

INTRODUCTION

The present-day composition of fish species in Twin Lakes is quite unlike the native fish fauna encountered by early investigators at the turn of the century. Manipulations of fish populations began as early as 1885, with spawn taking and stocking of yellow-fin cutthroat trout and "black-spotted" native trout (Wiltzius, 1985). In rapid succession, numerous other salmonid species were included in similar management activities from 1885 through 1917. Species stocked in subsequent years included:

Atlantic salmon	*Salmo salar*	1886
landlocked salmon	*S. salar sebago*	1886
rainbow trout	*Oncorhynchus mykiss*	1886
mackinaw	*Salvelinus namaycush*	1886
brook trout	*S. fontinalis*	1892
brown trout	*Salmo fario*	1893
rainbow × native hybrids		1894
Loch Leven brown trout	*Salmo trutta*	1895
lake whitefish	*Coregonus clupeaformis*	1907
grayling	*Thymallus arcticus*	1917

Records of existing fish species in 1889 from Jordan (1891) and later from Juday (1906) show Twin Lakes' fish population consisted of:

- yellow-fin cutthroat trout
- greenback cutthroat trout (*Oncorhynchus clarki stomias*)
- McCloud River rainbow trout (*O. irideus shasta*)
- landlocked salmon
- brook trout
- mackinaw
- sucker
- minnows

All of the native salmonids and many of the early introduced fish species are no longer present in the lakes. In 1957-59, the fish population consisted of (Nolting, 1968):

- white sucker (*Catostomus commersoni*)
- rainbow trout
- kokanee salmon (*Oncorhynchus nerka*)
- mackinaw (lake trout)
- longnose sucker (*Catostomus catostomus*)
- brown trout
- brook trout
- longnose dace (*Rhinicthys cataractae*)

Unsuccessful attempts were made to introduce several fish species to provide forage for lake trout. In 1967, fathead minnows (*Pimephales promelas*) were introduced in a single stocking attempt.

From 1971 to 1973, Bonneville cisco (*Prosopium gemmiferum*) and rainbow smelt (*Osmerus mordax*) were introduced (Finnell, 1977). From 1971 to 1984, predominant species included:

- lake trout
- rainbow trout
- white sucker
- longnose sucker

Other species less frequently encountered included: brown trout, brook trout, cutthroat trout, kokanee salmon, and longnose dace.

Stocking of kokanee salmon was conducted periodically through 1975 to provide additional sportfishing opportunity, though contributions of this species to the harvest were deemed poor to fair at best (Nesler, 1984). Since 1984, kokanee salmon fry have been stocked annually with the sole intention of serving as forage for lake trout.

The lake trout population is considered to be largely self-sustaining though annual plants of 25 000 fingerling-size fish were made by the CDOW (Colorado Division of Wildlife) to supplement the population. Brown trout, brook trout, both sucker species, and longnose dace maintain populations by way of natural reproduction. Rainbow trout are maintained in the lakes through large-scale stocking of hatchery-raised, creel-size fish by the CDOW.

Studies of fish populations were conducted from 1973 to 1984 by CDOW and the CCFRU (Colorado Cooperative Fishery Research Unit). The object of these studies was to determine life history attributes and population dynamics of Twin Lakes fishes prior to the planned start-up of Mt. Elbert Pumped-Storage Powerplant. Griest (1976, 1977), Finnell (1981), and Nesler (1984) investigated life history characteristics and population dynamics of lake trout. These studies spanned most of the 11 years from 1973 to 1984. Additional studies were conducted on lake trout movements and behavior (Walch 1980), and the life history of the two sucker species (Kreiger 1980). The intent of this chapter is to synthesize the results of these studies.

METHODS—LAKE TROUT

Population Characteristics

Sampling was conducted primarily with gill nets from 1970 through 1984 (excluding 1976 and 1980). To monitor lake trout distribution more accurately, a standardized design of gill-net sampling sites was developed in 1974 to sample systematically both Upper and Lower Twin Lakes. This approach provided the basis of data collection for Griest (1977), Finnell (1981), and Nesler (1984). Gill-net sampling was conducted during May, August, and November at 17 stations on the two lakes (fig. 12-1). Ten stations were located along a traverse line running from outlet to inlet in both lakes. Seven other stations were located at sites which were either greater or less than 15 m deep. The locations of these stations varied for each sample month, and were originally selected at random in 1974. Fourteen stations were located on the lower lake and three on the upper lake.

An experimental-mesh gill net (1.8 × 38 m) with five panels of different mesh size (19 to 44 mm) was set on the lake bottom at each station for approximately 13 hours overnight. This sample design was followed in 1974, '75, '77, '79, and '82. In 1978, gill-net sampling was conducted only at the

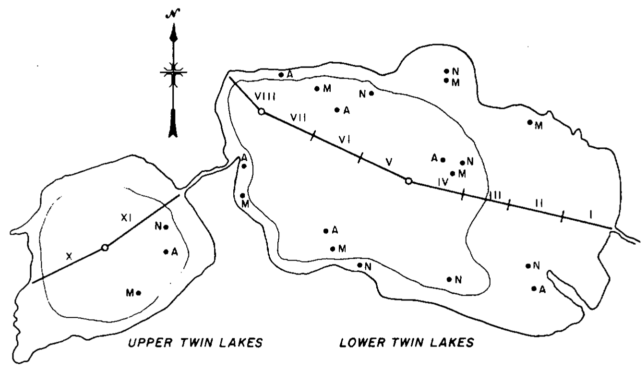


FIGURE 12-1.—*Standardized gill-net sampling locations at 17 stations Twin Lakes, May (M), August (A), and November (N).*

traverse line stations. In 1981, sampling was conducted only in November. In 1983 and 1984, sampling was delayed until June (if at all) because of prolonged ice cover.

Captured fish were enumerated by station:

- Lake trout were measured to the nearest millimeter (total length) and weighed to the nearest gram.
- Scale samples were taken from all lake trout in all years except 1983-84. These samples were collected from either side of the fish just below the dorsal fin and above the lateral line.
- Seasonal and spatial distributions of lake trout were examined by factorial analysis of variance on sample totals by station, depth category, months, and years.
- The mean number of lake trout captured per net was calculated for all stations by lake, month, year, and depth.
- Age determinations for lake trout were made by counting scale annuli according to criteria described by Cable (1956) and Everhart et al. (1973).
- Body-scale relationship was determined by two approaches:
  - For lake trout scales collected during 1971-75, Griest (1977) fit the data to a third-degree polynomial described by Carlander (1969).
  - For lake trout scales collected from 1977 to 1982, the body-scale relationship was assumed to be linear and a least-squares regression was used based on Lee's method (Lagler, 1956).
- Length-weight relationships were determined as logarithmic equations.
- Mortality estimates were made based on equations by Ssentongo and Larkin (1973), Ricker (1975), and Gerking (1978).
- Lake trout data, collected from 1977 to 1984, were accomplished by computer using the Fisheries Evaluation Program (Culver and Lentsch, 1984):
  - length-weight frequency distributions,

- length-weight relationship,
- age composition,
- body-scale relationship,
- back-calculated growth,
- mortality estimation between age groups, and
- various length and growth analyses per age group.

Food Habits

Stomachs were removed from a subsample of fish collected at the same time that total length, weight, sex, and scale samples were taken. The stomach was considered to be that portion of the gut from the esophagus to the pyloric valve. Stomach contents were preserved in a 10-percent formalin solution. The majority of stomach samples were collected during May, August, and November 1974 gill-net sampling periods, but some also were collected between these sampling periods and during the period of ice cover. Stomachs of 216 fish were examined using a dissecting microscope to identify the contents. Insects were identified to order and, in some cases, to family. Identification of fish remains was relatively easy due to the limited number of species present in Twin Lakes. Identifications were made on the basis of scales and bony parts. Remains that could not be identified were classified as unknown.

Volumes were reconstructed. Volumes of food other than fish that exceed 0.1 percent of the total volume in stomachs were measured by water displacement. Fish residues were reconstructed to preingested volume using the relationship of the combined length of last five vertebrae to the volume of whole rainbow trout or white suckers (Griest, 1977). All fish remains contained the necessary vertebrae to use this relationship. The only other fish species found in the stomach samples was one small lake trout. This fish had not been digested so that a determination of its preingestion volume by the water displacement method was possible.

Reproduction

Several attempts were made to determine the lake trout spawning period. Considerable boulder and rubble bottom is present along the north and south shores in the lower lake and these areas were presumed to be lake trout spawning areas. Lake trout typically spawn in lakes over large boulder or rubble bottoms which are cleared of sand and mud by currents and wave action (Royce, 1951; Eschmeyer, 1954; Martin, 1957; DeRoche, 1969; Machniak, 1975).

Spawning of lake trout in Twin Lakes has been observed near shore in 3 to 10 m of water (Nolting, 1968). During the last week in October 1974 and once a week starting in early October 1975, 3/4-inch bar-mesh gill nets were set in presumed lake trout spawning areas at Twin Lakes. Water temperatures during these periods ranged from 9 to 14.2 °C, depending on depth of set.

During 1977, gill nets were set in the lower lake each week from mid-October until December. Again, nets were set in suspected spawning areas, and also near the powerplant. During October and the first week of November 1978, 12 gill nets were set in the lower lake and five nets in the upper lake each week. Small-mesh gill nets were used when possible to minimize mortality. Lake Creek was sampled in October 1978 with a backpack electroshocking unit to determine if lake trout used the creek for spawning. Experimental mesh gill nets, vertical gill nets, and otter trawls were used in 1977-78 in attempts to capture juvenile and young-of-the-year lake trout.

Population Estimate

Several attempts were made to estimate the lake trout population size in Twin Lakes using mark and recapture methods in 1974 (Griest, 1977). These attempts were unsuccessful because capture with gill nets resulted in high mortalities; capture with hook and line resulted in an excessively large number of small fish, and finally; no marked fish were recaptured. An estimate of population size was made by using an estimate of the number of marked fish already in the lake from the stocking of marked fish in 1971, and the number of these fish caught during gill-netting efforts. An attempt to estimate population size by hydroacoustic surveys was made in 1980 (Thorne and Thomas, 1981).

Movements

Ultrasonic telemetry, gill netting, electrofishing, and trawling techniques were performed in 1977-78 to determine lake trout movement rates, home ranges, and general distribution patterns. All tracking equipment used was manufactured by Smith Root, Inc. Receiving equipment consisted of a model SR-70-H unidirectional hydrophone, a PA-74 preamplifier, a PC-74 pulse counter, and a TA-60 sonic receiver. Several types of Smith Root Inc. transmitters (tags) were used. Sizes ranged from 57 mm long, 14 mm in diameter and weighed 9 grams in water, to 120 mm long, 19 mm in diameter and weighed 40 grams in water. To increase the life of the tags, power output was decreased on most of them. Tags used during 1978—which included seven temperature-sensitive tags—had wire loops attached at each end to facilitate external attachment.

All lake trout used in the movements study were captured using gill nets of varying sizes. Gill nets were raised slowly with frequent pauses to minimize stress on fish caused by temperature and pressure changes. Special care was taken to select trout which had not been injured in the nets.

An external tagging method similar to that of Haynes et al. (1978) was used. The transmitter was attached to the fish with monofilament by passing a large surgical needle through the large musculature below the dorsal fin. The procedure is detailed in Walch (1980). Efforts were taken to keep the weight of transmitters in water from exceeding 2 percent of the weight of the tagged fish out of water. Most authors agree that the 2 percent or less tag-to-fish weight ratio minimizes the effect of the tag on fish stability and buoyancy (Morris, 1977; Haynes et al., 1978; Winter et al., 1978). For fish tagged at Twin Lakes, the tag-to-fish weight ratio rarely exceeded 2 percent.

This external method of attachment was tested in the laboratory using two rainbow trout. The two fish retained the tags for 10 weeks. Activity and feeding behavior of tagged fish were similar to control fish after several days. However, minor abrasions from the tags were noted. One externally tagged lake trout was recaptured 7 months later in a gill net and the transmitter was still attached. The fish appeared to be in good condition, although some abrasion was apparent.

Movement data were not collected from tagged fish for 2 days following tagging to minimize the chances of recording movements resulting from behavioral changes that could have occurred due to handling or stress from tagging. Typically, sightings (fish locations) were conducted on all fish during a randomly selected 24-hour period. However, in winter on the lower lake and year-round on the upper lake, sightings were obtained during daylight hours only. Deviations from the sampling schedule occurred caused by weather and lake conditions, equipment malfunctions, and inability to locate a desired fish during a sampling period. A complete description of the sampling schedule is provided in Walch (1980).

A grid search pattern with listening stops every 300 to 440 m was used to locate ultrasonic signals. Positioning the boat over tagged fish and determining the position of the boat was accomplished in a manner similar to that used by Diana et al. (1977). During winter, the location of tagged lake trout was determined using triangulation techniques described by Poddubnyi et al. (1970) and Diana et al. (1977). To determine the accuracy of location techniques, efforts to locate an activated transmitter placed on the bottom of the lake were conducted in 1977. The boat could be positioned within 10 m of a point directly over the transmitter. Wind and water conditions could reduce the accuracy of boat positioning by an additional 10 m.

Information recorded at each sighting included location, fish identification number, and time of day. In addition, if a lake trout was carrying a temperature-sensitive transmitter, the pulse rate of the tag was recorded. Pulse rates of temperature-sensitive transmitters were dependent on water temperature; thus, water temperature at the location of the fish was known and the depth of the fish could be determined from temperature profiles of the lake. Bottom depths were also obtained when a fish with a temperature tag was sighted. These depths were determined later using the sighting position and contour maps.

Tracking results were plotted on 1:4800 scale maps. Linear displacement rate (net change between fish locations in meters per time interval in minutes) was used as an index of activity when the time interval between locations was less than 270 minutes (Walch and Bergersen, 1982). Parametric and nonparametric analyses of variation were used to determine if differences in rate of displacement occurred seasonally or daily. Correlations between lake trout size and rate of displacement were examined with regression techniques. Means of displacement data were compared using the studentized range, Q, described by Snedecor and Cochran (1967). The calculation of Q was modified to adjust for unequal sample sizes by using the harmonic mean described by Bancroft (1968). Significant differences were determined at P=0.05.

Estimates of utilized home range (Odum and Kuenzler, 1955) were calculated from plots of individual fish locations. Utilized home range is the area in which fish are most commonly found and in which single additional relocations will not increase the total home range size by more than 1 percent. Any peripheral relocation points outside a 200-m radius of an adjacent relocation point recorded on a different day were not included in the calculation of utilized home range. Odum and Kuenzler's (1955) method eliminates bias introduced by unequal sample size. Position plots of individual lake trout were examined with respect to shoreward movements, vertical movements, spawning activities, homing, and habitat.

Netting efforts were conducted at least once each month from January through October 1978 at 10 locations on the upper and lower lakes with experimental-mesh gill nets to supplement the telemetry data (fig. 12-2). Data were examined

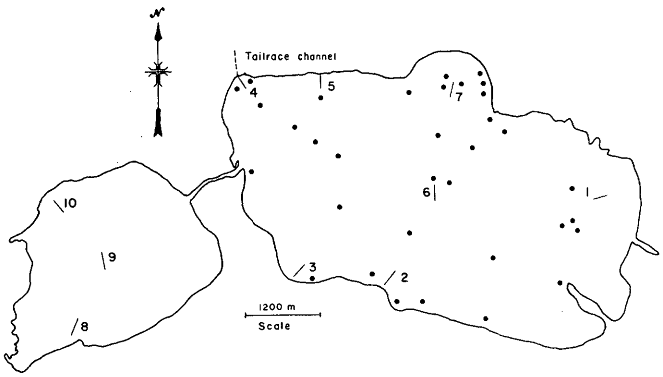


FIGURE 12-2.—*Locations of standard horizontal and vertical gill-net sets in Twin Lakes during 1978. Horizontal sets are numbers 1 to 10. Vertical sets are shown by solid dots.*

statistically for seasonal and within-season differences. Four vertical gill nets were used to obtain information to supplement telemetry data from lake trout tagged with temperature-sensitive tags. At least two of the four vertical nets were fished for three 24-hour periods each month from June through October 1978.

A boat-mounted electrofishing unit was used to gather data concerning distribution of lake trout in shallow water areas at night. Collections were made in June, August, and October 1978.

A Hydrolab Surveyor Multiparameter Probe was used to record temperature and DO (dissolved oxygen) profiles near the center of both lakes at least once every 2 weeks during most of 1977 and 1978. These data were used in linear regression analyses of fish movement rates to determine if movement rates were correlated to DO or surface water temperatures.

RESULTS AND DISCUSSION

Population Characteristics

Relative abundance.—Yearly totals of lake trout sampled in standardized netting ranged from 181 in 1979 to 318 in 1982 (table 12-1). Per sampling site, the number of lake trout caught in Lower Twin Lakes ranged from 0 to 22 fish in deep-water stations, and from 0 to 14 fish in shallow-water stations. In Upper Twin Lakes, 0 to 10 lake trout were captured per sampling site. Sample totals for lower lake stations showed that a notable increase in numbers of lake trout captured occurred in November 1981, and carried through to November 1984. Generally, throughout the study period, more lake trout were captured per net in the lower lake versus the upper lake, and in deep-water stations versus shallow-water stations (fig. 12-3). Catch rates in the lower lake ranged from 4.2 to 8.3 lake trout/net/year. In the upper lake, catch rates were 37 to 86 percent lower, ranging from 0.6 to 3.2 lake trout/net/year. For lower lake deep stations, monthly catch rates ranged from 4.1 to 12.6 lake trout per net, whereas monthly catch rates in the shallow stations were 16 to 91 percent lower at 0.0 to 0.4 lake trout per net.

Factorial analysis of variance was possible on lower lake sample totals for 1974, '75, '77, '79, and '82 (table 12-2). Considering all stations, sampling variation between years, months, and stations was significant. Stratification of sample totals according to depth category provided more meaningful results related to lake trout biology. Yearly sample totals were significantly different for deep stations but not for shallow stations. Sample totals among deep stations were not significantly different while those for shallow stations were. These results suggest significant differences occurred in lake trout abundance among years due to changes in the abundance of lake trout occupying the deep-water zone of the lake. Differences in their abundance in shallow station samples illustrated differences in the lake trout's use of shallow-water habitat of Twin Lakes between months and lake area. This difference may be related to fish size and feeding behavior.

Monthly sample totals were not significantly different for deep stations, but were for shallow stations. Monthly pattern of sample totals over years was significantly different for both depth categories (fig. 12-4). Monthly sample totals increased from May to November, decreased from May to November, or showed May and November highs surrounding an August low for the lower lake. The source of variation in these patterns was presumed to be related to monthly water temperatures. During May and November, sampling occurred under homothermous lake conditions soon after ice-cover break-up, or just prior to ice-cover formation. Variation in mean water temperatures among sampling dates relative to the lake trout's preferred optimum of 10 to 12 °C may have influenced lake trout activity and gill-net catch rates. The pattern of station sample totals over months or over years was not significantly different for either depth category, suggesting lake trout distribution over lake area was relatively homogenous, or more likely, randomly clumped.

Analysis of variance was also conducted on station VIII alone for the years 1974, '75, '77, '78, '79, and '82 (table 12-2). Significant differences were not noted in relative abundance of lake trout between years or months, suggesting concentrations of lake trout in the area of the powerplant tailrace channel have remained fairly constant prior to powerplant

Table 12-1.—Number of lake trout captured per station in standardized gill-net sampling during May, August, November, or June in 9 years from 1974 to 1984 at Twin Lakes, Colorado.

	Lower Twin Lakes sample stations														Upper Twin Lakes		
	Deep, >1 m									Shallow, <15 m							
	Transect						Random			Transect		Random			Transect		R*
	III	IV	V	VI	VII	VIII	1	2	3	I	II	1	2	3	IX	X	UT
1974																	
May	3	3	4	0	4	3	3	4	13	3	1	3	0	4	1	4	2
Aug.	5	9	2	3	7	8	3	7	4	0	1	4	4	0	3	1	1
Nov.	5	10	10	14	10	5	5	7	8	2	7	7	6	5	2	8	6
1975																	
May	2	7	7	1	8	6	4	4	3	3	3	0	3	3	3	2	1
Aug.	4	6	6	6	9	11	8	4	7	0	7	2	1	0	3	1	0
Nov.	6	9	9	10	8	4	13	4	8	4	6	4	3	5	2	7	10
1977																	
May	12	9	9	7	6	5	11	7	7	9	8	3	10	4	0	5	1
Aug.	9	5	7	6	4	7	2	3	3	0	0	0	0	0	3	0	5
Nov.	10	5	2	6	6	2	6	4	4	0	8	0	2	5	1	1	1
1978																	
May	4	3	12	11	1	4	–	–	–	3	3	–	–	–	–	–	–
Aug.	4	7	6	5	8	5	–	–	–	0	1	–	–	–	–	–	–
Nov.	10	9	5	6	7	5	–	–	–	2	5	–	–	–	–	–	–
1979																	
May	2	4	4	4	1	9	5	–	4	2	5	2	2	4	0	1	1
Aug.	1	5	5	4	6	1	–	6	3	1	8	0	1	0	0	0	–
Nov.	9	14	7	9	4	8	11	2	3	4	3	4	0	1	0	3	0
1981																	
Nov.	22	8	5	–	9	12	10	14	10	5	5	1	3	4	3	3	3
1982																	
May	6	16	7	11	13	7	8	11	10	9	6	2	1	6	5	4	3
Aug.	10	11	7	18	13	2	4	14	10	2	6	0	0	1	1	2	1
Nov.	12	7	8	2	12	6	8	8	7	2	4	1	1	2	2	3	6
1983																	
Aug.	9	8	4	13	16	7	13	8	7	1	6	2	0	3	0	0	0
Nov.	9	9	10	10	17	9	12	10	8	6	13	9	0	14	3	4	1
1984																	
June	8	12	7	4	11	7	12	7	7	6	3	2	2	3	1	1	1
Nov.	10	10	8	13	9	11	14	11	12	2	10	12	9	2	7	3	4

* R = random. UT = Upper Twin Lakes.

operation. Sample data for station VIII, in 1983 and 1984, suggest a similar status (table 12-1).

Length-Weight Characteristics.—Mean lengths of lake trout showed no trends over 1973-84—ranging from 319 to 396 mm total length. When depicted by deep and shallow stations for each month sampled in each year, lake trout mean lengths from shallow stations appeared larger than the corresponding mean length for lake trout captured at deep stations (fig. 12-5). This finding occurred in 18 of 20 samples, excluding one-fish samples. The mean difference in mean lengths of lake trout between shallow and deep stations was 76.5 mm in May, 24.4 mm in August, and 58.8 mm in November. This tendency may be viewed as a result of fish prey species' conditions at Twin Lakes, and the piscivorous nature of lake trout. The mean length results suggest the use of shallow-water habitat in the lakes by larger lake trout, especially during May and November, probably in search of larger prey such as sucker and rainbow trout, which are more available in the littoral zone of the lakes (Krieger, 1980). Smaller lake trout, which rely heavily on *Mysis* shrimp as food (Griest, 1977), would tend to remain in deeper waters normally inhabited by the majority of the shrimp population (Gregg, 1976; Nesler, 1981a).

Length-frequency distributions of lake trout showed no major trends from 1973 to 1984 (fig. 12-6). A large single peak dominated in each year, and shifted position within the length range from 300 to 450 mm. Outside this range, differences were noted as slight increases or near absence of larger lake trout, or as changes in relative abundance of smaller lake trout. These length frequency distributions were simplified into four size groups for

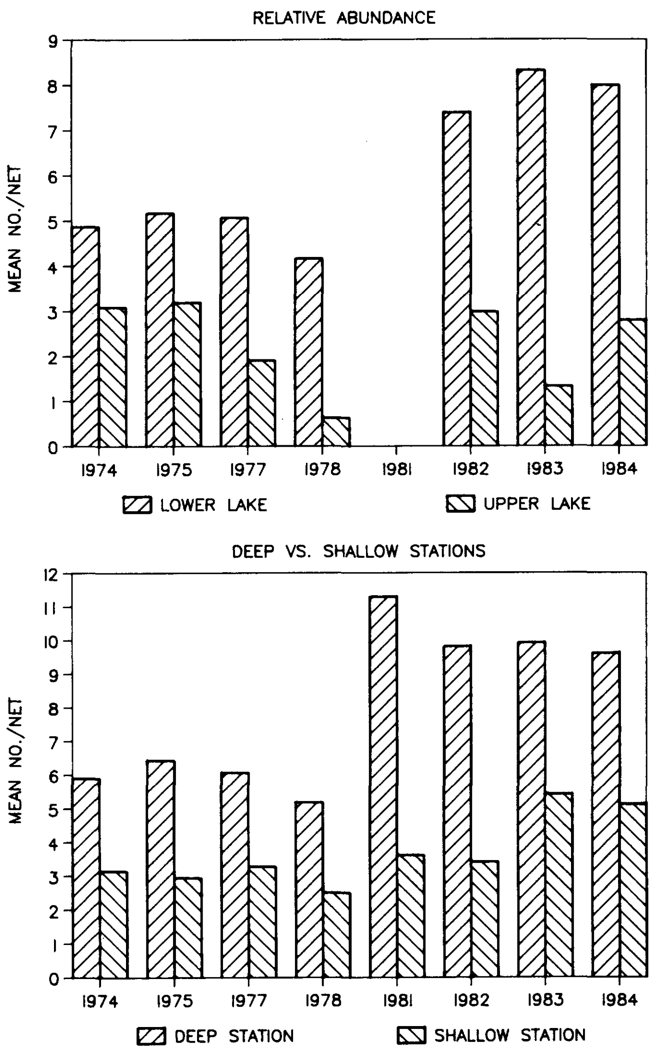


FIGURE 12-3.—*Relative abundance of lake trout in Upper and Lower Twin Lakes, and deep and shallow stations.*

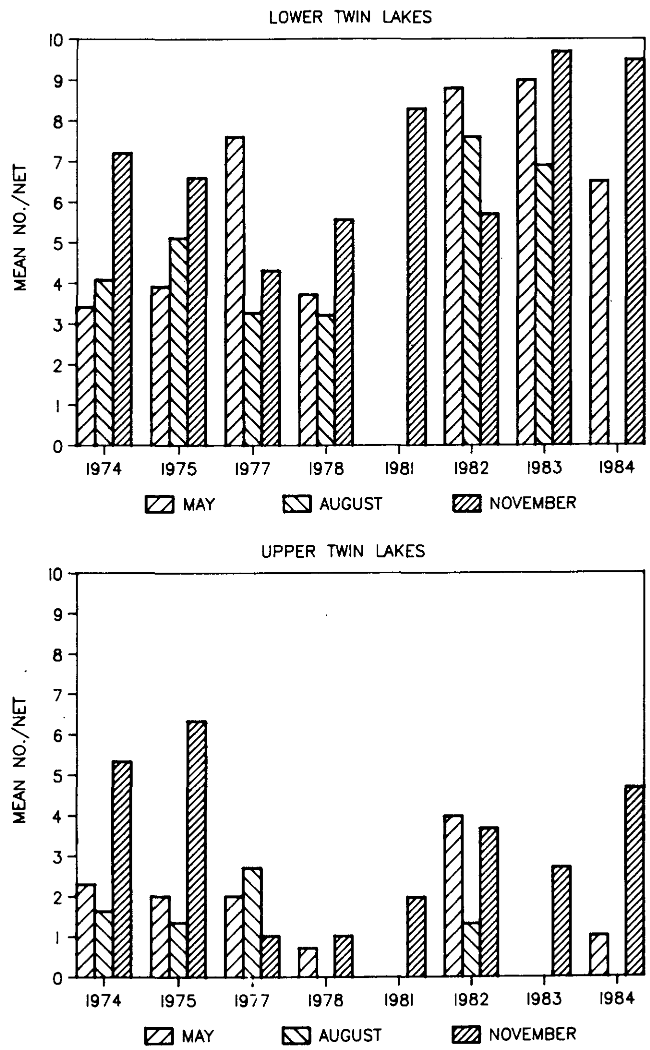


FIGURE 12-4.—*Monthly gill-net catches of lake trout in Upper and Lower Twin Lakes.*

Table 12-2.—Comparison of gill-net sample totals of lake trout using factorial analysis of variance, Twin Lakes, Colorado. Entries per source of variation and strata indicate either no significant variation (ns) or significant variation at the 0.01 or 0.05 probability level.

Source of variation	Sample Station Stratum All	Deep	Shallow	Sta. VIII
Years	0.01	0.01	ns	ns
Months	.05	ns	0.01	ns
Stations	.01	ns	.01	
Years × months	.01	0.01	.01	
Years × station	ns	ns	ns	
Months × station	ns	ns	ns	

Years included in analysis were 1974, '75, '77, '79, and '82.

chi-square analysis (table 12-3). Significant variation from the mean frequency distribution was observed for 1973, '75, '77, and '78. In 1973, relative abundance of lake trout 401 to 500 mm total length was significantly greater than average. Relative abundance of smaller lake trout (≤300 mm) in 1975 was also significantly greater than average. This abundant size group probably resulted in the significant differences noted in the 301 to 400 mm size groups in 1977 and 1978. Clear demarcations distinguishing age groups were not evident—suggesting considerable range in growth within age groups present. Comparison of lake trout length frequency data from this 10-year period with that from 1957 to 1961 (Nolting, 1968) revealed no large change in population structure.

The length-weight relationship for lake trout changed little between 1971 to 1975 and 1977 to 1983.

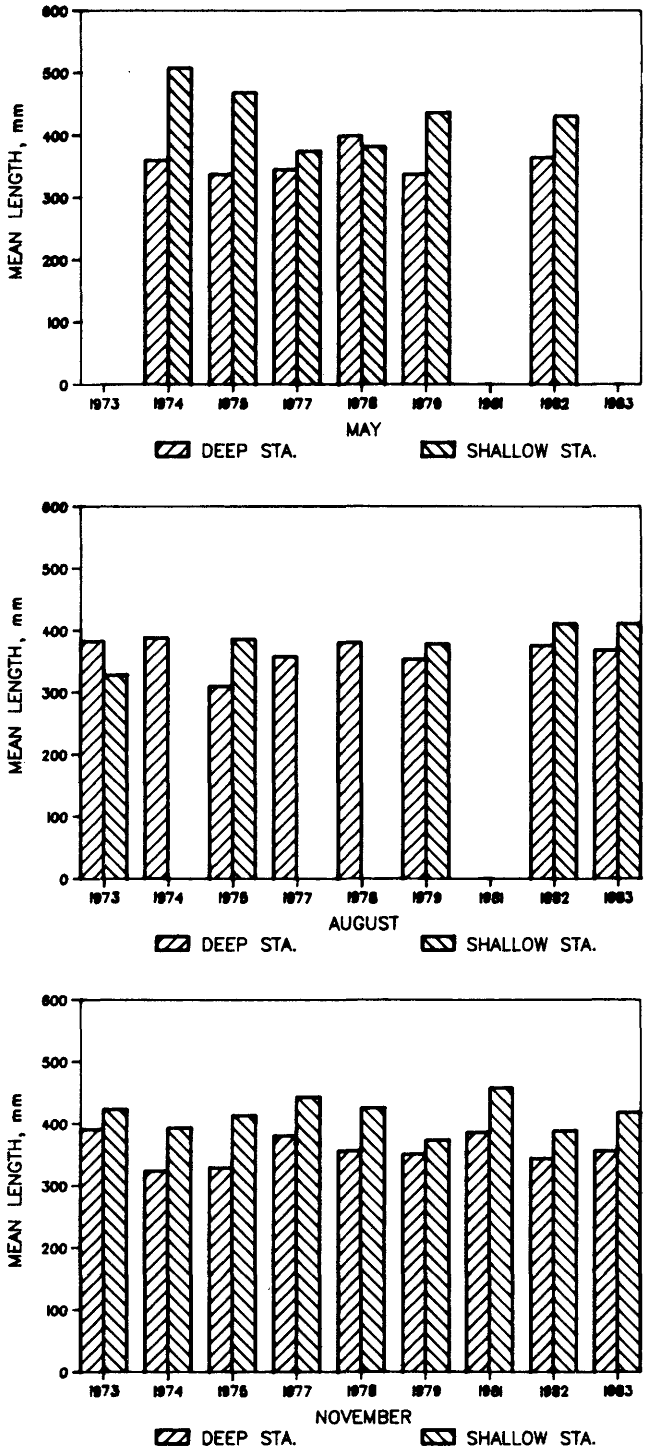


FIGURE 12-5.—*Mean lengths of lake trout from shallow and deep stations.*

Equations for the two time periods were as follows:

1971-75: $\log W = (3.18 \log L) - 5.56$ $r = 0.99$
Griest 1977)
and
1977-83: $\log W = (3.00 \log L) - 5.12$ $r = 0.97$
(Nesler 1984)

On a yearly basis, calculated slopes varied from 2.82 to 3.14 during 1977-83, with no trend evident over years. Lake trout gained weight in greater proportion to their length in 1977, 1978, and 1982, and gained weight in lesser proportion to their length in 1979, 1981, and 1983. The difference in slope calculated between 1971 to 1975 and 1977 to 1983 suggests little biological significance, especially considering the range of slopes evident during the latter period.

Age and mortality characteristics.—The age composition of lake trout samples in 1971-75 and 1977-82 was similar, and showed 84 to 89 percent of the sample population was represented by ages II-V (fig. 12-7). Relative abundance of older lake trout decreased rapidly from ages V through VII. Few lake trout age VIII or older were ever sampled by gill nets. Comparison of year-class strength showed a select year-class generally dominated the population as 3-, 4- or 5-year-old fish, and only once did a single year-class (1974) dominate 2 years in succession (fig. 12-8). The chronological order of dominant year-classes (and the sample year in which they appeared dominant) was 1971 (74), 1974 (77, 78), 1975 (79), 1976 (81), and 1978 (82). This sequence of dominant year-classes suggests a combination of relatively stable recruitment over 1974-82 with angler mortality affecting each year-class as it becomes vulnerable to the fishery at ages III through V.

Age at maturity was demonstrated to be a 3- to 4-year process for a given year-class of lake trout (Griest, 1977). The youngest mature male captured was 4 years old. Of all 4-year-old males captured, 20.9 percent were mature; 57.9 percent of all 5 year olds, 71.4 percent of 6 year olds, and 100 percent of all older males were sexually mature. The youngest mature female was also 4 years old, with 8.1 percent of all 4-year-old females being mature. Fifty percent of 5-year-old females and 66.7 percent of 6-year-old females were sexually mature. No samples of 7- or 8-year-old female lake trout were available, but all age 9+ females were sexually mature.

