Clean Lake Study of Dillon Reservoir in Summit County, Colorado

DILLON CLEAN LAKES STUDY
FINAL REPORT

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Summary

1. The purposes of the Dillon Clean Lakes Project were as follows: (a) to provide comprehensive limnological information on the lake, including its seasonal cycles and its basic physical, chemical, and biological features with special attention to trophic status; (b) to provide information on the present nutrient sources of the lake and their relation to land use; and (c) to construct a model that would be capable of predicting the trophic status of the lake given any likely combination of future development patterns for the Lake Dillon catchment.

2. The limnological study showed that the events in Lake Dillon can be divided according to the following seasons: (a) ice cover from January through April, (b) spring mixing from the beginning to the end of May, (c) summer stratification from June through October, and (d) fall mixing for most of November and all of December. During summer stratification, the mixed layer is stable at a thickness of 5-10 m until September, at which time it begins to thicken.

3. Transparency is lowest during the first half of the stratification period. There is a sudden decrease in transparency in June caused by the entry of large amounts of particulate materials from the watershed. Depending on the amount of runoff, this reduces the transparency at the surface from about 4 m to less than 1 m. A large proportion of the entering particulates settle out quickly, but rapid growth of algae causes transparency to stay low. Minimum secchi depths in July, at the time of the chlorophyll maximum, are between 1.5 and 2.5 m. Transparency increases steadily after the chlorophyll maximum

in July because of algal nutrient depletion. Fall transparencies are high (secchi, 3-5 m).

4. Runoff from the three rivers enters the lake at different depths depending on the wetness of the year (amount of runoff) and the time of year. During both 1981 and 1982, river water entered the lake at progressively greater depths during the stratification season. Entry level was uniformly higher in 1982, when there was much runoff, than in 1981. In both years runoff entered the upper water column in the early period of stratification and the middle water column during the middle and late portions of stratification.

5. Annual average total phosphorus over all depths was 6.6 ug/l in 1981 and 7.7 ug/l in 1982. Peak values at the surface during runoff were considerably higher than this in both years, however (12-17 ug/l). Soluble inorganic phosphorus was consistently present in very small amounts. The upper water column showed a steady loss of total phosphorus from the beginning of stratification until the increase in thickness of the mixed layer in September. Soluble organic phosphorus was almost totally depleted by the end of August.

6. Total nitrogen in the entire water column averaged 294 ug/l in 1981 and 447 ug/l in 1982. Lake Dillon is unusual in its very high ratio of total nitrogen to total phosphorus. This is partly due to tertiary treatment, which removes a lot of the phosphorus but not nitrogen, and to the presence of mining, which releases much nitrogen into upper Tenmile Creek. Soluble inorganic nitrogen is steadily depleted from the upper water column after the onset of stratification. Depletion is essentially complete by the middle of July.

7. The amount of phosphorus in the sediments of Lake Dillon does not differ from the amounts typical of natural oligotrophic lakes.

8. Total particulates in the upper water column (0-5 m) averaged 2.3 mg/l in 1981 and 3.85 mg/l in 1982. In both years there were two peaks of total particulates in the upper water column: one at the time of runoff caused by inorganic particulates, and a second in July caused by phytoplankton biomass.

9. The maximum chlorophyll a at the surface occurred during July of both years and fell between 11 and 13 ug/l. The average for the post-runoff stratification season was 6.7 ug/l in 1981 and 7.3 ug/l in 1982. The highest chlorophyll values in both years were between 17 and 18 ug/l.

10. Repeated elements of the annual composition cycle for phytoplankton were Asterionella, a diatom that is abundant under the ice and during early spring in Lake Dillon, and Synedra, a diatom that accounts for a major portion of the July peak of chlorophyll in Lake Dillon. Irregular but very large populations were observed of the blue-green alga Synechococcus and the diatom Rhizosolenia. No nitrogen-fixing blue-greens were found.

11. Low ratios of phosphorus, nitrogen, and chlorophyll a to organic matter during stratification suggest significant nutrient stress on the phytoplankton during stratification. The ratio of photosynthesis to chlorophyll a was highest during periods of deep mixing and lowest under ice cover and during the first half of stratification, indicating that the greatest nutrient stress occurs

under ice and between the middle of June and the end of August. Maximum daily photosynthesis was between 500 and 900 $mgC/m^2/day$.

12. Minimum oxygen 5 m above the lake bottom occurred in October of both years. In 1981 the minimum was 4.4 mg/l and in 1982 it was 4.6 mg/l. This represents about 50% loss of oxygen from the saturation values. The areal hypolimnetic oxygen deficit was well above the level expected for oligotrophic lakes.

13. Nutrient enrichment studies showed that phosphorus limitation prevails between 1 January and 15 July. After 15 July, the phytoplankton community is strongly nitrogen limited until September, when thickening of the epilimnion replenishes nitrogen supplies. At the beginning of October, strong phosphorus limitation is reestablished. Nutrient limitation is minimal during the month of May and during the last three weeks of September. Nitrogen and phosphorus limitations are rather closely balanced, as shown by the switch from one to the other over the seasonal cycle.

14. Studies of horizontal spatial variation over the lake show no statistical evidence of sustained differences between different parts of the lake. Randomly changing patterns of variation between stations are detectable but small for both biological and chemical variables. The lake can be treated as a functional unit.

15. A variety of trophic indicators indicate that Lake Dillon is mesotrophic at the present time.

16. Volume of water entering the lake by way of the three major rivers was almost double in 1982 what it was in 1981. The time-weighted concentration of both nitrogen and phosphorus was somewhat

higher in 1982 than in 1981. The time-weighted averages for total phosphorus concentration in inflowing rivers varied between 6 and 20 ug/l. Annual time-weighted nitrogen concentrations varied between 200 and 600 ug/l in the three rivers. The average total phosphorus concentrations in tertiary wastewater treatment plant effluents were substantially boosted for all such plants by brief or extended shutdowns and malfunctions.

17. The total amount of phosphorus entering the lake was 6300 pounds in 1981 and 10,700 pounds in 1982. The comparable figures for nitrogen were 189,000 pounds in 1981 and 333,800 pounds in 1982. Expressed per unit lake area, the phosphorus loading corresponded to 0.29 g/m^2/year in 1981 and 0.42 g/m^2/year in 1982.

18. The total nutrient loading was divided according to sources on the basis of a detailed study of small representative watersheds and watershed segments. Equations for the following sources were developed which relate the amount of nutrient yield from a given land surface to the amount of runoff: background yield (undisturbed), nonpoint yield from residential area on sewer, nonpoint yield from urban area on sewer, yield from area on septic systems, yield from ski slopes, and yield from interstate highway. Equations were also developed for the yield from the following sources that did not show dependence on the amount of runoff: Climax Molybdenum, secondary treatment plants (package plants), tertiary treatment plants.

19. It was shown from an analysis of the stream segments that river valley bottoms incorporating substantial amounts of wetland, standing water, and gravel beds accumulate significant fractions of the

nutrient yield in dry years and export accumulated yield in wet years. An equation was developed to account for this storage and purging effect.

20. The yield equations and the storage equations were tested on the total runoff nutrient yield observed in 1981 and 1982. The predicted runoff yield to the lake was 2800 kg and the observed was 2900 kg in 1981. In 1982, the predicted was 4600 and the observed was 4800. Nitrogen predictions were similarly good.

21. By use of the yield equations and other information on nonrunoff nutrient yield, a complete breakdown was made of the phosphorus and nitrogen sources for Dillon in 1981 and 1982. In 1982, which is most typical of a run of years under present trophic conditions, tertiary plants accounted for 15% of phosphorus loading, secondary plants for 2.1%, Climax Molybdenum mining for 2.2%, background runoff for 45.6%, precipitation for 13.1%, groundwater for 1.9%, dispersed nonpoint sources for 14.2%, and a major construction project in the Snake River bottom for 6.4%.

22. At the present time, total loading of the lake can be divided approximately into four quarters. The first two of these quarters are taken up by natural sources. The third quarter is taken up by sources that can be traced back to human waste (septic, wastewater treatment plants), and the fourth quarter is accounted for by human activities that cannot be traced back to waste disposal.

23. At most 25% of the present loading of the lake could be stopped by the most elaborate controls short of diverting wastewater elsewhere. A comprehensive list is made of possible additional

controls. The most important items on the list include major reduction in the shutdowns and malfunctions of tertiary treatment plants, conversion of septic systems to tertiary sewer systems, containment procedures for construction in valley bottoms, and containment of the present yields of interstate highway.

24. A model was developed with the following components: (a) a land use component that accepts information on land use and amount of runoff, (b) a trophic status component that accepts output from the land use component, and (c) an effects component that accepts output from the trophic status component. This model, which is called the Dillon Clean Lakes Model, predicts the nutrient yields by source and by watershed segments given any reasonable combination of future land uses. It also predicts the total loading and trophic status of the lake in terms of total phosphorus and chlorophyll a, and translates this trophic status prediction into economically significant measures such as transparency and hypolimnetic oxygen deficit. The model performs well on 1981 and 1982 data and is used to make predictions about 10 different development scenarios described at the end of the report.

The modelling indicates that Lake Dillon will move into the eutrophic category if diversion water rich in phosphorus is added in quantity to the lake or if high growth occurs without the adoption of nonpoint source controls or other measures not now in practice to reduce the phosphorus loading of the lake. Under low growth or high growth with additional controls, the condition of the lake could be held within the mesotrophic range, but would suffer some trophic

degradation or remain more or less the same, depending on the exact conditions.

Credits and Acknowledgments

This work was overseen by the Dillon Clean Lakes Steering Committee, whose members and their constituencies have been exceptionally supportive and helpful. We are especially indebted to the Committee's Secretary, Mr. Tom Elmore, who has answered innumerable queries and solved many problems for us, and to the Chairman of the Committee, Mr. Bruce Baumgartner, for his decisive leadership.

Our own technical staff deserves much credit for the smooth completion of a complex and sometimes difficult field and laboratory program. Chemists for the project included Ms. Katherine Ochsner, Mr. Stephen Hamilton, Ms. Margaret Robbins, and Mr. Lewis Dennis, all of whom were dedicated to their work well beyond what we had a right to expect. The field crew, which performed its duties well many times under the most arduous conditions, included at various times Dr. Robert Epp, Mr. Steven Murray, Mr. Donald Morris, Jr., Mr. George Kling, and occasionally our chemists and Mr. Terry Carter of the Colorado Department of Health, who was also helpful in many other ways. Ms. Mary Marcotte did a skillful job of preparing the manuscript and Mr. George Kling did the same for the drafting.

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did a considerable amount of sample processing. We also appreciate occasional volunteer contributions from Dr. Thomas Frost, Dr. Suzanne Levine, and Lic. Claudia Cressa, all of the C.U. Boulder Limnology Laboratory.

Among ourselves we have divided the labor roughly as follows. J. Saunders was field leader, ran the quality control and troubleshooting for all aspects of the project, and built and maintained the STORET and other data files. D.W. Crumpacker collected all of the land use data from a combination of primary and secondary sources and reduced the data to standard forms for modelling. C. Brendecke supervised the collection of discharge data, did the water balance calculations, and obtained the U.S.G.S. records. W. Lewis did the data analysis, writing, and modelling.

Introduction

Lake Dillon of Summit County, Colorado, is an impoundment of the Blue River just below its confluence with the Snake River and Tenmile Creek. The watershed drains elevations between lake level at about 2750 m and the mountainous headwaters of the three inflowing rivers at elevations as high as 4300 m. For at least the first decade after its creation in 1963, Lake Dillon was considered unequivocally oligotrophic, as shown by its high transparency. Because of its location, Lake Dillon would be expected to remain oligotrophic indefinitely if the watershed were uninhabited or very sparsely inhabited. Watersheds at high elevations in the Central Rockies are seldom rich in phosphorus because the parent material, which lies relatively close to the surface, consists mostly of hard crystalline rock that is resistant to weathering and poor in phosphorus. In addition, the natural vegetative cover effectively holds the particulate phosphorus inventory, as shown by the natural clarity of streams in undisturbed areas, even at times of peak runoff.

The water of Lake Dillon is under the direct control of the Denver Water Department, which uses Lake Dillon as the main storage facility for the City of Denver. Recreational activities on the lake are managed by the U.S. Forest Service. The lake is a popular sport fishery for rainbow trout, brown trout, and kokanee salmon, and, because of a significant spawning run of brown trout up the Blue River, is of use to the State as a source of brown trout eggs. Numerous permanent homes and vacation homes lie within sight of the lake and most of the owners of these properties would probably regard the blue

color and high transparency of Lake Dillon as an aesthetically or economically valuable amenity. Furthermore, the appearance of the lake is especially prominent to the general public because of the proximity of the lake to Interstate 70. Thus it is clear that major changes in the algal standing crop or transparency of Lake Dillon could be undesirable from diverse viewpoints, ranging from increased water treatment costs for the City of Denver and lower property values for lakeside residents to loss of aesthetic appeal for watershed residents and the general public.

The uses of the watershed are especially diverse. The U.S. Forest Service controls over half the land of the Lake Dillon watershed; its holdings are principally on the steeper slopes and at the higher elevations and are essentially undeveloped. At lower elevations, the watershed contains three major municipalities (Frisco, Breckenridge, and Dillon: Figure 1). There are many other smaller housing developments, some of which are served by small package plants, some by sewer through the four major treatment plants, and others by septic systems. Four important ski areas are also located in the watershed (Breckenridge, Copper Mountain, Keystone and Arapahoe Basin). Climax Molybdenum, a major mining enterprise, is located at the headwaters of Tenmile Creek. Finally, the watershed serves a large number of nonresident visitors, who come in greatest numbers for winter sports and in somewhat smaller numbers for summer recreation. In 1981-1982, the time-weighted average population of the watershed was 19,000 persons and the seasonal peak (winter) was about 84,000 persons. According to the Summit County Planning Department, Summit County, half

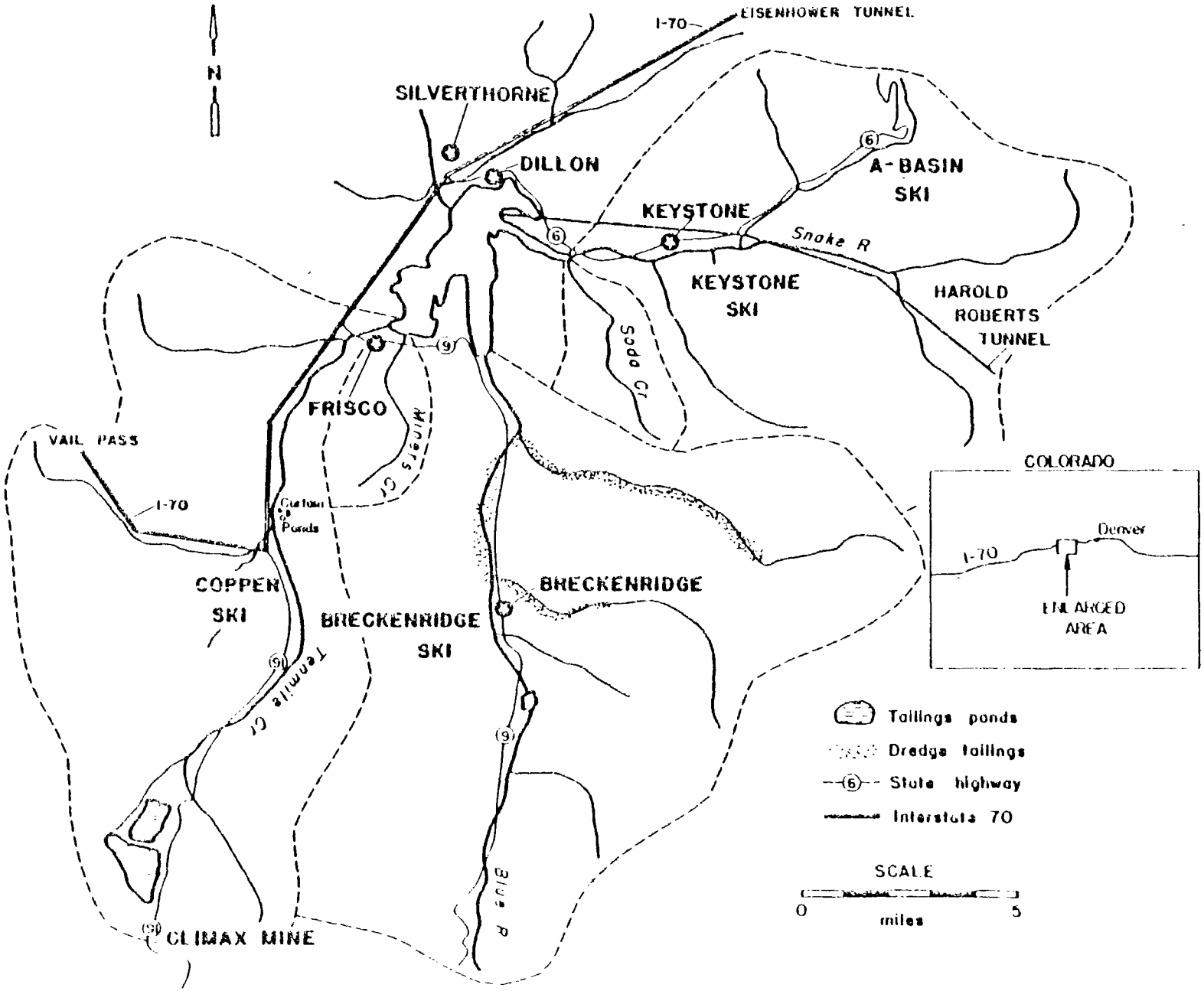


Figure 1. Generalized map of Lake County showing the Western Slope Mining Area and the Eastern Slope Ski Area.

of which is comprised of the Lake Dillon catchment, was the fastest growing county in the United States in 1981.

The motivation of the Dillon Clean Lakes Project is twofold. First, certain notable changes, specifically a decrease in transparency, have occurred in Lake Dillon in recent years. This, coupled with the known increase in population and intensity of land use in the watershed, suggests that a comprehensive evaluation of the trophic status of the lake and of the present nutrient sources would be timely if not essential. Secondly, additional development in the watershed is expected, while the incentives for high water quality remain the same or possibly even increase. The consequences of varying degrees of additional nutrient loading of the lake are not obvious and must be anticipated by some kind of predictive model capable of approximating the trophic status of the lake under various contingencies of watershed development. Thus the twin purposes of the Dillon Clean Lakes Study are evaluation of the present state of the lake and construction of a model that will predict the future trophic status of the lake given various assumptions about the direction of development.

A thorough study of Lake Dillon is well justified by some of its unusual features, which may cause it to respond to nutrient loading somewhat differently than lakes that have been studied elsewhere. It is important to keep some of these points in mind throughout the evaluation of data and modelling. First, the background total phosphorus concentrations for the Dillon watershed in complete absence of human activity would be about 5 ug/l P. Such low background

phosphorus concentrations can maintain lakes of very low algal standing crop and thus of high transparency and beautifully blue water. In contrast to the many lakes that occupy watersheds whose background total phosphorus concentrations are several times higher than this, relatively little human presence or human activity is required to double or triple the total phosphorus concentrations. The vast majority of lakes that have been studied with respect to eutrophication have higher background phosphorus concentrations than Lake Dillon, so good analogies are not always available. We will draw on information from the lakes of the Canadian Shield and parts of Scandinavia, since these lakes typically have low phosphorus concentrations, cool water, and short growing seasons reminiscent of Lake Dillon. Reservoirs and natural lakes in the Midwest and southeastern United States, while thoroughly studied, are less similar to Dillon in these important respects. Lake Tahoe comes to mind as a good comparison for Dillon and will occasionally be useful, but Tahoe is considerably more oligotrophic than Lake Dillon, owing largely to its great mean depth and small ratio of watershed area to lake volume.

The following report, which is based on field work extending from 1 January 1981 to 31 December 1982, deals first with the study design and methods of data collection. A comprehensive limnological analysis follows; the emphasis of this is on trophic status and nutrient chemistry. Next is an analysis of watershed nutrient yield, including quantification of total nutrient loading and separation of nutrient sources based on land-use analysis. The last portion of the report is devoted to description of the Dillon Clean Lakes Model and

its application to development scenarios provided by the Steering Committee.

Design of the Study

Lakes and watersheds show two major kinds of variability: temporal and spatial. Sometimes one category of variation can be studied much more carefully at the expense of the other, but a thorough assessment and modelling effort such as that required for the Dillon Clean Lakes Study would not be sound without inclusion of extensive information on both spatial and temporal variation. At the same time, very frequent sampling over an extensive network of stations for a large suite of variables is seldom financially feasible. Fortunately, such an approach is seldom required. For the Dillon Clean Lakes Study, the need for information on temporal and spatial variation was fulfilled by a dual sampling program, one part of which was designed primarily to provide detailed information for a few sites on many dates and the second to provide a detailed picture of spatial variation on a smaller number of dates. These two approaches were supplemented by special data collection programs of more limited scope whose purpose was to provide information that would not necessarily be forthcoming from the routine studies of temporal and spatial variation. The data base for the Lake Dillon Study thus consists of three components: (1) time series data set, (2) spatial survey data set, and (3) special studies data set.

Time Series Data Set

The time series data set consists of three parts: routine lake data, routine stream data, and precipitation chemistry data.

(1) Routine Lake Data. A key station was selected near the center of the lake in deep water. This station is referred to as the index station; its location is shown in Figure 2. On each routine lake sampling date, the water column at the index station was sampled in 5-m increments from top to bottom. Variables quantified at the index station are listed in Table 1. In addition, one station was selected at the mouth of each of the four major arms of the lake. The locations of these four stations, which are referred to as the main stations, are also shown in Figure 2. The main stations were always sampled as part of the routine lake sampling program. However, samples were taken only at the top and bottom of the water column rather than over the entire vertical profile as at the index station. Analytical coverage is summarized in Table 1.

The schedule of sampling for the routine lake data collection series is shown in Table 2. Samples were taken on 32 different dates. For most of the year, the collections were biweekly. During the period of ice cover, collections were less frequent.

(2) Routine Stream Data. All overland flow was sampled as close to the lake as possible. This required collections at eight stations; the resulting data make up the routine stream data set. The eight stations, with abbreviations as shown in Figure 3, were as follows: Snake River Mouth (SR1), Blue River Mouth (BR1), Tenmile Creek Mouth (TC1), Miner's Creek Mouth (MC2), Soda Creek Mouth (SC2), Blue River Outlet (BRO), Frisco Effluent, and Snake River Effluent. Other effluents were represented in the river mouth samplings because they do not enter the lake directly.

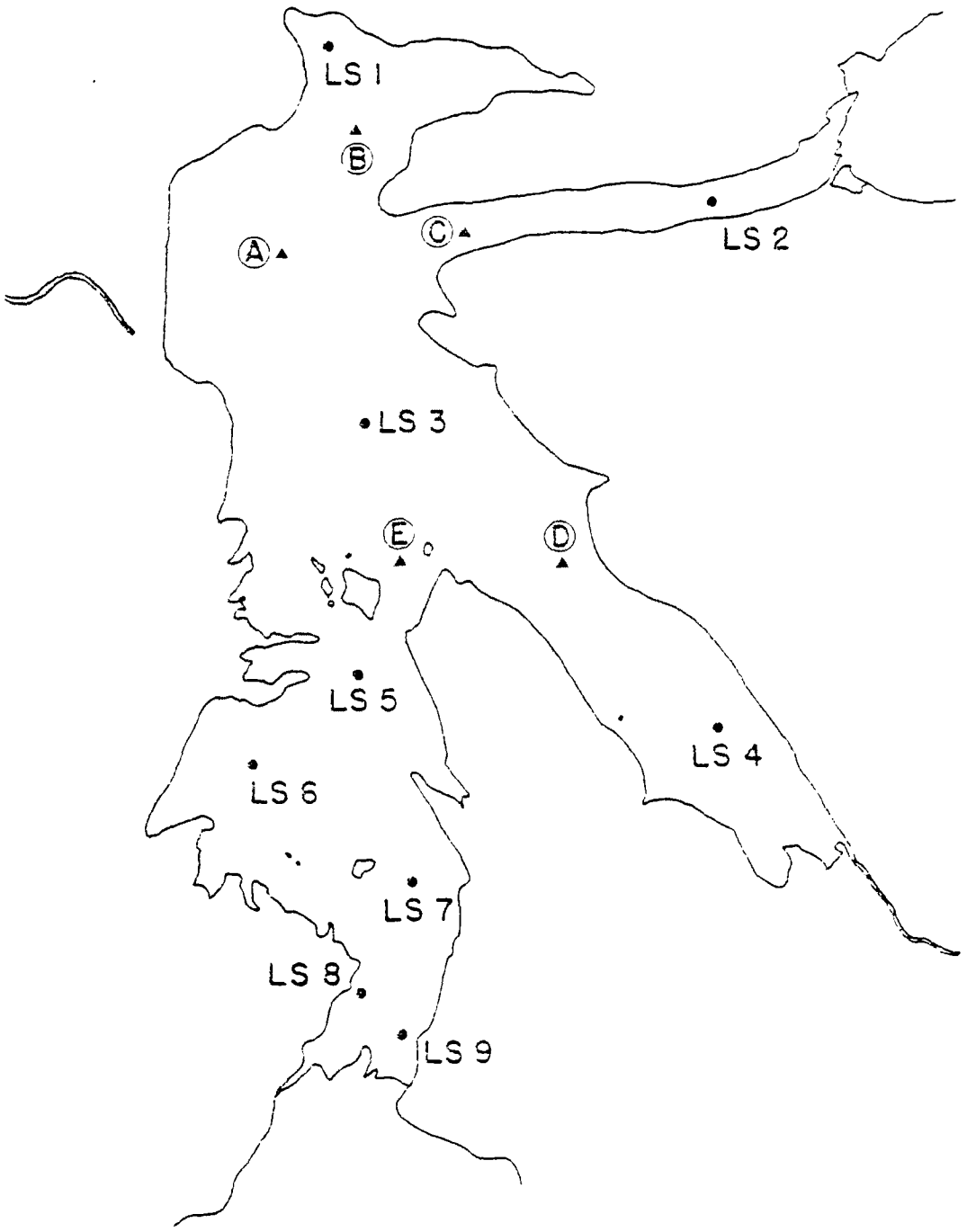


Figure 2. Map of Lake Dillon showing the locations of sampling stations, including the index station (A), the four main stations (B-E), and the survey stations (LS1-LS9).