Mortality estimates between age groups (Gerking, 1978; Culver, and Lentsch, 1984) were based on combined age data from 1977 to 1982, and suggested a general increase in mortality rate from age 3 to 9 (fig. 12-7). This pattern reflects an increasing effect of fishing mortality on top of a lower and presumably more constant natural mortality. Most 3-year-old lake trout were not legal harvest size (381 mm), and would be less vulnerable to fishing mortality other than some hooking mortality. Mortality rates between ages 5 to 6 and 7 to 8 correspond to lake trout sizes that are important to the fishery. The

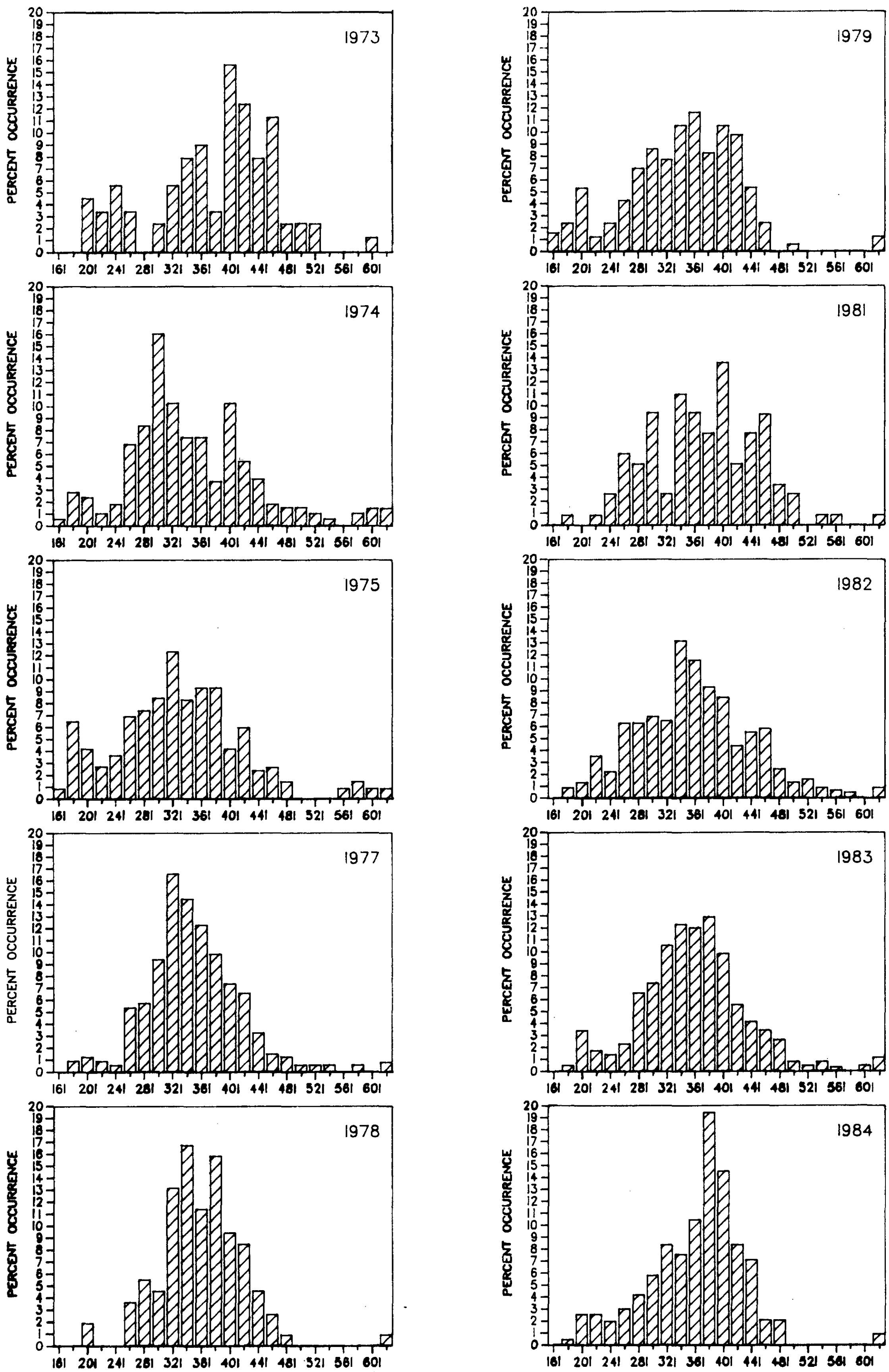


FIGURE 12-6.—*Length-frequency distribution of lake trout in Twin Lakes.*

Table 12-3.—Comparison of length-frequency distribution of lake trout given as percentages at Twin Lakes, Colorado.

Size interval, mm	1973	1974	1975	1977	1978	1979	1981	1982	1983	1984
≤300	16.9	23.8	32.1	4.8	11.2	23.8	15.5	20.4	15.4	15.4
301 to 400	28.1	45.6	47.3	62.6	61.8	46.5	40.1	47.3	54.9	51.0
401 to 500	49.4	23.8	16.5	19.7	26.1	27.9	39.1	26.8	25.8	33.8
≥501	5.6	6.8	4.1	2.9	0.9	1.8	5.3	5.5	3.9	0.8
Chi-square value	24.2 *	4.8 ns	14.8 *	8.1 *	9.2 *	2.5 ns	6.2 ns	1.1 ns	1.8 ns	4.4 ns

* = significant at P=0.06 ns = not significant.

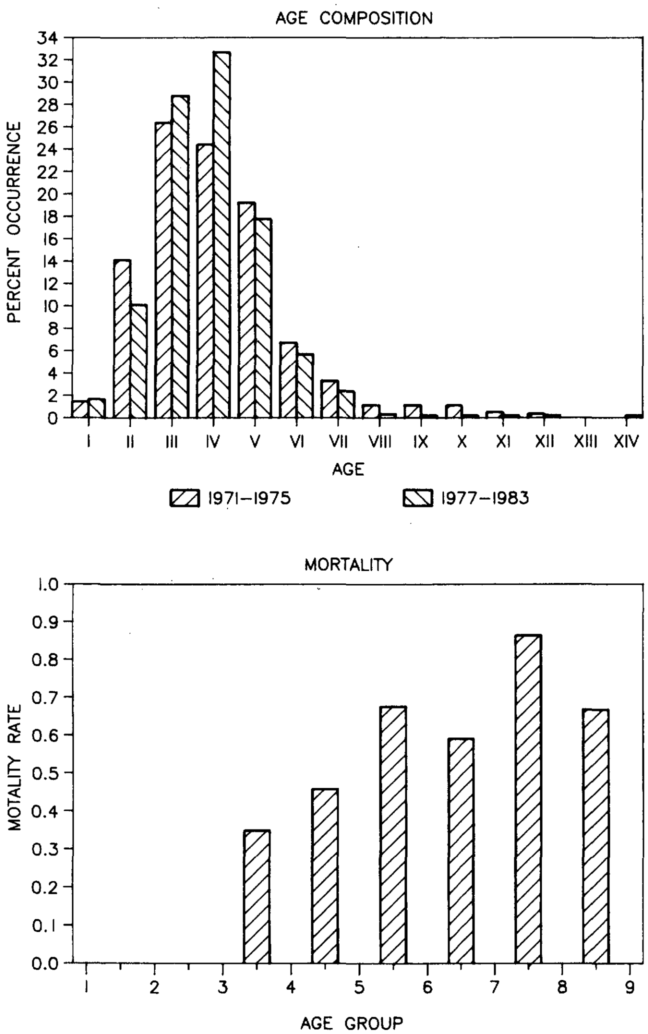


FIGURE 12-7.—*Age composition and age-specific mortality rates for lake trout. Mortality histograms denote mortality rate between age groups.*

majority of a given year-class of lake trout become vulnerable to fishing mortality between ages 5 and 6. All age 6 lake trout were larger than the legal size limit. Between ages 7 and 8, lake trout generally reached the 500-mm size-class, which was highly preferred by the angler as a "quality-size" fish. During periods of good success, anglers tended to catch and release lake trout, seeking larger fish. Most lake trout age 8 and older were kept by anglers. Using the mortality rates on figure 12-7, a population of 10 000 age 4 lake trout would be reduced by combined mortality to 32 fish at age 9.

Using the approach of Ssentongo and Larkin (1973), estimation of instantaneous and annual mortality hinges on a subjective interpretation of age of first capture. This method uses the relationship between age of first capture and average age of the sample population to estimate instantaneous mortality rate. The criterion defining age of first capture is all fish above this age are equally likely to be caught (in this case, by gill nets). Using age 2 as the age of first capture, the instantaneous mortality rate was 0.661 and the annual mortality rate was 0.484 for the lake trout population data from 1974 (Griest, 1977). Using the same method and age of first capture for lake trout population data from 1977 to 1982, the instantaneous mortality rate was 0.518 and the annual mortality rate was 0.404. Age composition data from 1977 to 1982 suggested that representative sampling of lake trout may first occur at an age closer to 3 years old. Using this age as that of first capture raised the estimate of instantaneous mortality to 0.869 and that of annual mortality to 0.581.

Ricker (1975) provided a method for determining annual mortality from a portion of an age series. Using this method, the mortality rates for lake trout from age 3 to 9 and 4 to 9 were 0.329 and 0.555, respectively. Again, the difference between these latter two mortality rates may be a reflection of angler mortality differences between 3-year-old lake trout and age 4+ fish.

The body-scale relationship used to determine back-calculated growth was calculated as curvilinear for 1971-75 lake trout data and fit to a third degree polynomial. For 1977-82 data, a least-squares regression analysis was used. The resulting equations

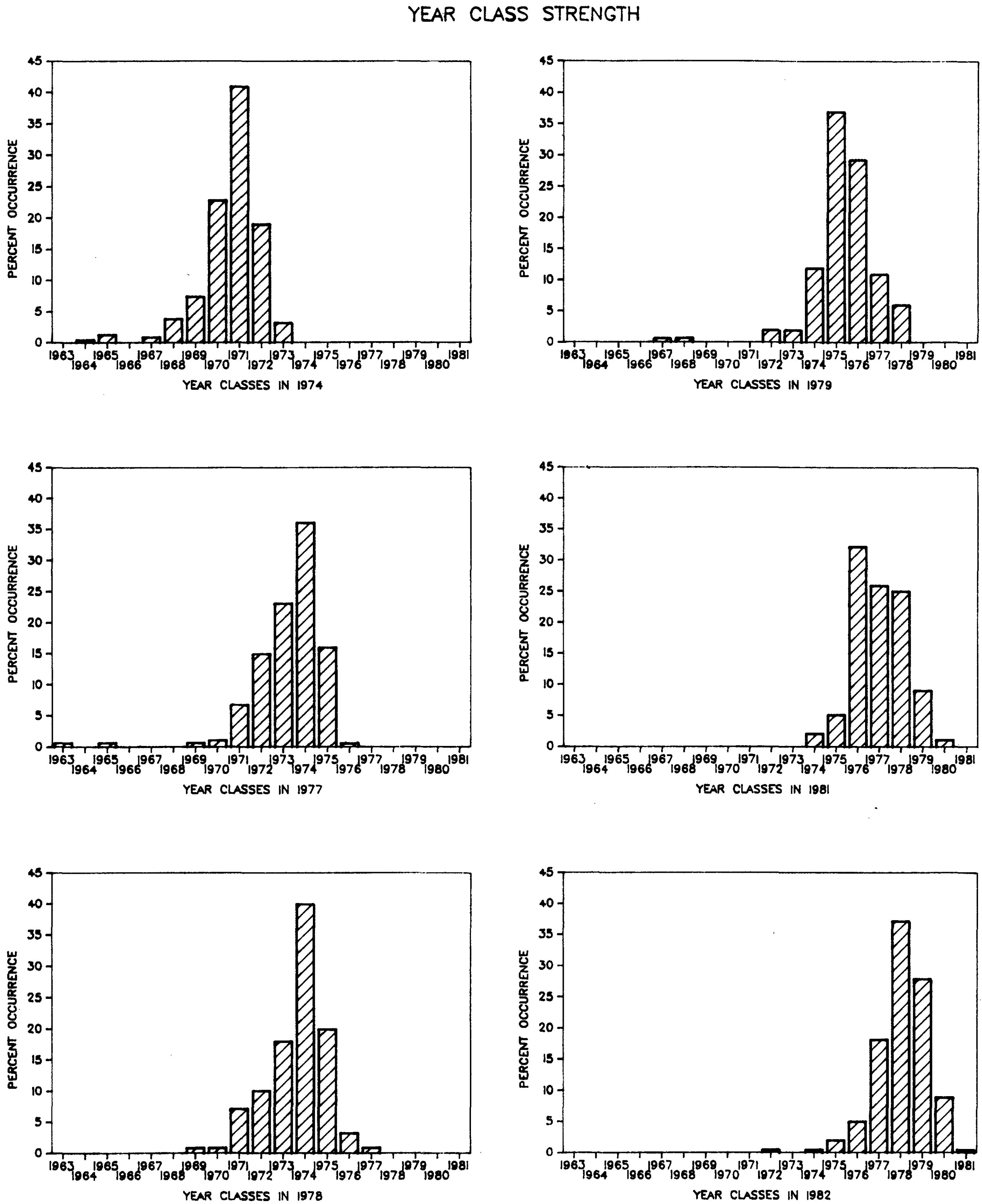


FIGURE 12-8.— *Year-class strength of lake trout by year of capture.*

for both methods were:

1971-75:

$$TL = 38.1 + 2.944\ S - 0.007\ 34\ S^2 + 0.000\ 0151\ S^3 \quad (r = 0.816)$$

1977-82:

$$TL = 97.35 + 6.12\ S \quad (r = 0.82)$$

where:

TL = total length, mm, and
S = scale radius, mm.

Correlation coefficients for the two approaches were virtually identical, suggesting they were similar in effectiveness for describing lake trout growth. Relative differences would be greater at the upper and lower ends of the lake trout size range. Back-calculated length at annulus for lake trout captured in each of the time periods showed little difference in growth through the first 5 years (fig. 12-9). Thereafter, lake trout growth in later years in life appear noticeably better during 1977-82, which may be due to limited sample size. Growth from annulus VIII and older was based on only 9 lake trout for 1977-82 and 25 lake trout for 1971-75. Shallow station gill-net data suggested no significant increase in abundance of potential fish prey species like white and longnose suckers. To the contrary, stocking levels for hatchery-raised rainbow trout decreased by 60 percent, and kokanee salmon fry stocking was eliminated between the two periods—significantly reducing this potential prey base.

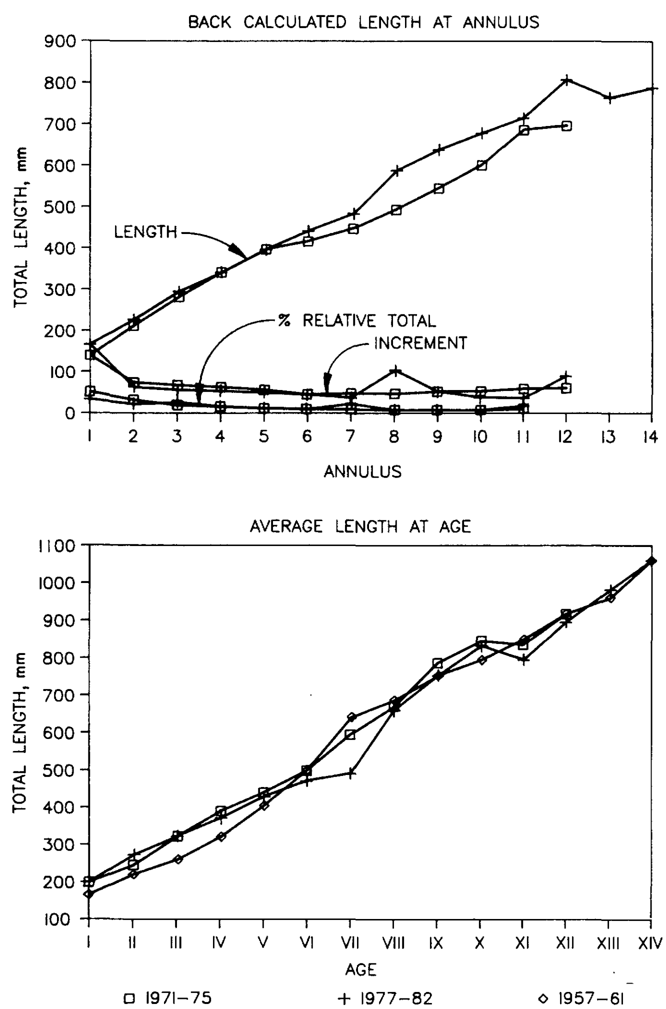


FIGURE 12-9.—*Back-calculated length at annulus, yearly growth increments and average length of age for lake trout as estimated in three time periods.*

Average length at age demonstrated lake trout growth has varied little since 1975 (fig. 12-9). Growth for lake trout from age 1 to 5 was consistently better during 1971 through 1982 than in 1957-61, and was attributed to the addition of *Mysis relicta* as a major prey species in the lake trout diet (Griest, 1977; Nesler, 1984). Growth for older lake trout during each of the three study periods varied considerably with no consistent trend evident. Improved growth of younger lake trout in 1971-82 resulted in earlier vulnerability to angling mortality under the 381-mm minimum size limit in effect. Lake trout became vulnerable to the fishery as 4-year olds in their 5th growing season, 1 year earlier than in 1957-61 (fig. 12-9).

Growth by age group from 1970 to 1982 showed relative stability in average lengths observed for age groups I through VI, while those for older age groups were much more variable (fig. 12-10). Average lengths of age groups VII and VIII declined considerably from 1970 to 1978 with some indication of improved size in later years. Average size of age IV-VI lake trout from 1970-82 also demonstrates the same pattern, but not as pronounced. These patterns probably reflect the effect of increasing angling pressure over the study period, lowering of the minimum size limit in 1974 from 508 to 381 mm, and reduction of the potential fish prey base due to reduced stocking of rainbow trout and kokanee salmon.

Food Habits

Food habits of lake trout in Twin Lakes have been investigated several times. In 1906, Juday (1906) examined stomachs of several lake trout, finding a trout (species unknown) in one and nothing in the others. Although *Mysis relicta* was introduced to Twin Lakes in 1957 (Klein 1957), lake trout stomach content analyses during 1958-59 indicated the shrimp were not included in the diet (Nolting 1968). It was uncertain at that time if the introduction had been successful. Dipterans and fish provided the bulk of the lake trout diet in 1958-59 (fig. 12-11). For comparative purposes, the three size

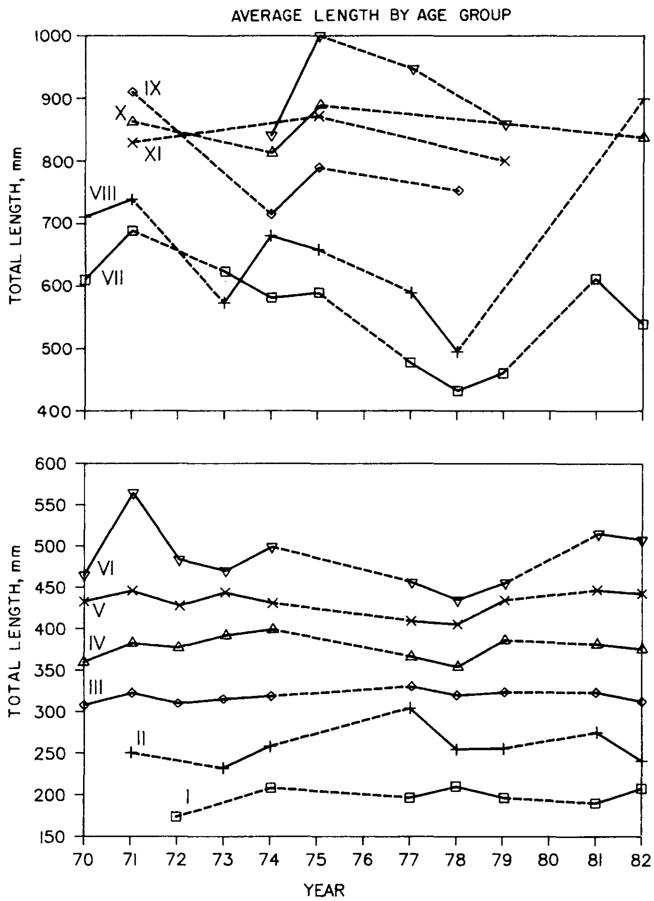


FIGURE 12-10.—*Average length of lake trout by age group by year.*

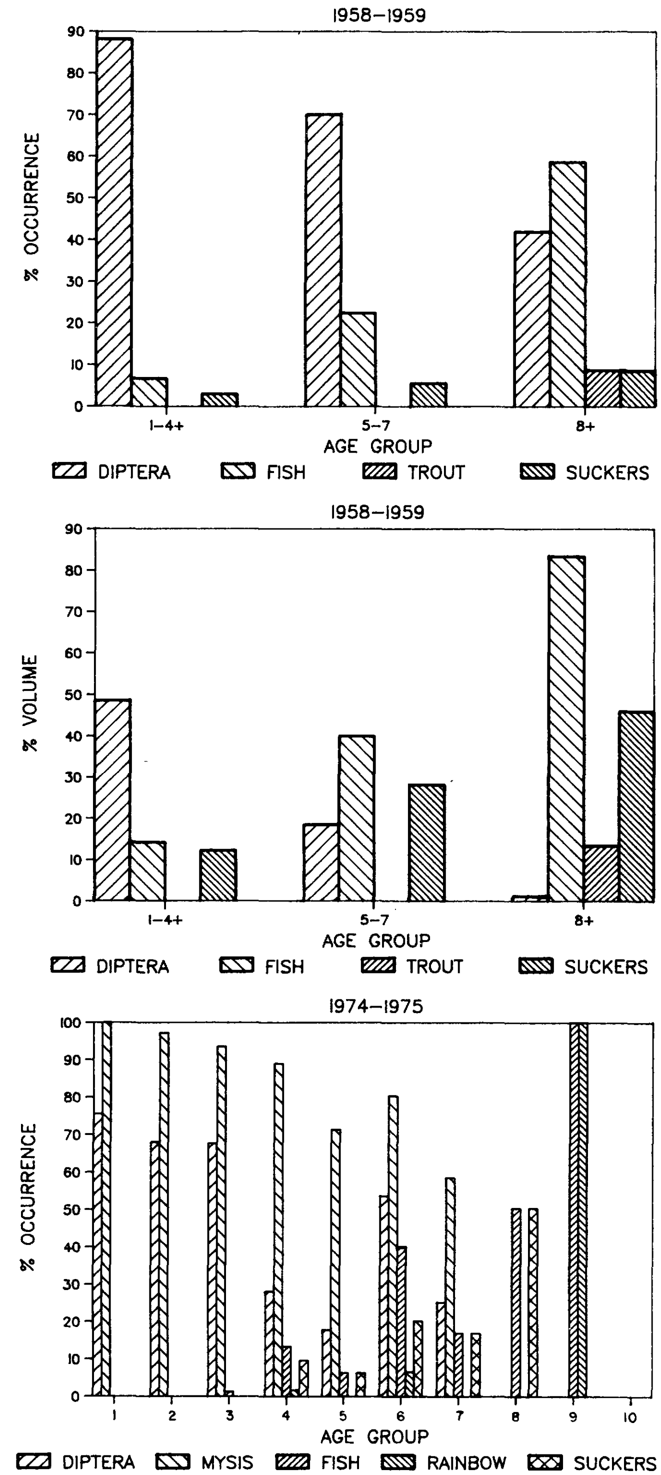


FIGURE 12-11.—*Lake trout food habits in Twin Lakes.*

classes reported in the food habit study in Nolting (1968) were translated to approximate age groups using age-growth data in the same report. As might be expected, the 1958-59 results indicated a decreasing use of Diptera and increasing use of fish. Trout and sucker appeared to be the primary prey fishes eaten. On a volume basis, 1958-59 results indicate fish increased as the primary prey item with age, and sucker provided the bulk of the lake trout diet.

A distinct change in the lake trout food habits had occurred by 1974. *Mysis relicta* had become established in the lakes, and comprised 23 percent by volume and 88 percent by frequency of occurrence in the lake trout diet (Griest, 1977). Fish comprised 71 percent by volume of the diet, but only 11 percent by frequency of occurrence for lake trout from ages IV through IX (fig. 12-11). Rainbow trout and sucker were the primary prey species. Chironomid larvae and pupae were the major constituent of the lake trout diet by numbers, and were common in the stomachs of lake trout through age VII. The presence of small freshwater clams and chironomid larvae suggested that lake trout often fed near the bottom.

During 1971-75, lake trout growth rate between ages 5 and 8 appeared to be slower than that for ages 0 to 5 and ages 8 to 12 (Griest, 1977). This leveling off of growth rate may have been due to continued reliance on *Mysis* as a major food item to the exclusion of fish. This reliance appeared to continue until around age IX, after which percent occurrence of fish was much greater than that of *Mysis relicta*. This growth pattern was not evident in 1977-82.

Reproduction

Mature fish were not captured during the 1975 fall netting efforts, indicating that either lake trout had not moved into the area to spawn or that they were using areas not sampled by the nets. Spawning must have occurred between the third week and end of October (about 9 days) because only spent fish were captured during the first week of November.

Ripe lake trout were captured during the last 2 weeks of October and the first week of November of 1977 and 1978 in depths ranging from 1.5 to 12 m. They were caught primarily along the southern shore and near the North Bay. The western one-third of the lake was not used by ripe lake trout. Nolting (1968) suggested that lake trout in Twin Lakes spawn at 2 to 10 m depths during October and November, and he also captured ripe lake trout in many of the same areas in which trout were found in the 1977-78 study. Two ripe lake trout tagged with temperature-sensitive tags preferred temperatures near 8.5 °C. Other investigators have found lake trout spawn when water temperatures were 8 to 10 °C in depths ranging from 150 mm to 61 m (Eschmeyer, 1954; Martin, 1955; Rawson, 1961; Galligan, 1962; DeRoche, 1969).

Because so few fish were captured during the spawning periods of 1974 and 1975, estimates of fecundity and sex ratios were not possible. Nolting (1968) was able to capture 10 spawners in 1958 and 14 in 1960. Total lengths ranged from 749 to 785 mm and average weight was 5.2 kilograms. Average number of eggs per kilogram of body weight was 782. Sex ratios were two to three males to one female.

Success of spawning of Twin Lakes trout was inferred by comparing composition of the catch with stocking records. Lake trout were not stocked in 1969, '72, '74, and '79, yet a year-class of lake trout for each of these years was sampled. From 1972 to 1974, fish of the 1969 year-class were caught in gill nets. In 1973, this year-class was about 25 percent of the catch. In 1974 approximately 20 percent of the catch was fish of the 1972 year-class. As shown earlier (fig. 12-8), the 1974 year-class dominated the gill-net catch in 1977-78. The 1979 year-class represented over 25 percent of the gill-net catch in 1982. Another indication of spawning success is found by comparing the number of fish of the 1971 year-class with fish of the same year-class that were stocked and had their adipose fin clipped. Of all samples taken during 1974, only 6 out of 94 fish had been adipose fin-clipped. This finding suggests that the remaining 88 fish may have resulted from natural reproduction.

Population Size

Although Twin Lakes were ideally suited to hydroacoustic sampling techniques, the low density and contagious distribution of lake trout resulted in wide confidence intervals that were ±72 percent of the mean. Three population estimates were made by combining intermediate and deep strata data collected in the upper and lower lakes during 1 day and 3 nights (table 12-4). Night estimates were pooled to produce a final estimate of the total population of both lakes of 14578±10433. This estimate is considerably smaller than the estimate calculated by mark-recapture in 1974 (73 556) and probably reflects both differences in methods and changes in the population. Despite these efforts, a useful estimate of lake trout population does not exist.

Movements

Movement rates.—Indexes of activity were highly variable in all seasons. Average movement rates for all fish from each season ranged from 1.08 to 1.61 meters per minute. Despite this variability, pooled data provided daily patterns of movement. Lake trout were significantly more active between 0830 to 1130 hours than between 2230 and 0500 hours during all seasons except winter. In addition, mean movement rate during late afternoon (1500 to 1800 h) in fall was significantly greater than the mean movement rate during the late night (2230 and 0500 h). Similar activity patterns have been found in other salmonids (Swift, 1964; Serchuk, 1976). Movement rates were not correlated with size of lake trout, surface water temperatures, or surface DO concentrations. Other investigators have found little correlation between fish movements and limnological, solunar, or weather factors (Warden and Lorio, 1975; Ager, 1976; Maclean, 1976; Dudley et al., 1977; McCleave et al., 1977).

Home ranges.—Home ranges were determined for 31 lake trout in Upper and Lower Twin Lakes using 1404 individual fish relocations. Utilized home range size varied between 11 and 173 hectares. Figures 12-12 through 12-18 illustrate examples of plotted home ranges and their general proximity to the powerplant tailrace channel. Additional plots of home ranges are in Walch (1980). There were wide variations in home range sizes among fish within the same season. Significant differences were not found between seasonal mean utilized home ranges (table 12-5). Lake trout tended to spend more time outside the utilized home range during spring or fall than during summer or winter. Although other

Table 12-4.—Population estimate of fish in deep and intermediate strata in Twin Lakes, Colorado.

Survey	Lake	Number	Relative sampling error	Relative sampling error, %	Confidence interval
Night 2	Upper	961			
Night 1	Lower	16 428			
	Total	17 389	5860	34	5259 to 34 778
Night 3	Upper	1 490			
	Lower	8 716			
	Total	10 206	3295	32	2957 to 17 455
Day 1	Upper	1 137			
	Lower	7 006			
	Total	8 143	3673	45	62 to 16 286
Night pooled	Total	14 578	5040	35	4145 to 25 010

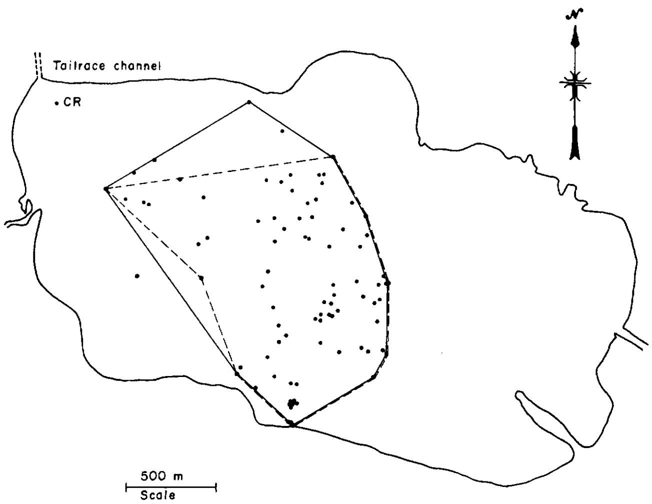


FIGURE 12-12.—*Sighting locations, cumulative, and utilized home range of fish No. 4 in Twin Lakes during the summer, 1977. Sighting locations, cumulative, and utilized home ranges are shown by solid dots, solid lines, and dashed lines, respectively. C represents the point of capture; R represents the release point.*

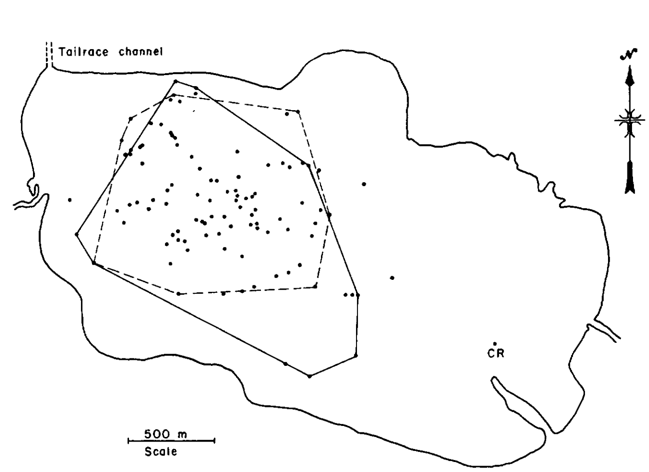


FIGURE 12-13.—*Sighting locations, cumulative, and utilized home range of fish No. 8 in Twin Lakes during the summer, 1977.* (See fig. 12 legend.)

Table 12-5.—Estimate of utilized home ranges in Upper and Lower Twin Lakes, Colorado, during 1977 and 1978.

	Home range, hectares
Overall mean	74 (n = 27)
Lower lake	67 (n = 25)
Spring	68 (n = 4)
Summer	99 (n = 11)
Fall	68 (n = 5)
Winter	27 (n = 5)
Upper lake	88 (n = 2)
Summer	80 (n = 1)
Fall	95 (n = 1)

n = number of observations.

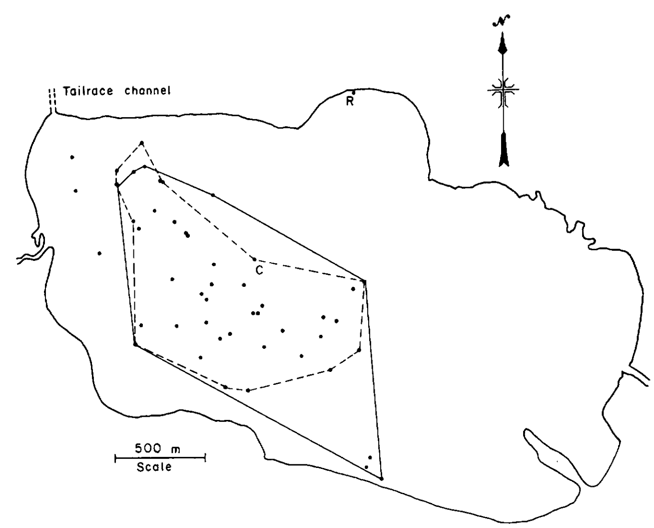


FIGURE 12-14.—*Sighting locations, cumulative, and utilized home range of fish No. 27 in Twin Lakes during the summer, 1978.* (See fig. 12 legend.)

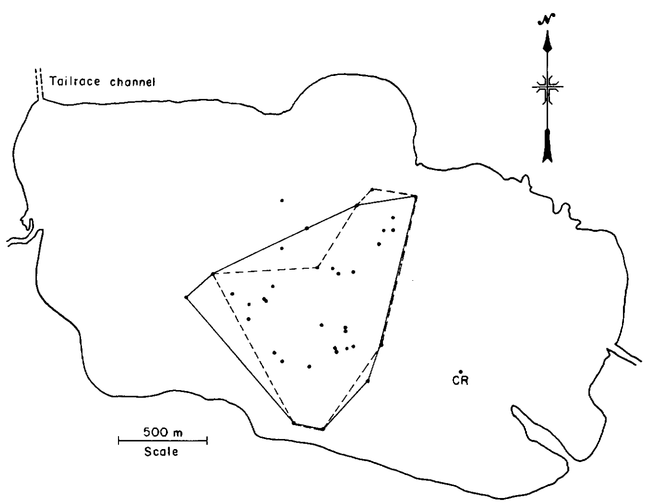


FIGURE 12-15.—*Sighting locations, cumulative, and utilized home range of fish No. 35 in Twin Lakes during the summer, 1978.* (See fig. 12 legend.)

FIGURE 12-17.—*Sighting locations, cumulative, and utilized home range of fish No. 38 in Twin Lakes during the summer, 1978.* (See fig. 12 legend.)

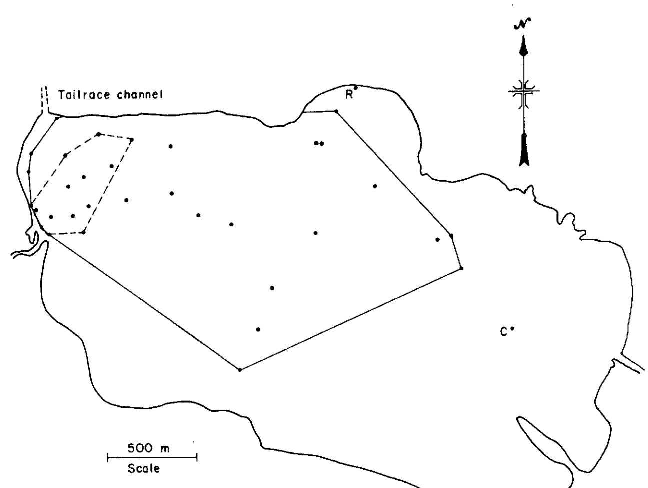


FIGURE 12-16.—*Sighting locations, cumulative, and utilized home range of fish No. 29 in Twin Lakes during the summer, 1978.* (See fig. 12 legend.)

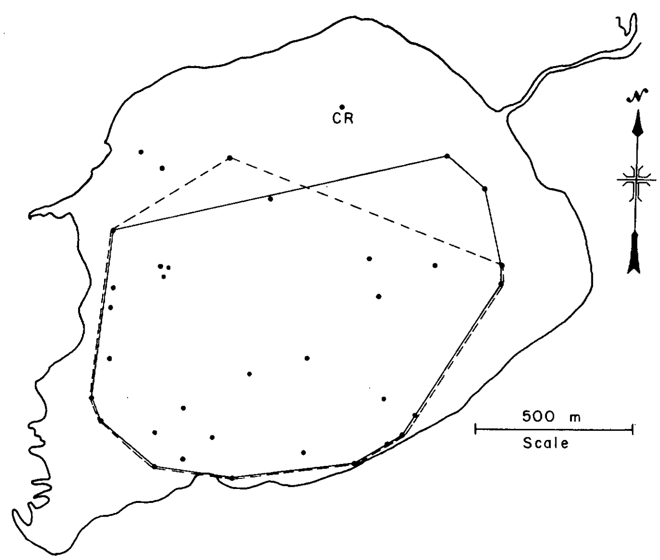


FIGURE 12-18.—*Sighting locations, cumulative, and utilized home range of fish No. 39 in Twin Lakes during the summer, 1978.* (See fig. 12 legend.)

researchers have found a direct relationship between home range size and lake trout size (Eschmeyer et al., 1952) such relationship was not found here. Lack of such a relation in Twin Lakes was probably due to movement patterns of individuals, relatively small size of the lakes, small number of fish sampled, and the limited size range of tagged fish.

Forty percent of the lake trout tracked in the lower lake during summer, fall, and winter had home ranges that extended to the vicinity of the powerplant tailrace channel. In spring, 75 percent of the tagged fish in the lower lake occupied home ranges near the powerplant. Most lake trout spent relatively little time in the eastern third of the lower lake. Only 6 percent of the lower lake sightings were in this area. Most of these sightings (57%) were obtained from suspected spawners or from one individual fish. Avoidance of the eastern one-third of the lower lake by most lake trout, except spawners, was probably due to an interaction of depth and water temperature preferences.

In contrast to fish from the lower lake, five of six fish from the upper lake used essentially all of the upper lake as home range. Tagged lake trout in the upper lake never moved through the connecting channel into the lower lake. However, two lake trout tagged in the lower lake moved into the upper lake. Movements of lake trout from the lower lake to the upper lake occurred when surface water temperatures were less than 11 °C.

Shoreward movements (movements to within 15 m of shore) were observed on 38 different occasions for 13 of 34 tagged lake trout. Over 60 percent of 22 sightings obtained close to shore during ice-free seasons occurred between 0600 and 1300 hours. Only during spawning were shoreward movements recorded at night (2230-0500 h). Larger (>550 mm) lake trout made shoreward movements more frequently than did small lake trout. This observation also occurred during standard gill-netting operations. Shoreward movements in winter were somewhat different. Fish of all sizes tended to move close to shore during the early morning and later afternoon hours.

Homing.—Six tagged lake trout that were displaced from their point of capture returned to the general area where they had been captured. In addition, two ripe male trout displaced during the spawning season exhibited homing behavior. However, three fish which were displaced from their point of capture did not exhibit homing tendencies. Possibly, these fish were on excursions outside their home range when they were captured. Shepherd (1973) found that cutthroat trout made extensive excursions throughout a lake even though the fish occupied restricted areas most of the time. Similar movements have been observed for largemouth bass (Winter 1977).

Temperature and depth preferences.—During the 1978 netting efforts by Walch (1980), significant differences in catch rates between gill-net locations were found only in the summer. During summer, the deepest station in the lower lake had a significantly higher catch rate than all other stations. Other investigators also have found that lake trout are most likely to be caught in deeper, cooler water during summer (Martin, 1951; Rawson, 1961; Nolting, 1968).

Differences in catch rates were found between seasons at a given station. Specifically, gill nets located at the mouth of the tailrace channel in shallow water had significantly higher catch rates during spring and fall than during summer or winter. The low summer catch rates in shallow water were probably due to avoidance of warm water by lake trout. Low winter catch rates could have been due to little nighttime activity or low use of the area by lake trout.

Avoidance of warm water was also indicated by telemetry data. Although lake trout carrying temperature-sensitive tags selected areas with temperatures ranging from 6.6 to 15.4 °C and depths ranging from 3.0 to 23.0 m, they clearly preferred temperatures below 12 °C; 91 percent of the tagged trout were found at these temperatures and 78 percent were found within a temperature range of 8 to 12 °C. Similarly, lake trout captured in vertical gill nets in the lower lake were taken at depths where water temperatures averaged 10.5 °C. Excursions into warm water may be related to feeding because the main prey species of large Twin Lakes lake trout—white sucker and rainbow trout—were typically in areas where the water temperature exceeds 12 °C (Krieger, 1980). Large (longer than 550 mm) trout were responsible for 63 percent of the sightings in water with temperatures greater than 12 °C. Other investigators have also suggested excursions by lake trout into warm water were for feeding (Martin, 1951; Galligan, 1962).

When depth could be determined from the tags and temperature profiles of the lake, trout were found to be within 3.2 m of the bottom during 66 percent of the sightings. Lake trout found in mid-water depths were most often small. Trends in daily vertical movements were not apparent for any lake trout tagged with temperature-sensitive tags; rather, changes in vertical distributions occurred during all times of day. Thus, movements into mid-water depths were not fully explained by searches for mysid shrimp that move into the water column primarily at night (Gregg, 1976).

Distribution of juvenile and young-of-the-year lake trout.—Only 12 lake trout (which had not been stocked by CDOW) between 150 and 230 mm in length were captured in over 400 hours of netting efforts with small-mesh gill nets. Nine of the small lake trout were captured during summer months in water deeper than 15 m and within 1 m of the bottom. The other three were captured during winter at locations where water depth ranged from 3 to 10 m. All small lake trout were captured in areas occupied by adult and large immature lake trout. Extensive otter trawling yielded one lake trout 60 mm long; this fish was captured near a rocky outcrop about 13 m deep.

Galligan (1962) and Griest (1977) also had little success capturing lake trout ranging in size from 50 to 240 mm with gill nets; but other researchers have captured lake trout in this size range by trawling near the bottom in water deeper than 18 m

(Eschmeyer, 1954, DeRoche, 1969, Pycha, 1977). Unfortunately, trawling was not conducted at depths greater than 14 m during this study.

Information was not collected on movements of lake trout fry in this study. In environments similar to Twin Lakes, hatching was found to occur in February or March, with fry leaving the spawning rubble by early June (Royce, 1951; Martin, 1955; DeRoche, 1962). DeRoche (1969) found that lake trout fry moved immediately to deep water after they left the spawning rubble. Other investigators have also captured lake trout fry in deep-water areas (Royce, 1951; Eschmeyer, 1954; Pycha 1977). Yearling lake trout have been found to prefer water temperatures of 11.6 to 11.8 °C (McCauley, 1970; Goddard et al., 1974). Numerous lake trout fingerlings stocked by CDOW were captured in the vicinity of the tailrace channel during electrofishing efforts in October 1978. These fingerlings had been stocked at a location about 1.7 km away from the tailrace channel.

METHODS—CATOSTOMIDS

Fish Collection

Only data on relative abundance and distribution of sucker species were collected from standardized netting during 1977–83. Extensive sampling was conducted in 1978 to acquire life history data on sucker, including age and growth, food habits, reproduction, and mortality. Experimental multifilament gill nets were set on the bottom and perpendicular to shore in 12 locations once each month from May to October 1978 to capture fish and gather data on depth preference and distribution. In addition, nine sets were set under the ice in February and March. Stations were selected to ensure sampling within a variety of depths and habitat types in each lake. Nets were set before sunset and pulled shortly after sunrise. Effectiveness of daylight capture by the net was negligible, as demonstrated by a number of daytime sets. Therefore, only hours of darkness were counted in length of set.

From June to October, three vertical gill nets, 3 m wide and from 10 to 20 m deep, were fished for several 24-hour periods in various locations to gain information on vertical distributions of the sucker. Bar mesh sizes ranged from 6 to 32 mm.

To capture white sucker less than 200 mm in length and all sizes of longnose sucker, shoreline areas were sampled at night with an electrofishing boat equipped with electrodes attached to a foreboom. The lower lake was sampled in June, August, and October and the upper lake in August and October.

Qualitative samples of sucker larvae were taken weekly from mid-July to late September in the north bay area of the lower lake by seining water less than 1.3 m deep with a large-mesh plankton net. To assess relative numbers of larvae appearing along shoreline of both lakes, a biweekly survey was conducted. About 200 m of shoreline were examined in 10 areas of the lower lake and 6 areas of the upper lake, and an estimate was made of number per square meter in each area. Later, an index number ranging from one to five was assigned to each area based on relative larval density compared to all other locations. This index was used to supplement investigations of areas used as spawning grounds within the lakes.

Data Analysis

Age and growth.—All captured fish were weighed to the nearest gram and measured to the nearest millimeter. Species, sex, condition of gonads, presence of clipped fins, gear, and data and location of capture were reported. Only fish with mature gonads were sexed.

Both opercular bones were removed from about 450 white and 200 longnose suckers and cleaned by methods described by McConnell (1951). Operculars were then viewed under a magnifying glass using transmitted incandescent light, and the center and annuli were marked on transparent tape laid along the opercular axis. The distance between these marks was measured when the tape was magnified 6.57 times on a projector. Opercular bones and scales were taken from 70 fish and used for age comparisons.

Total fish length was regressed against opercular radius; and first, second, and third degree equations were fitted to the data. Linear equations gave correlation coefficients of 0.98 and 0.97 for white and longnose suckers, respectively; these were used to describe both length-to-radius relationship:

$$TL = b_0 + b_1 R \qquad (12\text{-}1)$$

where:
TL = total length, mm,
R = opercular radius, mm $\times$ 6.57,
b_0 = constant describing the intercept, and
b_1 = constant describing the slope.

Body lengths at annulus formation for each age-class and grand averages were then computed using this relationship. The length-weight relationship was described by a log form of the linear equation:

$$\log W = \log b_0 + b_1 \log L \qquad (12\text{-}2)$$

where:
W = weight, g,
L = length, mm,

b_0 = constant describing the intercept, and
b_1 = constant describing the slope.

Separate regressions were calculated according to species, season (spring, summer, and fall), and state of maturity (immature, mature) because of the equation varied with maturity and season. Statistical comparisons of length-weight relations were made to test for significantly different coefficients among various groups. Unless otherwise stated, results were considered significant at the 5-percent level.

Food habits.—Gut contents from fish of the same species captured in the same month and lake and within a given size group were pooled into a single sample. This procedure eliminated the effort of examining individual stomachs while still allowing determination of the food habits of the population. Comparisons could then be made between different seasons. The number of stomachs used in each group ranged from 2 to 10. Size groups were: less than 100 mm, 100 to 200 mm, 200 to 300 mm, and (for white sucker) greater than 350 mm. These size groups were based on feeding habits associated with various sizes, as presented by Stewart (1926). Subsampling of gut contents and food item identification procedures were detailed by Krieger (1980).

Mass of food items and parts were reconstituted to obtain a better estimate of total mass eaten. Average live mass (wet) of organisms were determined from samples taken in Twin Lakes or from the stomach contents, if the item had not been crushed or digested. Reconstituted mass of the food item was then calculated by multiplying the number of individuals of species in a sample by its assigned mass. Gut contents of fish less than 100 mm in length were analyzed qualitatively. Fish were examined individually and the percentage of volume and percentage of occurrence of each food item were recorded.

The biological index, TU, of Keefe and Bergersen (1977) was used to compare the diversity of food items consumed by suckers of different species and sizes in both lakes at different seasons. The index number selected the degree of diversity of food items in sucker gut contents, and the evenness with which they were used. A high index number indicated that the fish were using a wide variety of food items and in nearly equal amounts. Differences were considered significant at the 5-percent level.

Reproduction, mortality, and population size.—Entire ovaries were wrapped in cheesecloth, identified by number, and placed in 10-percent formalin. Later ovarian tissue was removed, and total weight of loose eggs determined. The weight of eggs was subsampled to estimate total numbers of eggs which were regressed on length and weight for both species.

Mortality was estimated using the method of Ssentongo and Larkin (1973). This method enables calculation of a mortality coefficient from the mean length and the length at first capture. Only fish caught by similar gear (gill nets for white sucker, electrofishing for longnose sucker) were used in the computation. To test the validity of this method, a more traditional estimate using age-class frequencies was calculated for white sucker by methods of Robson and Chapman (1961). Aged fish from the June and July gill-net samples were used in the latter method.

RESULTS AND DISCUSSION

Age and Growth

Growth in length—white sucker.—Opercular bones were found to be a more suitable bony structure than scales for use in aging sucker because they function as a "key scale," eliminating the variation caused by using various-sized scales from the same fish. This effect increased the correlation coefficient for the body length-opercular radius regression. Similar conclusions were reached by Ovchynnyk (1965). Opercular bone ages agreed well with scale ages up to age group AG, VII after which the differences in age determinations increased with increasing ages. Usually the operculars revealed more annuli than did the scales.

The average length of any particular year-class varied considerably from year to year, but the variability decreased as the fish became older. It was assumed that this variation was due to unequal growth of young fish among the various age groups. Average lengths of age groups I and II were distinct, but beyond AG IV length was a poor indicator of age.

A linear regression was fitted by least squares to the opercular radius and fish length data:

$$TL = 0.3830\ R + 6.9531 \qquad (12\text{–}3)$$

where:
TL = total length, mm, and
R = opercular radius, mm $\times$ 6.57.

Average lengths at annulus formation for each age group were calculated by applying the opercular radius measurements to equation 12-3 (table 12-6).

Female white suckers were longer than males at all years of life. Difference in length between males and females were greatest between AG VII and IX, when male lengths averaged 93 percent of female lengths. Females also lived longer than males, which were not found to live beyond 11 years. Many females attained AG XII and a few 17-year-old individuals were found.

Table 12-6.—Calculated total lengths, at the end of each year of life, and annual growth increments for male and female white sucker. Captured from Twin Lakes, Colorado (1978).

Age group	Number of fish	Calculated length at end of year of life, millimeters											
		1	2	3	4	5	6	7	8	9	10	11	12
							Male						
VI	11	75	115	163	185	237	281						
VII	13	68	111	158	201	241	277	303					
VIII	9	61	99	134	173	219	254	284	316				
IX	2	69	104	149	192	231	260	283	307	325			
X	2	63	95	149	210	253	283	314	339	351	375		
Grant average		68	108	153	189	235	272	296	318	338	375		
Annual growth increment		68	40	45	36	46	37	24	22	20	37		
Percent of female length		99	95	97	94	96	97	94	91	93	99		
							Female						
V	3	79	130	181	226	263							
VI	7	74	119	172	210	250	297						
VII	20	67	113	155	197	218	262	299					
VIII	29	66	113	154	196	253	279	315	362				
IX	27	71	116	162	196	246	287	319	342	364			
X	17	69	109	143	199	237	277	308	337	355	375		
XI	10	62	108	168	217	251	295	324	353	370	384	397	
XII	8	72	125	172	217	207	308	332	355	372	385	397	405
Grand average		69	114	158	202	244	281	314	350	364	380	397	405
Annual increment		69	45	44	44	42	37	33	36	14	16	17	8

Growth in length—longnose sucker.—As found with the white sucker, the average length of a particular year-class varied from year to year, but the variability decreased in older age groups. Except for AG I, there was substantial overlap in length between all ages. Length intervals noted for AG IV and VI are entirely included within the intervals of age groups 1 year younger and older than these age groups. The opercular radius and fish length relationship was obtained from 140 fish:

$$TL = 0.3265\ R + 1.9561 \qquad (12\text{-}4).$$

where:
TL = total length, mm, and
R = opercular radius, mm × 6.57.

Average length at annulus formation for each age group were calculated by applying the opercular radius measurements to equation 12-4 (table 12-7).

Growth during the first year of life was rapid, followed by a declining rate of growth in later years. Growth of both sexes was nearly equal until the fourth year of life when growth of males declined rapidly. Males were typically shorter than females at all ages. Females appeared to live 1 year longer (to AG VII) and reached greater length than males.

Both sucker species in Twin Lakes grew more slowly and white sucker lived longer than suckers from most other areas reported in the literature. The relatively scarce food source (LaBounty and Sartoris, 1976) may be a factor limiting the standing crop of sucker. However, it is believed to be responsible for the slow growth because suckers in Twin Lakes almost always had abundant food in their guts and were plump. Cold temperatures and a short growing season may allow only minimal growth each year.

Growth in weight—white sucker.—Fish length-weight regressions were calculated and analyzed for six groups of fish representing fish from three seasons (spring, summer, fall) and two sizes (immature and mature) by Krieger (1980):

Spring:
immature log: $W = 3.126 \log L - 5.289$ (12-5)
mature log: $W = 3.027 \log L - 5.049$ (12-6)
Summer:
immature log: $W = 3.018 \log L - 5.027$ (12-7)
mature log: $W = 2.887 \log L - 4.693$ (12-8)
Fall:
immature log: $W = 3.046 \log L - 5.117$ (12-9)
mature log: $W = 3.047 \log L - 5.099$ (12-10)

Table 12-7.—Calculated total lengths, at the end of each year of life, and annual growth increments for male and female longnose sucker. Captured from Twin Lakes, Colorado (1978).

Age group	Number of fish	Calculated length at end of year of life, millimeters 1	2	3	4	5	6	7
					Male			
II	3	87	144					
III	13	87	135	182				
IV	6	81	114	161	194			
V	4	61	108	155	185	216		
VI	1	77	117	169	224	260	285	
Grand average		78	127	172	193	225	285	
Annual growth increment		78	46	45	21	32	60	
Percent of female length		96	97	98	93	91	102	
					Female			
II	3	92	156					
III	28	90	135	184				
IV	9	76	114	157	207			
V	17	80	127	171	210	245		
VI	7	74	130	171	209	248	278	
VII	3	85	136	177	201	250	280	304
Grand average		84	131	175	208	246	279	304
Annual growth increment		84	47	44	33	38	33	25

Growth in weight—longnose sucker.—Fish length-weight relationships were calculated and analyzed for six groups of fish:

Spring:

immature log: $W = 3.516 \log L - 6.124$ (12-11)

mature log: $W = 3.071 \log L - 5.183$ (12-12)

Summer:

immature log: $W = 3.187 \log L - 5.183$ (12-13)

mature log: $W = 2.974 \log L - 4.930$ (12-14)

Fall:

immature log: $W = 3.185 \log L - 5.428$ (12-15)

mature log: $W = 3.080 \log L - 5.184$ (12-16)

The slope and intercept values for immature fish were inversely related. Equations for mature sucker, however, differed in a parallel fashion; therefore, slope and intercept values changed in the same direction. A general trend developed for immature fish of a given species to have higher slope coefficients than mature fish within a given season. However, the differences were not usually significant. Apparently, immature fish tended to be slightly plumper than adults.

Slope coefficients, from the summer regressions, appeared to be lower within a given species and size, but only in one case were these differences significant. This finding might be expected for mature fish which had no gonadal development at that time, but it does not explain the differences in the immature fish. Possibly the fish were increasing in length at a rapid rate during the summer, thus, having less weight at a given length.

Length-Frequency Distribution

From the standardized netting samples, white sucker ranged from 120 to 500 mm total length, *TL*, while longnose sucker ranged only from 130 to 310 mm *TL* (fig. 12-19). Peaks in length-frequency distributions of both sucker species may be attributed to age groups, especially in the smaller size range. Peaks were less distinct in the size range greater than 300 mm. Using age-growth data for white sucker from above, the peaks at 140, 180, and 230 mm would represent age groups 2, 3, and 4. Age groups 5 and 6 appear to mix together in the 250- to 300-mm size range. The peaks at 320, 350, and 390 mm would represent age groups 7, 8 to 9 combined, and 10 to 11 combined. The presence of white sucker 420 to 500 mm *TL* indicates a small portion of the population reaches ages 12 to 17.

Age-growth data for longnose sucker indicate length frequency-distribution peaks for this species (fig. 12-19) at 150, 180, 210, 240, 290, and 300 mm would correspond to age groups 2, 3, 4, 5, 6, and 7. It is notable that, despite similarities in growth rates between the two sucker species, growth or survival

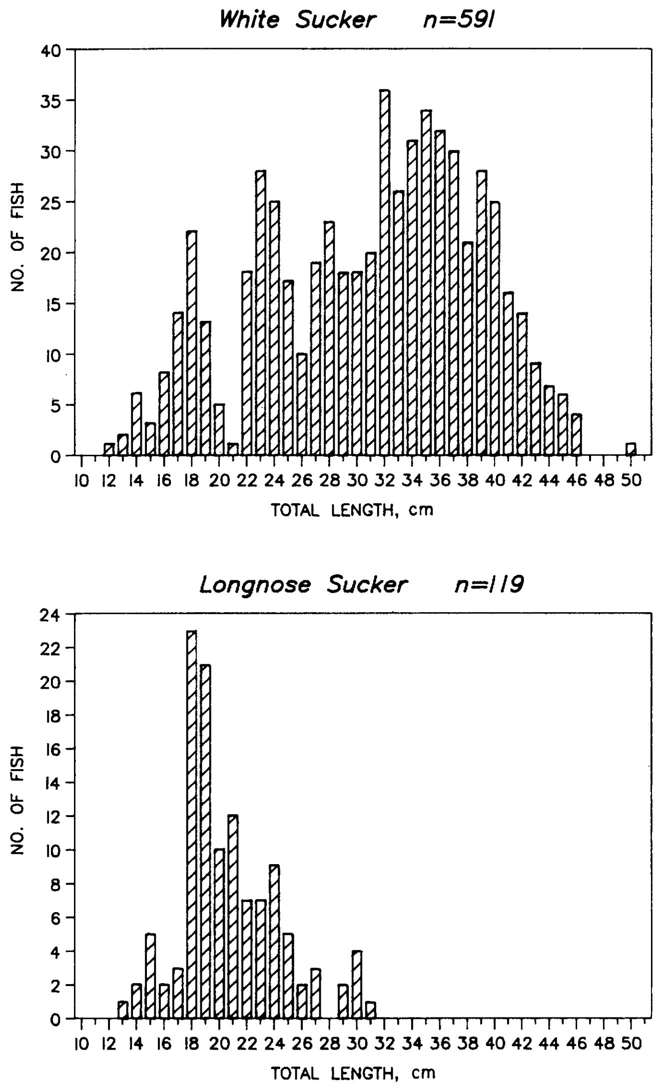


FIGURE 12-19.—*Length-frequency distributions of the white and longnose suckers in Twin Lakes.*

of longnose sucker beyond 310 mm and age 7 was negligible.

Food Habits

Fish less than 100 mm long.—The diets of white and longnose suckers were similar. Debris and silt were major constituents of the gut contents by weight and were more prevalent in fish that had eaten predominantly chironomid larvae. Cladocera were eaten most often, but small chironomid larvae were the dominant food item by weight.

Chironomids and cladocera were partitioned between the two sucker species by season. Chironomid larvae reached maximum numbers in the diets of white sucker, during the summer, but cladocera were important in the spring. In contrast, cladocera were found more often and in greater weight in gut contents of longnose sucker, during the summer. Chironomid larvae were not part of the longnose sucker diet during the summer but were important during the fall.

Fish 100 to 200 mm long.—Generally, the diets of both sucker species of this size were composed predominantly of chironomid larvae (figs. 12-20, 12-22). Cladocera also were eaten frequently by both sucker species and fingernail clams were important to the white sucker in the lower lake. From 10.8 to 25.7 percent of the total weight of the diet consisted of debris and silt/sand. Plant material was found in white sucker guts but not in longnose sucker guts, whereas trichoptera and homoptera were found in longnose sucker but not white sucker.

Sucker diet changed seasonally and were different between the populations of the two lakes. Both sucker species in the upper lake used more chironomid larvae than fish from the lower lake, which ate more cladocera and copepoda. Cladocera were most important to the lower lake longnose population, but were of little consequence in the upper lake longnose sucker diet. Zooplankton were prominent numerically in upper and lower lake white sucker diet but of little importance in terms of weight. In fall, small fingernail clams (1 to 5 mm) were abundant in the diet of Lower Twin Lakes white sucker but were not found in Upper Twin Lakes sucker diet.

The biological index, TU, values were significantly higher for the lower lake diets than upper lake diets in all seasons for both species (table 12-8). This difference is probably because the upper lake sucker population was forced to rely more heavily on chironomid larvae due to the low diversity of invertebrates in the upper lake. Although complete information on abundance of all invertebrates in Twin Lakes is lacking, LaBounty and Sartoris (1976) found in the upper lake one of the most sparse invertebrate fauna recorded in the literature. Despite a greater density of chironomid larvae in the lower lake sucker, diets (in the lower lake) were composed of a smaller percentage of chironomid larvae than diets of fish from the upper lake. Apparently, both sucker species will select a variety of food items if they are sufficiently abundant.

Fish 200 to 350 mm long.—Gut contents of both sucker species in this group from both lakes were again dominated by chironomid larvae in most seasons (figs. 12-20 to 12-22). As was observed in the smaller size group, white and longnose sucker from the upper lake consumed proportionally more chironomid larvae than did those from the lower lake. However, variations on the main diet were

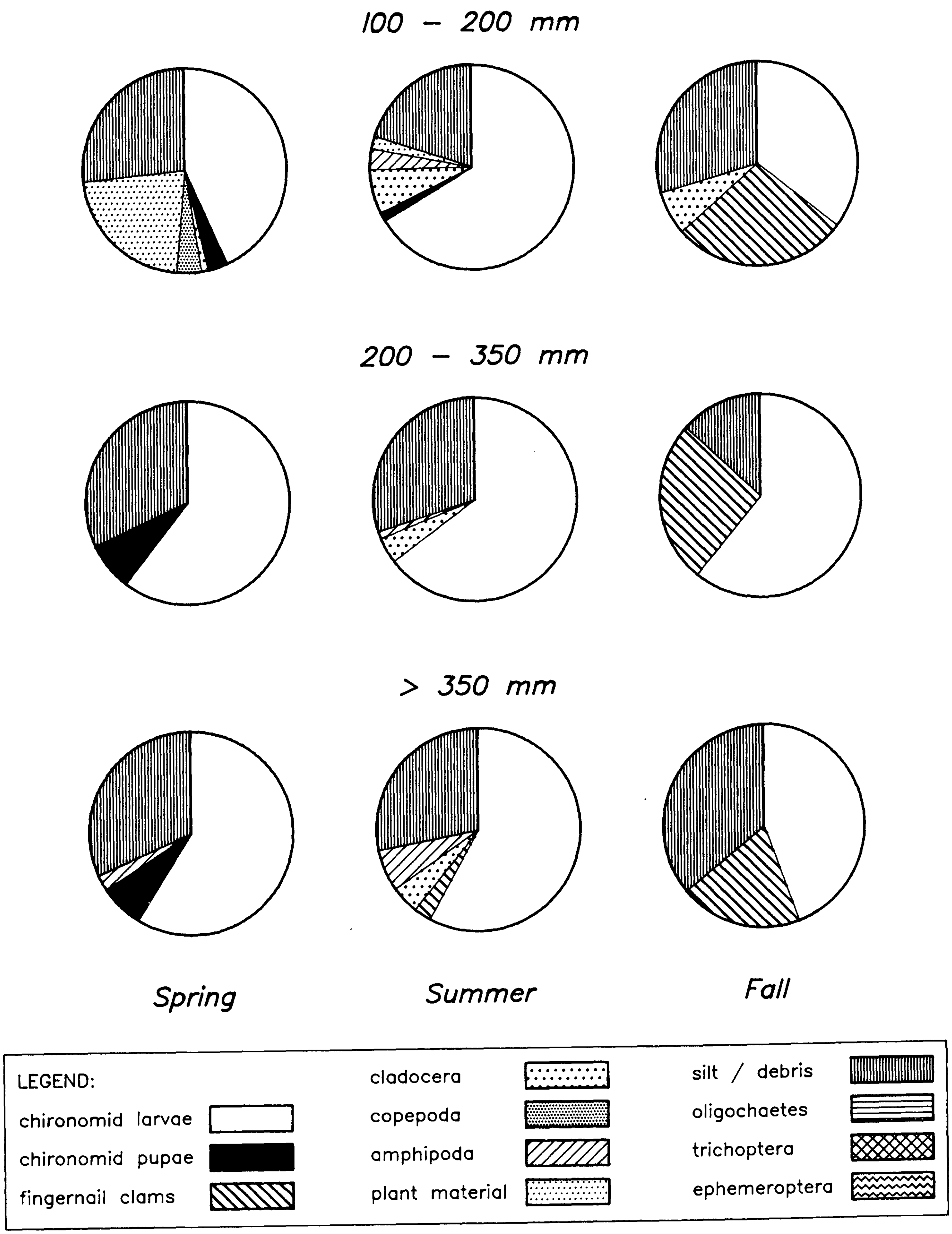


FIGURE 12-20.—*Food habits of the white sucker in Lower Twin Lakes, 1978.*

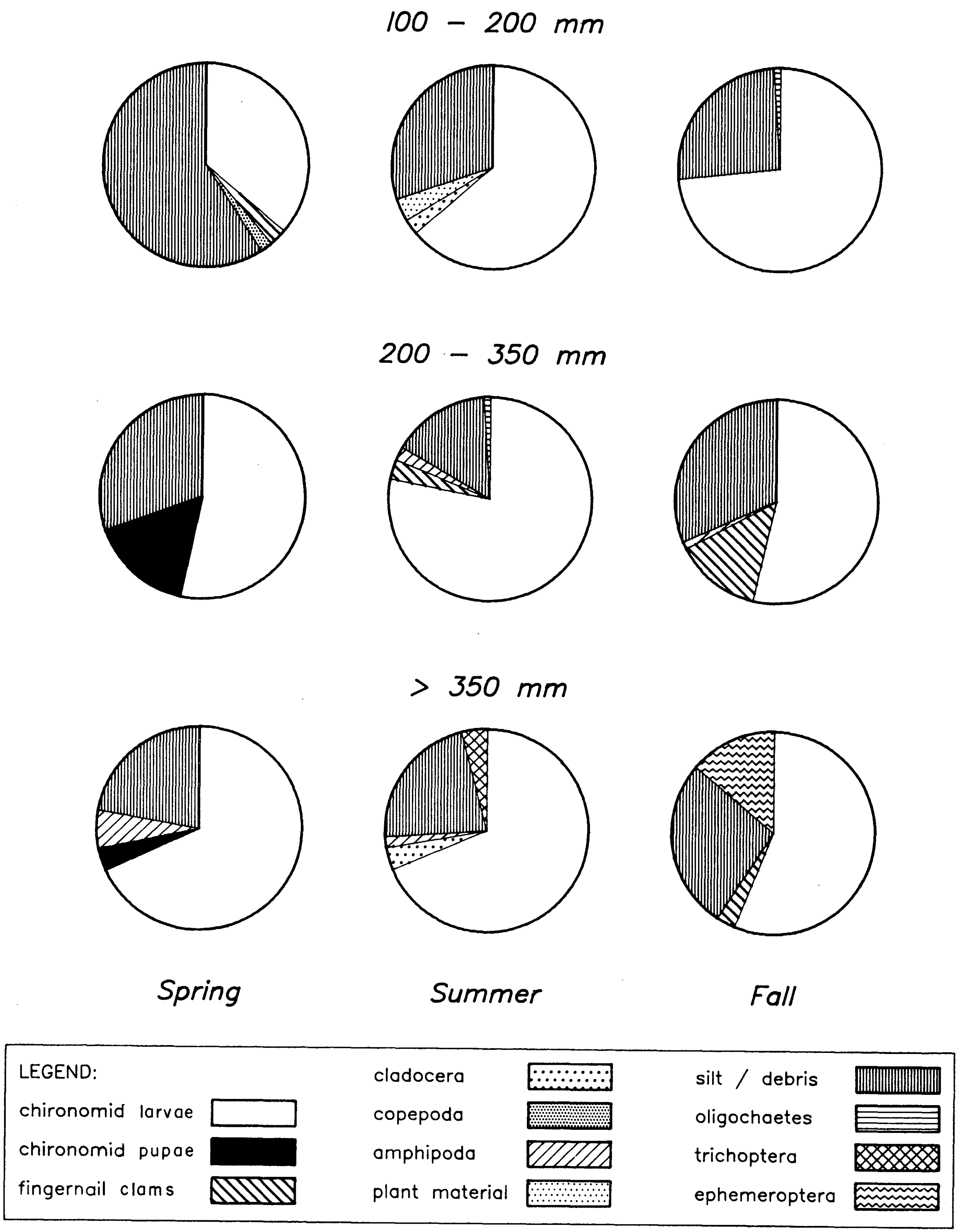


FIGURE 12-21.—*Food habits of the white sucker in Upper Twin Lakes, 1978.*

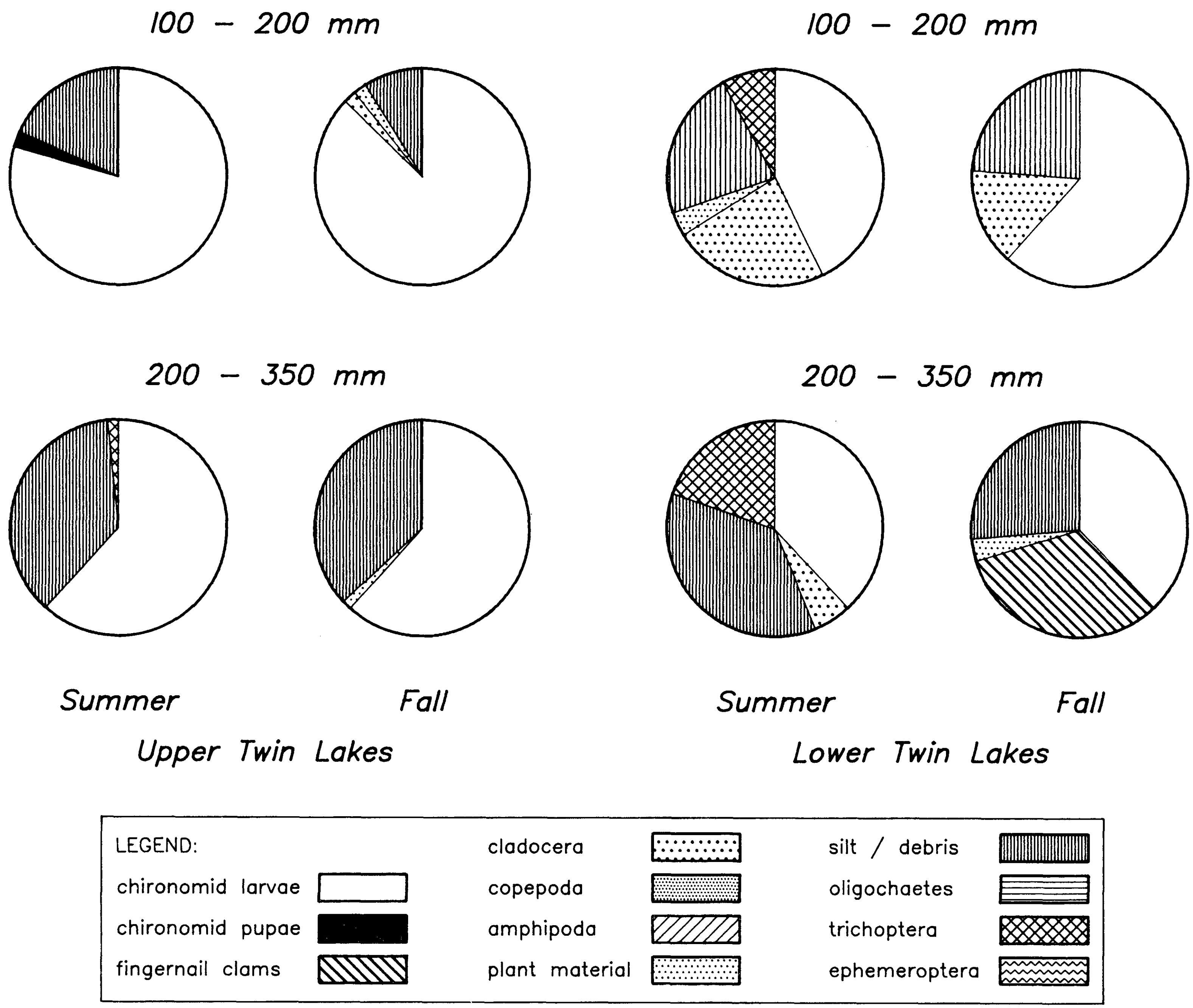


FIGURE 12-22.—*Food habits of the longnose sucker in Twin Lakes, 1978.*

Table 12-8.—Biological diversity index (TU), by fish size and lake, for food items consumed by white and longnose suckers. Confidence intervals (±95%) are in ( ), Twin Lakes, Colorado.