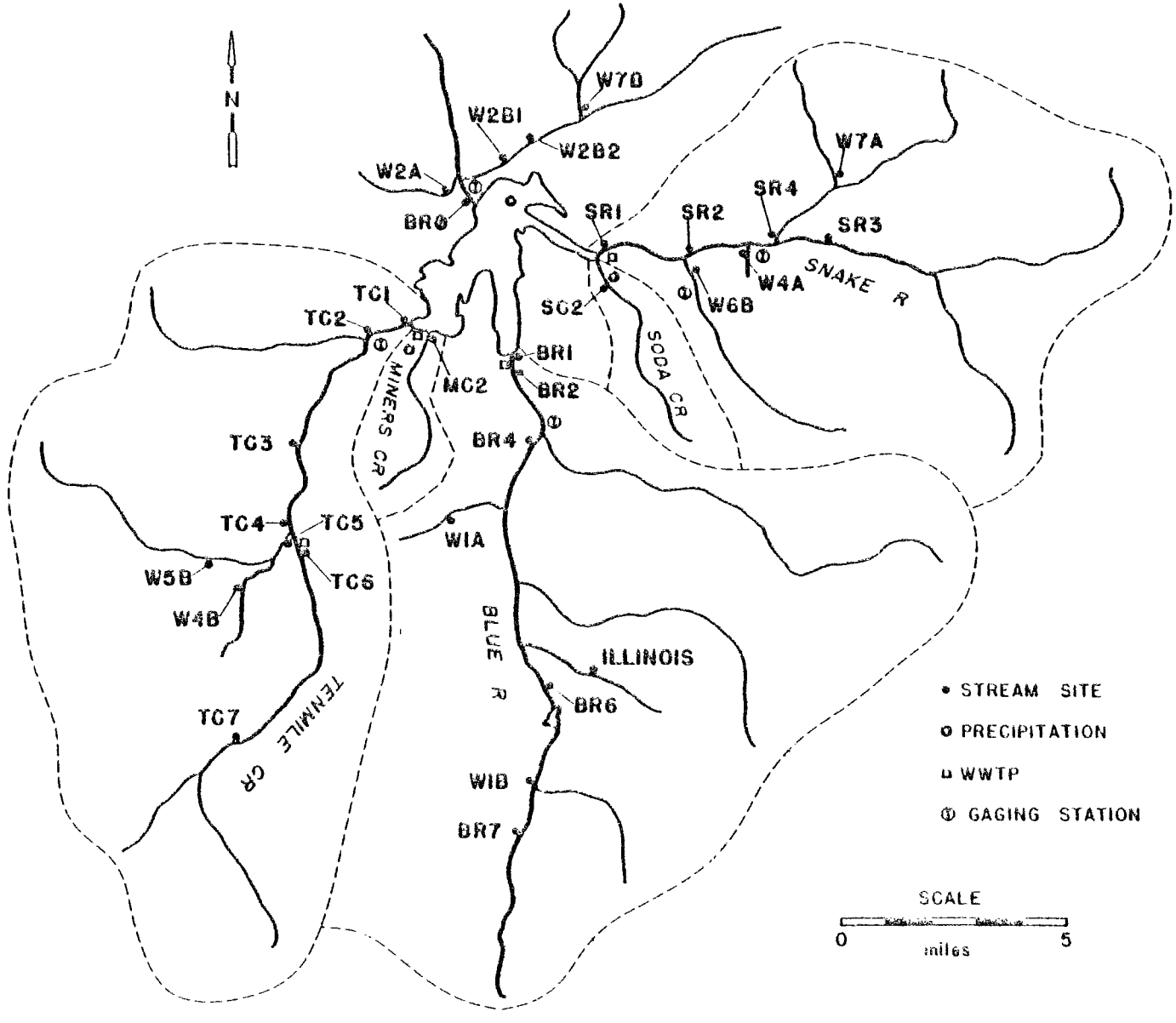


Figure 3. Locations of the stream sampling sites.

Variable	Time Series				Spatial Survey	
	Lake Index Station	Lake Main Stations	Routine Stream	Precipitation	Lake Survey	Stream Survey
Temperature	+	+	+			+
Conductance	+	+	+	+	+	+
Transparency	+	+			+	
Discharge			+			+
Dissolved O_2	+	+				
pH	+	+	+	+		
HCO_3^-	+	+	+	+		
PO_4^{-3}-P	+	+	+	+	+	+
NH_4^+-N	+	+	+	+	+	+
NO_2^--N	+	+	+	+	+	+
NO_3-N	+	+	+	+	+	+
Total Soluble P	+	+	+	+	+	+
Total Soluble N	+	+	+	+	+	+
Particulates	+	+	+	+	+	+
Partic. C	+		+			
Partic. P	+	+	+	+	+	+
Partic. N	+		+			
Chlorophyll a	+	+			+	
Primary Production	+					
Phytoplankton	+					

Table 1. Summary of analytical coverage for the different kinds of samples.

1981

Routine Lake/Stream	Stream Survey	Lake Survey	Diel	Enrichment
23 Feb	26 Jan	09 Feb	14 Jan	20 July
30 March	27 Apr	18 May	16 Nov	24 Aug
13 Apr	05 May	29 June		28 Sept
11 May	18 May	20 July		
26 May	01 June	24 Aug		
08 June	08 June			
22 June	15 June			
13 July	06 July			
27 July	03 Aug			
10 Aug	05 Oct			
31 Aug				
14 Sept				
28 Sept				
12 Oct				
26 Oct				
09 Nov				
23 Nov				
21 Dec*				

1982

Routine Lake/Stream	Stream Survey	Lake Survey	Diel	Enrichment
01 Mar	25 Jan	08 Feb		24 May
15 Mar	10 May	24 May		14 June
05 Apr	31 May	14 June		10 July
19 Apr*	07 June	26 July		26 July
03 May*	14 June	23 Aug		23 Aug
17 May*	21 June			07 Sept
07 June	28 June			18 Oct
21 June	12 July			
06 July	09 Aug			
19 July	11 Oct			
02 Aug				
16 Aug				
07 Sept				
20 Sept				
04 Oct				
18 Oct				
01 Nov				
15 Nov				
06 Dec*				

* Streams only. Lake work judged unsafe.

Table 2. Summary of sampling dates. Precipitation chemistry samples were collected on the same schedule as routine lake and stream samples.

The schedule of sampling for the routine stream data set was the same as for routine lake data collection, as shown in Table 2. Table 1 shows the list of variables analyzed for each sample. In addition to the chemical determinations, discharge was estimated at each sampling station on each date.

(3) Precipitation Chemistry. Bulk precipitation (the combination of dryfall and wet precipitation) was sampled continuously near the Snake River Wastewater Treatment Plant (Figure 3). The sample was collected whenever samples of any other kind were collected on the lake or in the watershed. The analytical coverage is summarized in Table 1.

Spatial Survey Series

The spatial survey series consists of two parts: lake survey and stream survey.

(1) Lake Survey. The lake survey stations include the index station, the four main stations, and nine additional stations scattered over the lake (Figure 2). Analysis for each station was based on an integrated sample taken from the top 5 meters. A bottom sample was also taken at the index station and four main stations. The analytical coverage is summarized in Table 1. The survey was repeated 10 times over the course of the two-year study interval, thus allowing for seasonal changes in spatial heterogeneity over the surface of the lake.

(2) Stream Survey. The stream survey series is based on samples taken at 33 sites in the watershed as shown in Figure 3. As with the

lake survey, all stations were sampled on the same day so that comparisons could be made between stations for a given date. The 33 stations include the stream survey stations already mentioned above and the 11 stations of the representative watershed sampling program (to be discussed below), plus 14 other stations distributed up and down the main stems of the three rivers flowing into Lake Dillon. The sampling dates are shown in Table 2. A few of the stations were not sampled over the entire two years; these will be identified in the course of data analysis.

Special Studies

Special studies include the following: (1) enrichment experiments to determine the identity of the limiting nutrient, (2) diel studies to determine the degree of 24-hour variation in loading, (3) sediment chemistry to determine the size of the nutrient inventory in the lake sediments, (4) a representative watershed sample series designed to provide export coefficients for major land use types in the Lake Dillon watershed, (5) supplemental rain chemistry to provide additional information on atmospheric loading, (6) a groundwater study to show influence of groundwater on loading, and (7) a land use survey.

(1) Enrichment Studies. On 10 different dates, water samples from the upper water column were enclosed in flexible containers and suspended within the lighted zone of the lake. Inorganic phosphorus was added to certain of these containers, inorganic nitrogen to others, and a combination of nitrogen and phosphorus to still others. Another set of containers was left unaltered as a control. Growth responses to the various treatments were determined by changes in the concentration of chlorophyll a within the containers. The objective of this work was

to determine whether the phytoplankton community at a given time was limited by phosphorus, by nitrogen, or by neither of these elements. The enrichment experiments were done more frequently in 1982 than in 1981 after preliminary evidence indicated a late-summer switch between phosphorus and nitrogen limitation. The schedule of enrichment experiments is given in Table 2.

(2) Diel Studies of Nutrient Chemistry. On two occasions the nutrient transport to the mouth of the Blue River, the Snake River, and Tenmile Creek was studied over a 24-hour period. The objective was to determine whether or not significant bias would result if loading of the lake with phosphorus and nitrogen were estimated on the basis of samples taken at a particular time of day or exclusively during the daylight hours.

(3) Sediment Chemistry. Lake sediment samples were taken at the four main stations and at the index station on 26 July 1982. The interstitial waters and particulate component of the sediments were analyzed separately for all phosphorus fractions. The objective of this work was to show whether or not substantial reserves of phosphorus are present in the sediments of Lake Dillon and, if so, whether or not significant amounts of these reserves are present as soluble phosphorus in the interstitial waters.

(4) Representative Watershed Program. Complete analysis of present phosphorus sources and modelling of future phosphorus loading both require sound knowledge of nutrient export coefficients associated with different land uses in the Lake Dillon watershed. Export coefficients cannot be derived from data on high-order streams because

land uses are typically mixed in the watersheds of such streams. For this reason, a set of representative watersheds was chosen, each of which is drained by a low-order stream and whose land use is readily assignable to a particular unmixed category. The land use categories chosen for representation are as follows: undisturbed forested watershed, residential areas served by septic systems, residential areas served by sewer systems, urban areas served by sewers, ski slopes, roads, mining, and interstate highway. Two watersheds were chosen for most types. In some instances, only one watershed for a particular type could be found, and some of the sites were moved after the first year. The discharges and concentrations of nutrients for each of the watersheds were determined for each stream survey sampling date. These data were used to compute the nutrient output per unit area for each of the watersheds.

(5) Supplemental Rain Chemistry. After the 1981 data showed that precipitation would be a significant contribution to the total phosphorus loading of Lake Dillon, it was decided that data on atmospheric phosphorus loading should be taken at several sites. One additional station was set up in the summer of 1982 near the Frisco Wastewater Treatment Plant, and another was placed on the Denver Water Department's raft on the lake surface. The purpose of the latter collector was to show whether the collection stations on shore were unduly influenced by terrestrial dusts.

(6) Groundwater Study. Water and nutrients enter Lake Dillon from the air, as overland runoff, and through groundwater. In view of the geology of the Lake Dillon watershed, groundwater movement was

considered to be probably a minor source of nutrient loading for the lake, but a study was nevertheless made of the groundwater flow in order to verify this impression.

(7) Land Use Survey. Partitioning of the present nutrient loading of the lake according to source requires a thorough quantitative understanding of land use. A land-use study was thus carried out with the objective of specifying the intensity, distribution, and total amount of various land uses in the Lake Dillon watershed.

Methods

Lake Sample Collection

Lakewater samples were always taken with an integrating sampler of the type described by Lewis and Saunders (1979). The sampler consists of a PVC tube 5 m long with closure devices at each end that can be triggered by a messenger. When the sampler is retrieved to the boat, the water is released through a piece of surgical rubber tubing into an integrating chamber, where the contents of the sampler are thoroughly mixed. Water is then drawn from the integrating chamber into sample bottles. The integrating chamber incorporates a baffle that rises with the water and thus prevents contact between the air over the sample and the water. The integrator can thus be used on samples to be analyzed for atmosphere-sensitive variables.

The advantage of the integrating sampler is that it eliminates sampling variance associated with vertical layering. When successive 5-m increments are taken from the top to the bottom of a water column no layers will have been missed. At the same time, since the sampler can be used on successive 5-m increments in the water column, information on gross vertical structure of the water column can be obtained with the sampler.

Samples of Lake Dillon were protected from heat and from direct exposure to sunlight after they were bottled in the field. Special precautions were taken to protect the chlorophyll samples from exposure to light. Sample bottles were always rinsed copiously with deionized water and dried at the end of each sampling period. A field log was

maintained of sampling conditions and of problems that arose during sampling.

Streamwater Sampling

Streamwater samples were collected by insertion of a wide-mouth plastic bottle just below the water surface in a reach of the stream where the flow was fast across the entire cross section. Care was taken not to stir up stream sediments in front of the sample.

Filtration of Water Samples

Filtration of water samples was carried out at the Snake River Wastewater Treatment Plant immediately after sample collection. Each sample was shaken prior to filtration and poured onto a filter tower over a Whatman GF/C glass fiber paper (47 mm diameter). The effective pore size of this paper is about 2 um. The purpose of the immediate filtration was to reduce the biological activity of the sample to a minimal level without the addition of any preservatives that might affect sample chemistry. The filtered samples were then returned to Boulder where they were stored under refrigeration until analyzed. Analysis of soluble constituents commenced the morning after sample collection. Analysis of labile constituents was completed the day after sampling.

Analysis of Particulates

One filter through which a known amount of water had been passed was used for the analysis of particulate phosphorus (see below), and,

if chlorophyll was to be measured, a second filter was used for this. The amount of water that could be filtered by a single filter depended on the amount of particulates in the sample, but typically fell between 100 and 1000 ml.

Another filter for each sample was dried to constant weight at 60°C and weighed with an analytical balance. A measured amount of water was poured over this filter, after which the filter was redried to constant weight at 60°C in an oven, cooled in a desiccator to room temperature and weighed to the nearest .01 mg on an analytical balance. The difference between the initial and final weights was recorded as the amount of particulate material on the filter, which was subsequently converted to mg/l total particulates. The filter was then saved for elemental analysis with a Carlo Erba Model 1102, which gave the amount of carbon and nitrogen on the basis of chromatography following automated high-temperature combustion. Because the amount of nitrogen in the particulate fraction was a very small proportion of the total nitrogen, especially for the streams, the elemental analysis was performed only on the samples from the river mouths, the lake index station, and selected additional samples from various locations.

Particulate P

One filter from each sample was analyzed for particulate phosphorus following the method of Solorzano and Sharp (1980a). This relatively new method is an especially sensitive and reliable alternative to older methods, which often present the problem of poor or uncertain recovery. The Solorzano and Sharp method relies on

decomposition of organic phosphorus compounds by pyrolysis. The filters are dried with magnesium sulfate, heated at 450-500°C for one to two hours, treated with hydrochloric acid to hydrolyze polyphosphates, and then analyzed for orthophosphate by the standard orthophosphate procedure (see below). When used in combination with a high-quality spectrophotometer, the method allows quantification of very low amounts of particulate phosphorus, thus eliminating problems arising from the interpretation of data that include many values below detection limits. Quality assurance information on this and other methods is given at the end of this section.

Soluble Reactive Phosphorus

Soluble reactive phosphorus was measured by the ascorbic acid-molybdate method (Murphy and Riley 1962). This is the most widely used method for low levels of soluble inorganic phosphorus. When combined with long-pathlength spectrophotometry, soluble reactive phosphorus as low as 1 ug/l can be measured. Subtraction of a turbidity blank at 885 nm is sometimes necessary but proved of negligible importance on the filtered samples of the Dillon study. Although the chemical species measured by this test will be referred to as PO_4-P, it is more properly characterized as soluble reactive phosphorus, since the test is sensitive not only to orthophosphate but also to organic phosphorus compounds of low molecular weight (Rigler 1968, Levine and Schindler 1980).

Total Soluble Phosphorus

A pyrolysis method similar to that used for particulate phosphorus was used for total soluble phosphorus (Solorzano and Sharp 1980a). For this method, a portion of the filtered sample is evaporated to dryness with magnesium sulfate. The sample is then combusted at 450-500°C and the resulting polyphosphates are hydrolyzed with hydrochloric acid. This is followed by determination of soluble reactive phosphorus by the molybdate method. Subtraction of the soluble reactive phosphorus from total soluble phosphorus gives soluble organic phosphorus. Addition of total soluble phosphorus to particulate phosphorus gives total phosphorus.

Ammonium

A modification of Solorzano's phenolhypochlorite method (1969) was used for ammonium (see Grasshoff 1979). This is a very sensitive technique capable of detecting as little as 1 ug/l NH_4-N with appropriate spectrophotometry. Hypochlorite is added to ammonium in alkaline solution. With the addition of phenol and nitroprusside, indophenol blue is formed. The indophenol blue concentration is determined spectrophotometrically.

Nitrite and Nitrate

The widely used diazotization method originally described by Bendschneider and Robinson (1952) was used for nitrite. The combination of nitrate plus nitrite was determined by passage of a portion of filtered sample through a reduction column containing a

cadmium-copper couple, after addition of a buffer. Nitrate is thus converted to nitrite. The amount of nitrite in the sample after reduction is the sum of nitrate and nitrite in the sample before reduction. Subtraction of the nitrite originally in the sample yields the amount of nitrate. The method is extremely sensitive; detection is below 1 ug/l of N.

After May 1982, all nitrate measurements were made by ion chromatography rather than wet chemistry. Ion chromatography was carried out with a Dionex Model 2110 chromatograph using a bicarbonate eluent. The automated ion chromatographic method is highly sensitive and offers the advantage of lower variance in measurements because of reduced sample handling.

Total Soluble Nitrogen

A new method outlined by Solorzano and Sharp (1980b) was used for total soluble nitrogen. Analysis of total soluble nitrogen when soluble organic nitrogen is present in small amounts has traditionally been unsatisfactory because of low sensitivity or poor recovery. The new method, which emphasizes better control over pH and oxidizing conditions, reduces these problems. The method is based on oxidation of soluble organic nitrogen by potassium persulfate in an autoclave. This converts all soluble nitrogen compounds to nitrate. Nitrate is subsequently analyzed by the reduction method described above. This produces an estimate of total soluble nitrogen. Soluble organic nitrogen can be obtained by subtraction of the ammonium, nitrate, and nitrite obtained in the other analyses. For the low levels of soluble

organic nitrogen present in certain circumstances for the Dillon study, recrystallization of reagents was necessary to improve the precision of the test.

pH, Alkalinity, and Oxygen

pH was determined with a Radiometer M29 pH meter and combination electrode on an unstirred sample. Alkalinity was determined by titration of a 100-ml aliquot to an endpoint of pH 4.4 with N/44 H_2SO_4.

Dissolved oxygen was determined by the azide modification of the Winkler method (APHA 1975).

Major Anions and Cations

Major ions not of direct nutritional significance were not determined routinely. However, a few determinations were made, and in these instances the anions chloride and sulfate were determined by ion chromatography (Dionex 2110) and the cations were determined by atomic absorption spectrophotometry (Varian AA6) after addition of lanthanum oxide and cesium chloride.

Chlorophyll a

Chlorophyll *a* was determined after hot methanol extraction. The basic procedure is that described by Talling (1969) but has been modified some in accordance with recommendations presented in the recent comprehensive methods evaluation edited by Rai (1980). The sample is filtered just after collection onto Whatman GF/C paper.

$MgCO_3$ suspension (0.2 ml) is added to the filter. The filter is then placed in a screwcap test tube with 40 ml of 90% methanol, heated to 65°C, and allowed to boil gently for 30-60 seconds. The tubes are then capped again and stored in the dark for 6 to 24 hours. The absorbance is measured at 665 and 750 nm relative to 90% methanol. The absorbance is then computed as follows: $12.9 \times (E_{665}-E_{750}) \times$ (volume solvent in ml/volume filtered in liters) x 1/pathlength in cm. The coefficient of this equation is lower than Talling's original coefficient (13.9). The lower coefficient brings the calculation into line with recent information on the specific absorbance of chlorophyll in methanol as reported in the summary by Marker et al. (1980). The data were also corrected for a problem that has not been brought out in the literature but which was discovered as a result of a seven-laboratory methods study coordinated by the University of Missouri in which we participated. Certain glass filters release a binder or fine colloidal fragment that does not centrifuge readily. These particles differentially increase the absorbance at 665 and 750 nm. Tests on our own filters showed that the fragments were present and increased the absorbance at 665 nm by 21.5% more than the absorbance at 750 nm. All of the calculations were corrected for this differential. The resulting difference in chlorophyll values was not very great in percentage terms, however.

The methanol method was selected because recent studies have shown its extraction capabilities to be better than those of any other method (Riemann 1980). We did a comparison of methanol and acetone specifically for Lake Dillon on 1 March 1982. Table 3 shows the

results. As expected from the literature, methanol is more efficient than acetone for samples that are steeped but not ground in a homogenizer. The differential averages about 15%. As a rule of thumb, values for Dillon should be divided by 1.15 if they are to be compared with future or past acetone-based values for Dillon derived from unground samples. Ground acetone samples would be very close to the methanol values, as shown by the recent interlaboratory study in which we participated.

Site	Chlorophyll a - ug/l		Ratio M/A
	Methanol	Acetone	
Tenmile arm	6.6	5.6	1.18
Index, 0-5 m	5.9	5.4	1.10
Index, 5-10 m	3.5	3.35	1.05
Index, 10-15 m	3.1	2.4	1.28

Table 3. Comparison of extraction by acetone and methanol for Lake Dillon samples on 1 March 1982 (samples were steeped, not ground).

Primary Production

Primary production (photosynthetic carbon fixation) was determined by the C-14 uptake method at the index station on each routine lake sampling date. The method was carried out essentially as described by Lewis (1974). Although the method is operationally somewhat complex, the principle is very simple. Water is removed from a specific depth, inoculated with C-14, and returned for a few hours to the depth from which it came. The sample is then retrieved, passed through a filter, and the filter is washed to remove any soluble C-14 label. The remaining C-14 label on the filter is that which has been taken up

photosynthetically by the algae during the period of incubation. From the total amount of inorganic carbon available in relation to the amount of C-14 added, one can calculate the total carbon fixation on the basis of the C-14 fixation.

Depths of sampling and incubation for the Dillon work were as follows: 0, 0.5, 1, 2, 5, and 10 m. Greater depths are of little interest from the viewpoint of carbon fixation because there is insufficient light to support much photosynthesis.

At each depth, two 125-ml transparent glass bottles were suspended ("light bottles"). For each depth one bottle was filled that was completely darkened on the outside ("dark bottle"). Since temperatures were nearly uniform over the mixed layer, all of the dark bottles were incubated together at a depth of 5 m. The dark bottles served as controls for nonphotosynthetic inorganic carbon uptake, although this was typically small. All three bottles were inoculated with 1 ml of C-14 in the form of sodium bicarbonate solution. The activity of the inoculum was 3 uCi/ml. The samples were suspended in situ for a timed period of 2.5-4 hours, typically between 10 a.m. and 2 p.m. On the day of the incubation, solar radiation reaching lake surface was recorded continuously by a Belfort pyrheliometer so that the amount of carbon fixation could be related to the amount of light.

At the end of the incubation, the bottles were removed from the lake and placed in a dark box so that no further photosynthesis could occur. The samples were then transported to the Snake River Wastewater Treatment Plant for filtration. As each bottle was removed from the dark box, it was shaken and a 100-ml subsample was withdrawn for

filtration. This subsample was placed onto a filter tower over a Millipore HA filter (effective pore size 0.45 um). The sample was filtered under gentle vacuum and the filter was washed two times with distilled water. The filter was then placed in a liquid scintillation vial containing 12 ml of scintillation cocktail (Aquasol II) and capped. The filtered water, still containing small amounts of C-14, was returned to Boulder for proper disposal through a regulated radioactive waste disposal system. The scintillation vials containing the labelled algae were then counted on a Beckman LS3133T scintillation counter. Counting efficiency was determined by addition of standard amounts of C-14-labelled toluene to selected samples. Counting efficiencies were very stable in the vicinity of 80 to 90%.