	White sucker			Longnose sucker		
	Spring	Summer	Fall	Spring	Summer	Fall
	100 to 200 mm			100 to 200 mm		
Lower lake	0.544	0.526	0.641		0.304	0.452
	(0.001)	(0.003)	(0.005)		(0.008)	(0.006)
Upper lake	0.061	0.485	0.140		0.299	0.238
	(0.008)	(0.014)	(0.018)		(0.009)	(0.020)
	200 to 350 mm			200 to 350 mm		
Lower lake	0.199	0.465	0.485		0.514	0.570
	(0.006)	(0.003)	(0.005)		(0.006)	(0.006)
Upper lake	0.446	0.287	0.368		0.166	0.088
	(0.010)	(0.006)	(0.010)		(0.008)	(0.010)
	>350 mm			>350 mm		
Lower lake	0.251	0.529	0.461			
	(0.006)	(0.002)	(0.004)			
Upper lake	0.185	0.453	0.158			
	(0.010)	(0.003)	(0.008)			

observed between seasons, species, and lakes; and a greater number of different food items was observed in fish this size than in the smaller fish.

Fingernail clams were a significant (20 to 45%) part of the diet of both species in the fall except for longnose sucker from the upper lake; clams were not found in these fish in any season. Cladocerans were important numerically, but not on a weight basis, to lower lake white and longnose suckers only in the summer. White sucker fed on more chironomid pupae and a large number of larger prey items such as trichoptera and mysids not eaten by smaller fish. Food items of lesser importance were copepods in white sucker from the upper lake, homoptera in longnose sucker from the upper lake, trichoptera in longnose sucker from the lower lake, and amphipoda in both sucker species from both lakes. Silt and debris (20 to 40%) made up gut contents of both species from both lakes.

The diversity index was significantly higher for diets from the lower lake in the summer and fall (table 12-8). Both sucker species from the lower lake ate numerous chironomid pupae, fingernail clams, and cladocera during various seasons, thereby offsetting a dependence on chironomid larvae. The numbers of fingernail clams in the fall diet of both species in the lower lake were sufficient to cause a notable increase in the fall diversity index.

Fish greater than 350 mm long.—Fish of this size group ate larger food items and had a more diverse diet than did fish of smaller size-classes (figs. 12-20, 12-21). Insect larvae and lake trout eggs became important items in the diet of upper lake sucker in the summer and fall, respectively; and amphipoda and gastropoda appeared in the gut contents of fish from the lower lake. However, numerically and by weight, chironomid larvae continued to be the chief animal food in fish gut contents from both lakes. Chironomid larvae constituted 57 percent of the food weight from lower lake fish and 66 percent from upper lake fish. On a weight basis, fingernail clams were important to lower lake sucker in the fall but not upper lake fish; clams were responsible for 6.1 percent of the total food weight in lower lake sucker. Amphipoda ranked third (3.2%) in importance to lower lake fish. Trichoptera and cladocera, however, were the second and third most important items, respectively, in the diet of upper lake sucker. Despite their overall ranking, trichoptera were found only in the summer.

The major difference in food habits of sucker from the upper and lower lake was the greater importance of chironomid larvae to the diet of sucker in the upper lake. In contrast, sucker diets in the lower lake were composed of larger percentages of less common food taxa resulting in higher TU diversity values (table 12-8). Based on catch data, the lower lake contained a higher density of sucker than the upper lake. In addition, the diversity of benthic organisms in Lower Twin Lakes is greater. The combination of these factors may intensify competition for the available food and result in increased consumption of less common food items. If this is true, lower lake sucker would especially be considered opportunistic feeders in taking whatever foods were competitively available. Food items found in the lower lake may not be present, or are in low abundance in the upper lake, which would account for the greater reliance on chironomid larvae in the upper lake (LaBounty and Sartoris, 1976). The sparse benthic fauna of the upper lake has been attributed to interactions between the biological and chemical dynamics of the upper lake and metals pollution entering from Lake Creek (LaBounty and Sartoris, 1976).

Seasonal differences in diets could be explained by the increased availability of certain invertebrate taxa at various seasons. Chironomid pupae were eaten in the spring when they were most available, while cladocera—which may reach highest densities during the warmer months—were found more often in summer diets. Fingernail clams may be eaten most in the fall because of some unexplained increase in availability or because chironomid larvae have become less abundant. Benthic studies by LaBounty and Sartoris (1976) did suggest that numbers of chironomid larvae generally decrease in late summer and early fall.

Lake trout egg predation by sucker.—Gut contents of large (more than 350 mm) white sucker captured in the fall from Upper Twin Lakes contained a total of 66 trout eggs, which made up 13.9 percent of total food mass for that group. In gill nets intended to capture spawning lake trout in the lower lake, 13 large white suckers were caught, of which, 6 contained trout eggs. Electrophoretic analyses of the eggs revealed that all were from lake trout.

The amount of egg predation by sucker in Twin Lakes was probably underestimated because the fish used for food analysis were captured in mid-October, just before the peak of lake trout spawning. If suckers captured in early November over spawning areas had been included, egg predation may have been greater.

It appears that sucker predation on lake trout eggs was minimal in Twin Lakes. Numerous nets were set over spawning areas but few suckers were taken. Of those captured, only the large white sucker had eaten eggs. Atkinson (1931) found 18 of 29 large white suckers captured over lake trout spawning beds to contain about 1500 eggs. However, Rawson and Elsey (1948) failed to find eggs in the stomachs of longnose sucker captured over lake trout spawning beds.

Because suckers are not able to obtain eggs from rock crevices efficiently, most eggs eaten were probably in more accessible locations. The probability of these eggs hatching properly—had they not been eaten—was in all likelihood reduced. DeRoche and Boyd (1955) suggested that lake trout eggs that were eaten were likely not lodged deeply in crevices of spawning rubble. They also determined that lake trout were predators on their own eggs.

Competition for food.—Food habits of sucker and lake trout overlap little in Twin Lakes. *Mysis relicta* was the predominant food item of the lake trout in Twin Lakes (Griest, 1977). Griest also found that chironomid larvae and pupae were eaten by 48.8 percent of the lake trout examined, but only constituted 6.1 percent of total volume of food eaten. Twin Lakes sucker seldom ate mysids but used chironomids extensively.

The greatest similarity in food habits occurred between sucker and young lake trout in the consumption of chironomid larvae and pupae. However, this may not have constituted competition for food between the two species. Food competition can occur only if the common food item is limiting to one or both of the fishes due to their combined predation. It is doubtful that the lake trout eat sufficient number of larvae to limit the sucker. Conversely, the lake trout could easily turn to mysids in the event that sucker predation substantially reduced the numbers of available chironomids.

Reproductive Biology

Sex ratio and maturity—white sucker.—Of the mature fish that were captured, 346 (40%) were males and 525 (60%) were females. *Chi-square* testing revealed these proportions were significantly different [$\chi^2 = 36.75$, 1 df (degree of freedom)].

Females first reached maturity at AG V and males at AG VI. However, because of the low number of mature females at AG V and VI, it is believed that they are not mature as a group until AG VII. More fish were sexed as males than females at AG VI. This occurred at no other age, and suggests that males reached maturity—on the average—at AG VI 1 year earlier than females. At AG VII, 68 percent of the fish were mature; and at AG VIII, 88 percent were mature. Some fish were not mature at AG IX.

The female white sucker in Twin Lakes lived much longer than males, and the female longnose sucker slightly outlived males. Differential mortality for the white sucker is reflected in the large female to male ratio. Although a high reproductive potential is possible with a population that has a large percentage of females, the delayed maturity of white sucker reduces this potential. Mortality acts for a longer time on the population before they are able to reproduce, thus reducing the size of the spawning population.

Sex ratio and maturity—longnose sucker.—Sexes were determined for 344 fish; 159 (46%) were males and 185 (54%) were females. A *chi-square* test indicated that there was no significant difference in sex proportions ($\chi^2 = 1.96$, 1 df).

Although both sexes had mature members at AG II, only 20 percent of AG fish were mature. By the fourth year, 91 percent of all fish were mature, and it was evident that both sexes had reached maturity at AG III. All fish AG VI and older were mature.

The longnose sucker sex ratio was probably equal because of the nearly equal longevity of the sexes. Geen et al. (1966) found the female to male ratio of a spawning group of longnose sucker to increase each year when they returned to the same area. He attributed this to different mortality. Hayes (1956) and LaLancette (1973) reported sex ratios of 1:1, as well as equal longevity, of the sexes in longnose sucker.

Other investigators have observed an earlier age at maturity of white sucker and a later maturation for longnose sucker than was observed in the study by Krieger (1980). Hayes (1956) found that male white sucker matured at AG III and females at AG V, and male longnose matured at AG III and females at AG IV. Both sexes of white sucker in Gamelin Lake, Quebec, matured during their third year (LaLancette, 1973). Male longnose matured 1 year earlier (IV) than females (V) in Yellowstone Lake (Brown and Graham, 1953).

Fecundity—white sucker.—Estimates of fecundity were made on 19 fish captured in June, ranging in size from a 316-mm (365-g) fish with 10 888 eggs to a 44-mm (844-g) female containing 21 970 eggs. Eggs per gram of body weight ranged from 24 to 33 and showed no trends with increasing weight. Ovary weight as percent of body weight (less ovary) was highest for mid-sized fish. Total length in millimeters, L, was regressed on total egg numbers, Y:

$$\log Y = 2.29 \log L - 1.70 \qquad (r = 0.97) \qquad (12\text{-}17)$$

The relationship of female weight in grams, W, to total egg count, Y, was linear:

$$Y = 20.74\ W + 2700 \qquad (r = 0.96) \qquad (12\text{-}18)$$

Fecundity—longnose sucker.—Number of eggs ranged from 2361 to 6750 for females of 176-mm (49 g) and 264-mm (181 g) length, respectively. Eggs per gram of fish (less ovary) ranged from 41 to 76. Ovary weight as percent of body mass increased with increasing size of the fish, except for the largest individual. The relationship of total length, L, to total egg count, Y, was logarithmic:

$$\log Y = 1.7 \log L - 0.292 \qquad (r = 0.97) \qquad (12\text{-}19)$$

Weight of fish, W, regressed on total egg count, Y, resulted in a linear equation:

$$Y = 26.80\ W + 2126 \qquad (r = 0.95) \qquad (12\text{-}20)$$

Spawning—white sucker.—At the end of May, all mature white suckers were gravid. By mid-June, all captured males were spent and all fish captured after mid-June were spent. Temperature of the water at the time of spawning ranged from 8.4 to 10.0 °C, depending on depth. White sucker did not use the Lake Creek inlet for spawning, but observations of a large congregation south of the powerplant suggest that they were spawning in that area. Larval fish first appeared along shoreline areas on July 12. During biweekly surveys of both lakes, highest numbers were noted in a north shore bay and along the southwest shores. Few larvae were seen in the upper lake.

Spawning—longnose sucker.—All mature fish captured in May were gravid. By mid-June, nearly 60 percent of the females and most males had spawned. Small percentages of gravid fish continued to be caught throughout August. Larval sucker could not be identified to species and all observations reported in the previous section for white sucker may be applied to the longnose sucker as well.

A correlation between initiation of spawning activity and water temperature was determined by Campbell (1935) for white sucker and by Brown and Graham (1953) for longnose sucker. Geen et al. (1966) counted fewer numbers of spawning suckers when stream temperatures declined. They also found that longnose sucker spawned earlier than white sucker. Hayes (1956) determined that white sucker used areas within Shadow Mountain Reservoir, Colorado, from July 1 to 14 when water temperatures were 14.7 to 18.5 °C.

Depth of spawning would have to be determined in Twin Lakes to reveal temperature preferences. Suckers may select specific depths, specific temperatures, or both. There did not appear to be any differences in time of spawning between fish in the two lakes, even though temperatures in the upper lake are usually from 2 to 4 °C colder at all depths because of the inflow from Lake Creek. It is plausible that the suckers may respond to stimuli other than temperature. Spawning in response to photoperiod may account for the coinciding spawning periods of the suckers in the two lakes.

Longnose sucker in Twin Lakes appeared to have a longer spawning period than did the white sucker. Hayes (1956) suggested that sporadic spawning occurred from late May until mid-August, based on occurrence of fry in Shadow Mountain Reservoir.

In Twin Lakes, the longnose sucker may have developed eggs and yet not have spawned, thereby accounting for the gravid females found in August.

Hayes (1956) described a typical white sucker spawning area as having water 0.5 to 2.1 m deep and a grave substrate. Also, he concluded that fry were observed only at locations that he had found to be spawning grounds for adult sucker. Assuming this relationship existed at Twin Lakes, the suckers were spawning near stations with shallow shorelines with sand and gravel substrates. Because all areas in Twin Lakes that had high abundance of fry were also considered sheltered, the possibility exists that fry had sought out these areas.

Sucker fry appeared 33 days after the first suckers spawned in tributaries to Yellowstone Lake, Brown and Graham (1953), and Geen et al. (1966) noticed fry migrating back to Sixteenmile Lake 30 days after the onset of spawning activity. Geen et al. (1966) observations included a 2-week hatching time at temperatures less than 10 °C followed by 1 to 2 weeks development in the gravel before moving. In Twin Lakes, larvae were seen on July 12; if development proceeded as noted, first spawning would have occurred June 9-14.

Relative Abundance and Distribution

A total of 1402 white suckers and 49 longnose suckers were captured in standardized gill-net sampling from August 1977 through November 1983. In 313 net sets, an average of 4.5 white suckers and 0.16 longnose sucker were caught per net. Overall, 82 percent of the white sucker and 98 percent of the longnose sucker were captured at stations I, VIII, and the three random shallow stations (fig. 12-1). At station I, 35 percent of the white sucker were collected. This strong preference of suckers for the littoral zone of Twin Lakes is hypothesized to be a function of predation pressure from lake trout, the lack of deep-water cover, and the presence of a large aquatic macrophyte community providing food and refuge in the northeast corner of the lower lake. Much of the deeper lake bottom consists of glacial silt with little structure to provide cover for fish prey species. This situation would limit the presence of all but the larger white sucker in the deep part of the lake basin.

Observations from gill netting and electrofishing suggested the presence and abundance of the relatively smaller longnose suckers were also linked to rocky shoreline habitat or submerged vegetation. Monthly catch rates for white sucker, at station I and the random shallow stations, ranged from: 11 to 21 fish per net in May, 42 to 53 fish per net in August, and 11 to 25 fish per net in November. The increased catch rate in August may have resulted from increased activity by suckers in the warmer water temperatures. These results were corroborated by similar results from intensive sampling in 1978.

1978 Distribution—white sucker.—These fish tended to be captured in nets set near the shore in both lakes, whereas nets in deeper pelagic areas caught relatively few fish. From June to October, most fish were found within 2 m of the bottom. However, in July, 54 percent of the suckers were caught 3 to 6 m from the bottom. White sucker avoided the areas near the powerplant tailrace.

Summer catches of white sucker were higher than spring catches at most stations; increased activity associated with warmer water may explain these results. Smaller fish in the shallow areas of the lower lake accounted for much of the catch at those stations, whereas large fish seemed to prefer areas of greater slope. Only large fish were caught in deep water.

Catch rate of white sucker in shallow areas greatly decreased during fall. LaBounty and Sartoris (1976) reported that the decrease in chironomid larvae numbers from April to November 1975 was due, in part, to predation by suckers, and that chironomid densities were higher at greater depths in all seasons. Suckers may have moved to deeper water after depleting chironomid larvae by predation in the shallows. This conjecture is supported by the increased catch of white sucker at the deep station in the lower lake. White suckers may have also moved offshore because lake trout were using the shallow areas to spawn. Spawning lake trout are often large enough to feed on suckers, and Walch (1980) determined that lake trout in Twin Lakes used shallow areas near Hartmann Point for spawning.

From June to October, white suckers were frequently found at depths less than 2 m in Twin Lakes. The reason for this depth selection is unknown. Gut contents of sucker caught in open water were compared to food of those captured on the bottom and obvious differences were not found. Horak and Tanner (1964) used vertical nets in Horsetooth Reservoir, Colorado, and also reported that white sucker preferred to be near the bottom; 68 percent were found within 1.5 m of the bottom, and 85 percent within 3 m of the bottom in an area of the reservoir that ranged from 5.5 to 12.2 m deep.

1978 Distribution—longnose sucker.—In contrast to the white sucker data, the gill-netting results for longnose sucker indicated that the area near the tailrace channel was favored by longnose sucker in spring and fall. Large numbers of larval longnose sucker near the channel suggest that this species spawns in the adjacent riprap. During the summer, these fish appeared to move to an area just east of the tailrace channel and to a cove along the southwestern shore of the lower lake. Because the fish caught in these areas were mostly adults, these may be spawning areas as well.

Longnose suckers were netted along the southern shore of the upper lake in the spring—but along the western and northern shores in the summer and fall. Preferred areas were characterized by riprap or small rocks. Suckers were not captured in pelagic sets or in vertical gill nets.

Because of the small numbers of longnose sucker that were captured, analysis of distribution is difficult for this species. Based on the gill-net efforts, a higher density of longnose sucker exists in the upper lake than the lower lake. Electrofishing data corroborated this observation because greater electrofishing effort was required to capture the same number of longnose sucker from the lower lake than from the upper lake.

A qualitative analysis of electrofishing catch indicated that longnose sucker and white sucker greater than 300 mm long share similar habitats. During electrofishing efforts in June, the highest numbers of both species were taken in the north bay. In October, more individuals of both species were found near the south and southwest shores, which have steeper slopes.

Gill nets were effective only during hours of darkness in Twin Lakes and information on daytime distribution was not gathered. Lack of catch during the day was likely because of less movement during the day rather than fish avoiding the net by sight. Spoor and Schloemer (1938) concluded that the visibility of the gill net did not result in a different catch rate between day and night sets, but Campbell (1935) found aquarium-held fish disperse more at night and show increased swimming rates.

Mortality

White sucker.—Ages of all white sucker in the June and July gill-net samples were determined. Fish were first fully captured at AG VIII. Mortality was calculated using these data and methods of Robson and Chapman (1961). The instantaneous mortality rate was determined to be 0.533 with a 95 percent confidence interval, CI, of 0.496 to 0.570. Survival was 0.587 (Ricker, 1975).

The method of Ssentongo and Larkin (1973) was also used to allow more fish (409 total) to be used in the computations of a mortality coefficient. The average length in the catch was 373 mm. The instantaneous mortality rate with 95 percent confidence limits was then 0.597±0.013. Survival rate was 0.550.

These survival estimates are within the range reported elsewhere. Survival rates for white sucker in Gamelin Lake, Quebec, during 1970–71, were found to be 0.550 for males and 0.560 for females (LaLancette, 1973). Coble (1967) analyzed tagging information to estimate survival of white sucker from Lake Huron. His estimates ranged from 0.600 to 0.758. He also used length-frequency data to correlate numbers at successive lengths with survival rate. Survival rate estimates ranged from 0.885 for fish 392 to 418 mm (fork length) to 0.517 for fish 494 to 519 mm long. Olson (1963) reported a survival rate of 0.869 for white sucker of all ages during 1957 and 1958 in Many Points Lake, Michigan.

Longnose sucker.—Because aging of a total catch of longnose sucker was not performed, the survival was estimated using the lengths of 173 fish and the method of Ssentongo and Larkin (1973). The average length at first capture was 170 mm and the average length in the catch was 237 mm. These numbers were used with the coefficients L and K, from the von Bertalanffy equation to calculate an instantaneous mortality rate of 0.532 with 95 percent confidence limits of 0.492 to 0.573.

Sucker as prey of lake trout.—Of approximately 150 lake trout stomachs examined during 1977–78, 4 contained a total of 5 suckers (3 white, 2 unknown). The lake trout ranged in size from 411 to 543 mm in total length. Griest (1977) examined 216 stomachs of Twin Lakes trout in 1974 and 1975 and determined that suckers were taken infrequently (6.2% by occurrence), but constituted 49.2 percent of total food volume. Suckers were found more frequently in lake trout more than 400 mm long.

Rawson and Elsey (1948) noted that less than 7 percent of 50 lake trout examined in Pyramid Lake, Alberta, contained longnose sucker. They believed that the percentage might have increased had larger fish been examined. Martin (1951) found that lake trout preferred yellow perch (*Perca flavescens*), but occasionally took white sucker. DeRoche (1962) found that white suckers were seldom eaten by adult lake trout weighing less than 2.7 kilograms.

Using biotelemetry techniques, Walch (1980) found that lake trout were equally distributed in all areas of the lake during spring and fall. Although they were generally more abundant in deeper, cooler water (12 °C) in summer, fish over 550 mm long invaded warmer waters. He attributed movements into warmer water to feeding behavior. Suckers were prevalent in habitat similar to that of the lake trout during spring and fall. Although sucker and lake trout were more segregated during summer, suckers were susceptible to predation due to the movements of trout into warmer water. Martin (1951) similarly concluded that although white suckers were usually separated from lake trout by temperature preferences in the summer, suckers eaten in the summer were taken by the lake trout that had entered warmer water to feed.

OTHER SALMONIDS

Relatively few numbers of other salmonid species were collected by way of standardized gill netting. Despite annual stocking of approximately 55 000 to 80 000 rainbow trout, only 97 were captured in 330 net sets during 1974-83 (0.29 fish per net). Of these trout, 94 were caught at station I, and the 3 random shallow stations (50 and 44 fish, respectively). Stomach content analyses conducted on some rainbow trout showed these fish appeared to use *Mysis* as food during the overwinter period; rainbow trout in the Mt. Elbert Forebay were exploiting *Mysis* entrained in "pump-back" flows. Overall, 62 of 108 rainbow trout checked during the study were found to have ingested *Mysis* (fig. 12-23). Although the length range of rainbow trout containing *Mysis* was similar to the length range of those fish without *Mysis*, a trend was apparent that use of *Mysis* as prey by rainbow trout may increase with size of the

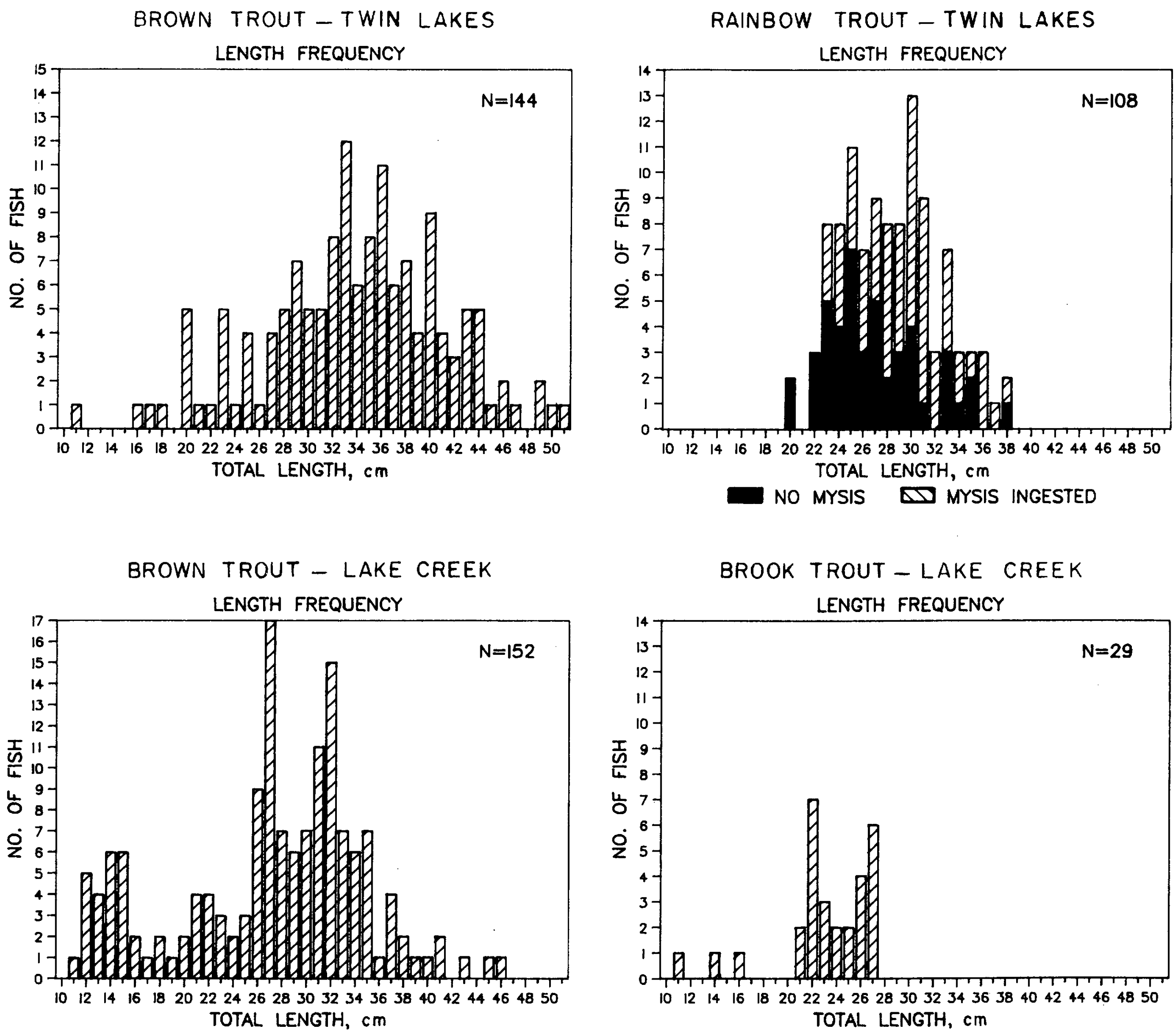


FIGURE 12-23.—*Length-frequency distributions of other salmonids in Twin Lakes and Lake Creek.*

fish. Primarily, rainbow trout appeared to prey upon *Mysis* when the opportunity was presented.

Brown trout also were captured infrequently in the standardized gill-net sampling. Selective sampling efforts indicated brown trout were only abundant in the inlet area of Upper Twin Lakes and in Lake Creek. Small congregations of reproductively mature brown trout were sampled each fall in the inlet area. Length-frequency distributions of the lake and stream populations suggested that smaller brown trout, especially young-of-the-year and juveniles, were relatively more abundant in Lake Creek. Larger brown trout were found in the littoral habitat of the inlet (fig. 12-23). Brown trout existing in Lake Creek below Twin Lakes Dam were also observed to ingest large number of *Mysis.* It was apparent these trout were exploiting an opportunity to prey upon *Mysis* entrained in outlet flows. Overall, 19 of 30 brown trout checked—including 9 Lake Creek fish and 21 lake fish—had ingested *Mysis.*

Brook trout were captured almost exclusively in the inlet bay of Upper Twin Lakes and in Lake Creek. These fish were usually small, ranging from 110 to 290 mm total length (fig. 12-23). Each year, reproductive effort was readily apparent by this species in Lake Creek. Some recruitment was evident, based on the collection of young fish 110 to 180 mm *TL.*

Other salmonids existing in the lakes included kokanee salmon and cutthroat trout. Before 1984, only incidental captures of kokanee were recorded, except in 1974, when 21 salmon ranging in length from 180 to 220 mm were examined for ingestion of *Mysis.* Six salmon contained shrimp. Only five cutthroat trout, ranging in length from 210 to 310 mm *TL* were collected during the study.

* * * * *

Sport Fishery

Chapter 13

T. P. Nesler and L. M. Finnell (retired)

Colorado Division of Wildlife
Fort Collins, Colorado

INTRODUCTION

Twin Lakes received attention as a lake trout fishery in June 1907 when a local angler caught a lake trout (*Salvelinus namaycush*) measuring 100.3 centimeters long, 54.2 centimeters girth, and 9.2 kilograms (*Outdoor Life*, Aug. 1907, p. 220). This specimen was reported at that time as the largest trout ever caught in Colorado.

An evaluation of the fishery in 1959 concluded the lakes were supporting "heavy" fishing pressure (Nolting, 1968). From 1941 through 1961, the fishing season at Twin Lakes was restricted to May through October. In 1961, the daily bag limit for lake trout was set at four, and a minimum size limit was set at 381 mm. Year-round fishing was permitted in 1962, heralding the beginning of ice-fishing at the lakes (Nolting, 1968). These regulations remained in effect until 1972, when the bag limit was reduced to two fish and the size limit was raised to 508 mm. In 1974, the size limit was evaluated as ineffective, and lowered to 381 mm. A two-fish bag limit and 381-mm minimum size limit for lake trout and an eight-fish bag limit for all other trout species in aggregate with no minimum-size limit remained in effect at Twin lakes from 1974 through the end of the study period in 1983.

Rainbow (*Oncorhynchus mykiss*) and lake trout are the major game fish species in Twin Lakes. Lake trout provide the major attraction for the summer boat and winter ice fisheries. Stocked, creel-size rainbow trout provide a put-and-take summer shore fishery that is considerably larger than the lake trout fishery, but Twin Lakes are primarily recognized for their annual production of large lake trout to the creel. Stocking of rainbow trout by the CDOW (Colorado Division of Wildlife) has occurred annually since 1953 (Nesler, 1981b). Creel survey studies of the sport fishery were conducted during the summer seasons from 1973–82, and during the winter seasons 1973–74 through 1982–83.

METHODS

Procedures for angler counts, creel census interviews, and estimation of angler effort and harvest were based on the instantaneous count method described in Neuhold and Lu (1957). The stratified-random creel survey technique used at Twin Lakes represented a modified version of the instantaneous count method, and was described in Powell (1975). Modifications within the systematic structure of the creel survey sampling program were made between major creel survey periods of 1973–75 (Finnell, 1981), 1977–79 (Nesler, 1981b), and 1981–83 (Nesler, 1984). These variations concerned:

- Stratification of sample periods,
- Number of days sampled per strata,
- Number of counts per sample day, and
- Stratification of angler interview periods within a sample day.

A generalization of the creel survey technique used for winter and summer periods is described, but details of modifications can be found in the reports cited previously.

Summer Creel Survey

Sampling periods were stratified into 2-week blocks from May through September. The start of the sampling period in May varied by year. Sampling was further stratified by weekdays, weekend days, and holidays. Two weekdays, 1 Saturday, and 1 Sunday were selected randomly per block. On 3-day holidays, 2 or 3 of the days were sampled. Total shore anglers and boats were counted four times during the 13-hour fishing day (0700–2000 h). The fishing day was split into four equal time periods, and initial count times for each sample day were selected randomly from 13 possible times (using 15-min intervals) within the first count period (0700–1015 h). Three subsequent counts were made every 3.25 hours after the first count time. In September, the fishing day was reduced to 12 hours (0700–1900 h). Shore angler and boat counts were made from a boat circling the perimeter of both lakes.

Angler interviews per sample day were split into a morning period (0700–1330 h) and an afternoon period (1330–2000 h). Shore anglers were interviewed during one period and boat anglers during the other. Interviews for each angler type were split equally per block between morning and afternoon periods, but the sequence of interview sampling in the two time periods was selected randomly. During 1977, interview periods were stratified into four periods, including early and late morning and afternoon time intervals. A fishing party was used as the interview unit because members of the same party were likely to begin and end their fishing trip collectively. Interview data sought included:

- Sample date,
- Number of anglers in the party,
- Starting fishing time for each person,
- Interview time,
- Total fishing time for the party, and
- Number of each game fish species kept.