The total activity in each scintillation vial was computed from the observed activity and the machine efficiency after correction for machine blank (typically a very small correction in the vicinity of 50 cpm). The available inorganic carbon in the sample was determined from the pH and alkalinity of the sample and the sample temperature according to the tables of Saunders, Trama, and Bachmann (1962). Carbon fixation in each bottle was computed from the following information: (1) the activity of the sample, (2) the volume of water filtered, (3) the available carbon in the sample, (4) the isotope discrimination factor (1.06), and (5) the amount of C-14 added to the sample. The dark bottle fixation for a given depth was then subtracted from the light bottle fixation for the same depth to yield the observed photosynthetic carbon fixation for the incubation period

(mgC/m^3/incubation period). This is considered to be a good estimate of gross photosynthesis (Harris 1978).

From the gross photosynthesis at each depth, a vertical profile could be made of photosynthesis at the index station over the incubation period. A profile of this type was drawn on a piece of graph paper for each one of the incubation dates. The amount of fixation over all depths was obtained by integration of the area under this volume-specific fixation curve. This yielded an estimate of gross fixation per unit area (mg carbon/m^2/incubation period). Because the incubation periods were of slightly different lengths and the sunlight conditions were different from one day to the next, fixation per unit area was converted to fixation per day and to fixation per unit sunlight as follows. The amount of sunlight striking the surface of the lake over the incubation period was obtained from the pyrneliometer sunlight trace by planimetric integration. The fixation per unit area for the incubation period was divided by the amount of incident sunlight during the incubation period to yield the fixation rate/unit area/unit of sunlight (mg carbon/m^2/langley). The sunlight striking the surface of the lake over the entire day was then obtained by planimetric integration from the pyrheliometer charts. The total amount of sunlight for the entire day was multiplied by the fixation per unit of light to obtain the total estimated fixation per unit area for the day (mg carbon/m^2/day). This last calculation assumes that, on a given day, the photosynthesis below a unit area is proportionate to the amount of sunlight striking the surface. This is not precisely correct (Talling 1971), but is sufficiently true for present purposes.

Because photosynthesis has been measured by the C-14 method in a wide variety of lakes of different trophic status, the C-14 measurements provide a method for comparing the photosynthetic activity of Lake Dillon with that of many other lakes.

Temperature, Conductance, and Transparency

Temperature profiles were measured with a YSI thermistor and submersible probe. Conductance was determined in the laboratory with a Labline Model MC-1 meter standardized against KCl. All conductances were temperature corrected to 25°C.

Transparency of the lake was determined by secchi disk, and sometimes also by submersible photometer (Whitney-Montedoro with submersible selenium photocell and opal filter). The photometer readings were taken from the surface to the limit of detection in increments of 60 cm. The secchi disk is a standard white plate, 15 cm in diameter, which is lowered to the limit of visibility. In Lake Dillon, the secchi depth was on the average equal to the depth of 19% surface irradiance as measured by light meter (standard deviation, 5.7%).

Discharge Measurements

Discharge was estimated at stream sampling stations at the time of sampling. In addition, continuous discharge measurements were obtained from four U.S.G.S. gauging stations. The U.S.G.S. gauging station measurements serve two purposes: (1) they allow us to check our own methods of measuring discharge, and (2) they are the main basis for

estimating total amount of water entering the lake from each of the three major rivers. The gauging station locations are shown in Figure 3.

The Snake River gauge is well above the mouth of the Snake River. In order to estimate the amount of water entering the lake from the Snake River, we added the values from the U.S.G.S. gauges on the Snake River and on Keystone Gulch and then made a small correction for the remaining ungauged portion of this drainage. Similar corrections were made for the other gauge readings to account for small areas below gauges (Figure 3). Also for the Blue River we added the sewage treatment plant discharge to the gauge reading. The sewage treatment plant discharge was obtained from plant operators on sampling days.

We made discharge measurements when we took stream samples in ungauged reaches of stream for the routine stream series and for the stream survey series. Some of the stream sampling sites were sufficiently close to culverts or bridges that the discharge could be estimated by the use of standard tables (Chow 1959, BPR 1965). This was done for about eight sites in 1982 and slightly more in 1981. For most of the stream sites, standard tables could not be used because of the irregularity of the channel. Consequently, estimates were made by the method of cross-sectional current profiles. The depth profile of the stream channel was mapped at a location where the bed was likely to be stable. The cross-sectional area of the stream as shown by this map was divided into several segments (three to eight, depending on the width and irregularity of the stream). Within each segment, a current meter measurement was taken with a Gurley Pygmy current meter at a

depth equal to 60% of the distance from the surface to the stream bed. This is the depth at which the velocity is considered to be about average for a segment. The discharge of each segment was then computed on the basis of its area and its velocity of flow. The discharges for the different segments were added to produce the estimate of total discharge for the site. The current meter method is compared with the U.S.G.S. data in Table 4. The deviations fall within the daily range of fluctuations for a given site and are thus as close as could be expected.

Station	Number of Determinations	U.S.G.S. Mean, cfs	Current Meter Mean, cfs	Diff. %
Tenmile Cr.	20	223	216	-3
Blue River	21	173	167	-4
Snake River	21	145	131	-11

Table 4. Comparison of discharge measurements made by U.S.G.S. and near the same location by the current meter method used in the field sampling program (1982).

After a range of discharges had been obtained at a given site, a depth-discharge relationship was established in the form of a rating curve. After the rating curve was available, discharge estimates were sometimes based on the curve rather than current meter measurements.

The Denver Water Department provided daily estimates of water leaving the lake through the Roberts Tunnel and by the Blue River, and daily estimates of water entering the lake.

Precipitation Sampling

Atmospheric transport of nutrients was estimated at the Snake River Wastewater Treatment Plant site by means of a large bulk precipitation chemistry sampler mounted on a tower 3 m above the ground. The design of the sampler is given in detail by Lewis and Grant (1978; Figure 4). The opening of the sampler is 0.20 m^2, which is sufficiently large to supply measurable amounts of phosphorus and nitrogen over weekly or biweekly collection periods at any time of the year, even when there is no wet precipitation. The collecting surface consists of an inverted plexiglas pyramid which funnels materials into a narrow opening that leads to an insulated wooden box. Wet precipitation flows through the opening into a plastic receiving funnel. The receiving funnel is covered with a fine nylon screen so that insects reaching the funnel can crawl from it. The funnel is connected to a surgical rubber hose that is looped in order to form a vapor barrier, and the surgical rubber hose is connected to a hard glass receiving vessel with a collapsible plastic overflow container. The box housing the containers is heated by means of thermostatically controlled light bulbs. The heat is sufficient to melt snow that strikes the surface of the collector during the winter so that these contributions are not blown from the sampler.

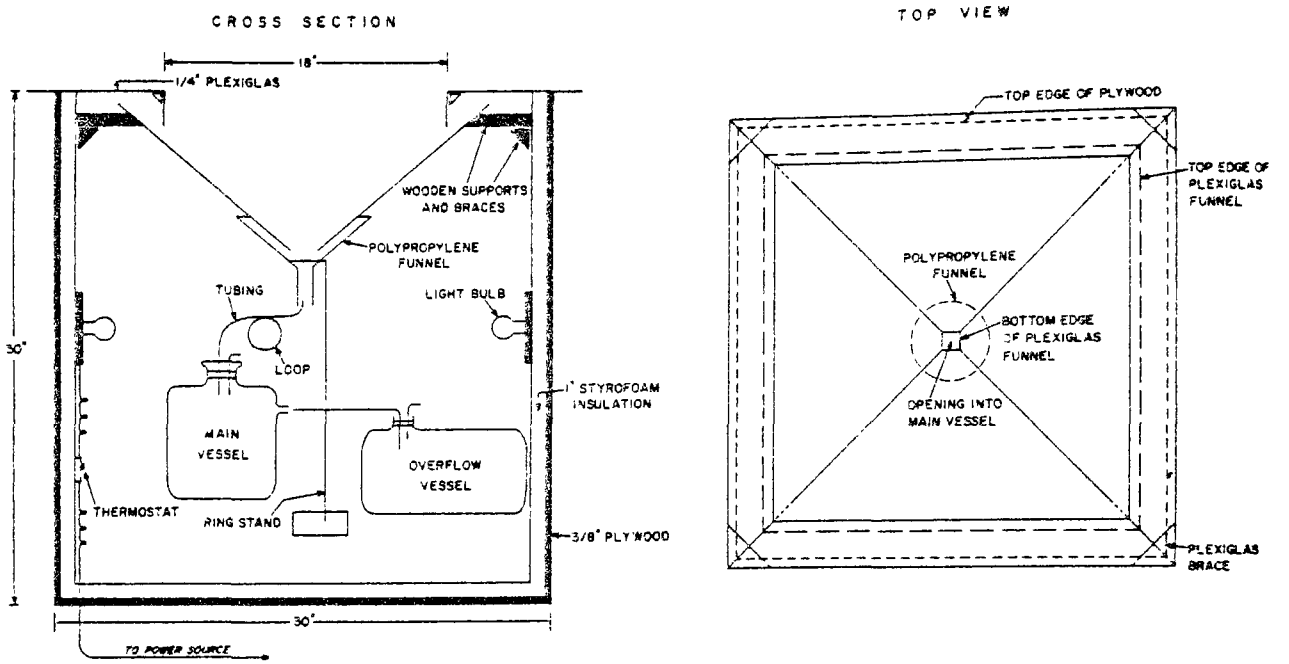


Figure 4. Design of precipitation chemistry sampler (from Lewis and Grant 1978).

The samples were collected at weekly to biweekly intervals depending on the schedule of other sampling programs. On each occasion, the collection vessels were removed from the box and mixed in a common container. An aliquot of 100 ml was removed for determination of pH and alkalinity. The surgical rubber hose was then hooked to a rinse container and 500 ml of deionized water was poured over the collecting surface, thus rinsing the collecting surface, collecting funnel, and tubing. After the volume of the rinse and the volume of wet precipitation were noted, the two were mixed together. The water was then analyzed according to the protocol for lake samples. The data were corrected for the amount of deionized water added and for the removal of the 100-ml aliquot for pH and alkalinity determination.

Concentrations were converted to loading rates on the basis of the area of the collecting surface and the elapsed time since the last sample.

Special Studies

The protocol for each of the special studies is described along with the results in later parts of this report. All chemical analyses and sampling were carried out by the methods described above unless otherwise indicated.

Quality Assurance

The quality assurance program is intended to provide quantitative information on the reliability and accuracy of analyses for major variables in the Lake Dillon Study. Three types of information are included: duplicates, spikes, and EPA standards.

(1) Duplicates. Approximately 10% of the analyses were duplicated. Table 5 summarizes the resulting information. The degree of variability is most easily evaluated from the ratio of standard deviation to mean (coefficient of variation, expressed as percent), which is shown in the last column of Table 5. The degree of variation for all variables is acceptable, and for most it is exceedingly low and thus warrants no discussion. Estimation of total particulates stands out as having the highest relative variation. We attribute this largely to irregularities in the distribution of particles within the stream (stream duplicates are based on two samples taken sequentially) rather than to variation in drying or weighing. This conclusion is supported by a difference in replicate variability for particulates in

Variable	Number of Duplicates	Mean		Standard Deviation	Coefficient of* Variation, %
Oxygen	80	7.59	mg/l	0.095	1.3
Conductance	188	164.0	umho/cm	4.4	2.7
pH	110	7.48		0.043	0.6
Alkalinity	115	36.8	mg/l	0.76	2.1
Chlorophyll a	101	7.29	ug/l	0.52	7.1
Total Particulates	212	3.15	mg/l	0.56	17.8
NO_2-N	235	6.76	ug/l	0.37	5.5
NO_3-N	308	151.61	ug/l	5.91	3.9
NH_4-N	283	91.63	ug/l	5.59	6.1
Total Sol. N	143	618.0	ug/l	78.0	12.6
PO_4-P	240	4.56	ug/l	0.40	8.8
Total Sol. P	303	7.45	ug/l	0.56	7.5
Part. P	224	4.95	ug/l	0.58	11.6

* Ratio of standard deviation to mean, times 100.

Table 5. Summary of statistics on duplicate analyses.

stream samples and lake samples. Lake samples, which come from the same integrator, and which are more likely to have an evenly dispersed particulate load of fine grain, show lower variability, whereas stream samples show higher variability than the average. Even so, determination of the particulate load of streams will not be affected in any important way by this variability because of the large number of samples taken over time at each site. Replicate variability in particulate load is reflected to some degree in the particulate phosphorus analyses, as expected.

PO_4-P shows slightly higher relative variation of duplicates than most other constituents. This is explained by the extremely low levels of PO_4-P characteristic of the entire system (note that the average PO_4-P concentration for all duplicates is just slightly over 3 ug/l). Since the relative variation of a determination is to some degree affected by the proximity of the analysis to the analytical limit for the test, the slightly higher relative variation for PO_4-P is not unexpected.

Total soluble nitrogen also shows slightly higher variability than most other variables. We attribute this mostly to the pH sensitivity of the test, but do not consider it important in the interpretation of results.

(2) Spikes. Table 6 summarizes the percentage recovery of spikes added to samples throughout the course of the study. The recovery is typically quite close to 100%. Only the total soluble phosphorus and nitrate recovery warrant discussion. For total soluble phosphorus, two types of spikes were used: dibasic inorganic phosphorus (K_2HPO_4)

and ATP. The recovery of K_2HPO_4 was lower than for ATP, which was characteristically recovered near the 100% level. Since K_2HPO_4 is not expected in nature, whereas organic phosphorus compounds more similar to ATP are expected in nature, we believe that the percentage recovery for ATP is more representative of the recovery of natural organic phosphorus by the test. In any event, the difference is small and thus does not warrant correction.

Variable	Number of Spikes	Mean % Recovery
PO_4-P	213	91.44
Total Sol. P	186	87.13
NO_2-N	187	104.7
NO_3-N	192	91.34
NH_4-N	196	96.7
Total Sol. N	113	94.29

Table 6. Summary of information on spikes for quality assurance.

For nitrate, the efficiency of recovery is about 90%. Once again, we feel that this percentage slightly underestimates actual recovery of nitrates in unspiked samples. Addition of the spike could reduce the efficiency of the reduction column by increasing the load on it and in this way leads to a small loss in recovery that is not likely to be characteristic of real samples. Thus we do not believe that the data should be corrected. The effect of any correction would be very minor, however.

(3) EPA Standards. Analysis of EPA standards showed good agreement of observed and expected values (Table 7).

	EPA	Detected Value
NH_4-N, ug/l	19	19.3
NO_3-N, ug/l	31	33.4
PO_4-P, ug/l	3.1	3.0
pH	7.4	7.7

Table 7. Data on EPA standards (raw standards were diluted to appropriate range).

Physical Variables and Major Ion Chemistry of the Lake

Geology

The geology of the Lake Dillon region need be covered here only briefly, since it has been well documented in connection with the construction of the Roberts Tunnel and the exploitation of mineral resources in the watershed. For details, the reader is referred to the publication of Wahlstrom and Hornback (1962), which contains references to earlier studies.

Lake Dillon lies on a bed of stream and glacial deposits of Quaternary age, especially in the old riverbed, and on sedimentary rocks of Mesozoic age. Except near the valley bottoms, the Snake River and Straight Creek drainages and small adjoining portions of the Blue River drainage are composed of igneous and metamorphic rock of Precambrian age. Nearer to the riverbed at lower elevations the Snake River drainage contains stream and glacial deposits of Quaternary age. In the upper Snake and Peru Creek there is an exposure of Tertiary quartzes (monzonite and monzonite porphyry).

There is a sharp break in the geology along a line that runs north and south to the east of the Blue River. This line represents the Williams Range Thrust Fault. To the west of this fault, we find Mesozoic sedimentary rocks and roughly equal exposures of Tertiary quartzes and felsite porphyry. Along the valley bottoms there are also extensive deposits of Quaternary age.

Morphometry

A bathymetric map of the lake is shown in Figure 5. It is useful to compare the central portion of the lake to the lake arms because differences between the arms and the main body of the lake will be of some interest throughout the course of this report. Dillon Bay is relatively deep and opens broadly onto the main body of the lake. The Snake River arm is narrowest but has a greater mean depth than the other arms. The Blue River arm is shallow in its upstream half, where sediments are exposed at times of the year when drawdown is most severe. The Blue River arm opens broadly onto the main body of the lake, and this promotes exchange with the center of the lake. The Tenmile Creek arm has the greatest extent of shallow water. Shallow water is found in the recessed area to the west and in the upstream end of the arm. Considerable areas of sediment surface are exposed during periods of extensive drawdown. Furthermore, the circulation between the main body of the lake and the Tenmile Creek arm could be restricted by the presence of islands near the mouth of the arm. Clearly the greatest opportunity for biological or chemical divergence from the main body of the lake is in this arm.

Figure 6 shows the depth/volume relationship, Figure 7 shows the hypsographic curve, and Table 8 summarizes morphometric statistics related to water level. Although a small amount of storage above spillway level is possible, spillway level is for practical purposes the maximum lake level, so other depths are expressed in relation to spillway level. The Roberts Tunnel diverts water from the lake through an intake pipe 52 m below the spillway. The outlet to the Blue River

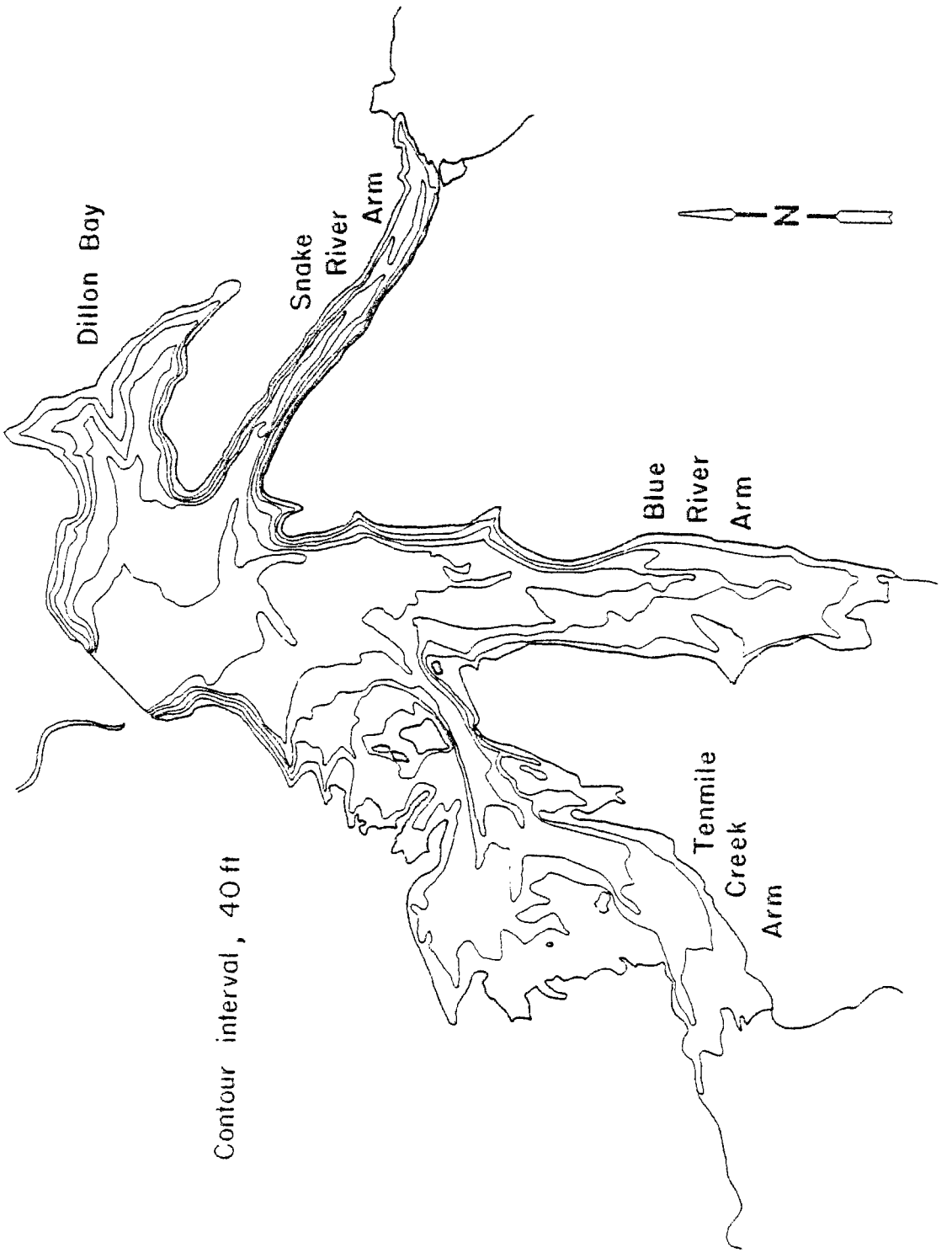


Figure 5. Bathymetric map of Lake Dillon.

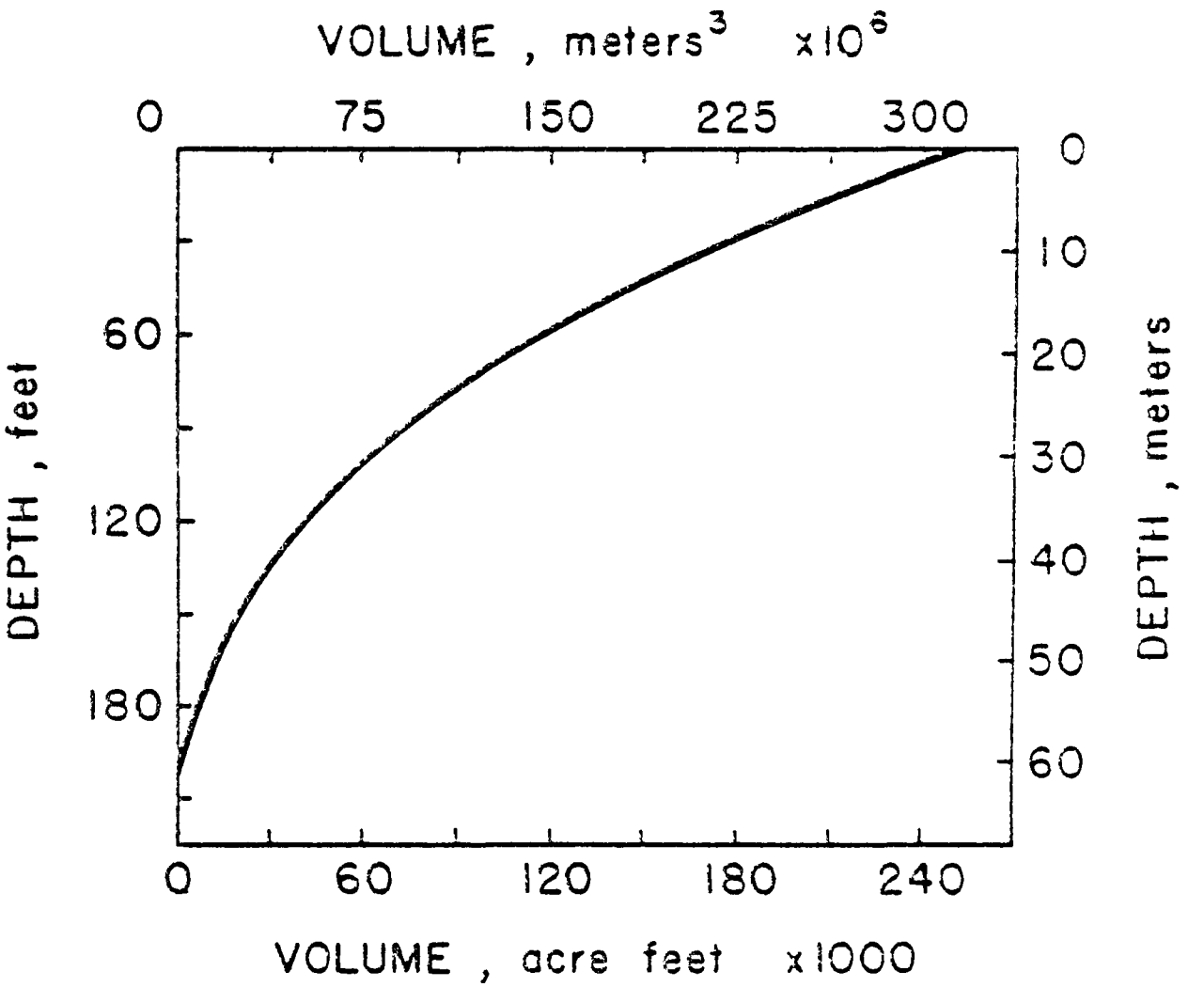


Figure 6. Depth-volume curve for Lake Dillon.

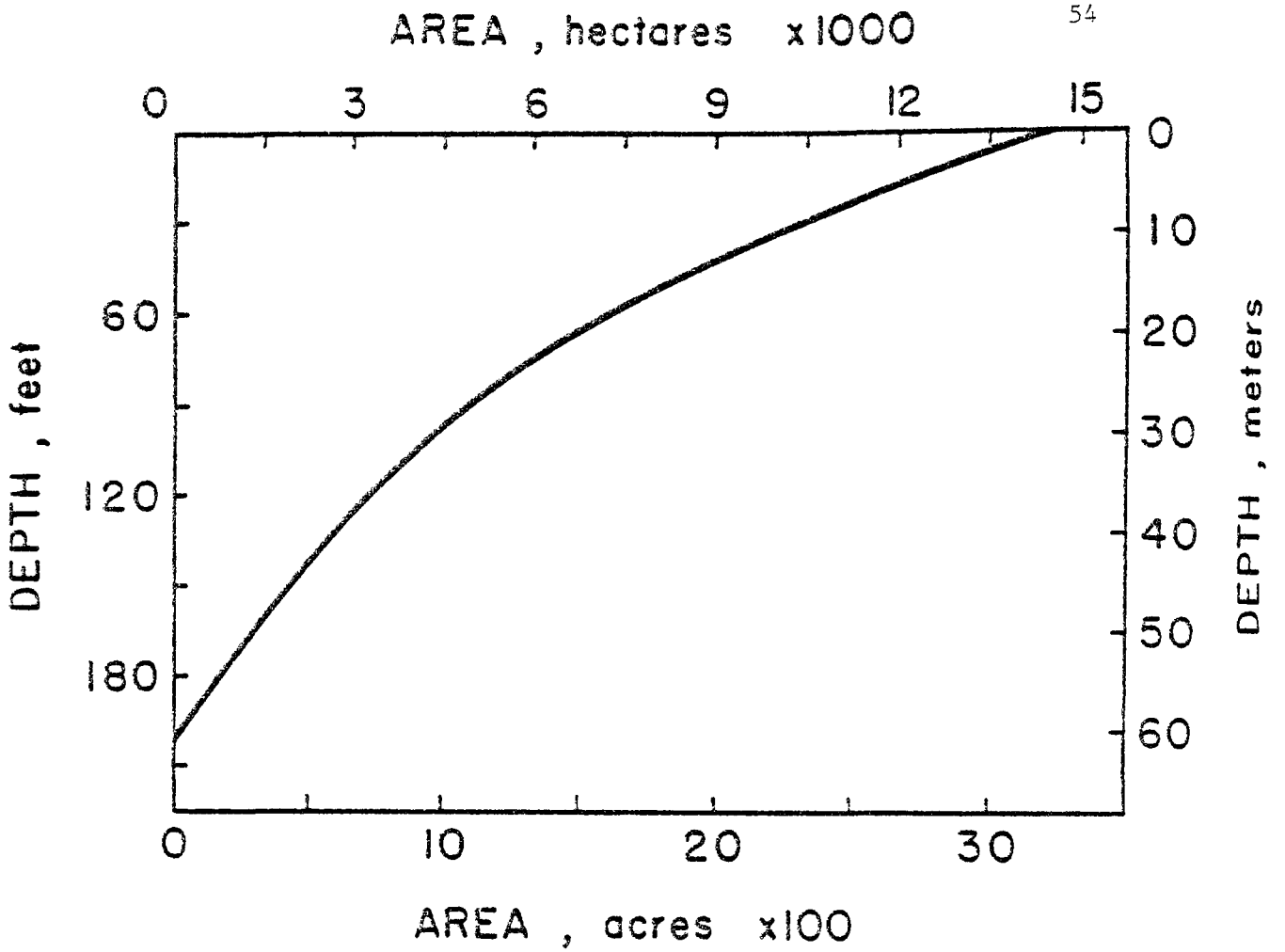


Figure 7. Hypsographic curve for Lake Dillon.

is slightly below this, at a depth of 57 m when the water is at spillway level. The original maximum depth of the lake was 68 m, which would allow for just over 1% dead storage. The current maximum depth is probably somewhat less than this. The greatest maximum depth measured during the course of our study was 61 m but, because of the presence of the old river channel, some deeper spots are probably still present.

	Sea Level Reference		Spillway Reference	
	feet	meters	feet	meters
Spillway	9017	2748	0	0
Maximum storage (flood)	9025	2751	+8	+2.4
Roberts Tunnel	8846	2696	-171	-52
Blue River outlet	8829	2691	-188	-57
Original maximum depth	8795	2681	-222	-68

Table 8. Morphometric statistics for Lake Dillon. Source: engineering documents provided by the Denver Water Department.

Table 9 summarizes the lake volume, area, length of shoreline, and mean depth when the water is at spillway level.

	English	Metric	
Lake Volume	262,000	acre-feet	0.323	km^3
Lake Area	3300	acres	13.35	km^2
Length of shore	24.5	miles	39.4	km
Mean depth	79.3	feet	24.1	m
Watershed Area	212,900	acres	85,160	ha

Table 9. Morphometric statistics on Lake Dillon, assuming water at spillway level. Source: Denver Water Department.

Hydrology

We summarize here only the changes in lake level and total flow into and out of the lake. A more detailed consideration of the hydrology of individual watersheds will be taken up later in connection with the analysis of nutrient income to the lake.

Figure 8 shows the total calculated inflow to the lake for the study period as estimated by the Denver Water Department. There are differences in the actual inflow and the calculated inflow because the calculated inflow is estimated as the sum of the outflow (which is measured) and the change in water volume (which is estimated from the depth/volume relationship). Errors in equating calculated inflow with actual inflow are not of concern here but will be discussed in another section. It will also be shown that groundwater inflow is at most 5-10% of the total inflow.

The calculated inflow to the lake for 1982 was about twice the amount for 1981. In both years the peak inflow occurred in the month of June, but the distribution of inflow around the peak was not quite the same in the two years. In 1981 the total inflow was quite symmetrical around the month of June. In 1982, high flow extended well into July. In both water years the months November through March showed a stable minimum inflow of about 2,000 to 4,000 acre-feet per month.

Figure 9 shows the total calculated inflow since 1963 (calendar years). The purpose of this figure is to put the two years of the present study into perspective hydrologically. For the period of record, 1981 was well below the average (only 1963 was lower) and 1982

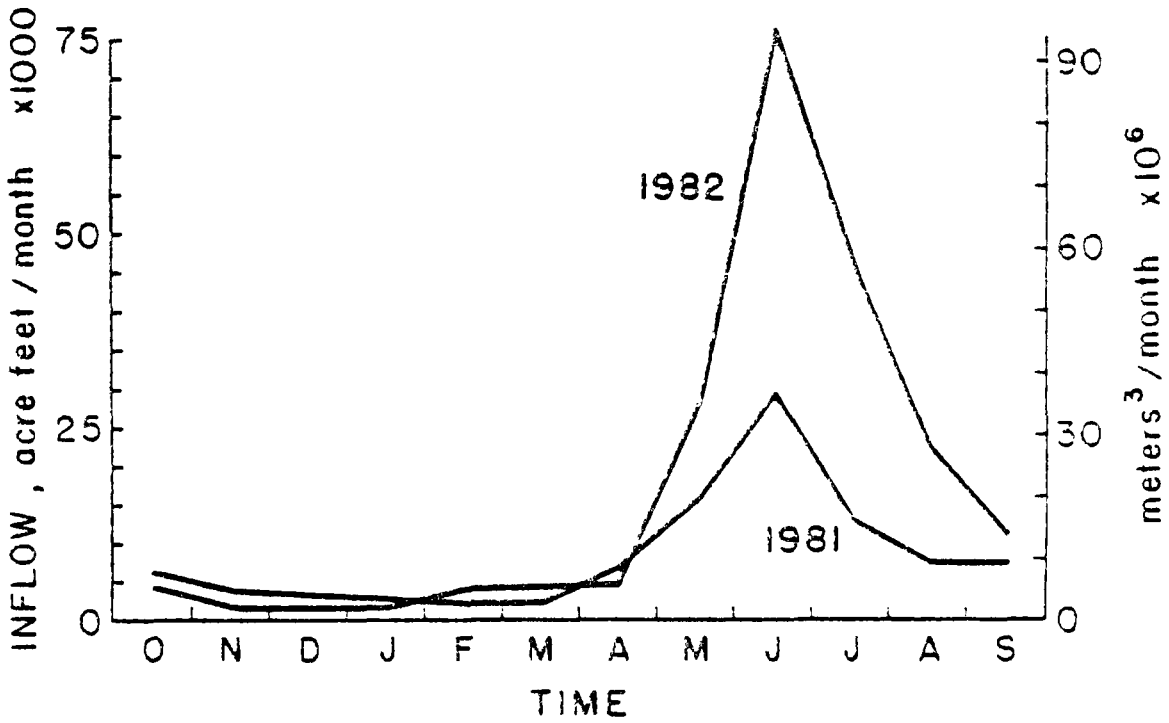


Figure 8. Calculated inflow as obtained by the Denver Water Department for Lake Dillon over the study period on a monthly basis (water years).

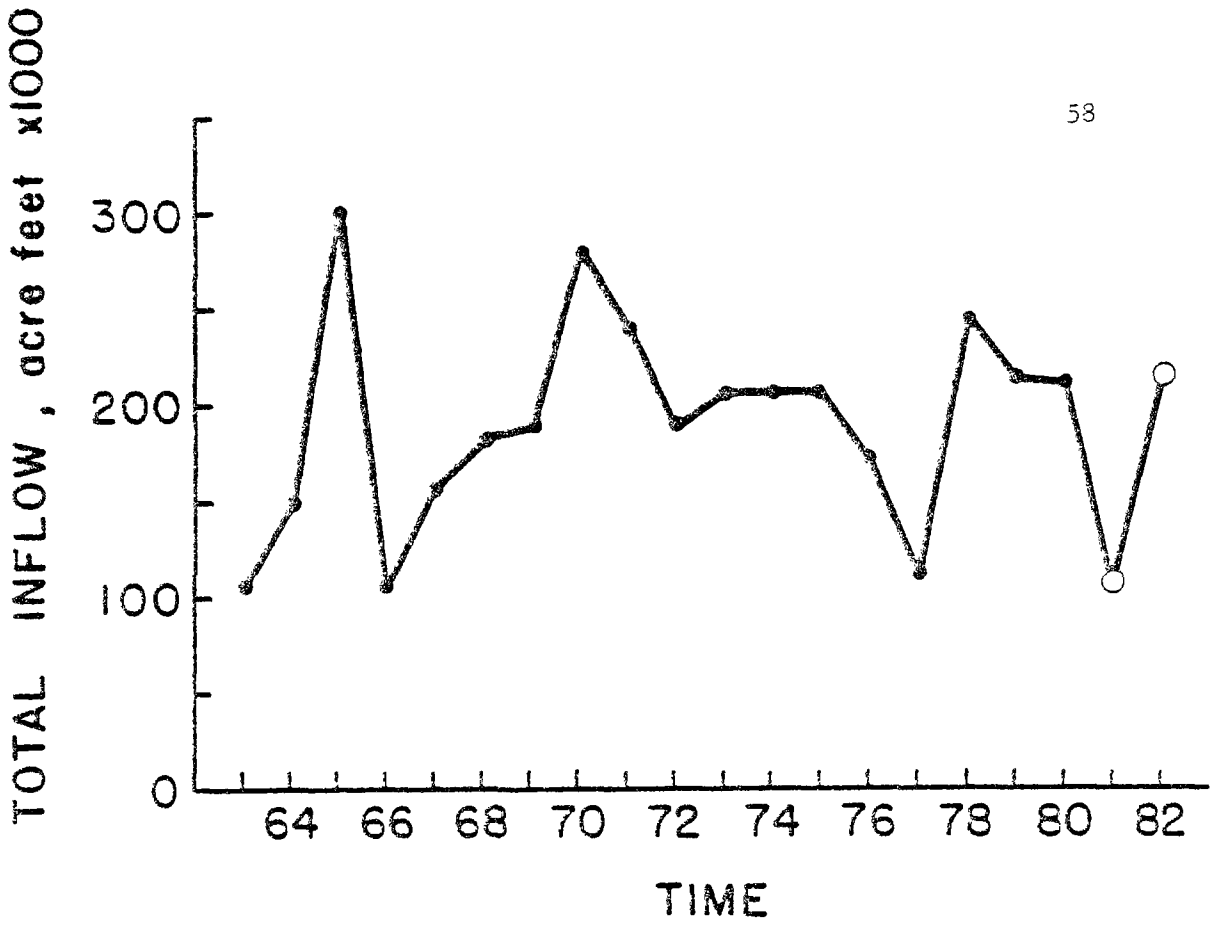


Figure 9. Calculated inflow since 1963 for Lake Dillon (Denver Water Department).

was somewhat above the average (13 of the 19 other years are below 1982).

Figure 10 shows the total outflow per month for 1981 and 1982. Water leaving the lake follows some combination of three pathways: (1) the Roberts Tunnel, (2) the Blue River outlet, and (3) the spillway. Since the Roberts Tunnel and Blue River subsurface outlet remove water from points near the bottom of the lake, they have not been separated in Figure 10, nor need they be treated separately in any of the analyses of this report. Spillway losses must be separated from other losses, however. In 1981 no water left the lake over the spillway, but in 1982 water left the lake over the spillway between 9 July and 4 November. Since water going over the spillway removes water from the epilimnion of the lake, it is important to recognize the timing and extent of spillway flow. The Denver Water Department does not keep records of the apportionment of outflowing water between the spillway and the outlet pipe, although it is known that spillway flow typically dominates when the two occur together. Because the surface and deepwater nitrate and conductance differ considerably at the time of spillway flow, we can estimate the relative contribution of spillway outflow to total outflow as follows: $C_c = XC_t + (1-X)\ C_b$ where C_c is the composite concentration of NO_3-N or conductance in the Blue River below the dam, C_t is the concentration at the top and C_b at the bottom of the water column, and X is the proportion of water going over the spillway. The values vary some, but the estimate shows that about 80% of the water reaching the Blue River between 9 July and 4 November 1982 came over the spillway. Using this percentage as a

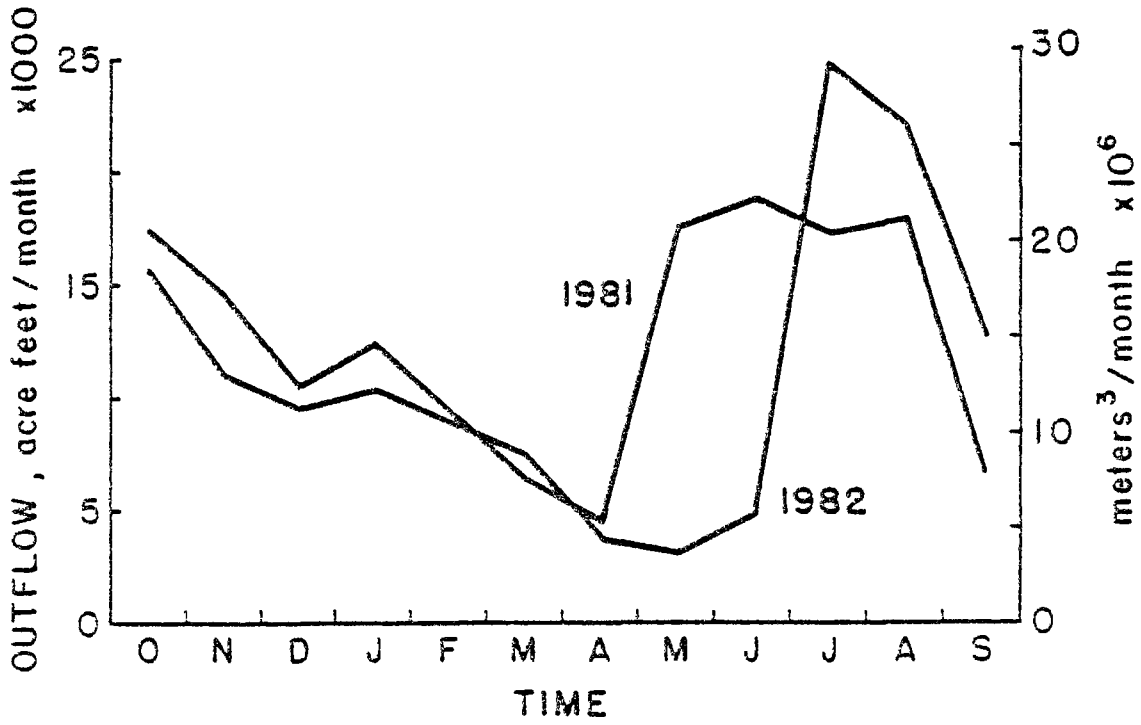


Figure 10. Outflow from Dillon over the study period on a monthly basis. (Denver Water Department).

constant, we have computed the spillway loss on a weekly basis over the period of spillway flow (Table 10).

Figure 11 shows the change in lake level during 1981 and 1982. The lake is typically drawn down in the winter months to meet demand and legal requirements for minimum flow in the Blue River during the period of low winter streamflow. The rate of drawdown in the two years was essentially the same, but the effect on lake level was more drastic in the water year 1982 because the snowpack of water year 1981 was not enough to fill the lake to spillway level.

The combined effect of lake volume changes and changes in total inflow are illustrated in Figure 12 in terms of the percent of volume added to the lake per month for each month of the study. Dilution is most significant in June, less so in May and July, and minimal in other months.

Table 11 summarizes the flow of water into and out of the lake on an annual basis during 1981 and 1982. Both calendar year and water year figures are shown, but the difference between these two conventions is not very great.

Temperature

Like most high temperate lakes of moderate to great depth, Lake Dillon is dimictic. Lakes of this classification show two annual episodes of complete mixing per year, one of which is in the fall and the other in the spring. The year can thus be divided into four intervals: (1) winter ice cover, (2) spring mixing (spring overturn), (3) summer stratification, and (4) fall mixing (fall overturn).

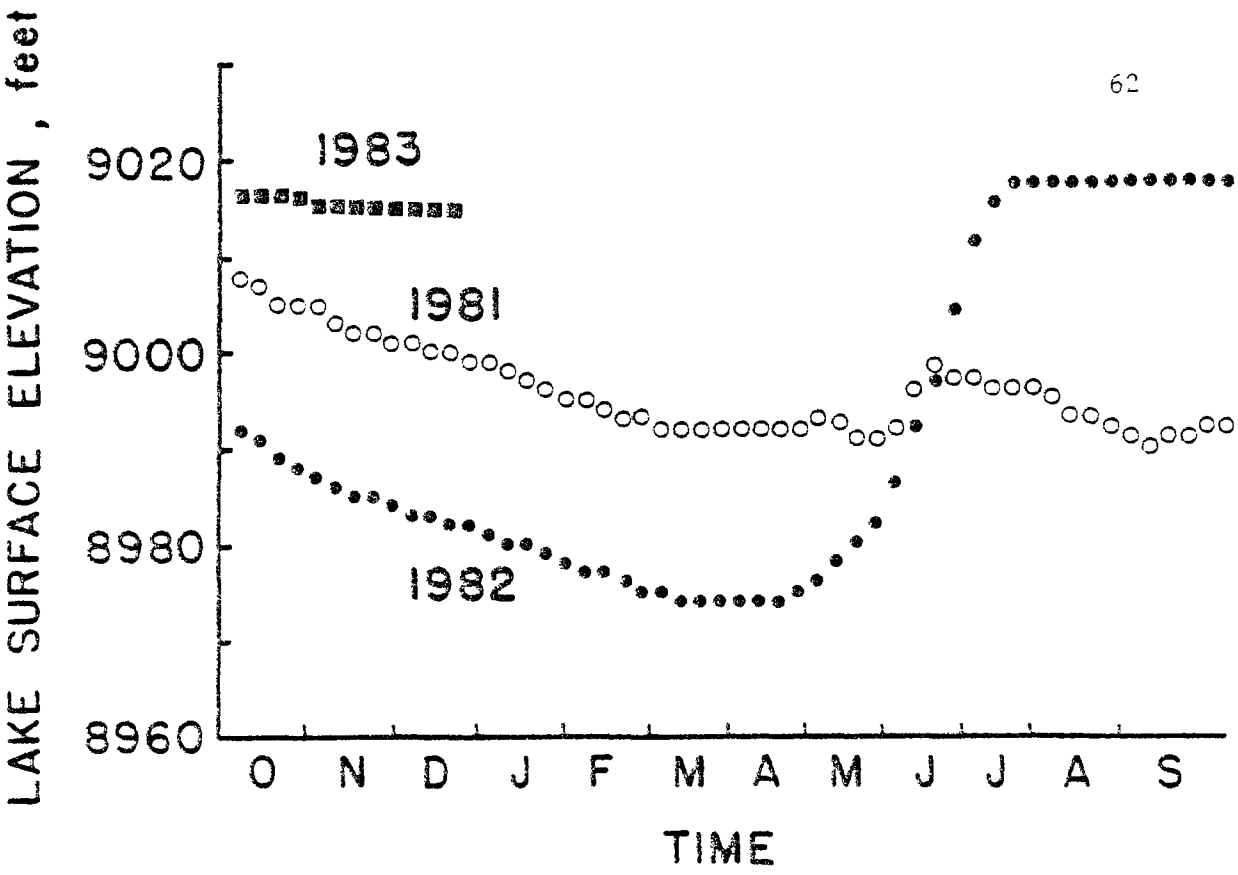


Figure 11. Change of lake level with time over the study period. Data points are midweek values (Source: Denver Water Department).

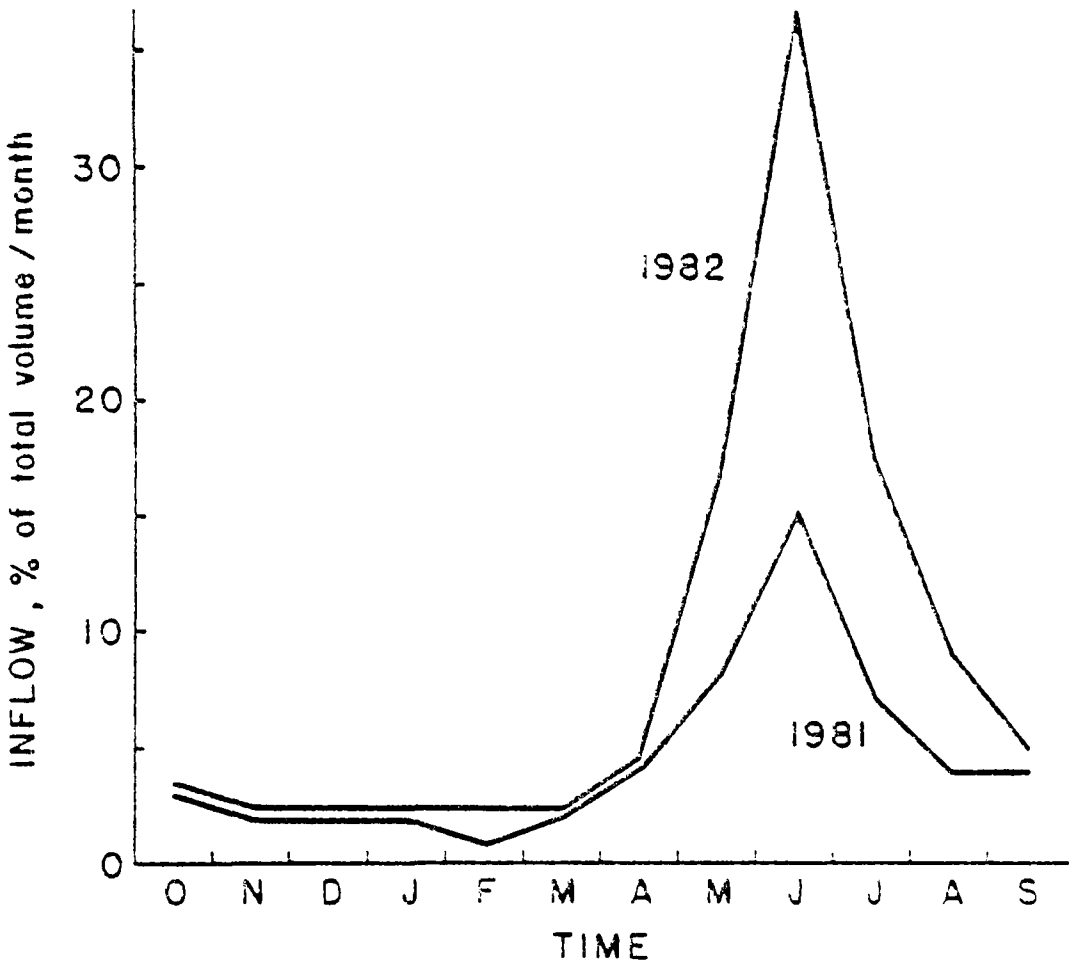


Figure 12. Volume of monthly inflow as a percentage of actual lake volume.

Week	Acre feet per week			
	Total Outflow	Outflow from Bottom*	Outflow over Spillway	Spillway Outflow as %/wk of 0-5 m layer
9 Jul - 15 Jul	4478	3376	1102	2.5
16 Jul - 22 Jul	2458	3951	2507	5.6
23 Jul - 29 Jul	5024	3795	1229	2.8
30 Jul - 5 Aug	7228	5546	1682	3.8
6 Aug - 12 Aug	3162	679	2483	5.6
13 Aug - 19 Aug	5396	1076	4320	9.7
20 Aug - 26 Aug	5322	1064	4258	9.5
27 Aug - 2 Sept	8942	1788	7154	16.0
3 Sept - 9 Sept	3008	602	2406	5.4
10 Sept - 16 Sept	3194	639	2555	5.7
17 Sept - 23 Sept	2830	566	2264	5.1
24 Sept - 30 Sept	2476	495	1981	4.4
1 Oct - 7 Oct	2069	415	1654	3.7
8 Oct - 14 Oct	1858	372	1486	3.3
15 Oct - 21 Oct	1844	369	1475	3.3
22 Oct - 28 Oct	1774	355	1419	3.2
29 Oct - 4 Nov	1744	538	1206	2.7

* Includes both Roberts Tunnel and Blue River outflow pipe.

Table 10. Estimated partitioning of outflow over the period of spillway loss (1982).