Presence of tags or clipped fins was noted for all game fish. Length and weight measurements and a scale sample were taken from all lake trout. For rainbow trout, only numbers and clipped fins were noted. To determine fishing time, census clerks were instructed to ask what time the angler began fishing, and if he/she had been fishing continuously up to the interview time. All nonfishing time approximating 1/2 hour or more that could be accounted for was discounted from the angler's total time. Also, clerks were instructed to interview anglers in proportion to their relative concentration over the lake area. Most interviews represented incomplete trip data.

Winter Creel Survey

Sampling periods were stratified into 1- or 2-week blocks from December through March. The start of the creel survey in December varied with ice-cover formation. Only the lower lake was included in the winter creel survey. Sampling periods were also stratified by weekday and weekend day, with 1 or 2 weekdays and 1 weekend day selected randomly from each block. Angler counts were made four times at equal time intervals throughout the sample day similar to summer seasons. Length of the fishing day was adjusted monthly from 10 hours in December and 11 hours in January, to 12 hours in February and March (0700–1900 h). Count times were selected in the same manner as described for summer seasons, but adjusted accordingly to fishing day length. Interviews were conducted throughout the sample day as anglers completed their fishing trips. Only completed trip information was collected. Interview data sought was similar to summer season interview data.

Angler Effort and Harvest Estimations

Angler count and interview data were grouped by angler type (shore, boat, ice), weekend days, and weekdays for each month. Estimates of angler hours and fish harvested were calculated for four combinations of angler-type/day-type during the summer months for both the upper and lower lakes. Only weekday and weekend day estimates of hours fished and fish harvested were calculated during the winter months. Monthly estimates of hours fished for shore, boat, or ice anglers by weekdays or weekend days were calculated by determining mean number of anglers per count (see Summer Creel Survey) per weekend or weekday stratum, multiplying the mean by the number of hours in the fishing day, and then multiplying by the number of available days (weekend or weekdays) in a given month. Mean numbers of boat anglers per count per sample day-type had to be estimated from boat angler interview data. A mean number of anglers per boat was calculated each month using interview data for both weekdays and weekends, then multiplying the boat count data by this factor.

During the winter, anglers fished through the ice inside portable, enclosed shelters. Consequently, direct observation of anglers was not always possible at a given count time. The entire lower lake was observable from one of two main checkpoints on the roads accessing the lake; and often, a running total of anglers on the ice could be recorded. In some instances, however, an exact count was uncertain, but could be corrected when anglers from a particular fishing shelter passed through the interview checkpoint.

Harvest estimates were calculated by determining catch rate as number of fish per angler-hour from interview data within a stratum, and multiplying by the hours fished estimate for the same stratum (i.e., harvest = catch rate × hours fished estimate for June shore anglers, weekdays). Catch rates and harvests were estimated for each stratum. Estimates for individual strata were summed to provide total shore, boat, weekday, or weekend day estimates of angler hours and harvest for each month and for the season. The calculation of these estimates and their variances was performed by computer program (Powell and Bowden, 1979). Interview data also provided:

- Harvest species composition,
- Mean length of lake trout caught,
- Percent return to the creel of stocked rainbow trout, and
- Residence time in the lake of stocked rainbow trout.

Seasonal estimates of hours fished and fish harvested were compared using the standard normal deviate, or *Z*-test (Snedecor, Cochran, 1967), to test for significant differences. The equation used for the *Z*-test was:

$$Z = \frac{x_1 - x_2}{[(sex_1)^2 + (sex_2)^2]^{1/2}}$$

where:
x_1 = larger estimate of hours or harvest,
x_2 = smaller estimate of hours or harvest, and
sex_1 or sex_2 = respective estimated standard errors of x_1 and x_2.

RESULTS

Hours Fished and Fish Harvested During Summer

Estimated total hours on the lower lake fluctuated from about 90 000 hours in 1973 to a low of 67 000 hours in 1977, and back to slightly above 90000 hours in 1982 (fig. 13-1). Estimated total angler hours on the upper lake varied between 20 to 29 000. Total angler hours fished by shore anglers constituted the major segment of the summer fishery on both lakes, and fluctuations in the shore angler estimates were responsible for the yearly fluctuations observed. Average monthly estimates of hours fished by shore anglers demonstrated a July peak during the

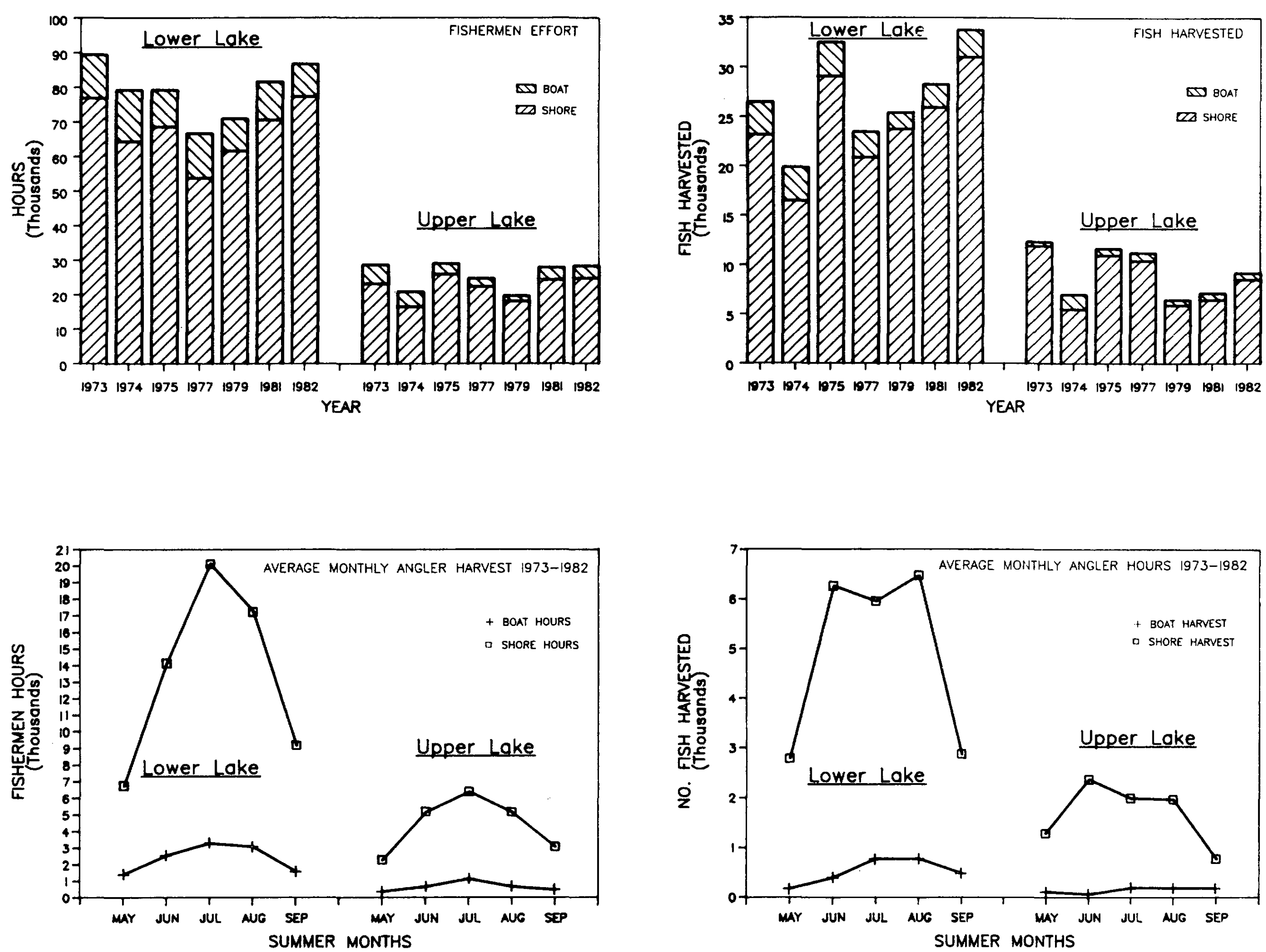


FIGURE 13-1.—*Annual and average monthly estimates of anglers' effort and fish harvested at Twin Lakes during summer.*

summer season for both lakes, though differences were less pronounced on the upper lake. Average monthly estimates of shore angler hours ranged from 6800 to 20 200 hours on the lower lake and 2300 to 6400 hours on the upper lake. Average monthly hours fished by boat anglers also demonstrated July peaks on average for both lakes. Estimates of average monthly boat angler hours ranged from 1400 to 3300 hours on the lower lake and from 400 to 1100 hours on the upper lake.

Estimated total fish harvested on the lower lake ranged from a low of about 20 000 in 1974 to a high of near 34 000 in 1982 (fig. 13-1). Estimates of total fish harvested on the upper lake fluctuated between 6400 and 12 500. Shore anglers were also responsible for a dominant share of the harvest on both lakes, and were responsible for the yearly fluctuations observed. Average monthly estimates of shore angler harvest on the lower lake demonstrated the bulk of the harvest was distributed equally over June, July, and August, with between 6000 and 6500 fish harvested each month. Average monthly estimates of shore angler harvest decreased to just less than 3000 fish each in May and September. A similar trend was noted for the average monthly shore angler harvests on the upper lake, although average harvests ranged from 800 to 2400 fish. Estimates of average monthly boat angler harvests on the lower lake ranged from only 200 fish in May to a peak of 800 fish each in July and August. Estimates of average monthly boat angler harvest on the upper lake were always equal to or less than 200 fish.

Statistical Comparisons

Coefficients of variation were used as indicators of the relative precision of the creel survey estimates (table 13-1). The best relative precision was associated with estimates of total summer hours and total shore angler hours (6 to 10%), followed by total boat angler hours (7 to 15%), total summer harvest (7 to 17%), and total shore harvest (8 to 18%). The relative precision of estimates for total boat harvest and the monthly estimates (for both shore and boat hours and for harvest) were progressively less, reaching 29 to 72 percent for mean monthly boat harvests. This hierarchy of relative precision indicates that the more comprehensive totals describing the fishery were the most sensitive statistics for reflecting changes.

The Z-test comparisons were made between yearly totals of hours fished and harvest for both lakes (table 13-2). Significant differences were noted between the lowest estimates of hours fished on the lower lake in 1977 and 1979 and the highest estimates in 1973 and 1982 (fig. 13-1). For the upper lake, the low estimates of fishermen hours in 1974 and 1979 were significantly less than the estimates for the other five summer seasons. Estimates of fish harvested on the lower lake in 1975 and 1982 were significantly greater than the low harvests estimated in 1974 and 1977 (fig. 13-1). For the upper lake, the 1982 harvest estimate appeared to be a statistical midpoint between the three higher estimates (1973, 1975, 1977) and the three lower estimates (1974, 1979,

Table 13-1.—Coefficient of variation, %, for estimates of anglers' effort and fish harvested for Twin Lakes by strata.

							Total summer hours							
1973	8	8	6	9	6	7	7	6	6	7	9	6	10	9
							Total shore hours							
1974	9	10	6	0	6	8	7	6	7	7	9	6	10	9
							Total boat hours							
1975	9	7	8	9	9	10	8	10	15	11	14	15	15	11
							Mean monthly shore hours							
1976	17	17	13	20	15	17	15	14	16	15	22	16	21	24
							Mean mothly boat hours							
1977	27	17	19	22	22	21	19	23	26	22	34	28	33	28
							Total summer harvest							
1978	8	11	7	12	8	9	13	9	17	10	13	13	13	17
							Total shore harvest							
1979	8	13	8	12	8	10	14	9	14	11	14	13	13	18
							Total boat harvest							
1980	27	22	17	26	20	13	20	39	29	41	36	29	27	26
							Mean monthly shore harvest							
1981	24	21	19	26	19	21	27	24	32	25	29	29	26	40
							Mean monthly boat harvest							
1982	43	38	42	45	44	29	40	69	62	69	72	57	47	59

Table 13-2.—*Z*-test comparison of estimated angler hours and harvest for 1973-82 summer seasons.

Year	1973	1974	1975	1977	1979	1981	1982
			Lower Twin Lakes				
1974	c†						
1975		C					
1977	H		c				
1979	h		c				
1981		C					
1982		C		Hc	H		
			Upper Twin Lakes				
1974	HC						
1975		HC					
1977		c					
1979	HC		HC	C			
1981	C	h	C	c	H		
1982		H			H		

† H and C = significant differences in hours, H, and harvest, C, at $P = 0.01$; and h and c = significant differences in hours, h, and harvest, c, at $P = 0.05$.

1981). The differences between the high and low harvest estimates, for the upper lake, were all statistically significant.

A crude measure of the sensitivity of the *Z*-test results was determined by approximating the difference required between estimates of hours fished and harvest for both lakes to achieve statistical significance (table 13-3, midpoint). This approximate value was determined from the smallest significant difference and largest nonsignificant difference observed from all comparisons of hours fished and fish harvested during the study. Because of a relatively large standard error for the 1982 estimates, some differences between 1982 estimates and other years were nonsignificant, but were larger than some significant differences. Excluding these confounding comparisons from the sensitivity analysis, the indexes describing average differences required to achieve statistical significance were related as a percentage of mean estimate of the angler hours or fish harvested for Upper and Lower Twin Lakes for the study period. The objective was to determine a general range of statistical sensitivity associated with the evaluation of Twin Lakes creel survey estimates through *Z*-test analyses. Presuming that the mean estimates of angler hours and fish harvested would be considered characteristic of the fishery before Mt. Elbert Pumped-Storage Powerplant began operation, mean changes of 21 and 39 percent in angler hours and fish harvested would be necessary to demonstrate statistical significance. These relatively large percent changes reflect the nature of the recreational fishery at Twin Lakes. A significant degree of variation year to year is considered characteristic.

Composition of angler hours and harvest estimates remained quite similar for both lakes throughout 1973-82 (Nesler, 1984). Shore anglers contributed an average of 85 percent of the estimated total hours (range 78 to 91%), and 90 percent of the harvest (range 79 to 96%) for the two lakes over the seven seasons sampled. Weekend anglers accounted for an average of 55 percent of the total estimated hours (range 47 to 60%), and 54 percent of the total fish harvested (range 43 to 63%) during the study.

Summer Catch Rates

Shore anglers experienced higher catch rates than boat anglers on both lakes. The number of fish CPMH (caught per man-hour) fluctuated considerably on the lower lake from 1973 to 1975 for both shore and boat anglers, ranging from 0.25 to 0.43 fish per hour on shore and 0.19 to 0.28 fish per hour for boats (fig. 13-2). From 1977 through 1982, the CPMH for both angler types appeared more stable. Fluctuations in CPMH were relatively greater on the upper lake for both angler types, ranging from a high of 0.51 fish per hour in 1973 to a low of

Table 13-3.—Projected differences required between Twin Lakes creel survey estimates to demonstrate statistical significance at P=0.05.

	Angler, count	Hours	Fish harvested		
	Lower lake	Upper lake	Lower lake	Upper lake	Both lakes
Smallest significant difference, (A)	18 900	7 300	6 700	4100	10 800
Greatest nonsignificant difference, (B)	15 100	5 000	5 900	3200	9 100
Midpoint of A and B, (C)	17 000	6 150	6 300	3650	9 950
Mean study estimate 1973-82, (D)	79 800	25 700	27 200	9300	36 500
Statistical significance index, $(C/D) \times 100$	21	24	23	39	27

0.26 fish per hour in 1981 for shore anglers; and from a low of 0.10 fish per hour in 1973 to a high of 0.37 fish per hour in 1977 for boat anglers. The trends observed were probably the result of a combined influence of (1) the small magnitude of the upper lake fishery relative to the lower lake, (2) a significant change in numbers of creel-size rainbow trout stocked after 1973 in both lakes, and (3) the tendency of stocked rainbow trout to emigrate from the upper lake to the lower lake. Reduction in stocking levels probably had the greatest destabilizing effect. Also, changes in angler use in the Twin Lakes area, lowered stocking rate, or lesser fish abundance in the lakes would have a relatively greater impact upon the smaller fishery of the upper lake.

The frequency distribution of catch rates determined for interviewed anglers indicated 33 percent of the 12 344 shore anglers interviewed and 59 percent of the 2340 boat anglers interviewed had captured no fish of any species (fig. 13-3). Sixty-five percent of the shore anglers and 25 percent of the boat anglers had been successful at capturing at least one rainbow trout. Capture of lake trout by shore anglers was a relatively rare event, with only 2 percent being successful, while 16 percent of boat anglers had captured lake trout. For anglers successful at catching rainbow trout, the percentage experiencing catch rates higher than 0.40 rainbow trout per hour decreased rapidly for both angler types. Ninety-seven percent of the shore anglers and 95.5 percent of the boat anglers experienced catch rates of 0.01 to 2.0 rainbow trout per man-hour.

Anglers successful at catching lake trout were evaluated in the context of the two-fish bag limit. Among successful fishing parties, 39 percent of the shore anglers and 52.3 percent of the boat anglers captured one legal-size lake trout (fig. 13-4). Almost 17 percent of the boat anglers but only 3.2 percent of the shore anglers had captured a limit of two legal-size lake trout.

Harvest Composition

Composition of the summer harvests was consistent throughout 1973-82. For both lakes, 97.5 to 99 percent of the shore harvest was rainbow trout, followed by lake trout (1.0 to 1.5%) and incidental numbers of brown trout (*Salmo trutta*), brook trout (*Salvelinus fontinalis*), and kokanee salmon (*Oncorhynchus nerka*). Harvest composition for boat anglers on the lower lake consisted of rainbow trout (74.5%), lake trout (24.5%), brown trout (1%), and incidental captures of kokanee salmon. For boat anglers on the upper lake, harvest composition consisted of rainbow trout (87.1%), lake trout (10.5%), brook trout (1.4%), brown trout (0.6%), and kokanee salmon (0.4%).

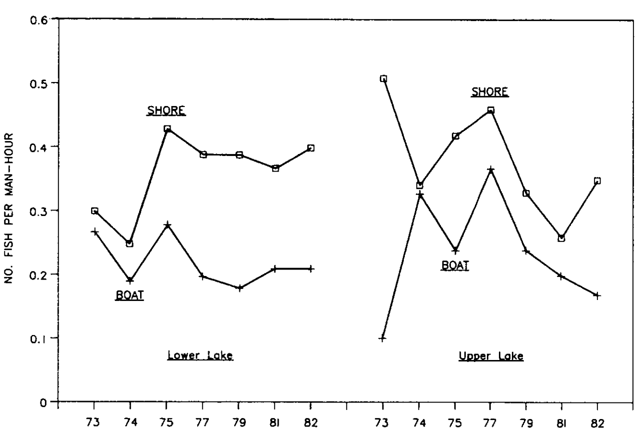


FIGURE 13-2.—*Summer season catch rates for shore and boat anglers at Twin Lakes.*

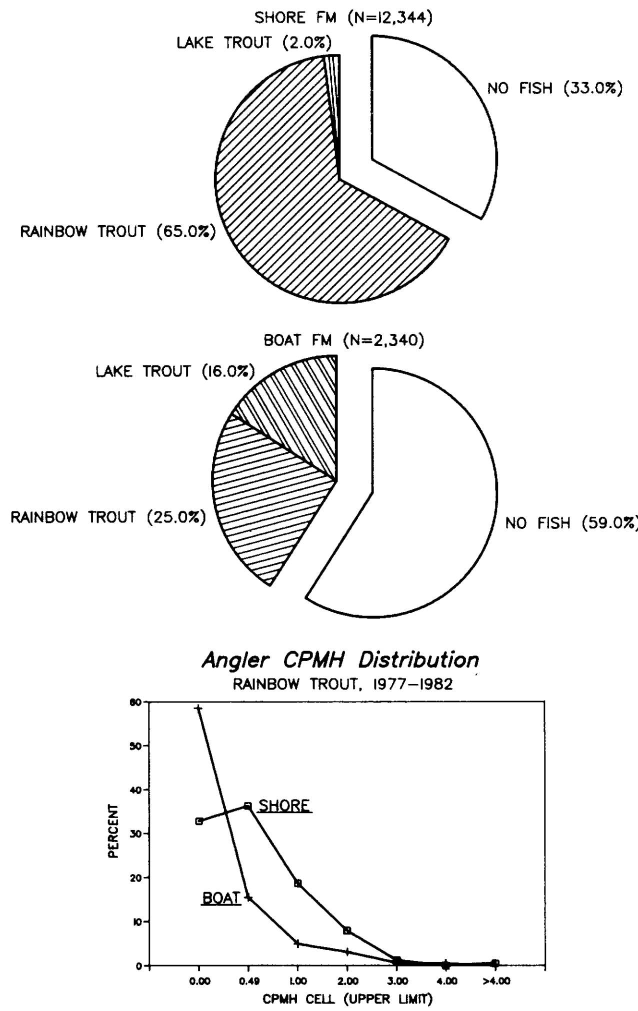


FIGURE 13-3.—*Success ratios for shore and boat anglers and distribution of catch per man-hours (CPMH) for shore and boat anglers harvesting rainbow trout at Twin Lakes (FM = fisherman).*

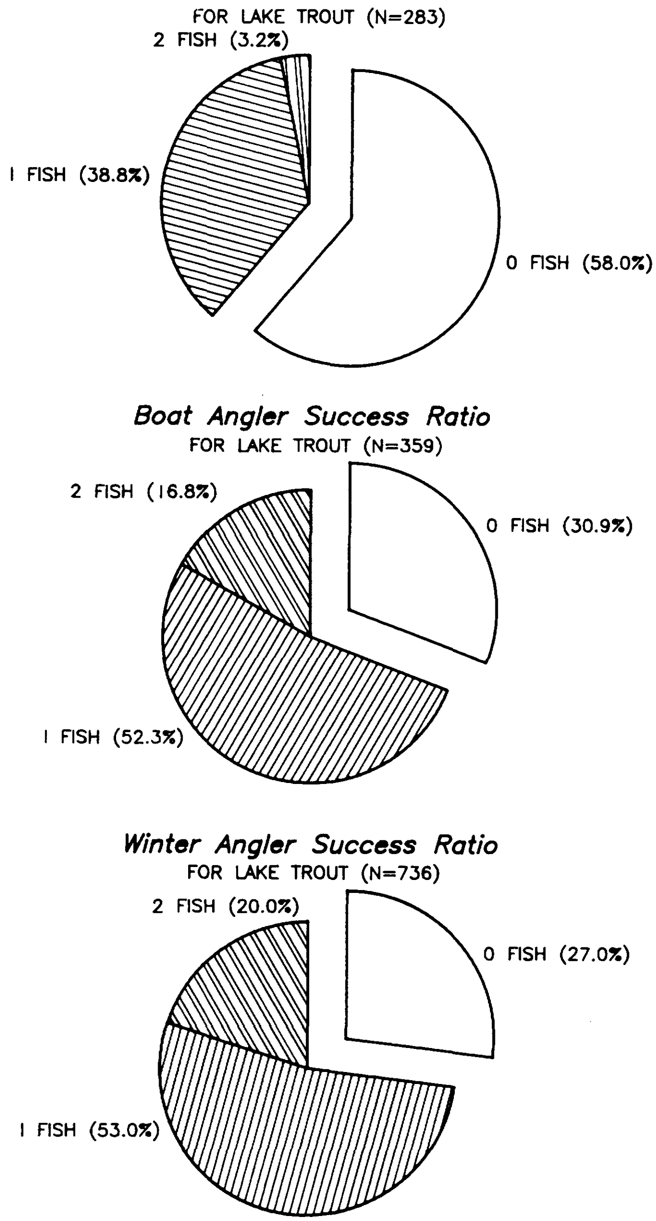


FIGURE 13-4.—*Success ratios for all angler types seeking lake trout under two fish per day bag limit at Twin Lakes.*

On a species basis, 91 percent of the rainbow trout harvested on both lakes were caught by shore anglers. Conversely, 67 percent of lake trout harvested during the summer from the lower lake were caught by boat anglers. On the upper lake, 46 percent of the lake trout harvested were caught by boat anglers.

To summarize the roles of the two angler types at Twin Lakes, shore anglers represented 85 percent of the summer fishery and accounted for 90 percent of the total summer harvest.

For the shore anglers, 65 percent were successful at catching at least one rainbow trout, and their harvest consisted of over 97 percent of this species. Shore anglers accounted for 91 percent of the rainbow trout caught during the summer season. Only 2 percent of shore anglers were successful at catching at least one lake trout, and their harvest of this species represented less than 2 percent of the total shore angling harvest. However, lake trout caught by shore anglers represented 33 percent of all lake trout harvested during the summer season in the lower lake and 54 percent in the upper lake. Boat anglers represented 15 percent of the summer fishery, and accounted for 10 percent of the total summer harvest.

For the boat anglers, 25 percent were successful at catching at least one rainbow trout, and their harvest consisted of 75 to 87 percent of this species. The boat harvest of rainbow trout, though, represented only 9 percent of the rainbow trout caught during the summer. Of the boat anglers, 16 percent were successful at catching at least one lake trout, and their harvest of lake trout represented 11 to 25 percent of the total boat harvest in the upper and lower lakes, respectively. The boat harvest of lake trout represented 46 to 67 percent of all lake trout harvested during the summer season on the upper and lower lakes, respectively.

Stocked Rainbow Trout Return to the Creel

Extensive marking of stocked, creel-size rainbow trout was conducted periodically throughout the study to determine percent return of these fish to the angler (table 13-4). The best returns were observed in 1975, 1979, and 1982, when 47 to 58 percent of stocked rainbow trout were harvested. These return rates were associated with stocking rates of 62 to 75.8 fish per hectare. A marking experiment conducted in 1982 was designed to evaluate (1) contribution of stocked fish to the same season harvest, (2) determine the contribution of overwintering fish, and (3) determine the degree of movement of fish between the two lakes.

Harvest of marked rainbow trout in 1982 demonstrated 95 percent of the total rainbow trout was finclipped fish stocked the same season. Only a small portion of the harvest was represented by the catch of rainbow trout in May before the first plant of fish. Thirty-eight percent of the rainbow trout stocked in the upper lake emigrated to the lower lake, and comprised 13 percent of the lower lake harvest. In contrast, only 4 percent of the rainbow trout stocked in the lower lake emigrated to the upper lake, but these fish also comprised 13 percent of the upper lake harvest. In 1977 and 1979, 5 and 3 percent of the marked rainbow trout, respectively,

Table 13-4.—Stocking rates, fish per hectare, with associated percent harvested for creel-size rainbow trout in Twin Lakes.

Year	Stocking rate, fish/ha		Percent return to creel		
	Lower lake	Upper lake	Lower lake	Upper lake	Total
1973	151.9	365.6			19.0
1974	72.6	109.7			38.0
1975	75.6	109.7			†50
*1977	62.0	136.3	41.4	5.2	46.6
*1979	62.0	69.4	46.9	3.0	49.9
1981	86.2	89.2			‡41
1982	67.9	75.8	56.0	58.0	56.0

* All marked fish stocked in lower lake.
† Estimated from season harvest equals 95 percent of trout stocked same year.
‡ Average return of three marked plants.

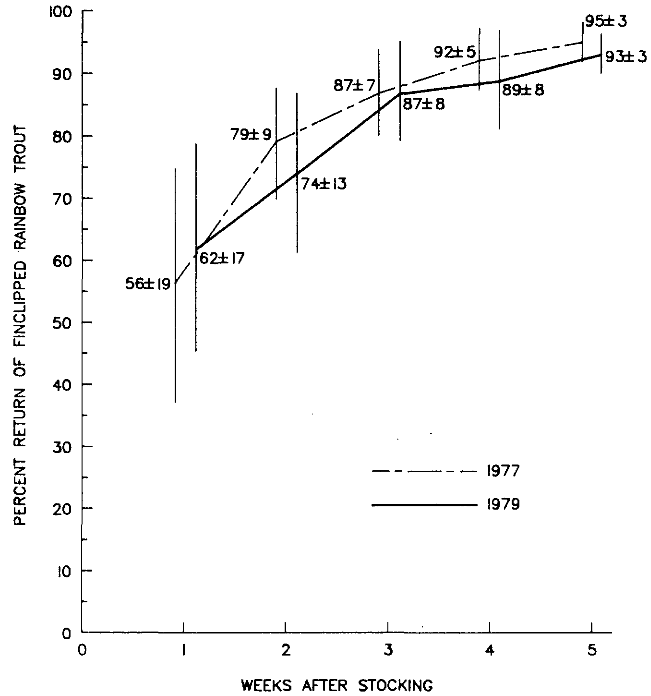


FIGURE 13-5.—*Mean weekly cumulative return rates of rainbow trout harvested in Twin Lakes with 95 percent confidence intervals.*

emigrated from the lower lake to the upper lake, and comprised 17 to 19 percent of the upper lake harvest.

The cumulative return rate of individual plants of marked rainbow trout were determined in 1977 and 1979 (Nesler, 1981b). A weekly mean return rate for each of eight plants was estimated in each year for 5 weeks following each stocking. An overall cumulative mean return rate was then estimated for each week after stocking for both 1977 and 1979 (fig. 13-5). Estimates were fairly similar for the 2 years, and indicated 50 to 60 percent of the rainbow trout harvested from a given plant of stocked fish would occur in the first week. Over 90 percent of the ultimate harvest of a given plant of stocked rainbow trout occurred after 4 weeks. Regarding percent return to the creel, 47 to 58 percent of the rainbow trout stocked in a summer season would be harvested, and 90 percent of these fish would return to the creel in less than 4 weeks.

The immediate harvest of stocked rainbow trout on stocking day plus 2 to 3 days following was also examined in 1982. Rainbow trout were usually stocked on the day before a weekend or 3-day holiday to maximize return to the creel. Interviews suggested a high success rate (4 to 8 fish/h) for anglers near the point of release on the stocking day, and good catch rates (2 to 3 fish per hour) for anglers elsewhere on the lakes on the following day. During a 3-day sample on a mid-August weekend, 31 percent of 5000 rainbow trout stocked in the lower lake and 34 percent of 2000 rainbow trout stocked in the upper lake were estimated to have been harvested. During the Labor Day holiday, which represented a 4-day estimate of harvest determined from a 3-day sampling effort, 38 percent of 5000 rainbow trout stocked in the lower lake and 48 percent of 2000 rainbow trout stocked in the upper lake were estimated to have been harvested.

In 1983, a short-term creel survey was conducted to determine the percent return of two different size-classes of stocked, creel-size rainbow trout. Almost 13 000 fish had clipped fins, and were separated into one size-class greater than 240 mm and another size-class less than 240 mm. Percent returns of the two size-classes were almost identical at 59 to 61 percent, and over 95 percent of these fish were harvested within 3 weeks of stocking. During the high-use, 3-day Memorial Day holiday, 66 to 70 percent of the fish were returned to the creel. The estimated holiday harvest comprised 40 percent of the total number of rainbow trout stocked, and occurred within 3 days of stocking.

Winter Seasons

The winter fishery at Twin Lakes occurred mainly on Lower Twin Lakes. Too few anglers were ever observed on the ice of Upper Twin Lakes to warrant including this lake in the creel survey. Angler hours were estimated for eight seasons, and ranged between 3900 and 13 200 hours (fig. 13-6). For the 1975-76 season, a creel survey was conducted only during December-January. Angler hours and harvest estimates for February-March in 1975-76 were subsequently projected by regression equations as a function of the known December-January

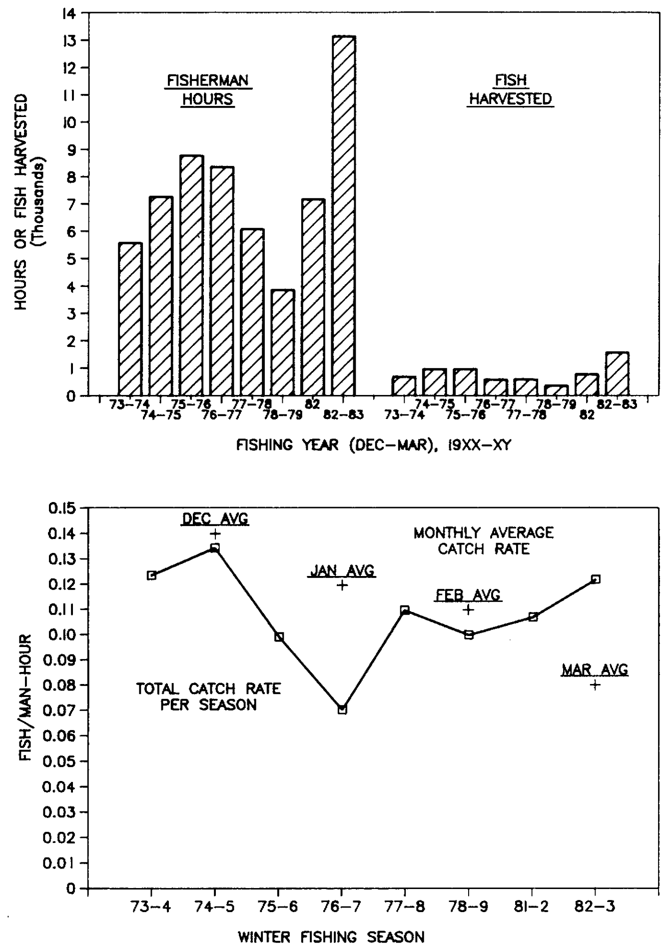


FIGURE 13-6.—*Estimates of angler hours, fish harvested, season total catch rate and monthly average catch rates at Lower Twin Lakes during winter seasons.*

estimates (Nesler, 1981b). Significant correlations between December-January and February-March estimates were observed among the 1974-79 winter seasons for angler hours (r=0.82) and harvest (r=0.99). References to 1975-76 estimates of hours and harvest are based on projected estimates in Nesler (1981b).

The winter fishery expanded for three seasons following reduction of the size limit on lake trout in 1974, but then decreased to its lowest point during the study in 1978-79. The 1981-82 and 1982-83 season estimates suggested a rapid increase thereafter. Ice-fishing harvests were low, ranging from 400 to 1600 fish. However, these harvests were dominated by lake trout greater than or equal to 381 mm in length. This species was the primary target of anglers almost to the exclusion of rainbow trout. In seven of eight seasons, 89 to 100 percent of the harvests were comprised of lake trout. Only in 1976-77 was harvest composition notably different, when lake trout comprised 57 percent of the harvest. The difference in that season relative to the others was that 1976-77 was a drought winter with little accumulation of snow. Under clear-ice conditions, anglers experienced their worst catch rate during the study (fig. 13-6). This factor, in conjunction with back-to-back heavy snow years that impeded access to the lakes in 1977-78 and 1978-79, may have resulted in the observed decline in angling effort through 1978-79. The trend in estimated winter harvests followed that observed for angler hours, with the lowest estimated harvest observed in 1978-79 and the peak harvest observed in 1982-83. Mean lengths of lake trout harvested ranged from 422 to 478 mm.

Catch rates were also low, ranging from 0.07 to 0.135 fish per hour (fig. 13-6). With regard to the quality of the lake trout fishery, harvest rates would be a more appropriate description because ice anglers (as well as some boat anglers) caught more lake trout than they kept. The highest catch rates observed also followed initial reduction of the minimum-size limit for lake trout in 1974 (part way into the 1973-74 season). Following the low catch rate in 1976-77, catch rates generally increased through the 1982-83 seasons. Monthly average catch rates (fig. 13-6) indicated anglers had the most success initially in December (0.14 fish per hour). Catch rates then gradually decreased over the winter season to 0.08 fish per hour in March. The frequency distribution of ice anglers' catches of lake trout for the 1976-77 season through the 1982-83 season (fig. 13-4) showed their success was similar to boat anglers. On average, 20 percent of ice anglers were successful in harvesting a limit of legal-size lake trout and 53 percent caught one legal-size lake trout. The best success ratios were observed in 1982-83, when 25 percent of ice anglers harvested limits of legal-size lake trout and only 20 percent caught none (Nesler, 1984).

Statistical comparisons of hours fished and harvest estimates were split by December-January periods and December-March periods as a consequence of the shortened creel survey in 1975-76 (table 13-5). From 1973 through 1979, the increase or decrease in estimates of hours fished from one season to the next was generally significant. The estimate of hours fished in 1982-83 was significantly different from all other season estimates regardless of time period considered. For harvests, the December-January estimates for 1974-75, 1975-76, and 1982-83 were similar. All remaining comparisons of the 1982-83 harvest estimate with other seasons showed the 1982-83 harvest was significantly greater, regardless of time period considered.

Winter Harvest Lake Trout Length-Frequency

Length-frequency data on lake trout harvested by ice anglers was available from 1974 to 1983. A general

Table 13-5.—*Z*-test comparison of estimated angler hours and harvest for the 1973-83 winter seasons on Lower Twin Lakes.

	1973–74	1974–75	1975–76	1976–77	1977–78	1978–79	1982	1982–83
				December–January				
1974–75	HC*							
1975–76	HC	H						
1976–77	H	C	C					
1977–78		HC	HC	H				
1978–79	hc	HC	HC	H	h			
1982		HC	HC	H				
1982–83	HC	H	H	HC	HC	HC	HC	
				December–March				
1974–75								
1976–77		C						
1977–78		HC		H				
1978–79		HC		H	H			
1982						HC		
1982–83		HC		HC	HC	HC	HC	

* H and C = significant differences in hours, H, and harvest, C, at $P = 0.01$; h and c = significant differences in hours, h, and harvest, c, at $P = 0.05$.

shift in length-frequency toward the smaller-size lake trout was observed after the 1974–75 season (fig. 13-7). The 1974-75 season was the first full ice-fishing season following the reduction of the lake trout minimum-size limit. From 1981 through 1983, lake trout length-frequency distributions appeared to shift toward larger lake trout. Lake trout greater than or equal to 508 mm total length comprised 26 percent of the 1974-75 harvest. Thereafter, the percentage of these larger lake trout in winter harvests continued to decline to 18 percent in 1975–76, and 6 to 8 percent in 1977–78 and 1978–79. In 1981-82 and 1982-83, the percentage of larger lake trout increased to 17 percent. These fluctuations in relative abundance of larger lake trout in winter harvests were correlated to fluctuations in relative abundance of larger lake trout in the standardized gill-net samples (Nesler, 1986). This correlation was statistically significant, and suggested that standardized gill-net samples (see ch. 12) and winter harvests of lake trout were representative of the size-frequency distribution of the lake trout population vulnerable to angling.

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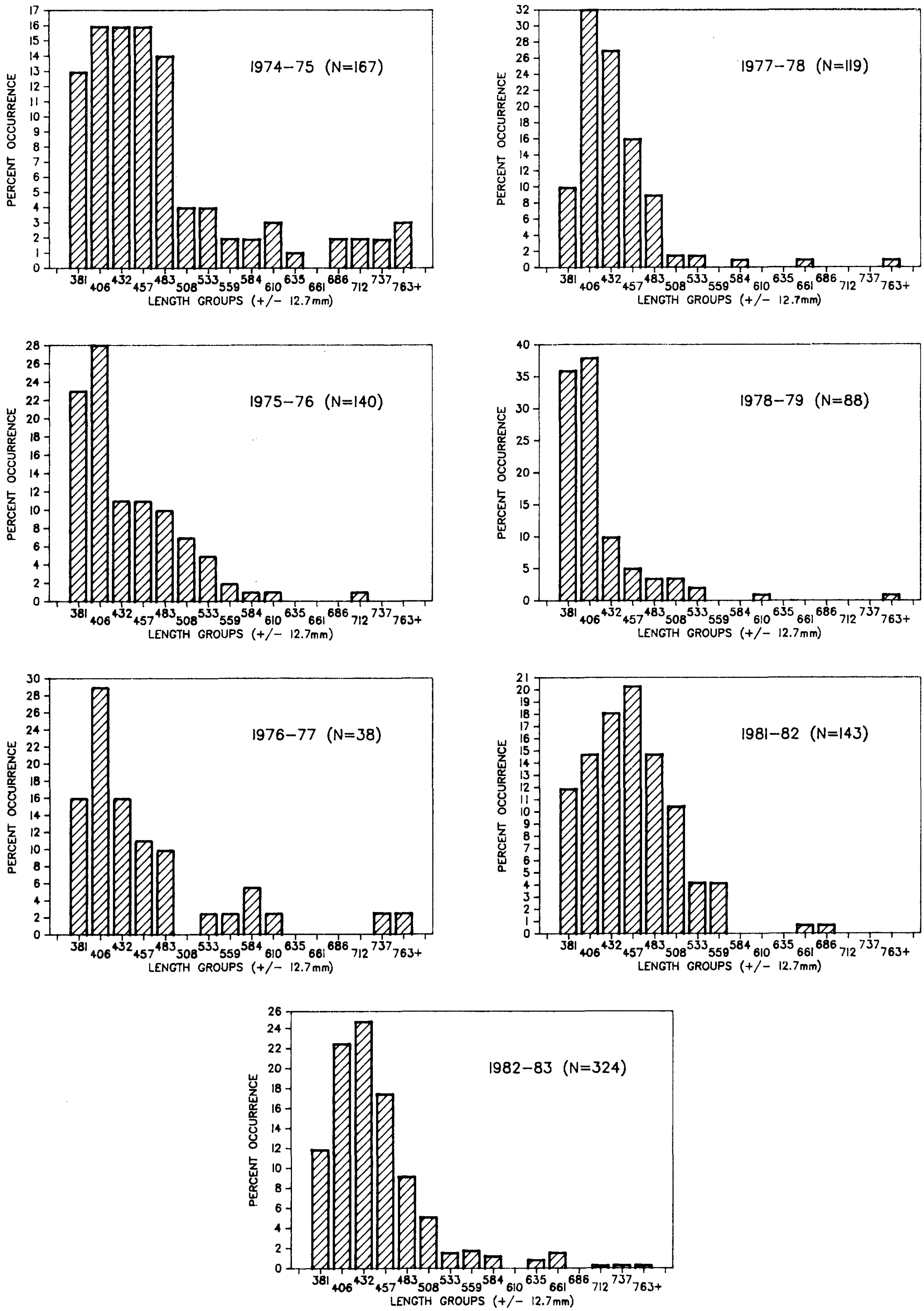


FIGURE 13-7.—*Length-frequency distribution for lake trout harvested by winter anglers during seven seasons at Lower Twin Lakes.*

Fish and Mysid Entrainment

Chapter 14

M. Maiolie, E. P. Bergersen, K. Phippen, and J. Wullschleger, with K. Hamilton

Colorado Cooperative Fish and Wildlife Research Unit
Colorado State University
Fort Collins, Colorado

INTRODUCTION

Over 1000 potential pumped-storage sites have been identified in the United States (Hauck and Edson 1976); many of these sites likely will be developed as energy demand increases. Because of this potential to disrupt existing aquatic resources, it is important to identify environmental impacts of this form of power production. Anticipated aquatic problems could take various forms: water level fluctuations, adverse changes in quality or quantity of water released, entrainment of aquatic organisms, and numerous other effects. Our study dealt with quantifying the entrainment of fish and opossum shrimp (*Mysis relicta*) at Mt. Elbert Pumped-Storage Powerplant. Results of our study and that of others in Twin Lakes are being used to forecast changes expected in the Twin Lakes system; they also should be applicable to evaluating actual or potential impacts at other similar sites.

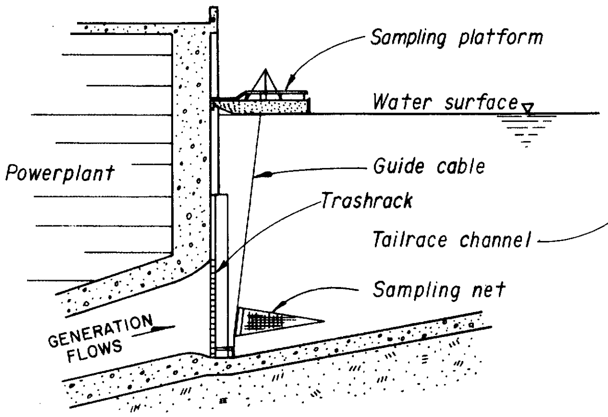
To our knowledge, this study was the first one intensively conducted on a relatively small, high elevation, cold water lake system. As such, effects of powerplant operation were more immediate and more quantifiable than in larger lake systems. Twin Lakes also has a rather abbreviated food chain, including opossum shrimp (*Mysis relicta*) and lake trout (*Salvelinus namaycush*). This simple community structure made it less likely that compensatory changes by other forage items or sport fish would mask powerplant-induced changes in the system. For these reasons, Twin Lakes was an ideal location to document environmental effects of pumped-storage powerplant operation.

METHODS

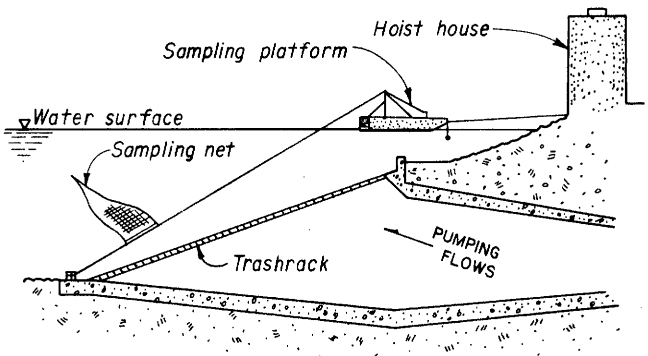
Pump-Turbine Operations

Generation sampling.—A work platform was built so that turbine discharge, during the generation cycle, could be sampled for opossum shrimp (*Mysis relicta*) and fish. Two surplus military bridge barges, 9 m long, served as pontoons for the platform. Nets were raised and lowered by hand winches through an opening in the center of the platform deck. Steel guide cables, 60 mm in diameter, extended from the platform to a removable attachment bar hooked onto the powerplant trashrack. Then, nets could be lowered along the taut guide cables to a position within 2 m of the trashrack as shown on figure 14-1a.

Shrimp nets, 3 m long, were made of 0.8-mm diameter nylon and had a 1- × 2-m mouth, 0.8-mm bar mesh; they terminated in a 200-mm diameter cod-end to which a sampling bucket was attached. A 300-mm wide vinyl collar surrounded the net mouth to strengthen the net and provide better flow. A similar net was made for sampling fish less than 150 mm long. A 10-m long net, with 25-mm bar mesh and a 2- × 2-m mouth, was used to collect fish longer than 150 mm. Two General Oceanics, Inc. flowmeters (model 2030) were positioned two-thirds of the way between the center of the net and the rim to accurately register the volume (Bowles et al., 1978). Flowmeters were read before and after each sampling period to quantify the volume of water filtered.



a. Mt. Elbert Pumped-Storage Powerplant—Lower Twin Lakes.



b. Mt. Elbert Forebay Reservoir Inlet-Outlet Works. A concrete inlet-outlet structure separates into two 4572 mm (15-ft) steel penstock pipes.

FIGURE 14-1.—*Floating platforms used to sample discharges during generating, a, or pumping, b, cycles.*

Pump-back sampling.—A sampling platform (for Forebay Reservoir) was built so shrimp could be collected during "pump-back" cycles. It was functionally similar to the platform in the tailrace channel, and the same size nets were used. Nets were raised and lowered along 60-mm diameter steel guide cables; the guide cables were hooked onto an attachment bar at the bottom of the trashrack. However, the Mt. Elbert Forebay trashrack was not vertical (as in the tailrace channel), but sloped as shown on figure 14-1b. Therefore, nets were raised and lowered over the back of the boat instead of through an opening in the deck. A cable-mounted counterweight secured the platform to the hoist house and allowed for changes in water level. Flowmeters were used as before to quantify water volumes sampled.

In many cases, it was possible to estimate the entire number of shrimp entrained during either pump-back or generation cycles. Linear interpolation was used to estimate the density of shrimp entrained between samples. A time-weighted mean shrimp density was calculated for the cycle and multiplied by the powerplant's total discharge, which gave an estimate of the total number of shrimp entrained.

Mysids

Entrainment models.—Several entrainment models to assess the effects of powerplants on aquatic organisms were evaluated by Maiolie (1987). Maiolie judged Goodyear's (1977) percent-loss equation to be most suitable for Twin Lakes.

This simple equation assumes instantaneous, complete mixing so that the intake concentration of organisms is continuously reduced by the proportion by which the population declined due to accumulated entrainment losses. Percent loss—calculated by this equation—represented the percent reduction of the population over what would have been present without the intake:

$$\text{Percent loss organisms} = 100\,(1 - e^{-pT}) \qquad (14\text{-}1)$$

where:

$p = f Q_p/v$,
Q_p = flow through the powerplant intake during generation in cubic meters per day,
T = number of days organisms are vulnerable to entrainment,
v = water in lake or Mt. Elbert Forebay in cubic meters,
$f = mC_p/C_v$,
m = mortality rate of entrained organisms,
C_p = mean concentration of organisms in intake water, and
C_v = mean concentration of organisms in lake or Mt. Elbert Forebay.

This technique assumed that natural mortality rates were unaffected by powerplant operation. Under this assumption, it was not necessary to estimate natural mortality in the field.

Fish

Entrainment mortality rate.—Two rainbow trout (*Salvelinus gairdneri*) were placed into each of 21 net bags that were attached to blaze-orange wood floats. Bags of fish were placed into weighted paper bags and lowered with a rope into the penstocks at the hoist house. Within minutes, the paper bags ruptured and the rush of water carried the fish in the net bags down through the powerplant. Bags were recovered from the tailrace channel, and the fish were transferred to an aerated and cooled tank. Recovered fish were confined for 4 days for observation of mortality. Control groups consisted of rainbow trout in the same net configuration. The controls were lowered by rope to the tailrace floor during a generation cycle. After the paper bags ruptured, the net bags of fish floated to the surface where they were then collected and transferred to the same holding tank as the test fish. The first trial was conducted in December 1983 during a 102-MW generation cycle, and the second trial was conducted during a 70-MW generation cycle in August 1984.

Net efficiency tests.—About 20 rainbow trout, averaging 225 cm long and marked with a caudal fin punch, were placed into weighted paper bags. The paper bags were lowered by rope into the penstocks. When the paper bags ruptured, the freed fish were released and carried by the flows through the penstock and into the lower lake. Nets equipped with flowmeters were in the position used for sampling during generation cycles. The nets were raised about 1/2 hour after the fish had been released.

Time, flowmeter reading, and Mt. Elbert Forebay elevation were recorded before and after the trial to determine the amount of water sampled and the volume of water the powerplant discharged. Two trials were conducted in December 1983, during a 70-MW generation cycle and again in January 1984, during a 102-MW generation cycle. In the first set of trials, 202 fish were released; 145 fish were released on the second date.

Attraction to the powerplant.—A Raytheon Co. JFF-560 straight line echo sounder with a 200-kilohertz transducer was used to record fish densities. A separate, fully charged 12-volt battery ensured an adequate and constant power supply during surveys. Settings on the echo sounder were kept constant so that the projected signal would remain the same between transects.

The "cone" or area below the transducer in which a fish could be detected was measured at different depths, using a 203-gram lead sphere in a manner suggested by Johnston (1981). One survey transect extended from the end of the tailrace channel to the powerplant and back without overlap. This transect could only be surveyed when the powerplant was not operating. Other transects were selected at random along the 2780-m contour of the lower lake, and were timed so they would be similar in length to the tailrace survey. Thus, all transects were of comparable depth, duration, and scanned volume. All surveys were conducted between 0830 and 1000 hours.

RESULTS

Mysids

Generation cycle.—Between November 1982 and June 1986, 502 604 m^3 of water from the generation cycle were filtered through the nets in 78 samples. In the generation cycle, the shrimp density ranged from 0.00 to 2.36 shrimp per cubic meter. Of the 78 samples, entire generation cycles were filtered on 13 occasions to estimate the total number of shrimp entrained during one cycle. The total number of entrained shrimp ranged from 14 607 to 933 096. The time period of generation cycles varied; on a per hour basis case, the range of numbers of entrained shrimp observed during entire generation cycles was 1445 to 274 615.

The extent of entrainment was affected by the time of day of generation and by the length of time from the onset of generation. A logarithmic equation was developed that expressed the relationship between density of shrimp entrained and minutes from generator start-up:

$$y = 1.87 - 0.352 \ln x \qquad (14\text{-}2)$$

where:
y = shrimp per cubic meter entrained, and
x = time from generator start-up in minutes.

The correlation coefficient for this relationship was 0.65, meaning that 42 percent of the variability of shrimp density could be explained by the time factor.

Highest numbers of shrimp were entrained during early morning and late evening cycles. The effect of light on entrainment was apparent because more shrimp were entrained on moonless than moonlit nights. Because more mysids move vertically during dark nights, it appears that the extent of entrainment was a function of the number of mysids in the water column during generation.

During daytime generation, the number of entrained shrimp dropped rapidly after the first hour of generation; at times, 98 percent or more of the total number of entrained shrimp were entrained during the first hour of generation. A quadratic equation that expressed the relationship between density of entrained shrimp and time of day was developed:

$$y = 15.6\, x^2 - 15.6\, x + 3.97 \qquad (14\text{-}3)$$

where:
y = shrimp entrained per cubic meter, and
x = time of day in percent.

Using the percent organisms loss equation, the estimated loss of shrimp from Mt. Elbert Forebay population was small during daytime generations. Although generation was conducted most often during the day, a sample of night generation revealed higher entrainment rates. The estimated loss of the shrimp population during nighttime generation in December 1983 was 13 percent.

Except for some winter samples, length frequency, age, and sexual characteristics were similar to the Twin Lakes population, as described earlier. For example, Gregg (1976) collected only 15 to 30 percent mature female shrimp during winter and early spring, yet samples obtained on January 11, 1983, February 7, 1983, and April 20, 1984 contained 63, 71, and 68 percent mature female shrimp, respectively. Mature male shrimp were unusually abundant in winter 1985 samples when they comprised 50 to 75 percent of the sample; Gregg (1976) found 12 to 34 percent mature males in January and February.

Pump-back cycle.—The entrainment was higher during pump-back than generation cycles. Between June 24, 1985 and October 2, 1985, 147 072 m^3 of pump-back flow were sampled. The first sample collected during pump-back cycles generally contained a relatively high number of shrimp, although this initial pulse was not as great as that observed during generation cycles. High numbers were entrained between 2300 and 0500 hours, but densities declined rapidly after 0500 hours. Usually, the pattern of entrainment during the night depended on the lunar phase. On clear, full-moon nights, the density of entrained shrimp was highest early and late in the night. On dark nights, high numbers were entrained in the first samples, and then a secondary peak occurred towards the middle of the night.

- Shrimp density ranged from 0.023 shrimp per m^3 on August 14, 1985, to 32.6 shrimp per m^3 on 1985.
- Mean shrimp densities ranged from 2.4 shrimp per m^3 in April–May 1986 to 17.2 shrimp per m^3 in June 1985.
- Mean density for the entire period of sampling was 8.3 shrimp per m^3.
- Estimates of total number of shrimp entrained ranged from 3 594 014 to 44 077 408.

During 1985, the reduction in the shrimp population caused by pump-back cycling was estimated to be 39 percent using the percent-loss organisms equation:

Q_p = 1 758 242 cubic meters per day, average volume discharge during a pump-back cycle,
T = 182, number of nights during which pump-back cycles occurred during 1985,
V = 112×10⁶ m^3, volume of Lower Twin Lakes at surface elevation 2802 m,
m = 1.0, this value assumes all shrimp were lost from the lower lake,
C_p = 9.93 shrimp per m^3, mean density of shrimp entrained during 1985, and
C_v = 58 shrimp per m^3, estimated using the Nov. 12, 1984, phototrawl estimate of 781×10⁶ shrimp and adding 18.5 young/mature female (39.5% of the population). This total was divided by the lower lake's volume.

Therefore, the percent-loss equation became:

$$100\,(1 - e^{-0.002\,688 \times 182}) = 38.69 \qquad (14\text{-}4)$$

This estimate agrees with the decline of 36 percent observed in the shrimp population of the lower lake in 1985. It suggests that natural mortality was constant, as was assumed in the organisms percent-loss equation.

Density estimates of shrimp entrainment were not made in 1984 because sampling equipment was unavailable. If shrimp were entrained at the 1985 rate (C_p = 9.93 shrimp/m^3), then the percent-loss equation would predict a 41 percent decline due to entrainment. The population of the lower lake was actually stable at 110 shrimp/m^2. This finding may be attributed to a decline in natural mortality, as newly inundated land caused by the new dam increased productivity and the lake's capacity. Thus, the shrimp population may have been able to compensate for losses from entrainment. When natural survival rates stabilized, the agreement between predicted and observed declines was improved.

These results indicate that increased electrical generation could have a profound impact upon the mysid population in the lower lake. Based on the percent-loss equation, the population would suffer an 88-percent loss if the powerplant operated at maximum capacity 75 percent of the nights in a year:

Q_p = 5×10⁶ m^3/day, maximum volume both units could pump for one night,
T = 274, assumes both units will operate 75% of the nights,
V = 112×10⁶ m^3, volume of Lower Twin Lakes at surface elevation 2802 m,
m = 1.0, this value assumes all shrimp were lost from the lower lake and all shrimp died, or because daytime behavior precluded their return to the lower lake during generation cycles,
C_p = 9.93 shrimp per m^3, mean density of shrimp entrained during 1985, and
C_v = 58 shrimp per m^3, estimated by using the Nov. 12, 1984, estimate of 781×10⁶ shrimp, adding 18.5 times the number of mature females (39.5%). This total was

divided by the volume of the lower lake.
f = 0.171 21, calculated from $f = mC_p/C_v$, and
p = 0.007 64, calculated from $p = fQ_p/V$.

Therefore, the percent-loss equation became:

$$100\,(1 - e^{-0.007\,643 \times 274}) = 87.7 \qquad (14\text{-}5)$$

Although this estimate does not take into account factors that are difficult or impossible to quantify, such as density-dependent compensation by the mysids, it does note the potential that plant operation has to seriously impact the dynamics of the shrimp population—and, indirectly, other populations in Twin Lakes.

Fish

Entrainment mortality rate.—All of the 40 trout in the control group were recovered from the tailrace channel during the 102-MW generation trial. After 4 days, the mortality rate of the control group was 10 percent. Of the 42 test fish which had passed through the penstock, 26 were recovered; upon recovery, mortality was 46 percent and increased to 61 percent during the observation period. With a correction for control mortality (Bell, 1981), the test group mortality was 57 percent.

During the 70-MW generation cycle, the control group exhibited 10 percent mortality and the test group exhibited 70 percent mortality. The corrected test group mortality was 67 percent. The mean survival rate from these two trials was 38 percent.

Net efficiency.—Data from the first trial, which took place during the 70-MW generation cycle, indicated that the net was an effective sampler. In the first sample, 7.9 percent of the powerplant discharge was filtered and 8.4 percent of the marked fish were caught. In the second sample, 7.9 percent of the discharge was sampled and 7.5 percent of the fish were caught.

However, during the 102-MW generation cycle, the fish sampling net efficiency averaged 58 percent. Because net avoidance was not indicated during the 70-MW cycle, it is unlikely the reduced efficiency was due to avoidance at a higher generation level. The problem could have been an over-estimation of the amount of water filtered. When two flowmeters were used, readings were discovered to be up to 47 percent higher on the flowmeter near the center of the trashrack than the one closer to the outside of the trashrack. At higher flows, the discharge spiraled through the conduit so that greater discharge occurred on one side than the other. Therefore, subsequent shrimp and fish entrainment sampling was performed with two flowmeters.

Pump-back sampling.—Between June 18, 1985, and September 21, 1986, 1.21×10^6 m^3 of pump-back discharge were filtered. Entrainment rates were based on pooled data from this period:

- One rainbow trout per 151 431 cubic meters,
- One lake trout (*Salvelinus namaycush*) per 340 719 m^3,
- One longnose sucker (*Catostomus catostomus*) per 340 719 m^3, and
- One white sucker (*Catostomus commersoni*) per 123 898 m^3.

Entrained game fish were typically of harvestable size; rainbow trout were 15 to 31 cm long, and lake trout were 27 to 50 cm long. All fish less than 150 mm caught in the small net were juvenile kokanee salmon. The entrainment rate for these fish was one salmon per 10 505 m^3. Entrainment of fish during the pump-back cycle was significantly greater than entrainment during generation ($P<0.001$).

Based on the amount of water pumped during the 1985 and 1986 sampling periods, these rates were extrapolated to estimate the total number of entrained fish. Between June 1985 and October 1985:

- 20 282 salmon
- 1 112 rainbow trout
- 889 longnose suckers
- 666 lake trout
- 223 white suckers were entrained

Between May and September 1986:

- 59 228 kokanee salmon
- 109 rainbow trout
- 826 lake trout
- 826 longnose suckers
- 5 022 white suckers were entrained

At full operation, 1.37×10^9 m^3 could be pumped per year, which would exchange the Lower Twin Lakes' volume 12 times. Assuming these calculated entrainment rates, the following would be entrained annually:

- 143 000 kokanee
- 6 290 rainbow trout
- 6 290 longnose suckers
- 4 710 lake trout
- 3 140 white suckers

Assuming mortality during pump-back cycles was similar to that calculated during generation (62%), the 6820 rainbow and lake trout mostly of harvestable size and 88 660 young-of-the-year kokanee would be lost from the fishery. Annual harvest from Lower Twin Lakes was estimated to be 23 534 fish during 1977 and 25 423 fish in 1979 (Nesler, 1981). Entrainment losses, therefore, could approach 30 percent of the fishery harvest. Based on the amount of water pumped in 1985 alone, 34 000 kokanee were entrained. Losses of this magnitude would hinder establishing kokanee as either a forage or game fish in the Twin Lakes system.

Attraction to the powerplant.—Fish density in the tailrace channel of the Mt. Elbert Pumped-Storage Powerplant was considerably higher than that in the main body of the lower lake; fish density was estimated to be 1228 fish per million cubic meters in the tailrace channel compared to 24 fish per million cubic meters at similar depths and times in the lake. Fish in the tailrace channel water column were in two distinct groups. One group was above 8 m and one was near the bottom. Rainbow trout and longnose suckers made up the surface catches. Longnose sucker dominated (81%) the bottom catches, which also included white sucker (9%), lake trout (4%), rainbow trout (4%), and brown trout (*Salmo fario*) (2%). The number of fish in the tailrace channel was estimated by multiplying the fish density per 2-meter depth strata by the tailrace channel volume. The result was 244 fish in the tailrace.

Although fish were attracted to the tailrace channel, they apparently avoided entrainment. The shallow distribution of rainbow trout kept them above the intake powerplant structure. Lake trout were found in relatively low densities (15 lake trout/million m^3) by hydroacoustic efforts and were entrained in even lower densities (3.4 lake trout/million m^3). The possibility of seasonal changes in the distribution and vulnerability of fish could not be determined because too few fish were collected during pump-back cycles.

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Ecosystem Overview and Project Effects

Chapter 15

Twin Lakes Research Cooperators

Bureau of Reclamation
Colorado Division of Wildlife
Colorado Cooperative Fish
and Wildlife Research Unit

Mt. Elbert Pumped-Storage Powerplant construction began in 1971; the first 100-MW pump-turbine unit became operational in 1981. The second unit went "on-line" in 1984. The closure of new Twin Lakes Dam (in late 1983) raised the maximum water surface elevation in both lakes 2.2 meters. This closure transformed the two connected impoundments into a single reservoir with two distinct basins. The powerplant includes the constructed Mt. Elbert Forebay—located on the ridge to the north of Lower Twin Lakes.

Project water enters Forebay through the Mt. Elbert Conduit from Turquoise Reservoir, which is located 17 kilometers north of Twin Lakes. Turquoise Reservoir is the eastern terminus of the Boustead Tunnel which diverts water from the Western Slope of the Rocky Mountains' Continental Divide. The source of inflow to the lakes was considerably altered by the import of water from different watersheds into Lower Twin Lakes. Our studies were designed to assess the effects of this pumped-storage hydroelectric development on the ecology of Twin lakes. The studies began in 1971 and ended in 1986.

The physical and chemical effects observed can be grouped under three general headings related to:

1. Raised lake levels behind the new Twin Lakes Dam,
2. Increased inflow to the lakes via Mt. Elbert Conduit, and
3. Operation of Mt. Elbert Pumped-Storage Powerplant.

Raised lake levels have changed the morphometry of the lakes, and particularly the morphometry of Upper Twin Lakes. Upper Twin Lakes' maximum surface area increased 45 percent, while maximum volume increased 34 percent, but the mean depth actually decreased by 7 percent—a reflection of the fact that most of the lake's expansion was into the shallow western floodplain.

In general, the bathymetry of the upper lake basin changed from a roughly circular shape to a more oblong configuration, with the deep end near the connecting channel (see fig. 1-5). This new shape altered the pattern of summer inflow to the lake. Formerly, the cooler water of Lake Creek entered Upper Twin Lakes as a plunging inflow that cooled and oxygenated the hypolimnion. Currently, warming in the extensive shallow area causes the creek water to enter the pelagic area of the lake as a near-surface inflow, warming and mixing the epilimnion. The shallow inflow area may also act as a sediment trap and nutrient buffer for the main body of the lake.

In both lakes, the inundation of new terrain resulted in an increase in internal nutrient loading, which is typical of new reservoirs. Total phosphorus and nitrate concentrations increased significantly in both lakes. Raised lake levels also exposed previously unflooded soils to erosion by wave action and water surface fluctuations, particularly along the isthmus separating the two lakes, where steep banks of fine silty soil were undercut and eroded into a new shoreline. This erosive action contributed to a decline in water clarity in the lakes.

During the postoperational period, increased inflow to the lakes not only resulted from project operation but also because annual runoff in the watershed was above normal in 3 of the 4 years after operation began. The average hydraulic residence time in both lakes decreased in the postoperational period after the importation of Fryingpan-Arkansas Project water; Lower Twin Lakes experienced the largest decrease—from 229 to 131 days. Average hydraulic residence time in Upper Twin Lakes declined from 88 to 86 days. Several factors complicated the comparison between the preoperational (1977-81) and postoperational (1982-85) period hydrology of the lakes:

- 2 record low runoff years in the preoperational period,
- 3 record high runoff years in the postoperational period,
- Enlargement of both lakes in the latter part of the postoperational period, and
- Decreased diversions through the Twin Lakes Tunnel during the last 2 years of the postoperational period.

However, Lower Twin Lakes was more affected by the additional inflow from the Mt. Elbert Conduit than was Upper Twin Lakes because the water imported by the Fryingpan-Arkansas Project caused a significant dilution of the lower lake. Mean TDS (total dissolved solids) concentrations of the lower lake decreased to about 40 mg/L—with $CaCO_3$ hardness averaging 27 mg/L.

Powerplant operation effects were especially noticeable at station 3 in the tailrace channel. At generation startup, an initial decrease in water temperatures occurred, followed by an increase, which reflected the mixing of cooler Mt. Elbert Forebay water with the warmer water of Lower Twin Lakes. During pumping, water temperatures at station 3 generally increased as warmer surface water of Lower Twin Lakes was drawn into the tailrace channel.

Pumping and generating operations at Mt. Elbert Pumped-Storage Powerplant increased the intensity and changed the pattern of water circulation in Twin Lakes. For example, during generation, flow direction in the channel connecting the two lakes was sometimes reversed. The induced mixing shortened the duration of thermal stratification in both lakes. In Lower Twin Lakes, the onset of summer stratification occurred later, although fall turnover occurred at nearly the same time as during the preoperational period. The onset of summer stratification was also delayed in Upper Twin Lakes, and fall turnover occurred earlier as well. At the same time, the strength of thermal stratification was reduced, particularly in the lower lake, by increased mixing of surface heat downward into the water column.

Winter conditions were affected also by powerplant operation. The tailrace, and connecting channel, remained ice-free during powerplant operation, and the water level fluctuations caused by pumping and generating weakened the ice cover around the edges of the lakes. Winter hypolimnetic anoxia did not occur in either lake during the postoperational period, which may have been caused by increased circulation of water induced by pumping and generating.

Light penetration in both lakes decreased significantly during the postoperational period. A combination of factors was responsible for decreasing light penetration. Those factors included:

- Inundation of new shoreline areas and their subsequent erosion by wave action and fluctuating water levels in both lakes,
- Resuspension of fine bottom sediments by a lakewide, powerplant-generation induced current gyre in the lower lake, and
- Increased chlorophyll *a* concentrations in the lower lake main basin.

During the postoperational period, changes in biota in the lakes that are attributable to powerplant operation—increased inflow, or the raising of the lakes—were not as clearly identified as changes in either physical or chemical parameters. Biological responses to changing environmental conditions tend to be integrative when the change is not immediately lethal and the length of time for response to chronic change is dependent on the life cycles of the organisms. Primary production tends to be governed by organisms with relatively short life cycles of a few hours to several weeks. Secondary production organisms have longer life cycles, from a few weeks or months to several years. However, certain changes in primary production, i.e., chlorophyll *a* concentration, carbon fixation rate, and phytoplankton populations—and secondary production; i.e., zooplankton, benthic organisms, mysid shrimp, and fish populations, were documented during the postoperational period.