	Calculated Inflow		Measured Outflow	
	Acre-feet	km^3	Acre-feet	km^3
1981				
Calendar Year	105,302	0.130	145,912	0.180
Water Year	106,193	0.131	152,565	0.188
1982				
Calendar Year	217,386	0.268	119,838	0.148
Water Year	210,255	0.259	131,989	0.163

Table 11. Summary of water flow into and out of the lake, based on daily statistics supplied by the Denver Water Department (cumulated monthly and daily statistics from the same source differ slightly).

In both 1981 and 1982, ice cover formed on Lake Dillon in late December or early January (Figures 13, 14). In both winters the maximum ice thickness reached about 0.5 m in midlake. In 1981 the ice was exceptionally transparent (i.e., without bubbles or snow layers), and, because of the small amount of winter snowfall, was not covered with significant amounts of snow during the first two months of ice cover. Because an ice cover of this type transmits light almost as well as water (Ragotzkie 1978), unusually large amounts of light reached the water column during the winter of 1981. By April of 1981 light penetration was blocked more effectively because of some snow accumulation and thaw cycles at the surface, making an opaque granular layer at the top. In 1982 there was snow cover from the very beginning, and it lasted through the entire winter. Snow on the lake surface did not accumulate to great depth because of wind exposure, but there were few open areas in the winter of 1982. Since snow has an absorption coefficient of about 0.24 cm^{-1} (Ragotzkie 1978), 10 cm of snow, which is approximately the amount on the lake in 1982, would remove about 90% of the light.

In both 1981 and 1982, the ice cover broke up in early May. Temperatures under the ice in both years were between 1 and 3°C over most of the water column, as expected. Temperatures closer to the ice were slightly lower than near the bottom. This is a stable condition because the density of water decreases from 4°C to the freezing point.

Warming of the surface occurred slowly in both years after the ice cover disappeared, but especially so in 1981. As long as a water column is below 4°C, warming at the surface actually creates

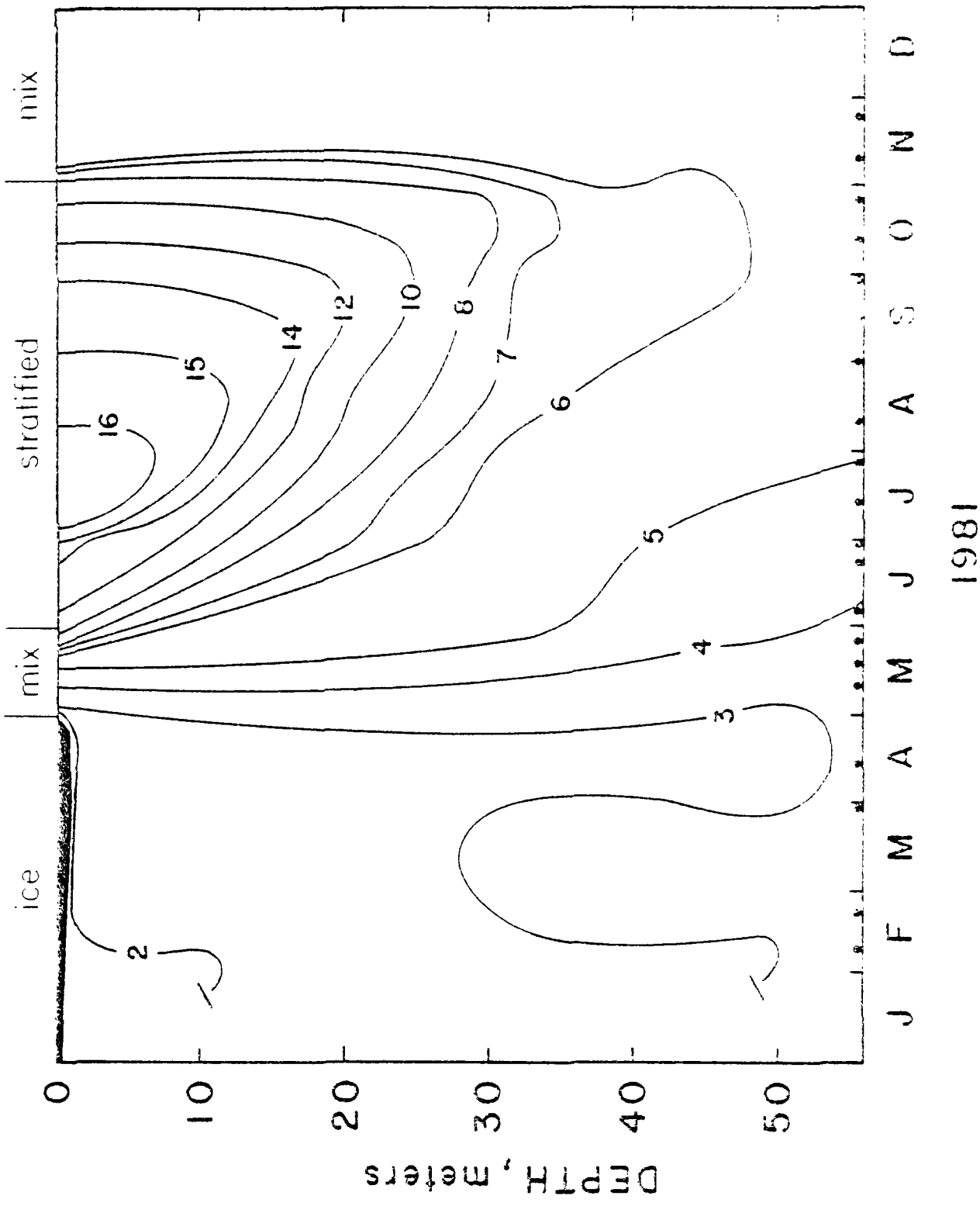


Figure 13. Time-depth diagram for temperature at the index station, 1981 (°C)

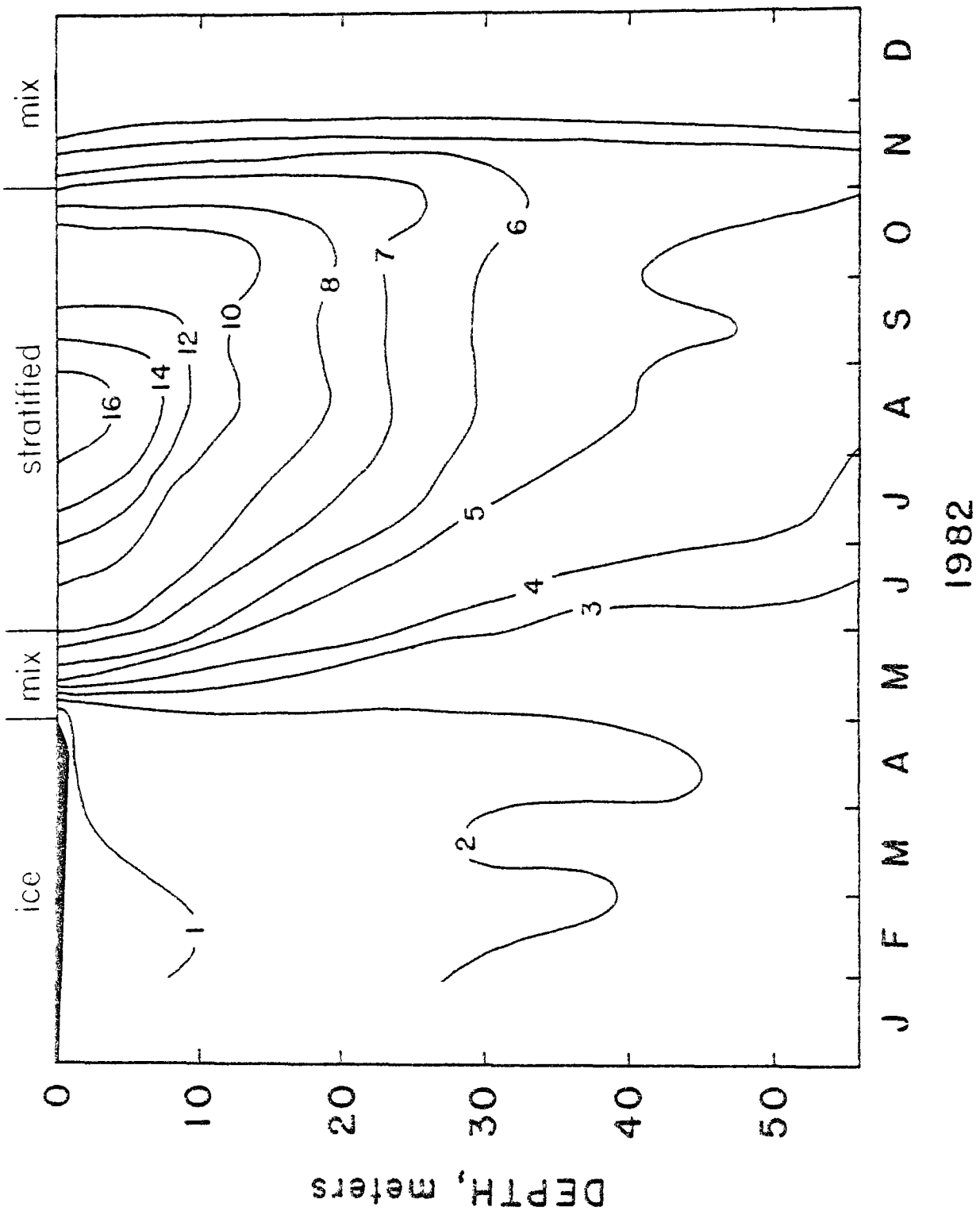


Figure 14. Time-depth diagram for temperature at the index station, 1982 (°C)

instability because of the increase in density of water as it is warmed from the freezing point to 4°C. Even after the water column warms to 5°C there is little resistance to mixing because of the very slight change in density of water in the vicinity of 4°C. Thus there is no tendency to stable layering until the surface water exceeds 5 to 6°C. Until then, heat is efficiently distributed through the entire water column by the wind, and thus the water warms only slowly at the surface.

In Lake Dillon some beginning indications of layering could be seen as the lake exceeded 5°C in the last half of May both years. Although stratification was very weak at first, complete mixing of the water column ended near the last of May in both years. June 1 thus marks the beginning of the stratification season.

By the last half of June in both years thermal layering had become very pronounced. The upper water column reached 12°C by the last half of June in both years, and the thickness of the upper mixed layer at this time was about 5 m (we equate the upper mixed layer with the epilimnion, although the epilimnion is sometimes defined somewhat differently: Hutchinson 1957). The June thermocline was relatively thick, extending from 5 m down to about 20 m. In both 1981 and 1982 the upper water column continued to take up heat well after stratification was first established. In 1981 the lake reached maximum heat content in the last week of July, when the temperature of the epilimnion rose to just over 16°C. In 1982 the warming occurred slightly more slowly, producing maximum epilimnetic temperature in the first week of August, but the maximum epilimnetic temperature was

essentially the same as in 1981. In both years the thickness of the mixed layer remained 5-10 m for the first half of the stratification season, i.e., until the middle of August, but the average thickness was somewhat greater in 1981 than in 1982 (Figure 15). In both years cooling of the epilimnion began in the last half of August. Cooling of the upper water column reduced the stability of stratification sufficiently to allow gradual erosion of the thermocline by late August or early September in both years. In 1981 the thickness of the epilimnion had reached 15 m by the middle of September and 25 m by the middle of October. In 1982, the epilimnion had thickened to 10 m by the middle of September and 25 m by the middle of October.

The stratification season ended with complete mixing during the first week of November in both 1981 and 1982. By the time complete mixing occurred, the lake had already cooled to a temperature of about 6°C. Isothermal cooling accompanied by complete mixing of the water column then proceeded through November and most of December until the establishment of ice cover at the end of December.

Table 12 gives a synopsis of the layering and mixing events in Lake Dillon in 1981 and 1982. There were no significant differences in the timing of major events in the two years.

Season	Duration
Ice cover	January, February, March, April
Spring mixing	May
Summer stratification	June to first week of November
Fixed epilimnetic thickness	June, July, August
Increasing epilimnetic thickness	September, October
Fall mixing	Last three weeks of November, December

Table 12. Seasons of layering and mixing in Lake Dillon.

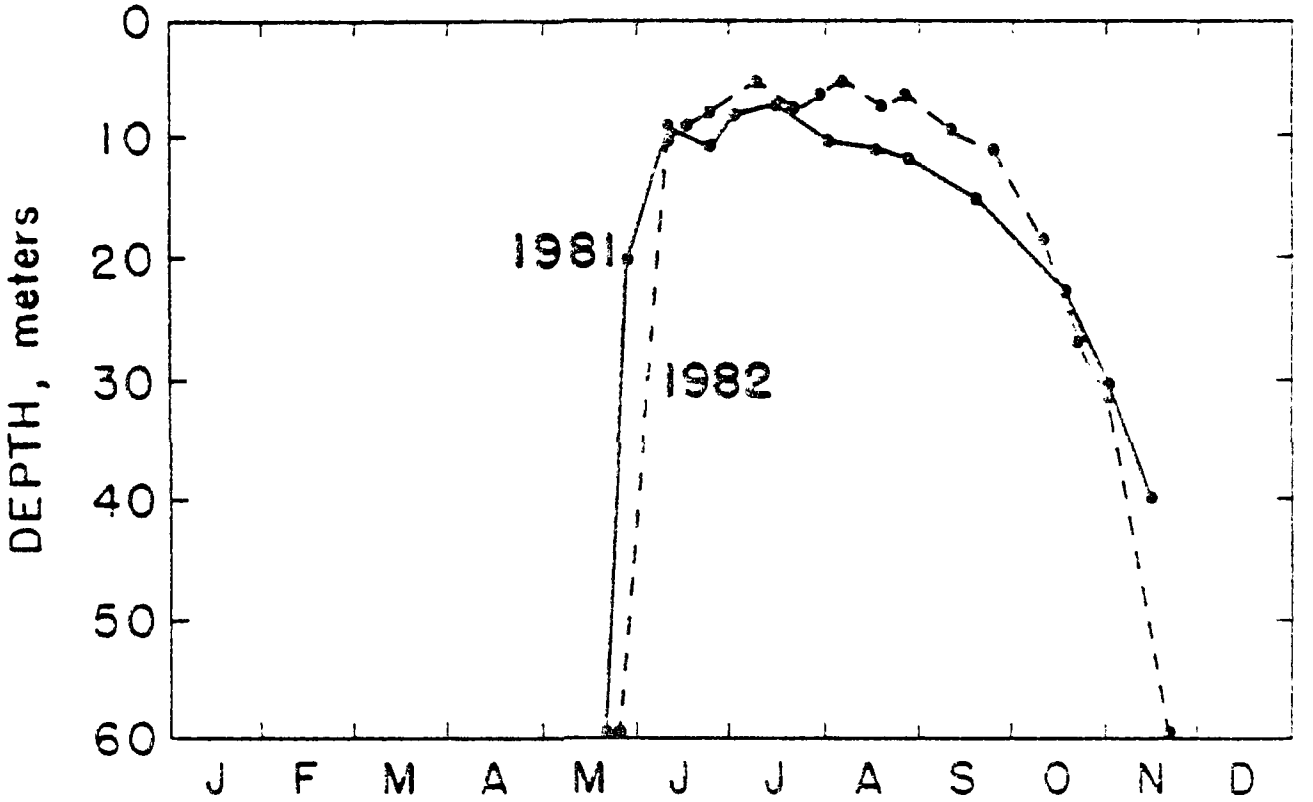


Figure 15. Thickness of the mixed layer during the ice-free seasons of 1981 and 1982.

Transparency

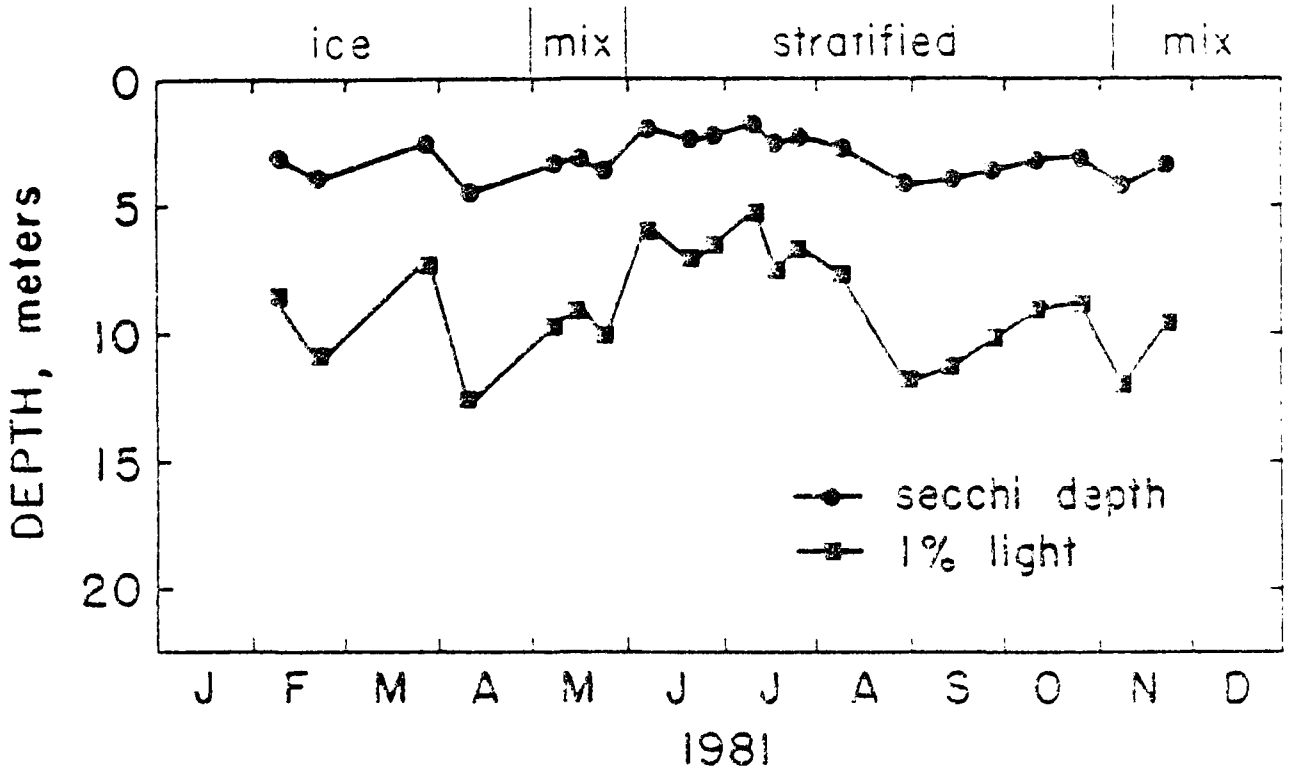
Figure 16 shows the transparency of the lake over the two-year study period both in terms of the secchi depth and the depth of 1% light. The depth of 1% light was approximated from the secchi depth as follows:

$$\eta_t = -\ln(0.19)/z_s$$

$$z_{.01} = -\ln(0.01)/\eta_t$$

where η_t is the extinction coefficient of photosynthetically active radiation (350-700nm, m^{-1}), z_s is secchi depth (m), $z_{.01}$ is the depth at which 1% of the surface photosynthetically active radiation (PAR) is found (m). The approximation assumes that the secchi depth always falls at 19% of the surface irradiance. The basis of this assumption is a series of comparisons between light meter readings and secchi depth readings as reported in the methods section. The purpose of the conversion from secchi depth readings to depth of 1% light is to show the approximate depth at which photosynthesis could occur rapidly enough to match the respiration of algal cells (Talling 1971).

Figure 16 shows that significant photosynthesis in Lake Dillon was always limited to the top 15 m and was usually limited to the top 10 m of the water column. The main features of the seasonal change in transparency could be summarized as follows: (1) transparency was moderate to high under ice, (2) transparency was consistently low over the entire first half of summer stratification, and (3) transparency was consistently high during spring and fall mixing and over the last half of summer stratification. The readings under ice may be



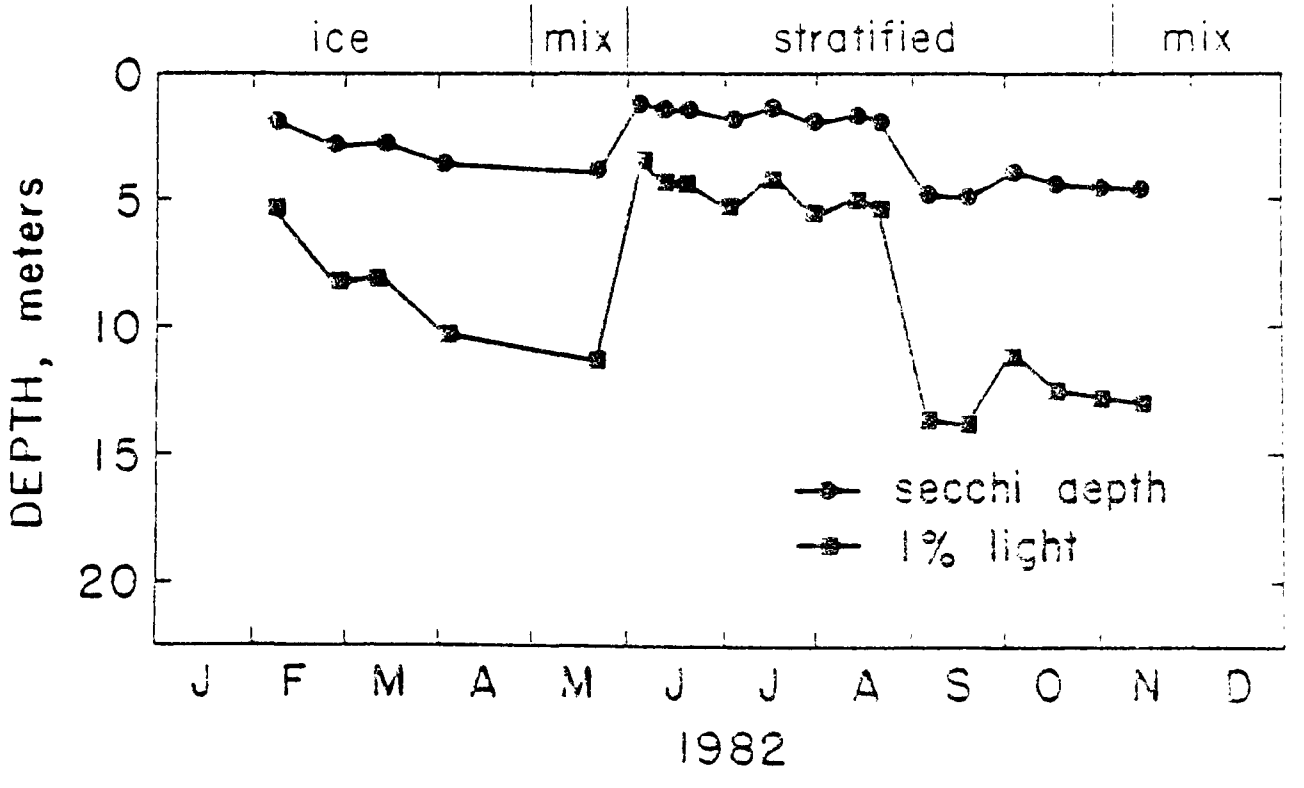


Figure 16. Secchi depth and depth of 1% light in Lake Dillon (index station).

underestimates, especially in 1982 when there was significant snow cover. Because the readings were taken through a hole in the ice, light could not reach the secchi disk as well as it would under ice-free conditions.

Four factors can affect the transparency of the lakewater: (1) the extinction of light by pure water, (2) the extinction of light by organic materials dissolved in the water, (3) the extinction of light by chlorophyll, and (4) the extinction of light by organic and inorganic particulate materials. The extinction coefficient (η_t) can thus be broken into components:

$$\eta_t = \eta_w + \eta_d + \eta_c + \eta_p$$

where w represents pure water, d represents dissolved substances, c represents chlorophyll, and p represents particles. η_t is the exponential coefficient in the equation that approximates the vertical attenuation of light:

$$I_z = I_o e^{-\eta_t z}$$

where z is depth (m), I is irradiance, and η_t has units m^{-1}. Values of η_t and its components are always somewhat approximate because of the change in spectral composition of light with depth (Smith 1968).

The effect of pure water is relatively constant through the seasons, although it does vary some for complex reasons having to do with changes in spectral composition (η_w is about 0.1 m^{-1}). Dissolved organic materials contribute to the extinction of light in the lake but are seasonally quite constant, as indicated by absorbances of filtered samples from Dillon. Dissolved organics, which are

responsible for the extinction component η_d, are not present in large amounts in Dillon. From the absorbances at 360nm and the equation of Lewis and Canfield (1977), the dissolved organic carbon is estimated as 1.5-1.6 mg/l. From the absorbances of filtered water and the approximation method developed by Talling (1971), we estimate that, in complete absence of all particulates, including phytoplankton, the secchi depth would be at least 10 m (based on $\eta_w + \eta_d = 0.17\ m^{-1}$; this is a minimal figure, since η_w is smaller for more transparent lakes). This corresponds to a 1% light level of 27 m. For comparison, the average secchi depth in Lake Tahoe, a lake exceptionally free of particulates and soluble organics, is about 28 m (Goldman 1974).