Differences in primary production in the lower lake during the postoperational period seemed equally affected by induced mixing of the euphotic zone, a direct result of powerplant operation, and the trophic upsurge observed in both basins in 1984 immediately after the water surface elevation was raised. Mean monthly chlorophyll *a* concentrations in the lower lake were significantly greater during the postoperational period. Increased chlorophyll *a* concentrations in the lower lake probably reflected a change from slower-growing, conservative algal species to fast-growing, opportunist species that invested in reproduction of new individuals. The typical algal species' succession or reversion from a spring-diatom to a fall-chrysophycean dominated population was disrupted in both lakes during the postoperational period. In the preoperational period, stable stratification conditions in mid- to late-summer favored organisms such as the chrysophycean alga, *Dinobryon* sp., which is adapted to resist permanent sinking losses and is more competitive at lower nutrient concentrations. During the postoperational period, turbulent mixing in the euphotic zone, resulting from increased circulation of the lakes, seemed to favor algal organisms such as diatoms *Asterionella* sp. and *Synedra* sp.

Although carbon fixation rates in both lakes declined during the postoperational period, biomass (as estimated by chlorophyll *a*) or phytoplankton densities did not decline significantly in either basin. The decline in carbon fixation rates seemed directly related to the decrease in light penetration that occurred in both lakes during the postoperational period. Carbon fixation rate is an instantaneous measurement of biological activity under current environmental conditions rather than an integrative estimate of conditions that have occurred over several days or weeks prior to measurement, as are both chlorophyll *a* and phytoplankton density. Therefore, it is entirely possible that carbon fixation rates, although generally lower than in the preoperational period, were sufficient for accumulation of biomass or growth and reproduction of algal organisms to occur.

Upper Twin Lakes' benthic community was dominated by oligochaetes throughout both periods, while Lower Twin Lakes' was dominated by chironomids. Total benthic biomass per unit area increased in Twin Lakes between pre- and postoperational periods. The benthic community in the upper lake was most affected by raising the water surface elevation of the lakes. The lower lake seemed relatively unaffected by either pumping or generating operation, or by increased volume and area after the lakes were raised in 1984.

The potential for operational impacts on the benthic community in the upper lake includes an increased vulnerability to winter anoxia resulting in a toxic heavy metals release due to additional input of organic material from the inundated marshland on the west shore. Mean bottom dissolved oxygen concentrations in the upper lake decreased, indicating an increased oxygen demand in the bottom sediments. If the powerplant, for some reason, did not operate in late winter, anoxic conditions in the upper lake might occur more quickly, last longer, and adversely affect bottom-dwelling organisms to a greater extent than in the preoperational period. Benthos in the lower lake could potentially be affected by powerplant operation if the hydraulic retention time decreased significantly during the period of chironomid pupation and emergence. Thus far, the increased volume of the lakes has prevented a further decrease in retention time; however, future demands for water and power may change this situation at any time.

Zooplankton densities in the lower lake were significantly greater than those in the upper lake throughout both periods. A significant decline in zooplankton densities occurred in both lakes between the pre- and postoperational periods. *Diacyclops bicuspidatus thomasi*, *Diaptomus* spp., and rotifers were the particular zooplankters exhibiting a significant population density decrease in the lower lake during the postoperational period. The largest decrease in zooplankton densities occurred in both lakes during 1982 and was related to increased flushing of the lakes during the growing season. When the lakes were raised in 1984, the greater volumes in the lake basins increased hydraulic retention time, even at higher inflows or throughflows, and zooplankton densities began to increase at the end of the postoperational period.

The observed decrease in zooplankton densities in the postoperational period was caused by a combination of factors. Hydraulic retention time in both lakes decreased significantly and zooplankton populations were subjected to more rapid flushing in the postoperational period—particularly in the lower lake. The duration of summer thermal stratification and the euphotic depth also decreased significantly in both lakes during the postoperational period which, in turn, decreased habitat availability for the zooplankton population. Finally, a dilution effect caused by the increased water volume in the lakes was determined. When the lakes' volume was factored into population estimates, total zooplankton densities, particularly in the upper lake, actually increased.

Like zooplankton densities, *Mysis* shrimp densities in Twin Lakes dropped to the lowest levels on record during 1982, coincidental with an outflow volume increase of 78 percent over the previous year. The lack of powerplant operation during the ice-free season in 1983 appeared to allow the *Mysis* population to recover further. Lake flushing rate was also less than in previous years—aiding recovery of the population.

In Lower Twin Lakes, mysid densities paralleled zooplankton densities, suggesting a dependence on the zooplankton. *Mysis* food habits studies elsewhere have demonstrated this relationship. In the upper lake, however, the two population densities diverged in 1984 with *Mysis* densities surpassing those in the lower lake for the first time during the studies. In the upper lake, mysid densities reached almost 200 per square meter in 1985 and 1986, while zooplankton densities changed little from past years. This apparent divergence was attributed to the zooplankton sampling protocol. As lake levels increased following completion of the dam on the lower lake, an increase in zooplankton densities was observed in the newly inundated areas of the upper lake. Dense swarms of zooplankton were visible throughout this area. Unfortunately, these localized increases were not detected at the standardized zooplankton sampling stations. The mysids, however, appeared to capitalize on this new food supply and their densities increased.

Both zooplankton and *Mysis* populations responded positively (directly or indirectly) to increasing hydraulic retention time associated with powerplant operation; likewise, both populations declined with increased flushing of the lake system.

Mt. Elbert Pumped-Storage Powerplant daytime generation cycle typically entrained 50 000 to 300 000 mysids in an 8-hour period. During 8-hour pumpback cycles, 6 to 44 million shrimp were entrained. Based on these entrainment rates, an estimated 88 percent of the shrimp population could be entrained annually once the powerplant reaches full operating capacity. Not all mysids die during entrainment; however, underwater "windrows" of dead mysids observed on the bottom of Mt. Elbert Forebay indicate immediate mortality is high. In combining these losses with those caused by flushing, the potential exists for severe reduction of the *Mysis* population in Twin Lakes—a situation that already may be occurring. *Mysis* trawl samples collected during the summer of 1989 indicated greatly reduced mysid numbers in Twin Lakes. This drop follows several years of routine pumping and generating operations.

The high *Mysis* densities in Forebay are largely due to the operation of the powerplant. Import of nutrients from Turquoise Lake and Lower Twin Lakes also contribute to the increased productivity in Forebay. The "new reservoir phenomenon" also may have played a role. In addition, large numbers of mysids are "parked" in Forebay during pumpback cycles, and few of these mysids leave during generation cycles. Together, these factors have resulted in *Mysis* densities in excess of 300 per square meter—the highest densities recorded anywhere in the Twin Lakes system.

The fish population in Mt. Elbert Forebay has also responded positively to this new environment with exceptional growth and condition. This situation has not gone unnoticed by local anglers; fishing pressure in these waters has increased.

Ultimately, the accumulated effects of the Mt. Elbert Pumped-Storage Powerplant upon the physical and chemical limnology of Twin Lakes and their phytoplankton, zooplankton, and benthos communities will be reflected in the fish community. In this regard, it is important to remember that the postoperational period, or impact phase, studies covered only a short period of regular pumping and generating operations.

Apart from immediately observable powerplant entrainment mortalities, the fuli effect of ecosystem changes summarized will not be evident until year-classes produced during the postoperational period reach maturity. The fish species in Twin Lakes that are self-sustaining will be affected through survival, food supply, growth, recruitment, and reproductive success. The influence of an altered lake morphometry because of increased lake levels and changing management of game fish species in the sport fishery complicate any final assessment of impacts attributable to pump-turbine operation.

In addition to impacting the lake trout forage base, powerplant operation also results in direct entrainment of the species. Entrainment probability is a function of several factors, including:

- Seasonal and daily movement rates,
- Time of pumping,
- Seasonal movements into the tailrace area,
- Home range locations and size,
- Extent of excursions out of the home range,
- Timing of shoreward movements,
- Water temperature preferences,
- Attraction to currents, and
- Feeding activities.

Individual movement rates were found to be similar, but seasonal rates differed significantly. Because the mean summer movement rate was greater than fall rates, entrainment of trout would be more likely in summer than fall. Periods of greatest movement, and thus vulnerability to entrainment during the day, would probably be between 0600 and 1300 hours during spring, summer, and fall, and between 1500 and 1800 hours during winter. Vulnerability of lake trout at night during winter is unknown.

Considering the large home range size and the number of excursions outside their home ranges lake trout make in the spring, it seems reasonable to assume that probability of lake trout entrainment will increase in spring. Vulnerability in fall will probably be similar to that in spring, as sizes of home ranges and frequency of excursions outside of home ranges were similar. A smaller percentage of lake trout occupied home ranges near the powerplant in fall than spring, but this situation could have been caused by a combination of small sample and individual behavior related to spawning activity. Evidence that lake trout were more likely to be found in the vicinity of the tailrace channel in spring and fall was also revealed during gill-netting efforts.

Lake trout will probably be least vulnerable to entrainment during summer because a majority of the population appears to establish home ranges near the center of the lower lake, and movements outside the home ranges are not as great during summer. Although the probability of summer entrainment is reduced compared to spring or fall, entrainment is still a threat. Cold, deep-water areas exist near the tailrace channel and 40 percent of the tagged fish had established summer home ranges near the tailrace.

Home range sizes of lake trout were much smaller during winter than during other times of the year, and few excursions outside of the home ranges occurred. Because of this situation, the probability that a given lake trout would move into the vicinity of the tailrace channel during winter is less than during other times of the year.

How susceptible lake trout in the upper lake will be to entrainment is unknown. Only six fish from Upper Twin Lakes were tracked and none of them moved to Lower Twin Lakes. Water temperatures probably limit lake trout movement through the shallow connecting channel during midsummer and early fall; however, water temperatures are not limiting at other times of the year. Unless some factor causes the habitat of the upper lake to become unsuitable, it does not seem likely that any movements from the upper to the lower lake will occur. Any lake trout movements between the lakes that do occur will probably occur between October and late June. The role of increased lake levels on lake trout movements is uncertain.

Because ripe lake trout were not found in the western third of the lower lake, it seems unlikely that spawning fish will be entrained. Although the riprap area in the tailrace channel may develop as a spawning area in the future, it is likely that existing lake trout spawners will continue using the same areas as in the past. The fish in Twin Lakes, like most lake trout, were found to return to the same spawning area each year.

The magnitude and timing of water level fluctuations caused by pumping or generating operations are important to lake trout spawning success. Historically, water levels in Twin Lakes have been lowest in the fall—with little further change until spring runoff. If the entire Mt. Elbert Forebay was drained to produce power, a 1.2-m increase in the water level of the lower lake could occur. Because natural reproduction has accounted for as much as 88 percent of the gill-net catch for a given year-class, serious losses of lake trout eggs could have a significant effect on the population. Although more information is needed about specific depths of egg deposition, most lake trout in Twin Lakes probably spawn at depths greater than 2 m. Thus, serious losses of lake trout eggs from the proposed maximum daily fluctuations (0.67 m) of the water level are unlikely, assuming no long-term drawdown in excess of 2 m occurs between October and May.

Major water currents formed during either pumping or generation modes of the pump-turbines were predicted before constructing the powerplant. A large gyre was observed in Lower Twin Lakes during generating cycles. The effect these currents will have on fish distribution is unclear. It is not certain lake trout will be attracted per se to the currents produced by the pump-turbines, although some evidence exists they are attracted to the Lake Creek outlet during high flushing flows. What influence the configuration of the outlet structure on the new dam will have on this apparent attraction to currents remains to be resolved.

During power generation, dead and injured fish were found in or near the tailrace channel. Lake trout may feed on dead fish if the dead fish are readily available; thus, they may be attracted to the tailrace area. Using hydroacoustic methods, significantly greater numbers of fish were found in the tailrace channel than elsewhere in the lake, this aggregation can be attributed to a more abundant food supply in this area. Gill-net catches confirmed the presence of both lake and rainbow trout in this channel.

Lake trout eggs probably will not be vulnerable to entrainment because the spawning areas are not near the tailrace channel. Larval lake trout will be safe for the same reason until they emerge from the spawning substrate. In other lakes, larval lake trout have been found to move to deep water after emerging from the substrate. Unless these small fish move into the vicinity of the tailrace channel, it is unlikely that many larvae will be entrained. In fact, relatively few larvae were entrained. Increased mortality could be expected if the tailrace channel becomes a lake trout spawning area.

The most detrimental impact to the Twin Lakes fishery will occur during pump-back cycles. Before constructing Mt. Elbert Pumped-Storage Powerplant, at least 40 percent of lake trout home ranges in Lower Twin Lakes included an area close to the powerplant. This area has experienced changes since the powerplant became operational. These changes make the area more attractive to lake trout and an increased likelihood of lake trout entrainment can be expected as:

- Attraction currents form,
- Injured fish and shrimp discharge during the generation cycle, and
- Lake level increases (3 m) provide deeper and cooler water in the tailrace channel.

Longnose sucker (an important forage species) have inhabited the riprap tailrace channel. Results of hydroacoustic surveys in the channel indicated the presence of about 15 lake trout per million cubic meters of water. Entrainment of lake trout was about 3.4 fish per million cubic meters—indicating that lake trout were able to resist entrainment once in the tailrace channel.

Lake trout, probably the slowest growing species in Twin Lakes, required 6 to 8 years during the preoperational period for recruitment of a year-class to the spawning population. The lake trout population exhibits stability in its age and size structure and growth rate, but characteristic slow growth and older age at maturity make this population unable to compensate quickly to impacts that reduce the abundance of mature adults. However, the longevity of adult lake trout does provide a survival strategy for compensating for short-term impacts affecting reproductive success or recruitment. Most of the lake trout population is found in deep water, away from the tailrace channel entrainment zone. The species' preference for deeper, cooler waters—especially during the summer—and the fact that they were most active during the daytime in spring, summer, and fall appears to reduce their vulnerability to pump-back entrainment. The large riprap substrate along the tailrace channel and its entrance creates suitable cover habitat for fish, especially longnose sucker and longnose dace. Proliferation of small fish populations in the channel's area may further attract lake trout.

Angler catch rates indicate lake trout are attracted to generation flows, but lake trout larger than 508 mm appear resistant to entrainment, as none of this size or larger was captured in powerplant entrainment sampling nets. Estimated entrainment and mortality to lake trout during the 1985 level of pumping and generation operations represent 56 and 35 percent, respectively, of the mean annual angler harvest of this species. Pumping-generating induced mortality in the lake trout population may be density-independent due to the attractive elements of the tailrace channel area and food supply, and thus may be largely additive to natural mortality and angling mortality. At full operation (with both pump-turbine units) entrainment and mortality to lake trout were projected at 239 and 148 percent, respectively, of the mean annual angler harvest. Angler harvest regulations were made stricter in 1984 to reduce angling mortality on lake trout less than 508 mm total length. The projected levels of powerplant mortality could offset benefits of reduced angling mortality. Water level fluctuations weakened winter ice cover, which delayed angler access during December—normally the best angler success month for lake trout. This situation would contribute to a reduced angler exploitation of lake trout, but attraction of lake trout to generation flows throughout the year concentrates fish in a well-defined area and makes them vulnerable to anglers, especially in winter.

Rainbow trout appear relatively resistant to entrainment because of a tendency to remain near the surface; thus, keeping them away from the inlet/outlet structure located 12 m below the water surface in the tailrace channel. The shoreline orientation of stocked rainbow trout serves to bring this species into certain contact with the tailrace channel as they move along shore following stocking. The presence of riprap cover habitat and entrained and vulnerable prey act to concentrate rainbow trout in this area, resulting in improved angler success there. Angler use patterns will change significantly if higher catch rates are sustained in the western end of the lower lake, concentrating angling effort in that area. On the positive side, this change may result in increased harvest of rainbow trout, and improve the return rate of stocked fish.

Winter generation flows have created an open-water tailrace fishery for rainbow trout that did not previously exist. Entrained *Mysis*, both in tailrace flows and lakewide induced currents, may provide a supplementary food source for rainbow trout that was previously unavailable, thus improving fish condition and perhaps increase growth.

The Twin Lakes summer fishery is primarily a shore fishery for rainbow trout that is characterized by low catch rates for 70 to 75 percent of the anglers. The bulk of Twin Lakes' angling public is already experiencing marginal but acceptable fishing success. Water level fluctuations contributing to shoreline turbidity in summer in the broader, more shallow littoral zones will reduce angler success and satisfaction.

In conjunction with additional powerplant related mortality of rainbow trout because of their concentration in the tailrace channel area, these factors may create unacceptable fishing conditions for a large number of "average" anglers. It remains to be seen if improved angler success in the lower lake's western end effectively counters decreased angler catch rates elsewhere in the lakes. Fortunately, the rainbow trout population derives its existence from periodic stocking. Any significant depletions in their abundance may be supplemented by increased stocking.

The increase in littoral habitat in Twin Lakes may have the greatest positive effect on the lakes' fish populations if water levels in these zones remain relatively stable. Pumping-generating induced water level fluctuations and annual drawdowns may increase turbidity or expose these areas—destabilizing habitat availability and macroinvertebrate food production. Increased productivity in these shallow, warmer lake areas will result in an energetic gain passed through the food web from zooplankton and *Mysis* to both catostomid fish and salmonid predators. White sucker demonstrated the lowest entrainment rate of all species, and may benefit the most from the newly created littoral habitat. The new littoral zones may improve the diversity of the fishery with increased populations of brown and brook trout.

Increased littoral zone areas may result in increased food production, prey fish abundance, and lake trout growth, but this may be offset by the relatively greater vulnerability of principal lake trout prey species, *Mysis*, longnose sucker, and kokanee salmon, to pumping-generating plant entrainment and mortality. *Mysis* represent an important component in the lakes' ecological pyramid, affecting both zooplankton and fish populations. A key factor may be the ability of *Mysis* to maintain a viable population in the face of increased mortality from several sources. Lakewide currents induced by generation flow may entrain *Mysis* and transport them over a large lake area, increasing their vulnerability to rainbow trout and kokanee salmon predation. Although this situation is beneficial to these fish species, it represents a new source of mortality. Local depletion of *Mysis* in the tailrace channel caused by pump-back flow entrainment would reduce entrainment potential over time; but lakewide currents creating the large gyre in the lower lake may redistribute shrimp from other lake areas into the tailrace area with each generation cycle—exposing *Mysis* populations to chronic entrainment and mortality. Wintertime entrainment of *Mysis* in pump-back flows appears selective towards mature females, affecting next spring's reproductive stock and production of young. The response of *Mysis* to this additional mortality would influence both zooplankton species' composition and production, and the growth and survival of juvenile lake trout.

Riprap cover habitat in the tailrace channel and Forebay could benefit longnose sucker, but this increased habitat availability also occurs in the lake areas with the highest entrainment potential for fish. The outcome of this predicament for longnose sucker remains uncertain. Kokanee salmon had the highest entrainment rate of all fish species. This vulnerability could limit the effectiveness of stocking kokanee as a lake trout prey base. The estimated entrainment and mortality rates under full pumping or generating capacity would reduce one million stocked kokanee to near zero abundance. Stocking of kokanee would have to be increased three to four fold to maintain an abundant prey population.

When the limnological studies ended in 1985 and the fishery and *Mysis* studies ended in 1986, Twin Lakes had not yet had time to settle into a post-impact equilibrium state. Therefore, a somewhat speculative look at the possible future equilibrium state of Twin Lakes follows.

The "trophic upsurge" in Twin Lakes was expected to level off or even decline somewhat as new habitats and niches were filled and easily leached nutrients were used up, a process surmised to take about 5 years after the lakes' normal water surface elevation established behind the new dam. Shoreline erosion was already subsiding in 1985 as new beaches and benches were forming. However, from late August through mid-November 1989, the water level in Twin Lakes was drawn down to the minimum elevation of 2795 m to complete the removal of the old outlet works and various constructed dikes. This elevation is 9 m below the new maximum and nearly 3 m below the new normal water surface elevation, and it is equal to the preoperational minimum water surface elevation for Upper Twin Lakes. The impact of this disturbance on the "equilibration" of the Twin Lakes ecosystem, and particularly that of Upper Twin Lakes, remains to be determined.

Under normal project conditions, Upper Twin Lakes is now the less flushed of the two lakes, and the extensive new shallow inflow area should diminish the impact of spring runoff on the main body of the lake. Also, circulation induced by powerplant operation could lessen the probability of recurrence of severe winter hypolimnetic anoxia in the Upper Twin Lakes. All these factors could combine to make Upper Twin Lakes the more productive of the two lakes, and a refugium for fish and shrimp populations. However, raising the lake's water surface elevation inundated an estimated 4000 metric tons of vegetation on the western end of Upper Twin Lakes (Bergersen and Maiolie, 1981). This organic contribution, plus that contributed by any increase in lake productivity will increase the biochemical oxygen demand of the lake's bottom sediments. If Mt. Elbert Pumped-Storage Powerplant were to be inoperable during a severe winter, Upper Twin Lakes would be more vulnerable to anoxia and "winter kill" than it was during the preoperational period.

Lower Twin Lakes seems destined to become an intensely flushed and manipulated reservoir. For example, Fryingpan-Arkansas Project operation studies have anticipated hydraulic loading more than 60 percent greater than that observed during these studies. The effects of entrainment, flushing, and pumping or generating induced circulation will probably limit the productivity of Lower Twin lakes to a level lower than that of the Upper Twin Lakes.

Some implications for the sport fishery of these changes in the Twin Lakes system are:

Once the trophic upsurge, which resulted from raising the level of the lakes, has settled into an equilibrium state, Lower Twin Lakes may become a less favorable habitat for lake trout than it was during the preoperational period. Because studies at Twin Lakes were discontinued before a new equilibrium state had established, and because it takes 6 to 8 years for the recruitment of a lake trout year-class to the spawning population, any chronic impacts of project operation on the lake trout fishery still need to be evaluated.

Increased productivity in Upper Twin Lakes, along with the new hydraulic regime and basin morphometry, may create more favorable conditions for lake trout than existed during the preoperational period. Because the studies focused on deep, midlake conditions, the habitat value, water quality, and trophic status of the newly inundated area on the western end of the upper lake were not adequately investigated. However, some evidence shows that this area has become a locus of high productivity. Therefore, Upper Twin Lakes could possibly become a higher quality lake trout fishery than it was before project operation began, but again, this requires further evaluation.

The opportunity for managing both lakes as high quality sport fisheries still exists. A well-planned, long-term environmental monitoring program at Twin Lakes is needed, however, to provide the necessary information for making the appropriate management decisions.

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Literature Cited

Ager, L. M. 1976. A Biotelemetry Study of the Movements of Walleye in Center Hill Reservoir, Tennessee. M.S. thesis. Tennessee Tech. Univ. 96 pp. Cookeville TN.

Ahlstrom, E. H. 1937. Studies on Variability in the *Genus Dinobryon* (Mastigophora). *Trans. Am. Micros. Soc.* 56(2):139-159.

Anonymous. 1986. SPSS-X User's Guide. 2d ed. SPSS, Inc., 988 pp. Chicago IL.

Armitage, K. B. 1961. Species Composition and Seasonal Distribution of Limnetic Crustacean Zooplankton of Northeastern Kansas. *Trans. Kansas Acad. of Sci.* 46:27-35.

Atkinson, N. J. 1931. The Destruction of Grey Trout Eggs by Suckers and Bullheads. *Trans. Am. Fish. Soc.* 61:183-188.

Bancroft, T. A. 1968. *Topics in Intermediate Statistics.* vol. I. Iowa State Univ. Press. 129 pp. Ames IA.

Baxter, R. M. 1977. Environmental Effects of Dams and Impoundments. *Ann. Rev. Ecol. Syst.* 8:255-283.

Bayne, D. R., J. M. Lawrence, and J. A. McGuire. 1983. Primary Productivity Studies During Early Years of West Point Reservoir, Alabama-Georgia. *Freshwater Biol.* 13:477-489.

Beeton, A. M. 1960. The Vertical Migration of *Mysis relicta* in Lakes Huron and Michigan. *J. Fish. Res. Bd. Can.* 29:1701-1708.

Bell, M. C. 1981. *Updated Compendium on the Success of Passage of Small Fish Through Turbines.* U.S. Army Corps of Engineers. DACW-68-76-C-0254.

Bergersen, E. P. and M. Maiolie. 1981. *Colorado Cooperative Fisheries Research Unit Studies at Twin Lakes, Colorado: 1980 Report of Finding.* Bureau of Reclamation Report No. REC-ERC-82-8. 59 pp. Denver CO.

Berrill, M. 1969. The Embryonic Behavior of the Mysid Shrimp, *Mysis relicta. Can. J. Zool.* 47:1217-1221.

Boreman, J. 1977. *Impacts of Powerplant Intake Velocities on Fish.* U.S. Dep. of the Interior, Fish and Wildlife Service. FWS/OBS-76/20.

Bowles, R. R., J. V. Merrier, and G. C. Grant. 1978. *Factors Associated With Accuracy in Sampling Fish Eggs and Larvae.* U.S. Dep. of the Interior, Fish and Wildlife Service. FWS/OBS-78/83.

Brannon, J. M., D. Gunnison, P. L. Butler, and I. Smith. 1978. *Mechanisms That Regulate The Intensity of Oxidation-Reduction in Anaerobic Sediments and Natural Water Systems.* Tech. rep. Y-78-11, U.S. Army Corps of Engineers Waterways Exp. Sta. 113 pp. Vicksburg MS.

Brazo, D. C. and C. R. Liston. 1977. *A Study of the Fishes Associated With the Offshore Rock Jetties of the Ludington Pumped-Storage Powerplant on Lake Michigan During 1976.* Ann. rep. Ludington Proj. Fisheries Res. vol. I, No. 2.

Brinkhurst, R. O. 1974. *The Benthos of Lakes.* St. Martin's Press. 190 pp. New York.

Brown, C.J.D. and R. J. Graham. 1953. Observations on the Longnose Sucker in Yellowstone Lake. *Trans. Am. Fish. Soc.* 83:38-46.

Brownell, W. M. 1970. *Studies on the Ecoloy of* Mysis relicta *in Cayuga Lake,* MS thesis. Cornell University. Ithaca NY.

Brylinsky, M. 1980. Estimating the Productivity of Lakes and Reservoirs. pp. 411-454. In: E. D. LeCren and R. H. Lowe-McConnell. eds. *The Functioning of Freshwater Ecosystems.* Cambrige Univ. Press. New York.

Buckman, H. O. and N. C. Brady. 1969. *The Nature and Properties of Soils.* 7th ed. The Macmillan Co./Collier-Macmillan Ltd., London, UK.

Buikema, A. L., Jr. and P. H. Loeffelman. 1979. Effects of Pumpback Storage on Zooplankton Populations. pp. 109-124. In: *Proc. of the Clemson Workshop on Environ. Impacts of Pumped-Storage Hydroelectric Operations.* U.S. Dep. of the Interior, Fish and Wildlife Service, Southeast Reservoir Investigations, Clemson SC.

Cable, L. F. 1956. Validity of Age Determination From Scales, and Growth of Marked Lake Michigan Lake Trout. U.S. Dep. of the Interior, Fish and Wildlife Service. *Fish. Bull.* 107. vol. 57. 59 pp.

Campbell, S. G. 1985. *Investigating Error in Calculation of Areal Chlorophyll* a *Concentration in Twin Lakes, Colorado.* Bureau of Reclamation Report No. REC-ERC-85-9. 11 pp. Denver CO.

__________. 1987. *Size Fraction Analysis of Chlorophyll* a *in Twin Lakes, Colorado, 1983-85.* Bureau of Reclamation Memo. 88-2-2. 12 pp. Denver CO.

__________ and J. F. LaBounty. 1985. *Chlorophyll* a *Concentration and Distribution in Twin Lakes, Colorado, Prior to Operation of Mt. Elbert Pumped-Storage Powerplant, 1977-81.* Bureau of Reclamation Report No. REC-ERC-85-6. 41 pp. Denver CO.

Campbell, R. S. 1935. A Study of the Common Suckers, *Catostomus commersoni* (Lacepede), of Waskesui Lake. M.S. thesis. Univ. of Saskatchewan, Saskatoon, Saskatchewan, Can.

Carlander, K. D. 1969. *Handbook of Freshwater Fishery Biology.* vol. 1. Iowa State Univ. Press.

Ames IA.

Carruthers, J. W. 1958. A Leaning-Tube Current Indicator ("Pisa"). *Bull. de l'Inst. Oceanogr.* No. 1126. Monaco.

Chronic, H. 1980. *Roadside Geology of Colorado.* Mountain Press Publ. Co. 322 pp. Missoula MT.

Coble, D. W. 1967. The White Sucker Population of South Bay, Lake Huron, and Effects of the Sea Lamprey on It. *J. Fish. Res. Bd. Can.* 24:2117-2136.

Cole, G. A. 1979. *Textbook of Limnology.* The C. V. Mosby Co., 426 pp. St. Louis MO.

Culver, S. R. and L. D. Lentsch. 1984. *Fisheries Evaluation Program (F.E.P.) User's Manual.* Colorado Div. of Wildlife. 8 pp. Fort Collins CO.

Dandy, G. C. and S. F. Moore. 1979. Water Quality Sampling Programs in Rivers. pp. 695-712. In: *J. Environ. Engineering Div.* ASCE.

DeRoche, S. E. 1962. *A Comparison of Lake Trout,* Salvelinus namaycush, *Populations in Two Maine Lakes.* Final rep. Dingell-Johnson Proj. F-13-R. Dep. of Inland Fish. and Game. 32 pp. Augusta ME.

__________. 1969. Observations on the Spawning Habits and Early Life of Lake Trout. *Progress. Fish. Cult.* 31:109-113.

__________ and L. H. Boyd. 1955. The Lake Trout of Cold Stream Pond, Enfield, ME. *Trans. Am. Fish. Soc.* 85:257-270.

Diana, J. S., W. C. MacKay, and M. Ehrman. 1977. Movements and Habitat Preferences of Northern Pike (*Esox lucius*) in Lac St. Anne, Alberta. *Trans. Am. Fish. Soc.* 106:560-565.

Dodson, S. I. 1981. Morphological Variation of *Daphnia pulex* Leydig (Crustacea: Cladocera) and Related Species From North America. *Hydrobiologia* 83:101-114.

Dudley, R. D., A. W. Mullis, and J. W. Terrell. 1977. Movements of Adult Striped Bass (*Morone saxatilus*) in the Savannah River, Georgia. *Trans. Am. Fish. Soc.* 106:312-322.

Dumont, H. J. and J. Green. 1980. Rotatoria: Proceedings of the 2d International Rotifer Symposium. 17-21 September. *Hydrobiologia* 73:1-263. Gent, Belgium.

Duncan, A. C., R. C. Ugland, J. L. Blattner, and R. D. Steger. 1985. *Water Resources Data for Colorado, Water Year 1984,* vol. 1. Missouri River Basin, Arkansas River Basin, and Rio Grande River Basin. USGS. rep. USGS-WRD-CO-84-1. USGS. 421 pp. Lakewood CO.

Eder, S. and C. A. Carlson. 1977. Food Habits of Carp and White Suckers in the South Platte and St. Vrain Rivers and Goosequill Pond, Weld County, Colorado. *Trans. Am. Fish. Soc.* 106:339-346.

Edmondson, W. T. and G. G. Winberg. 1971. *A Manual on Methods for the Assessment of Secondary Productivity in Fresh Waters.* Blackwell Sci. Publs. 358 pp. Oxford UK.

Elliott, J. M. 1977. Some Methods for the Statistical Analysis of Samples of Benthic Invertebrates. *Freshwater Biol. Assn. Sci. Publ.* No. 25.

Emery, A. R. 1973. Preliminary Comparisons of Day and Night Habits of Freshwater Fish in Ontario Lakes. *J. Fish. Res. Bd. Can.* 30:761-774.

Eschmeyer, P. H. 1954. The Reproduction of Lake Trout in Southern Lake Superior. *Trans. Am. Fish. Soc.* 84:47-74.

__________, R. Daly, and L. F. Erkkila. 1952. Movement of Tagged Lake Trout in Lake Superior, 1950-1952. *Trans. Am. Fish. Soc.* 82:68-87.

Everhart, W. H., A. W. Eipper, and W. D. Youngs. 1973. *Principles of Fishery Science.* Cornell Univ. Press. 288 pp. Ithaca NY.

Finnell, L, M. 1972. *Fryingpan-Arkansas Fish Research Investigations.* Colorado Div. of Wildlife. Proj. rep. No. 1:34-40. Fort Collins CO.

__________. 1977. *Fryingpan-Arkansas Fish Research Investigations.* Final rep. Colorado Div. of Wildlife. 96 pp. Fort Collins CO.

__________. 1981. *Results of Fisheries Investigations at Twin Lakes, Colorado 1973-1976.* Bureau of Reclamation Report No. REC-ERC-80-5. 47 pp. Denver CO.

__________ and G. L. Bennett. 1974. *Fryingpan-Arkansas Fish Research Investigations.* Colorado Div. of Wildlife. Proj. rep. No. 3. 44 pp. Fort Collins CO.

__________ and __________. 1975. *Fryingpan-Arkansas Fish Research Investigations.* Colorado Div. of Wildlife. Proj. rep. No. 4. 23 pp. Fort Collins CO.

Flynn, K. J. 1988. The Concept of "Primary Production" in Aquatic Ecology. *Limnol. Oceanogr.* 33(5):1215-1216.

Fraley, J., B. Marotz, J. Decker-Hess, W. Beattie, and R. Zubik. 1989. Mitigation, Compensation, and Future Protection for Fish Populations Affected by Hydropower Development in the Upper Columbia System, Montana, U.S.A. Regulated Rivers. *Res. Manage.* 3:3-18.

Fultz, D. 1960. Experimental Models of Rotating Fluids and Possible Avenues for Future Research. pp. 71-77. In: R. L. Pfeffer, ed. *Dynamics of Climate.* Pergamon Press. New York.

Furst, M. 1965. Experiments on the Transplantation of *Mysis relicta* Loven Into Swedish Lakes. *Rep. Inst. Freshwater Res.* Drottingholm 46:79-87.

Galligan, J. P. 1962. Depth Distribution of Lake Trout and Associated Species in Cayuga Lake, New York. N.Y. *Fish and Game J.* 9:44-68.

Geen, G. J., T. G. Northcote, G. F. Harman, and C. C. Lindsey. 1966. Life Histories of Two Species of Catostomid Fishes in Sixteenmile Lakes, British Columbia, With Particular Reference to Inlet Stream Spawning. *J. Fish. Res. Bd. Can.* 23:1761-1788.

Gerking, S. D. 1959. The Restricted Movement of Fish Populations. *Biol. Rev.* 34:221-242.

_________. 1978. *Ecology of Freshwater Fish Production.* John Wiley and Sons. New York.

Gilmer, D. S. 1971. Home Range and Habitat Use of Breeding Mallards (*Anas platyrynchos*) and Wood Ducks (*Aix sponsa*) in North Central Minnesota as Determined by Radio Tracking. Ph.D. diss. Univ. of Minnesota. 110 pp. Minneapolis MN.