Separation of the effect of chlorophyll from the other effects on transparency involves some assumptions, but is feasible. First it is useful to make a few qualitative observations concerning the relative importance of algae and nonliving suspended matter. In 1982 a major and precipitous decline in transparency occurred just at the onset of stratification but before chlorophyll levels in the upper water column had begun to climb (Figure 16). For example, the secchi depth of the lake in 1982 changed from 4.1 m on May 24 to 1.3 m on June 7. The effect was observable at the same time in 1981 but was less dramatic. This quick decrease in transparency coincided with the spring runoff, and, since the chlorophyll had not increased much at this point, the sudden decrease in transparency must be attributable to nonliving particulate materials added to the lake by spring runoff. The transparency of the lake prior to the onset of runoff shows that the

mixing of the lake was not responsible for the decrease in transparency. All evidence thus points specifically to the major role of spring runoff in decreasing the transparency of the lake in early June.

Transparency remained low in the lake both years into August, despite the much earlier decline in amount of surface water entering the lake. Low transparency after June must therefore have been in large part attributable to the buildup of phytoplankton biomass. This is confirmed by data on chlorophyll, to be presented in the next section, and by microscopic examination of samples, which showed a major decline in non-living particulates after runoff had passed.

In the last half of the stratification season of both years, beginning in August, there was a return to high transparency. This was accompanied by drastic declines in the surface chlorophyll which continued through the termination of stratification and into the fall mixing season. The reasons for this return to higher transparency are related to the nutrition of the algal populations, so the complete explanation is deferred until the chlorophyll data are discussed.

Quantitative separation of the effect of chlorophyll from other effects on transparency can be achieved by estimation of the chlorophyll-specific absorption coefficient, e_s, which is related to the extinction coefficient for chlorophyll as follows: $e_s = n_c/B$ where n_c is extinction due to chlorophyll (ln units per m), e_s is the chlorophyll-specific coefficient (ln units $\cdot$ m^2/mgChl*a*) and B is chlorophyll *a* concentration (mgChl*a*/m^3). e_s is traditionally treated as a constant for a given lake, although it is known to vary

some with depth (due to changes in spectral composition) and with seasonal changes in phytoplankton composition (Kirk 1975). The estimate of e_s for Lake Dillon is based on linear regression of extinction coefficient (dependent variable) on chlorophyll concentration (independent variable). The slope is then e_s. The regression for Dillon yields a slope of 0.016 m^2/mg, which will be taken as the value for e_s. This is within the midrange of literature values, which generally fall between 0.008 and 0.021 m^2/mg and show a mode around 0.015 (Talling 1982).

From the value of e_s and the chlorophyll concentration on any given day, the amount of light extinction due to chlorophyll (η_c) can be approximated. If the effect of pure water and dissolved organics, assumed constant at $\eta_w + \eta_d = 0.17\ m^{-1}$, is added to η_c, the remaining extinction must be the non-chlorophyll particulate effect, the last of the four components of extinction. The separation made in this way is shown in Figure 17.

Figure 17 confirms the foregoing interpretation based on qualitative considerations. Low transparency came suddenly in early June and lasted into August. At first the effect was attributable almost solely to particulate materials brought in from the watershed, but these materials rapidly settled out and were not replaced as the runoff passed its peak. Low transparency after runoff was sustained by buildup of algal biomass. Low transparency persisted until algal biomass was reduced by nutritional deficiencies toward the end of August. The increasing importance of phytoplankton in contributing to the low transparency of the lake toward the middle of the

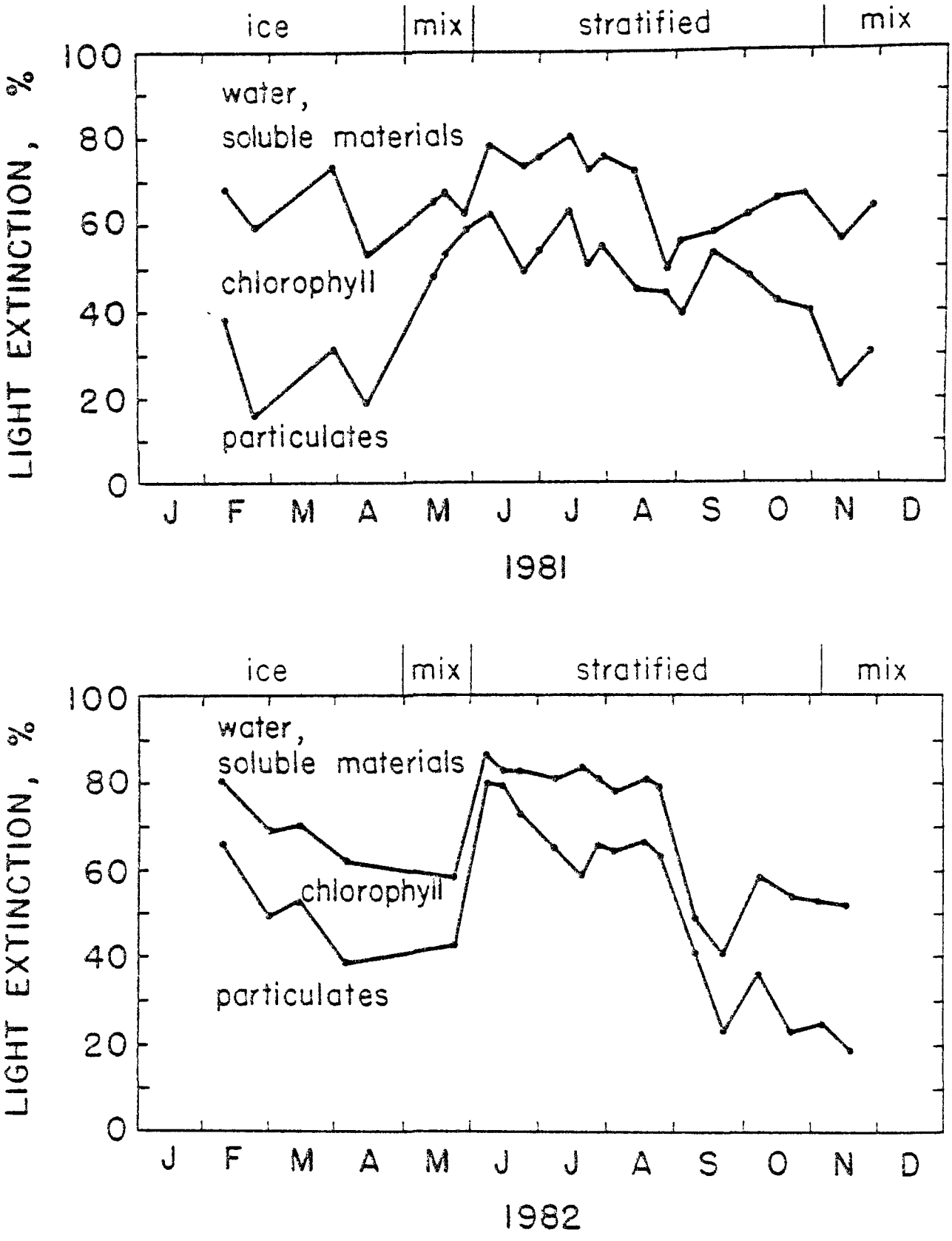


Figure 17. Cumulative percent light extinction due to particulates, chlorophyll, and soluble material fractions in Dillon for 1981 (above) and 1982 (below).

stratification season was accompanied by a change in water color from blue to green. Whereas nonliving particulate materials can change the transparency of a lake considerably without changing its color significantly, phytoplankton biomass responsible for an equivalent reduction in transparency has a marked effect on color (Hutchinson 1957).

Historical Changes in Transparency

There is some historical information on the transparency of Lake Dillon. The secchi depth was measured by the National Eutrophication Survey in 1975. In August 1975, the secchi depth, as determined from the mean of a number of measurements at different sites on the lake, was 7.8 m. In September of the same year the mean was 8.3 m. Since the secchi depths at the same times in 1981 and 1982 were only about half these values, it would appear that a major decrease in transparency occurred between 1975 and 1981, probably due to eutrophication. Secchi depth data are also available from Wesley Nelson, Colorado Division of Wildlife (personal communication). Nelson's values average lower than readings obtained at the same time by the Eutrophication Survey or the present study, possibly because of differences in discs or in the location of data collection. However, his series spanning 1975-1982 for secchi depth on or near August 25 shows some evidence of a trend toward lower transparency: 1975, 4.6 m; 1978, 3.6 m; 1979, 2.0 m; 1980, 2.3 m; 1981, 3.1 m; 1982, 1.6 m.

Composition of Major Ions

Table 13 shows the composition of major ions in water from Lake Dillon. There are some seasonal changes in major ions, particularly associated with the runoff in the month of June, but these are not great enough to have much biological significance, so the amounts of major ions were not quantified on a routine basis. Table 13 indicates that the water of Lake Dillon is a bicarbonate type exceptionally rich in sulfate. Cations are dominated by calcium. The total ionic strength, as indicated by Table 13 and by the conductance measurements that were made routinely throughout the course of the study, is somewhat higher than might be expected for an undisturbed Rocky Mountain watershed of similar geology. The total ionic strength and ion composition of the lakewater has undoubtedly been augmented to some extent by mining and dispersed earth disturbance, as will become evident in the analysis of stream chemistry. Expected conductance computed from Table 13 and ionic mobilities is 179 umho/cm at 25°C, which checks well with the observed conductance (170 umho/cm).

	concentration	
	mg/l	meq/l
Cations		
Ca^{++}	21.3	1.068
Mg^{++}	3.58	0.295
Na^{+}	3.45	0.150
K^{+}	0.54	0.014
Total cations	28.89	1.528
Anions		
HCO_3^-	47.1	0.772
$SO_4^=$	39.1	0.814
Cl^-	3.90	0.110
NO_3^-	0.93	0.015
Total anions	91.03	1.711
Total ions	119.92	3.239

Table 13. Major ion composition of Lake Dillon, based on analysis of a sample taken at the index station, 4 October 1982.

Conductance

Conductance of lakewater was slightly higher in 1981 than 1982, as shown in Table 14, which gives annual averages for the top, middle, and lower water column. The difference between years was more pronounced in the upper water column than in the lower water column. Conductance showed significant seasonal variation and a degree of vertical layering under ice and during summer stratification. These features are illustrated by Figure 18, which shows the change of conductance through time at the top of the water column (0-5 m) and in the bottom 5 m for the two years of the study. In both years the surface water was lower in ionic solids than deep water under ice. This layering must be set up after ice cover, as fall mixing homogenizes conductance. Retention of melt water near the surface is the probable explanation. Surface temperatures under ice were in the vicinity of 0°C at the top and, since water is less dense at 0°C than at slightly higher temperatures, there would have been a slight resistance to mixing to help retain the layering as long as the ice was protecting the surface from wind.

	Depth, m		
	0-5	20-25	35-40
Conductance, umho/cm			
1981	168	167	178
1982	152	153	172
pH			
1981	7.45	7.36	7.24
1982	7.80	7.59	7.54
Alkalinity, mg/l CO_2			
1981	33.9	34.4	34.3
1982	34.5	34.3	35.4

Table 14. Means for conductance, pH, and alkalinity at three depths in the water column at the index station.

Vertical homogeneity of conductance was typical of the spring mixing period, as expected. With the onset of thermal stratification in June, there once again was a divergence in conductance of surface water and deep water. The divergence appeared rather suddenly, suggesting that spring snowmelt was at first held in greatest amounts in the upper water column after thermal stratification began. Not all of the runoff entered the upper water column, however, because the lower water column also showed steady dilution. Furthermore, the conductance of the upper water column was homogeneous down to approximately 30 m, a depth much greater than the upper boundary of the thermocline. Thus the dilution caused by spring snowmelt affected the entire water column but was more pronounced in the upper 30 m than the lower 30 m. This accounts for the higher average conductance of the lower water column as illustrated in Table 14. With the fall mixing, vertical homogeneity of conductance was established as shown in Figure 18.

It is possible to estimate the depth at which entering streams penetrated the lake, since a stream will tend to seek the layer of its own density. This is not an important issue when the lake is mixing freely, but is potentially of interest during the stratification season when the entering water may tend to flow selectively into the upper, middle, or lower water column. In order to make such an estimate, we computed the density profile of the water column on each sampling date during stratification. Taking into account not only temperature, but also dissolved and suspended solids, we also calculated the density of the three rivers on each date. By comparison

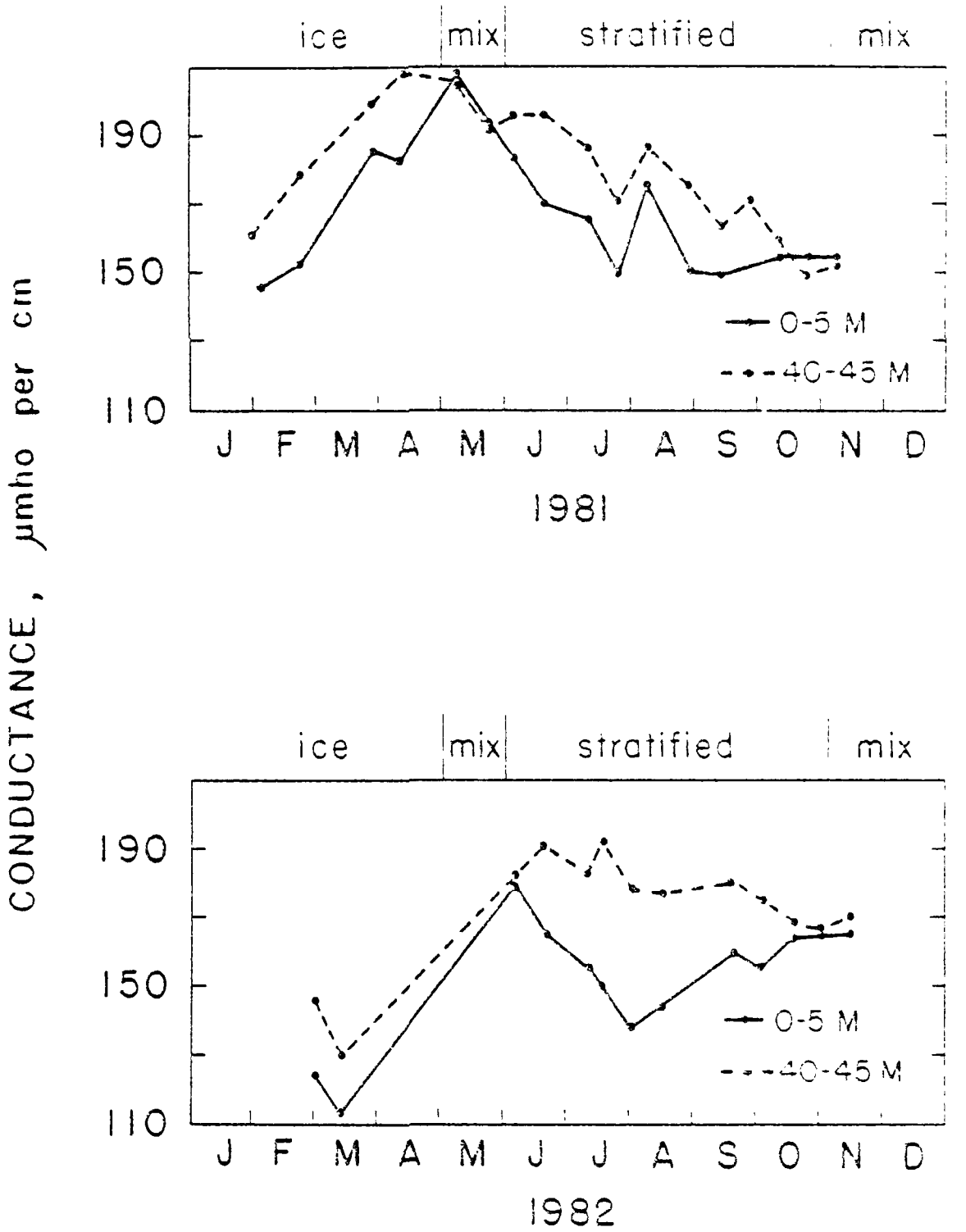


Figure 18. Conductance in Lake Dillon, 1981-1982 (index station).

we could then show the depth at which river and lake density would be matched, and this is the depth to which the rivers would tend to flow. The results are shown in Figure 19. In both years the bulk of flow was predominantly into the upper middle of the water column, and in both years the depth of penetration increased from June to October. In 1981 the penetration began at about 10 m, just below the thermocline, and increased to about 25 m by late stratification (due mainly to warming of the lakewater). In 1982, penetration was not so deep; it began at about 5 m, and slowly passed down to about 15 m. We postulate that the large flow at these depths, as well as heat loss over the spillway, was responsible for holding the thermocline at a higher level in 1982 than 1981.

pH and Alkalinity

Table 14 shows that the pH at all depths was slightly higher in 1982 than in 1981, and that in both years the mean pH at the bottom of the water column was slightly lower than at the top, as expected because of the influence of photosynthesis near the top of the water column. Figure 20 shows the seasonal patterns near the surface (0-5 m) and at the bottom of the water column. The pH levels at top and bottom diverged in a major way only after rapid photosynthesis began in late June. Photosynthesis elevated the surface pH values in proportion to the rate of removal of free CO_2, so the surface pH more or less paralleled chlorophyll. The pH levels were at no time extraordinary, however, and do not suggest extreme depletion of free CO_2, which is signified by pH levels above 9.0 (Talling 1976). In both years there

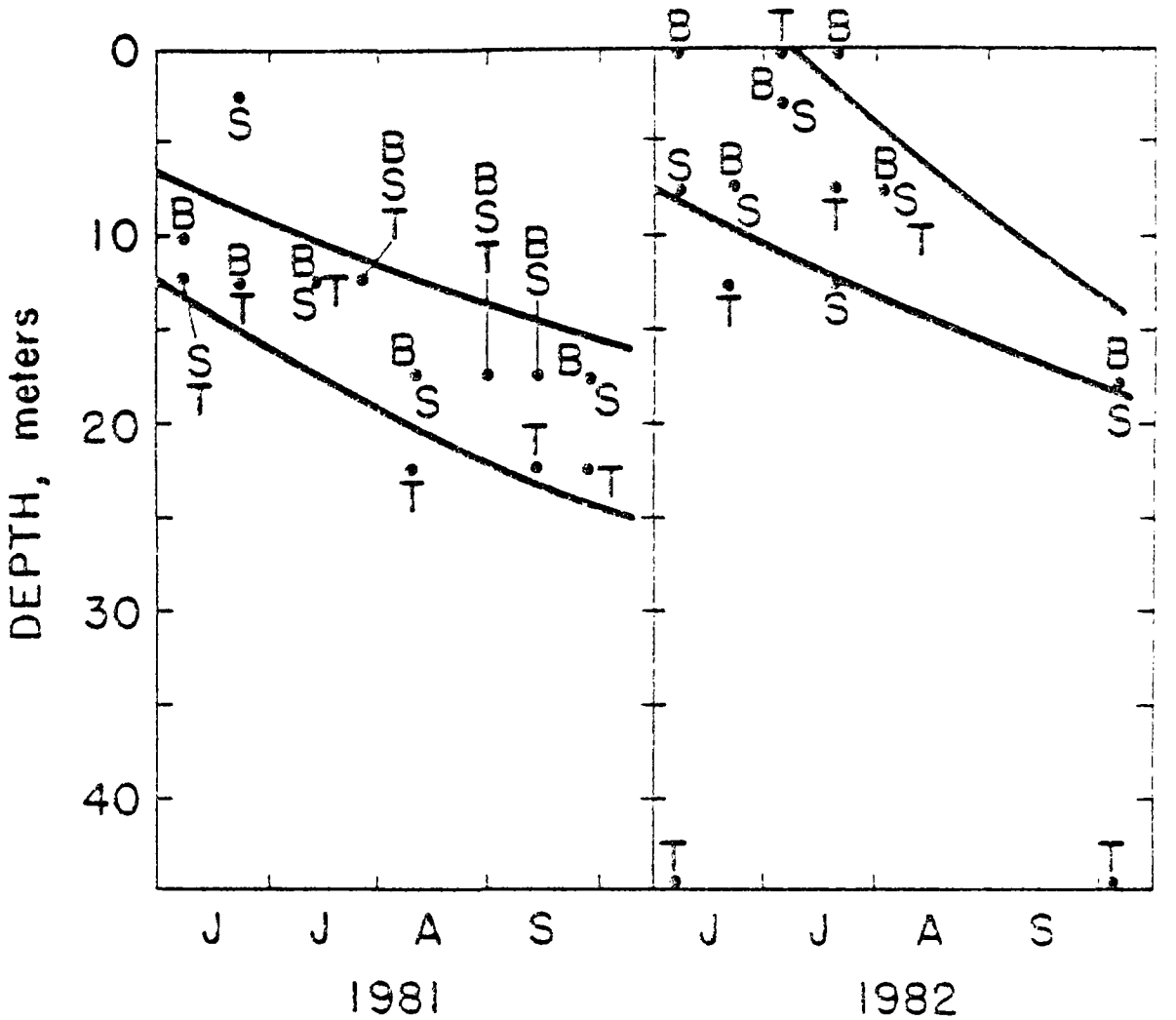


Figure 19. Calculated inflow depth of the Snake River (S), Blue River (B), and Tenmile Creek (T) during stratification in 1981 and 1982.

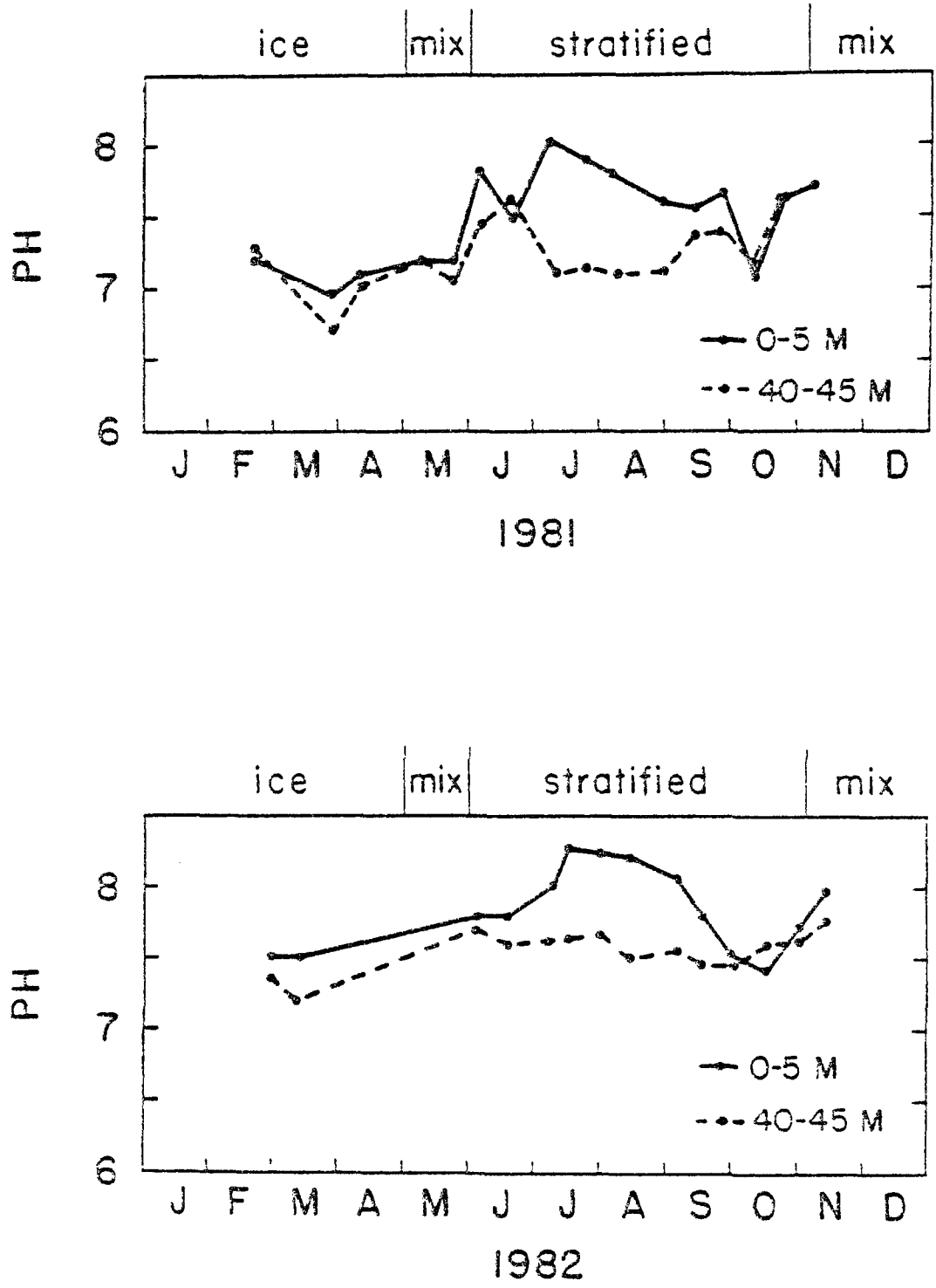


Figure 20. pH in Lake Dillon, 1981-1982, (index station).

was a return to lower pH levels as photosynthesis dropped off after July, and a resurgence after nutrient depletion was relieved by thickening of the mixed layer in fall.

Alkalinity values fell within a relatively narrow range, as shown by Table 14. Seasonal effects were slight (Figure 21), but the large runoff of 1982 can be identified as the cause of some reduction in alkalinity.

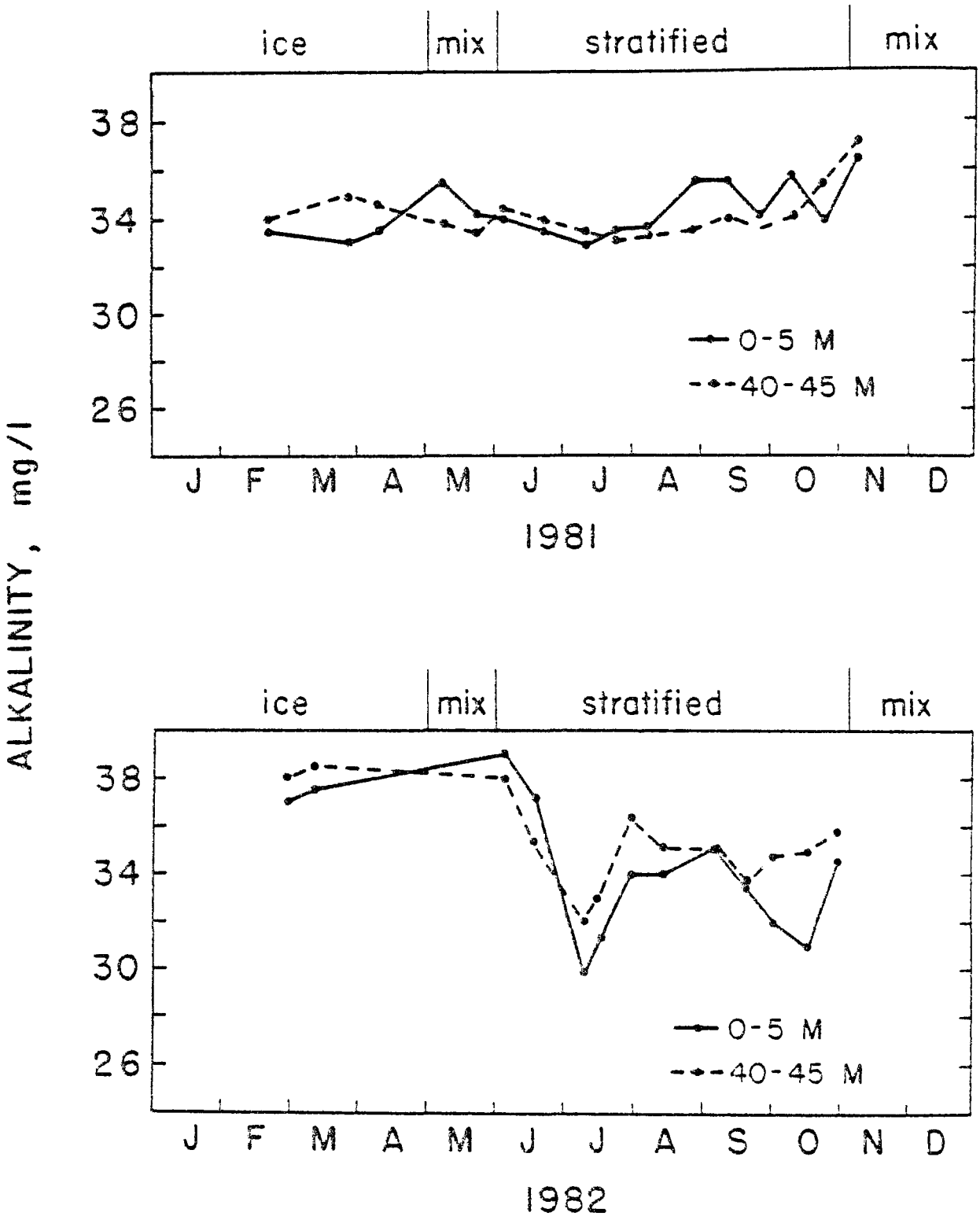


Figure 21. Alkalinity in Dillon, expressed as mg/l CO_2 (index station).

Phosphorus and Nitrogen in Lake Water and Sediments

Phosphorus in Lake Water

The concentration of total phosphorus and the contributions of orthophosphate, dissolved organic phosphorus, and particulate phosphorus to this total are summarized in Table 15. The average orthophosphate values were exceedingly low in both years and showed virtually no trend with depth. The averages for 1982 were unquestionably slightly higher, however. This is explained mainly by the greater flow of water into the lake in 1982, as described more fully below in connection with seasonal trends. Soluble organic P was similar to orthophosphate in showing no real trend with depth in the averages and in the slightly higher average for 1982. Particulate P, in contrast, was essentially the same the two years and showed a slight tendency toward higher values near the surface both years, probably due to the ability of phytoplankton, which are located mostly near the surface, to sequester phosphorus. As a result mainly of higher soluble P levels, total P values were slightly higher in 1982 than in 1981. Slightly less than half of the total P was soluble and slightly more than half was particulate on the average.

Table 16 shows the results of a two-way analysis of variance (ANOVA) in which the total P and P fractions were tested for statistically significant differences between years and depths. The 0-5 m layer was taken as representative of surface water and the 40-45 m layer was taken as representative of deep water. The years differed significantly only for PO_4-P and total P, and depths

	Concentration - ug/l							
	PO_4-P		Soluble Organic P		Particulate P		Total P	
Stratum	1981	1982	1981	1982	1981	1982	1981	1982
0-5	0.75	1.50	2.08	2.36	4.80	5.54	7.63	9.15
5-10	0.56	0.94	1.55	2.49	4.98	4.96	6.74	8.20
10-15	0.77	1.01	1.77	4.96	4.46	4.87	7.01	10.64
15-20	0.62	1.30	2.40	2.57	4.31	4.37	7.33	8.01
20-25	0.47	1.91	1.56	1.84	4.15	3.99	6.19	7.45
25-30	0.63	1.65	2.00	1.53	3.76	3.54	6.39	6.16
30-35	0.57	1.66	3.52	3.21	3.68	3.58	6.31	8.19
35-40	0.64	1.30	2.04	2.11	3.51	3.38	5.83	6.56
40-45	0.72	1.51	1.68	1.31	3.70	3.83	6.23	6.40
45-50	0.72	2.10	1.81	1.17	3.47	3.27	6.68	6.80
50-55	0.79	2.16	2.60	1.48	4.00	2.73	6.03	6.83
$\bar{x}$	0.66	1.55	2.09	2.28	4.07	4.00	6.58	7.67
Mean %	10%	20%	31%	29%	60%	51%		

Table 15. Mean concentrations of total phosphorus and phosphorus fractions at various depths between the surface and bottom at the index station. Fractions may not equal totals exactly due to occasional missing values for individual fractions.

	PO_4-P	Soluble Organic P	Particulate P	Total P
Difference bet. depths	Not significant	Not significant	Highly significant**	Highly significant**
Difference bet. years	Highly significant**	Not significant	Not significant	Significant*

**$p < 0.01$ * $p < 0.05$

Table 16. Results of a two-way ANOVA testing for differences in means between depths (0-5 versus 40-45 m) and between years (1981 versus 1982).

differed significantly only for particulate and total P. No interactions of year with depth were significant.

Seasonal changes in the total phosphorus concentration and in the contribution of the different phosphorus fractions in the upper water column (0 - 5 m) are summarized in Figure 22a. Comparable information is given in Figure 22b for the lower water column (40-45 m). There are some notable trends both in total phosphorus and in the contributions of the fractions, especially in the surface layer. These trends can be understood for the most part in terms of the seasonal events occurring in the lake.

Spring runoff coincided with some of the highest total phosphorus concentrations in both years. Rising levels of total phosphorus in spring were coincident to a large extent with an increase in the contribution of particulate phosphorus. We attribute this effect to the transport of large amounts of particulate material from the watershed at the time of runoff. Silt, which is an effective

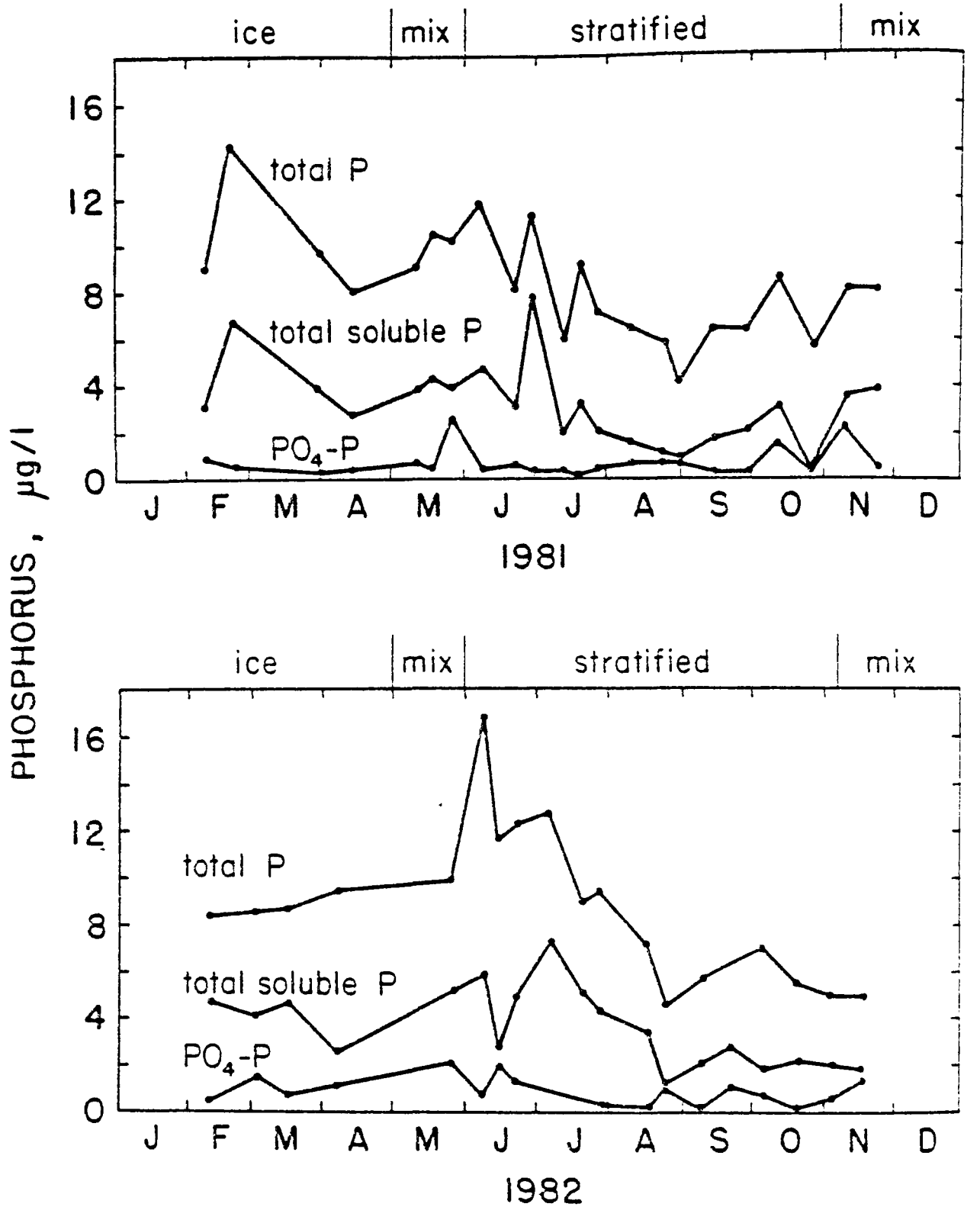


Figure 22a. Total phosphorus and phosphorus fractions in the top 5 m of Lake Dillon.

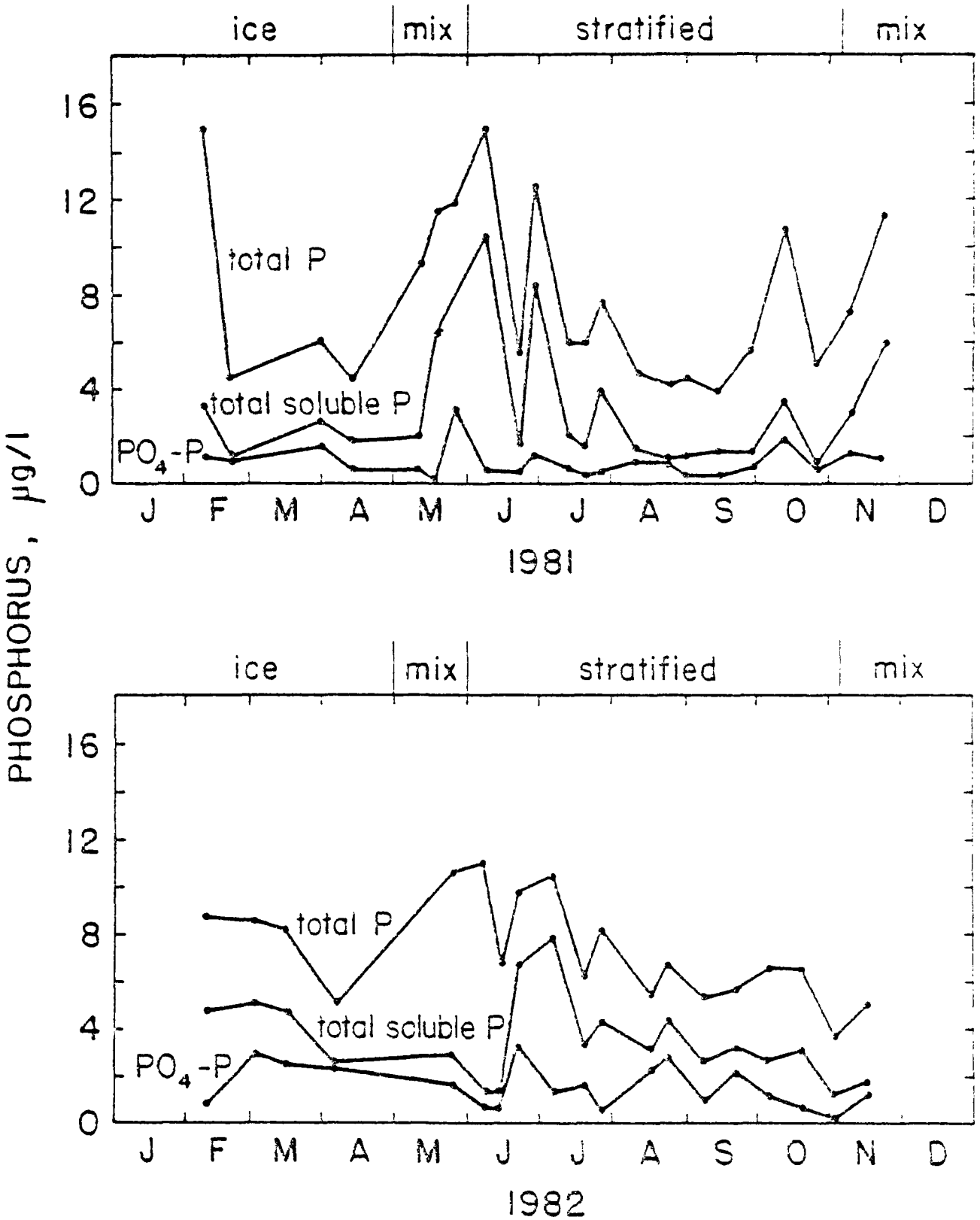


Figure 22b. Total phosphorus and phosphorus fractions in deep water (40-45 m) of Lake Dillon.

adsorption substrate for phosphorus, is particularly likely to make a major contribution to the particulate fraction of phosphorus at this time of the year. The concentrations declined rather quickly, however, and in July reached levels that were more or less average for the year. The peak in total phosphorus associated with runoff was considerably larger in 1982 than in 1981, as would be expected from the relative amounts of runoff, and thus of particulate transport, in the two years. Inorganic particulates added with runoff can be expected to sediment rapidly after entering the lake. This accounts for the decline in total phosphorus of the surface waters after runoff has peaked. Particulate phosphorus was not brought to the surface from the lake bottom in significant amounts at the time of fall overturn, despite complete mixing of the water column. From August until ice cover returned at the end of December, the total phosphorus hovered in the vicinity of 6 to 8 ug/l, most of which was particulate and was undoubtedly tied up in phytoplankton and bacteria.

Orthophosphate was seldom a major contributor to total phosphorus in either 1981 or 1982. For the most part, PO_4-P levels were in the vicinity of 1 ug/l at the surface and were somewhat higher near the bottom during stratification (but not as an annual average: Table 15). Higher PO_4-P at depth during stratification probably reflects lower biological demand there. Because of uncertainties as to the ability of the standard molybdate test to distinguish between true orthophosphate and small organic phosphorus compounds, it is quite possible that the actual PO_4-P was even lower than the apparent PO_4-P as indicated by the molybdate test.

Orthophosphate concentrations occasionally rose significantly in the surface water. In May of 1981, orthophosphate rose to about 4 ug/l, a concentration well above the annual average. It seems almost certain that this rise in PO_4-P was caused by spring mixing, when the winter phytoplankton had been dispersed from the surface and the spring phytoplankton had not been able to start surface accumulation because of the movement of the entire water column. Unfortunately there was only one May sample for 1982 because of a ring of ice around the lake which prevented launching of the boat during most of May.

There were also some minor but consistent increases in the amount of surface PO_4-P between September and December of both years. These probably were caused by the incorporation of hypolimnetic water, with higher PO_4-P, into the mixed layer as the thermocline descended.

Soluble organic phosphorus made a significant contribution to total P at all times of the year except at the end of the stratification season. There was a definite and consistent decline in the total amount and percentage of soluble organic phosphorus at the surface between the first of July and the end of August both years. This was almost certainly caused by strong biological demand for phosphorus, especially after the establishment of large phytoplankton populations by the end of June. Bacteria may have been involved as well, since the downward trend was also notable below the thermocline. Over a period of time, the phytoplankton and bacteria populations seemed to be able to sequester most of the soluble phosphorus, including the organic fraction, in the form of biomass. Alkaline

phosphatase on cell surfaces may account for this ability to tap the soluble organic pool (Fogg 1975, Nalewajko and Lean 1980).

Table 17 summarizes the historical information on surface total P in the main body of the lake. The late summer values show no evidence of change, but the early summer values trend upward. The initial spring phosphorus, which is responsible for the growth burst of early summer, has increased, but the epilimnetic supply is ultimately depleted by sedimentation to much the same levels now as in 1975, according to the table.

	Total P ug/l			
Year	June	July	August	September
1975[1]	-	-	6	9
1977[2]	5	6	60[4]	-
1978[2]	7	6	6	-
1981[3]	11	7	6	6
1982[3]	12	10	6	6

[1] National Eutrophication Survey
[2] Summit County/EPA study
[3] Present study
[4] Probably erroneous

Table 17. Comparison of total P values from various studies of Dillon (main body of the lake, surface samples).

Nitrogen in Lake Water

Table 18 summarizes the information on total nitrogen and nitrogen fractions at the index station. As would be expected in view of the presence of substantial amounts of dissolved oxygen at all depths throughout the year, the amounts of nitrite were exceedingly low, and nitrate was the dominant soluble inorganic form. There was a substantial depth gradient of average nitrate concentrations in both

epth tratum (m)	Concentration - ug/l													
	Nitrite N		Nitrate N		Ammonia N		Soluble Organic N		Total Soluble N		Partic. N*		Total N	
	1981	1982	1981	1982	1981	1982	1981	1982	1981	1982	1981	1982	1981	1982
0-5	1.5	2.5	73	67	19	15	127	306	221	391	28	30	248	422
5-10	1.7	1.9	76	66	14	19	104	384	196	472	28	30	224	502
0-15	1.2	2.3	74	67	13	15	104	216	192	300	28	30	220	329
5-20	1.2	2.3	83	79	13	22	135	286	232	388	28	30	260	418
0-25	1.2	3.8	101	106	18	33	109	165	230	308	28	30	258	338
5-30	1.3	4.3	118	139	18	32	107	196	245	372	28	30	273	402
0-35	1.2	5.0	155	174	15	30	108	243	280	452	28	30	308	482
5-40	1.1	3.6	166	188	13	26	126	252	311	470	28	30	339	500
0-45	1.1	2.9	187	224	13	26	113	227	315	480	28	30	343	510
5-50	1.3	2.8	202	259	13	16	222	259	438	537	28	30	466	565
lean	1.3	3.1	123	137	15	23	126	253	266	417	28	30	294	447
lean %	.4	1	42	31	5	5	43	57	91	93	10	7	100	100

Values below 5 m set equal to annual mean for 0-5 m

able 18. Average total nitrogen and nitrogen fractions at different depth strata in 1981 and 1982 (index station). Slight inconsistencies in the sums are due to occasional missing values for individual fractions.

years. The seasonal analysis will show that this was caused by the removal of nitrate nitrogen by phytoplankton in the surface waters during the summer. In 1982 the nitrate concentrations were very similar in the upper half of the water column to the 1981 concentrations, but were consistently somewhat greater in the lower water column than in 1981. Nitrate nitrogen accounted for most of the soluble inorganic nitrogen pool both years and made up 30-45% of the total nitrogen.

Ammonium nitrogen was present in measurable but very low amounts, accounting for only 5% of total nitrogen. Preferential uptake of ammonium by phytoplankton and continual oxidation of ammonium by microbes undoubtedly maintained the equilibrium ammonium concentrations at a low level.

Soluble organic nitrogen was always an important component of the total nitrogen pool. In 1981 its percentage contribution to total nitrogen was slightly lower than in 1982; the grand average for the two years was about 50%. Absolute levels of soluble organic nitrogen in 1982 were considerably higher at all depths than in 1981. This is probably explained by the larger amount of water entering the lake in 1982. It will be evident from the analysis of stream chemistry in later portions of this report that stream water entering Lake Dillon contains large amounts of soluble organic nitrogen, much of which is probably quite refractory, and that the concentrations increase with amount of runoff.

Table 18 shows that soluble nitrogen, including both organic and inorganic fractions, accounted for slightly more than 90% of the total

nitrogen as an annual average. Particulate nitrogen was only measured in the surface water (0-5 m); the amounts at other depths have been set equal to the mean in the surface water. This is only an approximation, but errors associated with the approximation are not likely to be important in the computation of total nitrogen because of the small contribution of particulate nitrogen to the total. Total nitrogen, which is shown in the last two columns of Table 18, was higher in 1982 than in 1981, as would be expected from the patterns in the most important nitrogen fractions.

Table 19 summarizes the results of a two-way ANOVA similar to the one that was carried out on the phosphorus data. The ANOVA confirms statistically the suspected significance of the difference between years in concentration of soluble organic nitrogen and total nitrogen. The difference between years in nitrate was not significant because it appeared only in deep water rather than over the entire water column. The ANOVA confirms statistically the differences in mean nitrate concentration with depth.

	NO_3-N	NH_4-N	Soluble Organic N	Total N
Difference bet. depths	Highly significant**	Not significant	Not significant	Not significant
Difference bet. years	Not significant	Not significant	Highly** significant	Highly** significant

** $p < 0.01$

Table 19. Results of a two-way ANOVA testing for differences in means between depths (0-5 m versus 40-45 m) and between years (1981 versus 1982).

Figure 23 shows the seasonal changes in concentration of all nitrogen fractions at the surface of the lake. Some missing values for individual fractions have been filled in by linear extrapolation. Figures 24 and 25 show the complete details over time and depth of the distribution of nitrate, which is the most informative nitrogen fraction from the viewpoint of phytoplankton nutrition. Figure 23 shows that ammonium was highest at the surface under ice. This probably reflects the lower demand for inorganic nitrogen and lower efficiency of nitrification at this time of year. Soluble nitrogen concentrations were highest in winter and spring. A spring increase was especially pronounced in 1982, probably because of the larger runoff that year.

The inorganic nitrogen fractions began a precipitous decline at the lake surface as soon as stratification occurred. There is little doubt that this was caused by phytoplankton demand for inorganic nitrogen. Complete depletion of inorganic nitrogen at the surface occurred by the middle of July in 1981 and by the beginning of August in 1982. In both years, inorganic nitrogen began to climb again in mid-September, in coincidence with major thickening of the mixed layer leading to complete breakdown of stratification. These trends in inorganic nitrogen are especially obvious from the time-depth diagrams for nitrate (Figures 24, 25). Both figures show very clearly that the nitrate depletion was associated specifically with the growth zone and did not occur below the thermocline where biological demand was much lower.