Goddard, C. I., J. W. Lilly, and J. S. Tait. 1974. Effects of M.S. 222 Anesthetization on Temperature Selection in Lake Trout, *Salvelinus namaycush. J. Fish. Res. Bd. Can.* 31:100-103.

Goldman, C. R. and A. J. Horne. 1983. *Limnology.* McGraw-Hill Book Co. 464 pp. New York.

_________ and B. L. Kimmel. 1978. Biological Processes Associated With Suspended Sediment and Detritus in Lakes and Reservoirs. pp. 19-44. In: J. Cairns, Jr., E. R. Benfield, and J. R. Webster, eds. Current Perspectives on River-Reservoir Ecosystems. *N. Am. Bentho. Soc.* Blacksburg VA.

_________, M. D. Morgan, S. T. Threlkeld, and N. Ageli. 1978. A Population Dynamics Analysis of Cladoceran Disappearance From Lake Tahoe, California-Nevada. *Limnol. and Oceanogr.* 24(2):297-298.

Golterman, H. L. 1975. *Physiological Limnology.* Elsevier Sci. Publ. Co. The Netherlands, Amsterdam.

Goodyear, C. P. 1977. Mathematical Methods to Evaluate Entrainment of Aquatic Organisms by Powerplants. U.S. Dep. of the Interior, Fish and Wildlife Service. FWS/OBS-76/20.3.

Gosho, M. E. 1975. The Introduction of *Mysis relicta* in Freshwater Lakes. *Fish. Res. Inst.* Circ. No. 75-2. Univ. of Washington, 66 pp. Seattle WA.

Graneli, W. 1979. The Influence of *Chironomus pluosmys* Larvae on the Exchange of Dissolved Substances Between Sediments and Water. *Hydrobiologia* 66(2):149-159.

Gregg, R. E. 1976. *Ecology of* Mysis relicta *in Twin Lakes, Colorado.* Bureau of Reclamation Report No. REC-ERC-76-14. 70 pp. Denver CO.

_________ and E. P. Bergersen. 1980. *Mysis relicta*: Effects of Turbidity and Turbulence on Short-Term Survival. *Trans. Am. Fish. Soc.* 109:207-212.

Griest, J. R. 1976. The Lake Trout of Twin Lakes, Colorado. M.S. thesis. Colorado State Univ. 68 pp. Fort Collins CO.

_________. 1977. *The Lake Trout of Twin Lakes, Colorado.* Bureau of Reclamation Report No. REC-ERC-77-4. 29 pp. Denver CO.

Grimard, Y. and H. G. Jones. 1982. Trophic Upsurge in New Reservoirs: A Model for Total Phosphorus Concentrations. *Can. J. Fish. Aquatic Sci.* 39:1473-1483.

Griswold, T. 1988. Transbasin Diversions. pp. 271-277. In: *Proc. 1987 Regional Meetings.* U.S. Comm. on Irrigation and Drainage. Denver CO.

Haffner, G. D., G. P. Harris, and M. K. Jarai. 1979. ch. 8. Physical Variability and Phytoplankton Communities. III. Vertical Structure in Phytoplankton Populations. *Arch. Hydrobiol.* 89(3):363-381.

Harris, G. P. 1980. The Measurement of Photosynthesis in Natural Populations of Phytoplankton. pp. 129-187. In: I. Morris, ed. *The Physiological Ecology of the Phytoplankton.* Univ. of California Press, Berkeley.

_________. 1984. Phytoplankton Productivity and Growth Measurements: Past, Present, and Future. *J. Plank. Res.* 6(2):219-237.

_________, G. D. Haffner, and B. B. Piccinin. 1979. ch. 8. Physical Variability and Phytoplankton Communities. II. Primary Productivity by Phytoplankton in a Physically Variable Environment. *Arch. Hydrobiol.* 88(4):393-425.

Harris, R.H.D. 1962. Growth and Reproduction of the Longnose Sucker (*Catostomus catostomus* Forster) in Great Slave Lakes. *J. Fish. Res. Bd. Can.* 19:113-126.

Hasler, A. D. and W. G. Einsele. 1948. Fertilization for Increasing Productivity of Natural Inland Waters. pp. 527-554. In: *Trans. 13th N. Am. Wildl. and Nat. Resour. Conf.*

Hauck, F. R. and G. A. Edson. 1976. Pumped Storage: Its Significance as An Energy Source and Some Biological Ramifications. *Trans. Am. Fish. Soc.* 103:158-164.

Hayden, F.V. 1874. Twin Lakes. pp. 47-54, figs. 10-11. In: Ann. rep. of the U.S. Geol. and Geogr. Surv. of the Terr., Embracing Colorado; Being a Report of Progress of the Exploration for the Year 1873.

_________. 1876. *Ancient Lake Basins-Glacial Lakes-Marainal Deposits in the Valley of the Upper Arkansas River and Along Both Flanks of the Sawatch Mountains.* pp. 47-53. In: Ann. rep. U.S. Geol. and Geogr. Surv. of the Terr., Embracing Colorado and Parts of Adjacent Territories; Being a Report of Progress of the Exploration of the Year 1874; and *Glacial Lakes.* pp. 48-50, 2 plates.

Hayens, J. M., R. H. Gray, and J. C. Montgomery. 1978. Seasonal Movements of White Sturgeon (*Acipenser Transmontanus*) in the Mid-Columbia River. *Trans. Am. Fish. Soc.* 107:275-280.

Haynes, M. L. 1956. Life History Studies of Two Species of Suckers in Shadow Mountain Reservoir, Grand County, Colorado. M.S. thesis. Colorado State Univ. Fort Collins CO.

Hayward, R. S. and M. J. Van Den Avyle. 1986. The Nature of Zooplankton Spatial Heterogeneity in a Nonriverine Impoundment.

Hydrobiologia 131:261-271.

Helsel, D. R. 1987. Advantages of Nonparametric Procedures for Analysis of Water Quality Data. *J. Hydro. Sci.* 32(2):179-190.

Hern, J. D. 1970. *Study and Interpretation of the Chemical Characteristics of Natural Water*, 2d ed. USGS. Water-Supply Paper 1473. U.S. Govt. Print. Off. Washington DC.

Hodgkiss, I. J. and L.T.H. Chan. 1976. Studies on Plover Cover Reservoir, Hong Kong. *Freshwater Biol.* 6:301-315.

Holm-Hansen, O. and B. Riemann. 1978. Chlorophyll *a* Determination: Improvements in Methodology. *OIKOS* 30:438-447.

Horak, D. L. and H. A. Tanner. 1964. The Use of Vertical Gill Nets in Studying Fish Distribution, Horsetooth Reservoir, Colorado. *Trans. Am. Fish. Soc.* 93:137-145.

Hudson, P. L. 1970. Quantitative Sampling With Three Benthic Dredges. *Trans. Am. Fish. Soc.* 99(3):603-607.

Hutchinson, G. E. 1957. A Treatise on Limnology, vol. I. *Geography, Physics, and Chemistry.* John Wiley and Sons, Inc. 1015 pp. New York.

Imboden, D. M. 1979. The Impact of Pumped-Storage Operation on the Vertical Temperature Structure in A Deep Lake: A Mathematical Model. pp. 125-145. In: Proc. of the Clemson Workshop on Environ. *Impacts of Pumped-Storage Hydroelectric Operations.* U.S. Dep. of the Interior, Fish and Wildlife Service, Southeast Reservoir Investigations. Clemson SC.

Jeffrey, W. V. and G. F. Humphrey. 1975. New Spectrophotometric Equations for Determining Chlorophylls *a*, *b*, c_1, and c_2 in Higher Plants, Algae, and Natural Phytoplankton. *Biochem. Physiol. Pflanzen.* (BBP). No. 2, Bd. 167:191-194.

Johnston, J. M. 1981. Development and Evaluation of Hydroacoustic Techniques for Instantaneous Fish Population Estimates in Shallow Lakes. Proj. F-68-R-4. Inland Marine Game and Fish Res. Washington State Game Dep. Olympia WA.

Jordan, D. S. 1891. Explorations in Colorado and Utah During the Summer of 1889, With an Account of the Fishes Found in Each of the River Basins Examined. *Bull. of the U.S. Fish Comm.* 9:1-40, 5 pl.

__________ and B. W. Evermann. 1890. Description of the Yellow-finned Trout of Twin Lakes, Colorado. pp. 453-454. In: *Proc. of the Natl. Museum*, vol. 12, 1889, No. 780.

Juday, C. 1906. A Study of Twin Lakes, Colorado With Special Consideration of the Food of the Trouts. *Bull. U.S. Bur. Fish.* 26(616):151-178.

__________. 1907. Studies on Some Lakes in the Rocky and Sierra Nevada Mountains. *Trans. Wisconsin Acad. Sci., Arts, and Letters.* 15(2):781-793.

Kaster, J. L. and G. Z. Jacobi. 1978. Benthic Macroinvertebrates of a Fluctuating Reservoir. *Freshwater Biol.* 8:283-290.

Keefe, T. J. and E. P. Bergersen. 1977. A Simple Diversity Index Based on the Theory of Runs. *Water Res.* 11:689-691.

King, D. L. and T. J. Rhone. 1975. Physical and Mathematical Model Studies of Pumped-Storage Reservoir Hydrodynamics for Determination of Environmental Effects. Bureau of Reclamation. Paper presented at the Intl. Assoc. for Hydraulic Res. Congress. Sao Paulo, Brazil.

Klein, W. D. 1957. An Experimental Plant of the Small Crustacean *Mysis.* Colorado Dep. of Game and Fish. Fish leaflet No. 53:1.

Krezoski, J. R., S. C. Mozley, and J. A. Robbins. 1978. Influence of Benthic Macroinvertebrates on Mixing of Profundal Sediments in Southeastern Lake Huron. *Limnol. Oceanogr.* 23(5):1011-1016.

Krieger, D. A. 1980. *Ecology of Catostomids in Twin Lakes, Colorado, in Relation to A Pumped-Storage Powerplant.* Bureau of Reclamation Report No. REC-ERC-80-2. 46 pp. Denver CO.

LaBounty, J. F. and S. D. Hiebert. 1981. *Manual for Operating the Hydrolab Meteorological-Limnological Monitoring Equipment.* Bureau of Reclamation. 44 pp. Denver CO.

__________ and L. O. Timblin, Jr. 1988. A Case History: Environmental Protection and Enhancement in the Planning, Construction, and Operation of the Mt. Elbert Pumped-Storage Powerplant. Bureau of Reclamation. Denver CO.

__________ and J. J. Sartoris. 1976. *Studies of the Benthic Environment of Twin Lakes, Colorado.* Bureau of Reclamation Report No. REC-ERC-76-12:1-28. 47 pp. Denver CO.

__________ and __________. 1981. *Studies of the Effects of Operating the Mt. Elbert Pumped-Storage Powerplant on Twin Lakes, Colorado: 1980 Report of Findings.* Bureau of Reclamation 56 pp. Denver CO.

__________ and __________. 1982. *Effects of Operating the Mt. Elbert Pumped-Storage Powerplant in Twin Lakes, Colorado: 1981 Report of Findings.* Bureau of Reclamation Report No. REC-ERC-83-2. 88 pp. Denver CO.

__________, __________, and D. M. Lieberman. 1984. *Effects of Operating Mt. Elbert Pumped-Storage Powerplant on Twin Lakes, Colorado: 1982 Report of Findings.* Bureau of Reclamation Report No. REC-ERC-84-23. 104 pp. Denver CO.

__________, __________, and S. G. Campbell. 1985. *Primary Productivity (C^{14}) at Twin Lakes, Colorado: 1973-81 Study Results.* Bureau of Reclamation Report No. REC-ERC-85-8. 35 pp. Denver CO.

__________, __________, __________, J. R. Boehmke, and R. A. Roline. 1980. *Studies of the Effects of Operating the Mt. Elbert Pumped-Storage Powerplant on Twin Lakes, Colorado: 1979 Report of Findings.* Bureau of Reclamation Report No. REC-ERC-80-7. 66 pp. Denver CO.

__________, C. A. Crysdale, and D. W. Eller. 1976. *Dive Studies of Twin Lakes, Colorado.* Bureau of Reclamation Report No. REC-ERC-76-15. 23 pp. Denver CO.

Lagler, K. F. 1956. *Freshwater Fishery Biology.* Wm. C. Brown Co. Publ. Dubuque IA.

LaLancette, L. M. 1973. Studies of the Growth, Reproduction, and Diet of the White Sucker, *Catostomus commersoni* (Lacepede) of Gameline Lake, Chicoutinin, Quebec. Ph.D. diss. Univ. of Waterloo. Waterloo, Quebec, Can.

Lambou, V. W., L. R. Williams, S. C. Hern, R. W. Thomas, and J. D. Bliss. 1976. Prediction of Phytoplankton Productivity in Lakes. pp. 696-700. In: Proc. Conf. Environ. *Modeling and Simulation*, April 19-22, 1976, EPA 600/9-76-016. Cincinnati OH.

Lawler, G. H. 1969. Activity Periods of Some Fishes in Heming Lake, *Canada. J. Fish. Res. Bd. Can.* 26:3266-3267.

Lieberman, D. M. 1983. *Common Plankton of Twin Lakes, Colorado.* Bureau of Reclamation Report No. REC-ERC-82-21. 17 pp. Denver CO.

Likens, G. E. 1975. Primary Productivity of Inland Aquatic Ecosystems. pp. 185-202. In: H. Leith and T. H. Whittaker, eds. *Primary Productivity of the Biosphere.* Springer-Verlag. New York.

Livingstone, D. A. 1963. Chemical Composition of Rivers and Lakes. USGS. Prof. pap. 440-G. U.S. Govt. Print. Off. 64 pp. Washington DC.

Machniak, K. 1975. The Effects of Hydroelectric Development on the Biology of Northern Fisheries (Reproduction and Population Dynamics), IV. Lake trout *Salvelinus namaysuch* (Walbaum). Dep. Environ. Fish. Marine Serv. Freshwater Inst. Tech. rep. 52 pp. Winnipeg, Manitoba, Can.

MacLean, A. L. 1976. The Use of Pressure-Sensing and Temperature Sensing Ultrasonic Telemetry to Study Depth Distribution of Rainbow Trout (*Salmo gairdneri*). M.S. thesis. Ohio State Univ. 140 pp. Columbus OH.

Main, R. B. 1981. *WATSTORE Data Link Users Manual.* Bureau of Reclamation. Unpublished.

Maiolie, M. 1987. Fish and Opossum Shrimp Entrainment in the Mt. Elbert Pumped-Storage Powerplant. Ph.D. diss. Colorado State Univ. 117 pp. Fort Collins CO.

__________ and E. P. Bergersen. 1979. *Twin Lakes Studies.* 1979 Annual Progress Report. Bureau of Reclamation. Contract No. 7-07-83-V200. Colorado Coop. Fish. Res. Unit Colo. State Univ. Fort Collins CO.

__________ and __________. 1984. *Fishery Investigations at Twin Lakes, Colorado.* Monitoring the Mt. Elbert Pumped-Storage Powerplant. Final segment rep. 1981-1984. Colorado Coop. Fish. Res. Unit. Colo. State Univ. Fort Collins CO.

Martin, N. V. 1951. A Study of the Lake Trout, *Salvelinus namaycush*, in Two Algonquin Park, Ontario Lakes. *Trans. Am. Fish. Soc.* 81:111-137.

__________. 1955. The Effect of Drawdowns on Lake Trout Reproduction and Use of Artificial Spawning Beds. pp. 263-271. In: *Trans. 20th N. Am. Wildl. Conf.* 683 pp. ed. James B. Trefethen. Montreal.

__________. 1957. Reproduction of Lake Trout in Algonquin Park, Ontario. *Trans. Am. Fish. Soc.* 86:231-244.

__________. 1960. Homing Behavior in Spawning Lake Trout. *Can. Fish. Cult.* 26:3-6.

Mason, W. T., Jr. 1973. *An Introduction to the Identification of Chironomid Larvae.* U.S. Environmental Protection Agency. National Environmental Research Center. Cincinnait OH.

McCauley, R. W. 1970. Preferred Temperature of Yearling Lake Trout (*Salvelinus namaycush*). *J. Fish. Res. Bd. Can.* 27:1729-1733.

McCleave, J. D., G. W. Labar, and F. W. Kircheis. 1977. Within-Season Homing Movements of Displaced Mature Sunapee Trout (*Salvelinus alpinus*) in Floods Pond, Maine. *Trans. Am. Fish. Soc.* 106:156-162.

McConnell, W. J. 1951. The Opercular Bones as An Indicator of Age and Growth of the Carp, *Cyprinus carpio linnaeus. Trans. Am. Fish. Soc.* 81:138-149.

Moenke, H. 1971. *Ecology of Colorado Mountains to Arizona Deserts.* Denver Museum of Nat. History, Museum Pictorial No. 20. 96 pp. Denver CO.

Morris, W. K. 1977. Evaluation of Methods of Attaching Simulated Ultrasonic Transmitter to Adult Male Walleyes, *Stizostedion vitreum vitreum*, Under Adverse Environmental Conditions. M.S. thesis. Univ. of Oklahoma, 116 pp. Norman OK.

Needham, J. G. and P. R. Needham. 1962. *A Guide to the Study of Freshwater Biology.* Univ. of California Press. 108 pp. San Francisco.

Nesler, T. P. 1981a. *The Relative Abundance of Opossum Shrimp,* Mysis relicta, *in Twin Lakes, Colorado, Using A Benthic Trawl.* Bureau of Reclamation Report No. REC-ERC-82-3. 22 pp. Denver CO.

__________. 1981b. *Twin Lakes Studies: A Characterization of the Twin Lakes Fishery via Creel Census With an Evaluation of Potential Effects of Pump-Storage Power Generation.* Bureau of Reclamation Report No. REC-ERC-82-4, 48 pp. Denver CO.

__________. 1981. *Studies of the Limnology, Fish*

Populations, and Fishery of Turquoise Lake, Colorado-1979-80. Bureau of Reclamation Report No. REC-ERC-82-5. 31 pp. Denver CO.

_________. 1984. *Twin Lakes—Mt. Elbert Pumped-Storage Studies.* Colorado Div. of Wildlife. 93 pp. Fort Collins CO.

_________. 1986. Prediction of Ice-Fishing Harvest of Quality-Size Lake Trout From Previous Gill-Net Sampling. *N. Am. J. of Fish. Manage.* 6:277-281.

Neuhold, J. M. and K. H. Lu. 1957. *Creel Census Method.* Utah State Fish and Game Publ. No. 8. 36 pp. Salt Lake City UT.

Nolting, D. H. 1968. *The Lake Trout in Colorado.* Fish. Res. Div., Colorado Div. of Wildlife. Unpublished. Denver CO.

Odum, E. P. and E. J. Kuenzler. 1955. Measurement of Territory and Home Range Size in Birds. *Auk* 72:128-137.

Olson, D. E. 1963. Role of the White Sucker in Minnesota Waters. *Proc. Minn. Acad. Sci., Arts, and Letters.* 31:68-73.

Ovchynnyk, M. M. 1965. On Age Determination With Scales and Bones of the White Sucker, *Catostomus commersoni* (Lacepede). *Zool. Anz.* 175:325-345.

Pennak, R. W. 1953. *Freshwater Invertebrates of the United States.* Ronald Press Co. 769 pp. New York.

_________. 1957. Species Composition of Limnetic Zooplankton Communities. *Limnol. and Oceanogr.* 2(3):222-232.

_________. 1963. Ecological and Radiocarbon Correlations in Some Colorado Mountain Lake and Bog Deposits. *Ecology* 44:1-15.

_________. 1966. Rocky Mountain States. pp. 349-369. In: D. G. Frey, ed. *Limnology in North America.* Univ. of Wisconsin Press. Madison WI.

_________. 1978. *Freshwater Invertebrates of the United States.* 2d ed. Ronald Press Co. 803 pp. New York.

Poddubnyi, A. G., L. K. Malenin, and V. V. Gayduk. 1970. Experimentation in Under-Ice Telemetric Observation of the Behavior of Wintering Fish. *J. Fish. Res. Bd. Can. Trans.* Series No. 1817. 9 pp.

Potter, D. V. and J. L. Meyer. 1982. Zooplankton Communities of a New Pumped-Storage Reservoir. *Water Resour. Bull.* 18(4):635-642.

_________, M. P. Steven, and J. L. Meyer. 1982. Changes in Physical and Chemical Variables in a New Reservoir Due to Pumped-Storage Operations. *Water Resour. Bull.* 18(4):627-633.

Powell, J. W. 1891. Eleventh Annual Report of the U.S. Geological Survey of the Secretary of the Interior, 1889-90, Part II—Irrigation. pp. 45-52 and 135-140, Plates LXXI and LXXXI.

Powell, T. G. 1975. *Urban Lake Creel Census.* Fed. Aid Proj. F-52-R-1, Job 2. Final report. Colorado Div. of Wildlife. 20 pp. Fort Collins CO.

_________ and D. Bowden. 1979. *Fishermen Survey and Harvest Analysis.* Fed. Aid. Proj. F-55-R-4. App. A. Colorado Div. of Wildlife. 49 pp. Fort Collins CO.

Powers, C. F. and A. Robertson. 1965. *Some Quantitative Aspects of the Macrobenthos of Lake Michigan.* Great Lakes Res. Div. Univ. of Michigan. Publ. 13:153-159. Ann Arbor MI.

Pycha, R. 1977. *Lake Superior Fish Stocks.* Quarterly Reports, July-September, 1977. Great Lakes Fish. Lab. Ann Arbor MI.

Rahrer, J. F. 1968. Movements of Adult Lake Trout in Lake Superior. *Trans. Am. Fish. Soc.* 97:481-484.

Rawson, D. S. 1961. The Lake Trout of Lac LaRonge, Saskatchewan. *J. Fish. Res. Bd. Can.* 18:423-462.

_________ and C. A. Elsey. 1948. Reduction in the Longnose Sucker Population of Pyramid Lake, Alberta in an Attempt to Improve Angling. *Trans. Am. Fish. Soc.* 78:13-31.

Reclamation, Bureau of. 1978. *Annual Operating Plan, Fryingpan-Arkansas Project, Colorado. 1977 Operations—1978 Outlook.* Lower Missouri Region. Denver CO.

_________. 1979. *Annual Operating Plan, Fryingpan-Arkansas Project, Colorado. 1978 Operations—1979 Outlook.* Lower Missouri Region. Denver CO.

_________. 1980. *Annual Operating Plan, Fryingpan-Arkansas Project, Colorado. 1979 Operations—1980 Outlook.* Lower Missouri Region. Denver CO.

_________. 1981. *Annual Operating Plan, Fryingpan-Arkansas Project, Colorado. 1980 Operations—1981 Outlook.* Lower Missouri Region. Denver CO.

_________. 1982. *Annual Operating Plan, Fryingpan-Arkansas Project, Colorado. 1981 Operations—1982 Outlook.* Lower Missouri Region. Denver CO.

_________. 1983. *Annual Operating Plan, Fryingpan-Arkansas Project, Colorado. 1982 Operations—1983 Outlook.* Lower Missouri Region. Denver CO.

_________. 1984. *Annual Operating Plan, Fryingpan-Arkansas Project, Colorado.* Lower Missouri Region. Denver CO.

_________. 1985. *Annual Operating Plan, Fryingpan-Arkansas Project, Colorado.* Lower Missouri Region. Denver CO.

_________. 1986. *Annual Operating plan, Fryingpan-Arkansas Project, Colorado.* Missouri Basin Region. Billings MT.

_________. 1987. *Annual Operating Plan, Fryingpan-Arkansas Project, Colorado.* Missouri Basin Region. Billings MT.

Reed, E. B. and J. R. Olive. 1958. Altitudinal Distribution of Some Entomostraca in Colorado.

Ecology 39:66-76.

Reynolds, C. S. 1984. Phytoplankton Periodicity: The Interactions of Form, Function and Environmental Variability. *Freshwater Biol.* 14:111-142.

Rhee, G. and I. J. Gotham. 1981. The Effect of Environmental Factors on Phytoplankton Growth: Temperature and the Interactions of Temperature With Nutrient Limitation. *Limnol. Oceanogr.* 26(4):635-648.

Rhone, T. J. 1976. Mt. Elbert Pumped Storage Project: Final Results and Conclusions on Physical Model Studies of Pumped-Storage Reservoir Hydrodynamics at Twin Lakes. In: J. F. LaBounty, ed. *Third Workshop on Ecology of Pumped-Storage Research at Twin Lakes, Colorado, and Other Localities.* Bureau of Reclamation. 107 pp. Denver CO.

Ricker, W. E. 1975. Computation and Interpretation of Biological Statistics of Fish Populations. *Bull. Fish. Res. Bd. Can.* No. 191. 382 pp. Ottawa.

Rinne, J. N. 1973. A Limnological Study of Central Arizona Reservoirs With Reference to Horizontal Fish Distribution. Ph.D. diss. Arizona State Univ. 350 pp. Tempe AZ.

Ritchie, J., J. McHenry, and A. Gill. 1973. Dating Recent Reservoir Sediments. *Limnol. Oceanogr.* 18:254-263.

Robson, D. S. and D. G. Chapman. 1961. Catch Curves and Mortality Rates. *Trans. Am. Fish. Soc.* 90L:181-189.

Royce, W. F. 1951. Breeding Habits of Lake Trout in New York. Bull. *U.S. Bur. Fish.* 52(59):59-72.

Ruffner, J. A. and F. E. Bair. 1978. *Climates of the States,* vol. 1. Gale Res. Co. 602 pp. Detroit MI.

Sartoris, J. J. 1989. Twin Lakes, Colorado, U.S.A: Ecological Studies of the Effects of Pumped-Storage Hydroelectric Development on a Pair of Montane Lakes. In: *Symp. for Biology in Hungary* 38:379-390.

__________, J. F. LaBounty, and H. D. Newkirk. 1977. *Historical, Physical, and Chemical Limnology of Twin Lakes, Colorado.* Bureau of Reclamation Report No. REC-ERC-77-13. 86 pp. Denver CO.

Serchuk, F. M. 1976. The Effects of the Ludington Pumped Storage Power Project on Fish Passage Through Pump-Turbines and on Fish Behavioral Patterns: Part I. Ph.D. diss. Michigan State Univ. 130 pp. Ann Arbor MI.

Shapiro, J. 1958. The Core Freezer—A New Sampler for Lake Sediments. *Ecology* 39:758.

Sharma, P. C. and M. C. Pant. 1984. Abundance and Community Structure of Limnetic Zooplankters in Kumaun Lakes, India. *Intl. Rev. der Gesamten Hydrobiologie* 69:91-109.

Shepherd, B. G. 1973. Activity Localization in Coastal Cutthroat Trout (*Salmo clarki clarki*) in A Small Bog Lake. *J. Fish. Res. Bd. Can.* 31:1246-1249.

Shope, W. G., C. O. Morgan, O. O. Williams, T. A. Wilson, N. G. Stuthmann, R. E. Booker, W. E. Price, and W. C. Meeks. 1975. *WATSTORE-National Water Data Storage and Retrieval System.* Open file report 75-426 + updates: 1979, 1981, 1983, 1984. USGS. 5 vols. + app. Reston VA.

Smith, R. L. 1966. *Ecology and Field Biology.* Harper and Row. New York.

Snedecor, G. W. and W. G. Cochran. 1967. *Statistical Methods.* 6th ed. Iowa State Univ. 593 pp. Ames IA.

Sommer, V. 1985. Seasonal Succession of Phytoplankton in Lake Constance. *Bio-Science* 36(6):351-375.

Spoor, W. A. and C. L. Schloemer. 1938. Diurnal Activity of the Common Sucker, *Catostomus commersoni* (Lacepede), and the Rock Bass, *Ambloplites rupestris* (Rafinesque), in Muskellunge Lake. *Trans. Am. Fish. Soc.* 68:211-220.

Ssentongo, G. W. and P. A. Larkin. 1973. Some Simple Methods of Estimating Mortality Rates of Exploited Fish Populations. *J. Fish. Res. Bd. Can.* 30:695-698.

Steele, R. G. and J. H. Torrie. 1960. *Principles and Procedures of Statistics.* McGraw-Hill Book Co. New York.

Steemann-Nielsen, E. 1952. The Use of Radioactive Carbon (C^{14}) for Measuring Organic Production in the Sea. *J. du Conseil.* 18:117-140.

Stemberger, R. S. and J. J. Gilbert. 1985. Body Size, Food Concentration, and Population Growth in Planktonic Rotifers. *Ecology* 66:1151-1159.

Stewart, H. H. 1926. Development, Growth, and Food Habits of the White Sucker, *Catostomus commersoni* LeSueur. *Bull. U.S. Bur. Fish.* 42:147-184.

Stockner, J. G. 1971. Preliminary Characterization of Lakes in the Experimental Lakes Area, Northwestern Ontario, Using Diatom Occurrences in Sediments. *J. Fish. Res. Bd. Can.* 28:265-275.

Straskraba, M. 1980. The Effects of Physical Variables on Freshwater Production: Analyses Based on Models. pp. 13-84. In: E. D. LeCren and R. H. Lowe-McConnell, eds. *The Functioning of Freshwater Ecosystems.* Cambridge Univ. Press. New York.

Strickland, J.D.H. and T. R. Parsons. 1972. A Practical Method of Seawater Analysis. *Fish. Res. Bd. Can.* 310 pp. Ottawa.

Swift, D. R. 1964. Activity Cycles in the Brown Trout (*Salmo trutta*). 2. Artificially Fed. *J. Fish. Res. Bd. Can.* 21:133-138.

Taraguchi, M. 1969. Diel Vertical Migrations of *Mysis relicta* in Green Lake, Wisconsin. Ph.D. diss. Univ. of Wisconsin. 229 pp. Madison WI.

Taylor, W. D., V. W. Lambou, L. R. Williams, and

S. C. Hern. 1980. *Trophic State of Lakes and Reservoirs.* U.S. Army Engineer Waterways Exp. Sta. Tech. rep. E-80-3.

Thorne, R. S. and G. L. Thomas. 1981. Hydroacoustic Surveys of Fish Abundance and Distribution in Twin Lakes, Colorado. Prepared for Colorado Coop. Fish. Res. Unit, Colo. State Univ. 24 pp. Fort Collins CO.

Ubbelohde, C. A. 1969. *A Colorado Reader.* Pruett Publ. Co. 342 pp. Boulder CO.

__________, C. A., M. Benson, and D. S. Smith. 1971. *Colorado History.* 3d ed. Pruett Publ. Co. 348 pp. Boulder CO.

U.S. Department of the Interior. 1975. Fryingpan-Arkansas Project, Colorado. *Final Environmental Impact Statement.* vol. I. Bureau of Reclamation. Denver CO.

U.S. Enivronmental Protection Agency. 1986. *Quality Criteria for Water 1986 (With Updates Through May 1, 1987).* EPA 440/5-86-001. Office of Water Regulations and Standards. Washington DC.

U.S. Geological Survey and Other U.S. Government Agencies. 1977. *National Handbook of Recommended Methods for Water-Data Acquisition.* Off. of Water Data Coord., USGS. Reston VA.

Valle, K. J. 1927. Okologische-Limnologische Untersuchungen uber die Boden und Tiefenfanna in einigen seen nordliche vom Lagodasee. *Acta Zool.* Fenn. 4:2-231. In: R. O. Brinkhurst. 1974. *The Benthos of Lakes.* St. Martin's Press. 190 pp. New York.

Verber, J. L. 1964. Initial Current Studies in Lake Michigan. *Limnol. Oceanogr.* 9:426-430.

Vollenweider, R. A., ed. 1969. A Manual on Methods for Measuring Primary Production in Aquatic Environments. *Intl. Biol. Prog.* Handbook. No. 12. Blackwell Sci. Publ. 225 pp. London UK.

Walch, L. A. 1980. *Movements of Lake Trout in Twin Lakes, Colorado, in Relation to The Mt. Elbert Pumped-Storage Powerplant.* Bureau of Reclamation Report No. REC-ERC-79-17. 71 pp. Denver CO

__________. 1979. Movements of Lake Trout in Twin Lakes, Colorado, in Relation to a Pumped-Storage Powerplant. M.S. thesis. 129 pp. Colorado State Univ. Fort Collins CO.

__________ and E. P. Bergersen. 1982. Home Range and Activity Patterns of Lake Trout in Central Colorado. *Fish. Res.* 1:311-318.

Walton, D.W.H. 1982. Instruments for Measuring Biological Microclimates for Terrestrial Habitats in Polar and High Alpine Regions: A review. In: *Arctic and Alpine Research.* pp. 275-286.

Ward, H. B. and G. C. Whipple. 1966. *Freshwater Biology.* John Wiley and Sons, Inc. New York. 1248 pp.

Warden, R. S., Jr. and W. J. Lorio. 1975. Movements of Largemouth Bass (*Micropterus salmoides*) in Impounded Waters as Determined by Underwater Telemetry. *Trans. Am. Fish. Soc.* 5104:696-702.

Waters, T. F. 1969. Subsampler for Dividing Large Samples of Stream Invertebrate Drift. *Limnol. Oceanogr.* 141:813-815.

Weber, C. I., ed. 1973. Biological Field and Laboratory Methods for Measuring the Quality of Surface Waters and Effluences. U.S. Environ. Prot. Agency. EPA-670/4-73-001. Cincinnati OH.

Weber, W. A. 1972. *Rocky Mountain Flora.* Colorado Assoc. Univ. Press, 438 pp. Boulder CO.

Welch, P. S. 1948. *Limnological Methods.* McGraw-Hill Book Co. 381 pp. New York.

Wetsel, R. G. 1975. *Limnology.* W. B. Saunders Co. 743 pp. Philadelphia PA.

Wilshire, H. G., J. T. O'Connor, and G. A. Swann. 1965. Geology of the Northern Part of the Twin Lakes Batholith, Lake and Chaffee Counties, Colorado. *Astrogeol. Stud. Ann. Prog. Rep: July 1, 1964 to July 1, 1965, Part B: Crater Investigations.* USGS. Denver CO.

Wiltzius, W. J. 1985. *Fish Culture and Stocking in Colorado, 1872-1978.* Colorado Div. of Wildlife. Rep. No. 12. 102 pp. Fort Collins CO.

Winer, B. J. 1962. *Statistical Principles in Experimental Design.* McGraw-Hill Book Co. 672 pp. New York.

Winter, J. D. 1977. Summer Home Range Movements and Habitat Use by Four Largemouth Bass in Mary Lake, Minnesota. *Trans. Am. Fish. Soc.* 106:323-330.

__________, V. B. Kuechle, D. B. Sineff, and J. R. Tester. 1978. Equipment and Methods for Radiotracking Freshwater Fish. Misc. rep. 152-1978, Agric. Exp. Sta. Univ. of Minnesota.

Wunderlich, R. C. 1974. Evaluation of Lake Pollution by Analysis of Sediments. M.S. thesis. Colorado State Univ. 77 pp. Fort Collins CO.

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