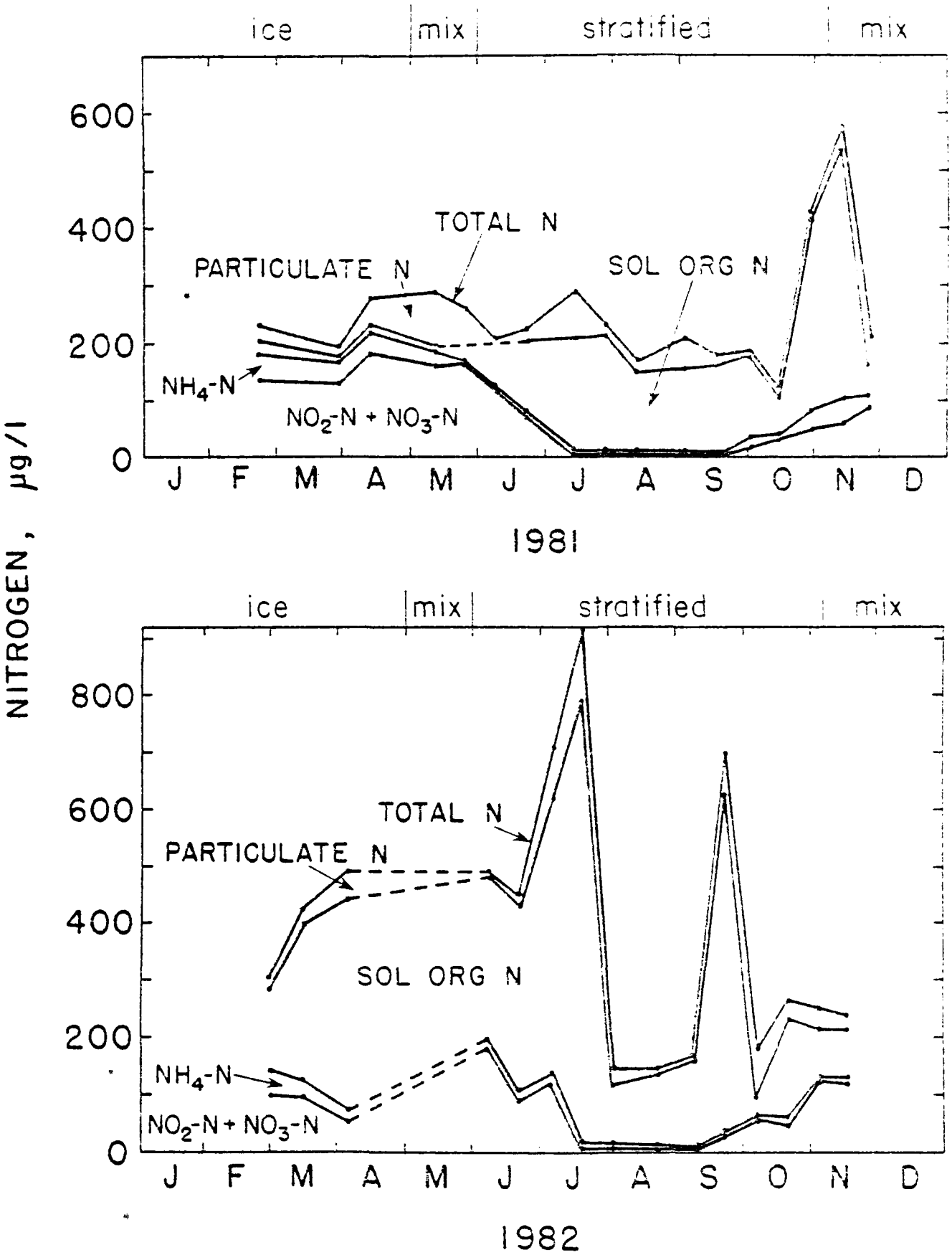


Figure 23. Total nitrogen and nitrogen fractions in the top 5 m of Lake Dillon.

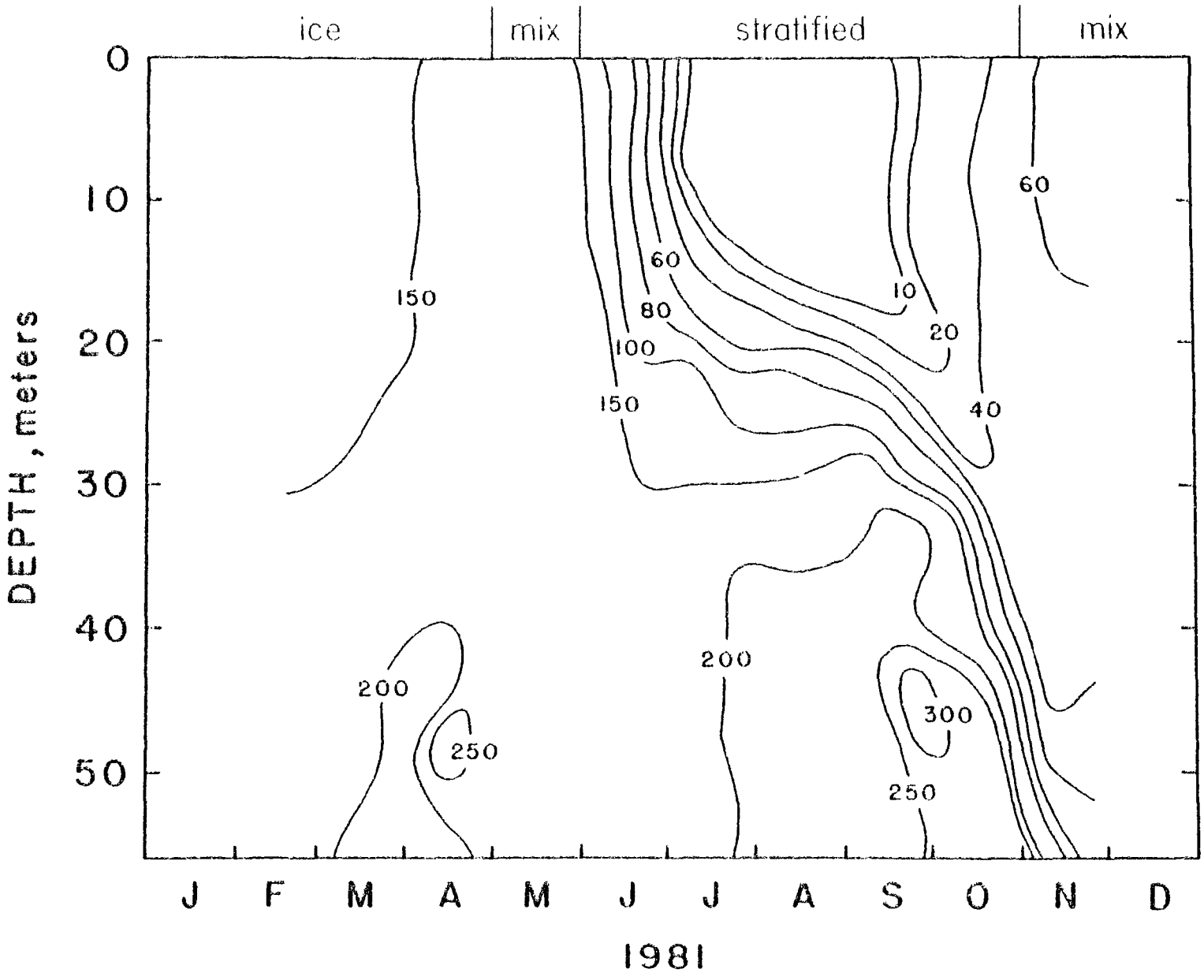


Figure 24. Time-depth diagram for nitrate in Lake Dillon, 1981 (ug/l).

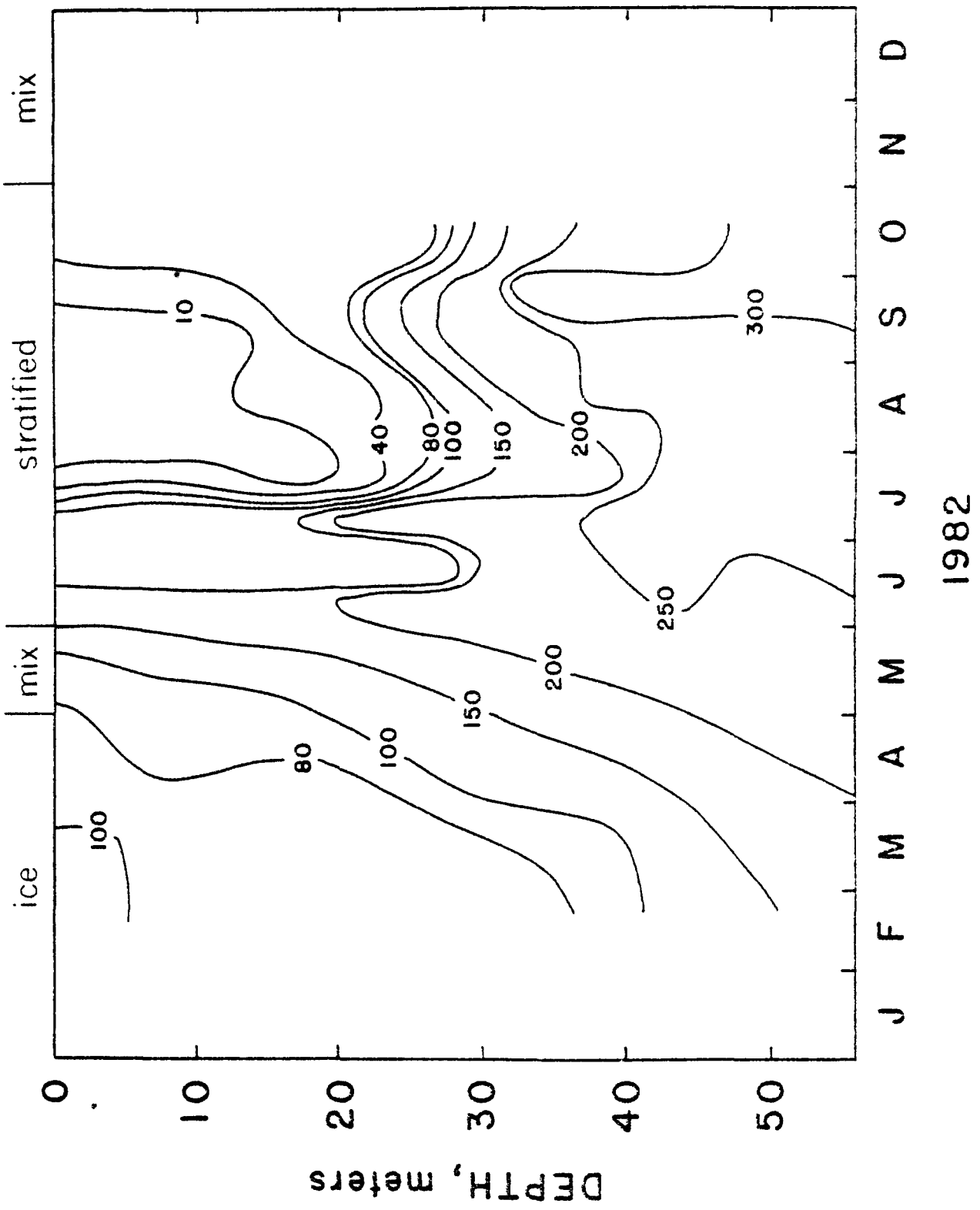


Figure 25. Time-depth diagram for nitrate in Lake Dillon, 1982 (ug/l).

Soluble organic nitrogen peaked very sharply in 1982 in coincidence with runoff, suggesting that a major amount of nitrogen entering with the peak runoff was in soluble organic form. In 1981, a year of low runoff, the organic fraction was lost, so we do not know the nature of the runoff spike.

During summer stratification of both years there was a reduction in the organic nitrogen pool at the surface. The decline was so drastic in 1982 as to suggest the importance of some abiotic processes, such as settling of flocculated organic matter. There was no evidence of a steady drain on the soluble organic nitrogen fraction as phytoplankton completely depleted the inorganic nitrogen. Thickening of the epilimnion in fall restored high levels of soluble organic nitrogen as soluble organic matter was brought to the surface from the hypolimnion.

No historical information is available on nitrogen in the lake. Total nitrogen was measured by the National Eutrophication Survey in 1975 but the nitrogen levels proved to be below their detection limit.

Table 20 summarizes the total nitrogen and total phosphorus data as an average over the entire water column for both years and shows the molar ratios of nitrogen to phosphorus. The ratio of nitrogen to phosphorus is exceptionally high. Municipal sewage typically has a ratio of 6 to 14, and most inland waters fall between 15 and 40 (Stumm and Baccini 1978). The very high ratios of Dillon are partly explained by tertiary treatment for P, which greatly raises the N to P ratio of

effluents. The ratio of N to P for phytoplankton biomass is typically

	1981	1982	Composite
Total N ug/l	294	447	370
Total P ug/l	6.6	7.1	6.8
Total N, umole/l	21	31	26
Total P umole/l	0.21	0.23	0.22
N:P ratio (molar)	100	134	118

Table 20. Summary of average N and P amounts for the whole water column at the index station.

between 10 and 17. Thus the ratios of Dillon are suggestive of phosphorus control. However, the differences in spatial distribution of nutrients and in the biological availability of the fractions of N and P require other and more direct evidence of phosphorus control.

Sediment Phosphorus

Sediment samples from eight different sites were analyzed for particulate phosphorus and interstitial phosphorus. The collection sites for sediment included the heads and mouths of the lake arms as well as the main body of the lake. The wet surface sediments from a given site were mixed and a subsample equivalent to 10 to 20 mg of dry material was dried to constant weight in a tared container and then analyzed for total phosphorus by the particulate phosphorus procedure described in the methods section of this report. Another subsample of wet sediment was centrifuged for 30 minutes at 8,000 rpm, after which 5 ml of the supernatant was withdrawn and analyzed for total soluble

phosphorus according to the procedures used for lake water. The results of these analyses are reported in Table 21.

There was surprisingly little variation in the percentage phosphorus per unit dry weight of sediment. Such variation as can be discerned in the data cannot be related in any way to particular lake arms or to particular depths. Thus it appears that the lake bottom can be treated as having 0.1% phosphorus on the average with random variation spanning about $\pm$0.02%. The percentage phosphorus in the dry sediments of the lake is not especially high. For example, the summary data of Brunskill et al. (1971) show an average for 23 Wisconsin lakes of 0.12% and for 30 eastern Ontario lakes of 0.11%.

The total soluble phosphorus in interstitial water was high by comparison with the overlying lakewater, as expected from the decomposition of organic matter liberating phosphorus in the sediments. These values, although much higher than lakewater, are not extraordinary, even for relatively unproductive lakes. For example, the summary by Brunskill et al. (1971) on interstitial water chemistry for unproductive lakes of the Canadian Shield shows a range of two orders of magnitude in the interstitial total soluble phosphorus and a mode of 220 ug/l P. In Lake Dillon the diffusion gradient between the interstitial waters, at about 135 ug/l of total soluble P, and the water column, at about 3 ug/l soluble P, is obviously quite steep. The sediments must therefore play some role in supplying phosphorus to the overlying water column, but there is no evidence that this internal source is in any way unusual in Lake Dillon.

Collection Site	Water Depth (m)	Sediment P % P in Dry Sediment	Interstitial P Total Soluble, ug/l
Lake Center (Index)	60	0.085	203
Mouth, Dillon Bay (Main)	32	0.102	113
Mouth, Snake Arm (Main)	33	0.092	103
Mouth, Blue Arm (Main)	30	0.103	152
Mouth, Tenmile Arm (Main)	33	0.115	132
Head, Snake Arm (Survey 2)	23	0.093	136
Head, Blue Arm (Survey 4)	18	0.124	132
Head, Tenmile Arm (Survey 7)	15	0.085	107
Mean		0.100	135

Table 21. Particulate P and interstitial P from surface sediments collected July 26, 1982 at eight sites on Lake Dillon.

Particulates and Phytoplankton Biomass

Total Particulates in the Water Column

The average values for total dry weight of particulates retained on Whatman GF/C filters are summarized in Table 22. The table shows that the upper water column had a considerably higher total particulate concentration than the lower water column in both years. The averages for depths below 30 m were about half the averages for the surface. In the upper 25 m, total particulates were considerably higher in 1982 than in 1981 as an annual average. In both years the upper 5 m was slightly higher than any other depth, a layer from 5 to 15 m was distinctively high but not quite so high as the top 5 m, and the stratum from 15 to 25 m was a transition from the higher concentrations of the upper 15 m to the lower concentrations of the bottom 30 m of the water column.

Variations in total particulate concentrations with depth and between years are not really understandable without some detailed consideration of seasonal variation. Figures 26 and 27 show time-depth diagrams of total particulate concentration, from which a number of major seasonal variations are evident. Total particulate concentrations under ice were low at all depths. In general, inorganic particulates settle out very thoroughly under ice because of the lack of turbulence and very small amounts of incoming water. Thus particulates that are present under ice are likely to be dominated by phytoplankton. Fine organic debris and bacteria may make small contributions, but microscopic examination of samples showed that

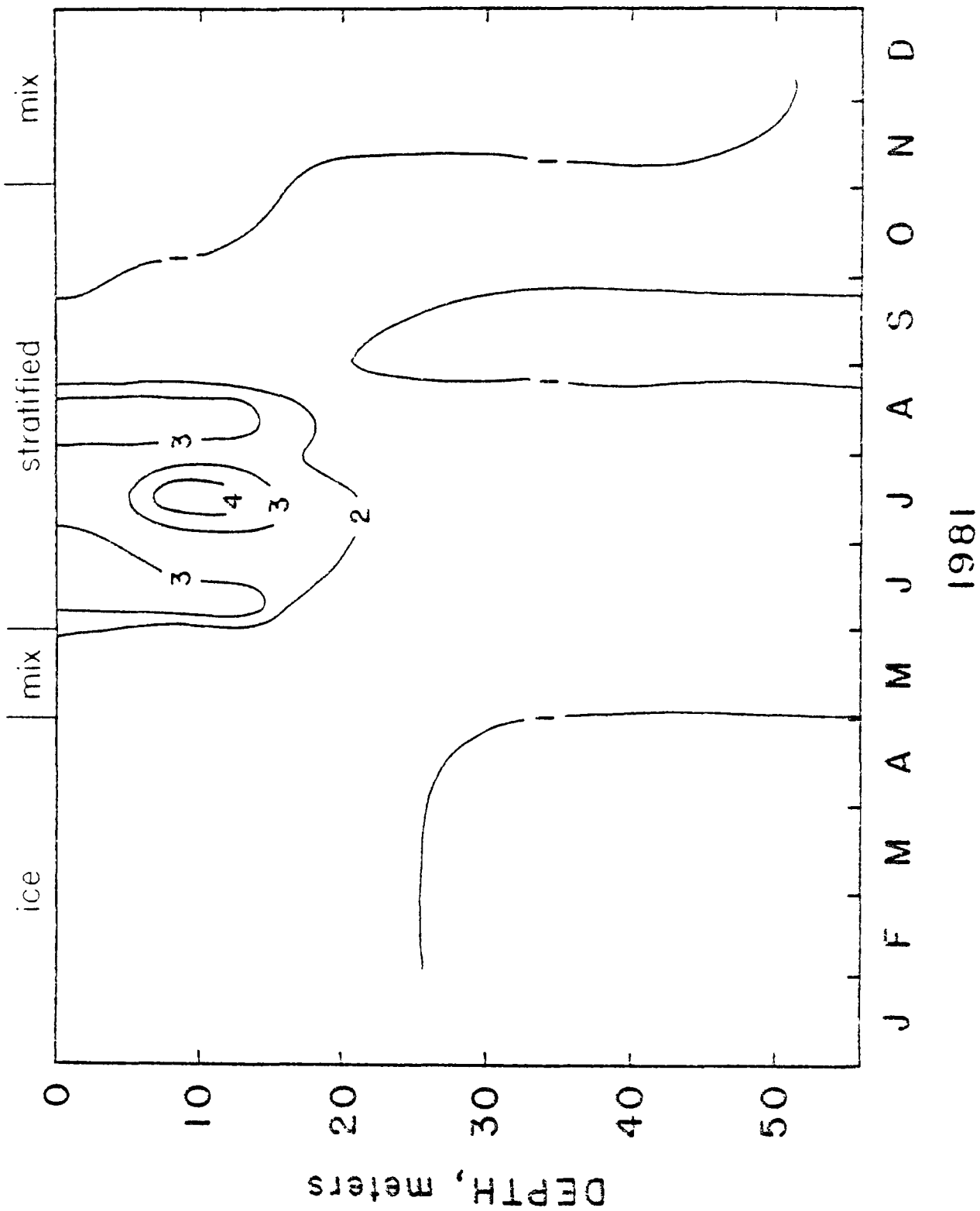


Figure 26. Time-depth diagram for total particulates in Lake Dillon, 1981 (mg/l).

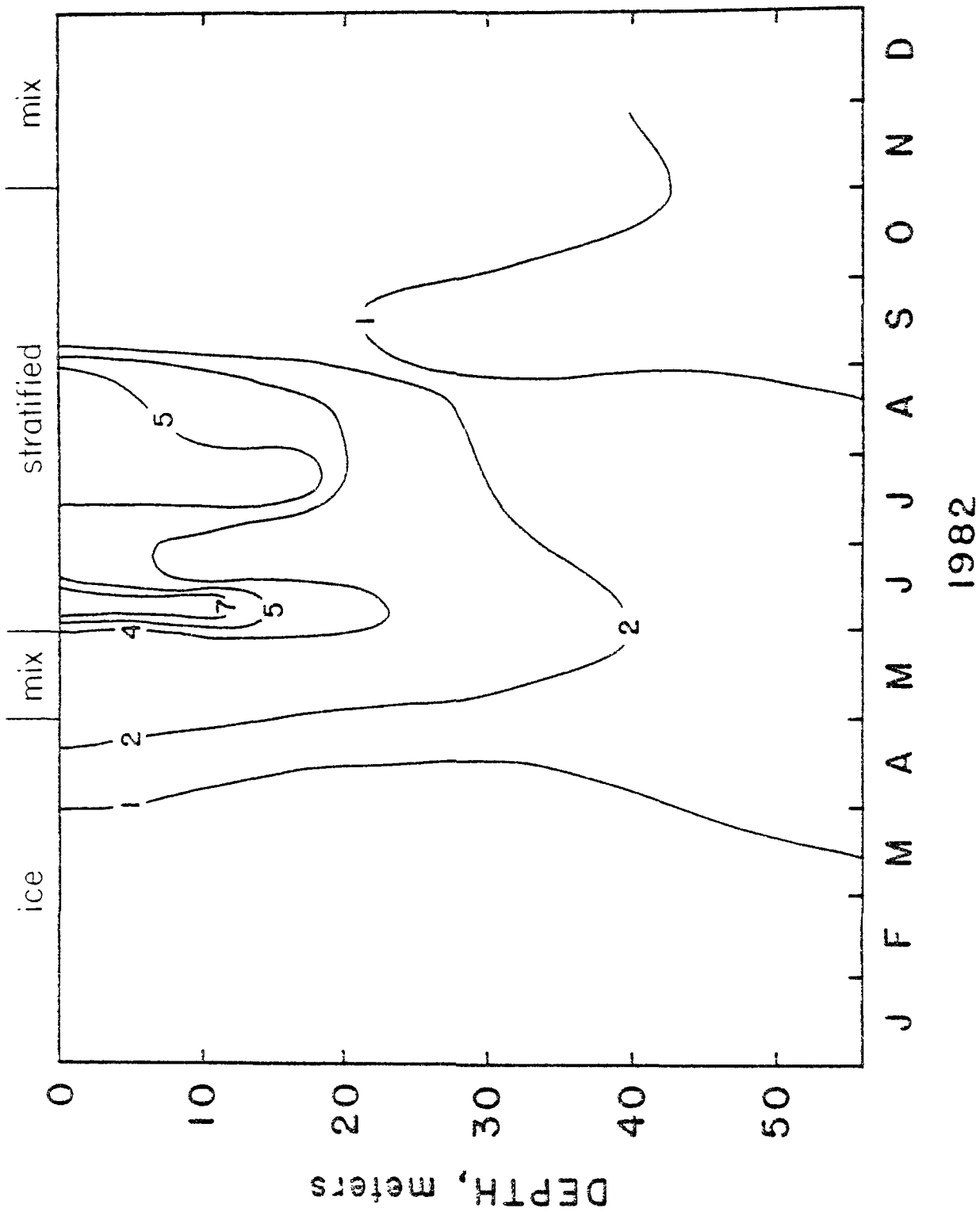


Figure 27. Time-depth diagram for total particulates in Lake Dillon, 1982 (mg/l)

Stratum (m)	Particulates - mg/l	
	1981	1982
0-5	2.23	3.85
5-10	2.12	3.39
10-15	2.14	3.64
15-20	1.77	2.84
20-25	1.59	2.31
25-30	1.44	1.62
30-35	1.23	1.57
35-40	1.19	1.27
40-45	1.42	1.98
45-50	1.24	1.52
50-55	1.66	1.40

Table 22. Mean total particulates of Lake Dillon in different strata of the water column (index station).

these did not compare with phytoplankton in accounting for particulates under ice in Dillon. As already noted, the ice was not covered completely with snow in the first months of the 1981 ice cover. The conditions for development of phytoplankton in the upper water column under ice were thus exceptionally good. This is reflected in somewhat higher total particulate concentrations in the upper water column under ice in 1981. Phytoplankton also dominated particulates under the ice in 1982 but were simply not abundant enough to create high particulate levels. As shown in Figures 26 and 27, total particulates in the upper water column under ice reached concentrations two to three times as high in 1981 as in 1982, although the total particulates in the lower water column where phytoplankton were not present were comparable to those in 1982.

During spring mixing, the winter phytoplankton were dispersed vertically and inorganic particulates from the watershed were added in increasing amounts. The stratification period, beginning in early June, was marked in both years by two peaks of particulate concentration occurring in the epilimnion. The first peak coincided with runoff, and was caused by the entry of large amounts of inorganic particulates from the watershed. As runoff declined, the entry of inorganic particulates declined and settling outstripped the addition of particulates. At the same time, as shown previously by Figure 19, incoming water from the rivers passed to greater depths as the summer progressed. Inorganic particulates entering deeper water would settle more quickly because of lower turbulence in deeper water. The peak for total particulates during runoff was considerably higher in 1982 than

in 1981, as would be expected. In both years there was a second peak of surface particulates in July after the peak runoff passed. The second peak of particulates was of essentially the same magnitude as the earlier one associated with runoff. The second peak was caused by the buildup of phytoplankton biomass and not by inorganic particulates. This was confirmed by microscopic examination of samples and by chlorophyll analysis.

The amounts of total particulates in deep water during stratification were low by comparison with the surface values. This is principally explained by the smaller amount of turbulence in water below the thermocline, which would promote more rapid settling in deep water. In 1982 the total particulate levels below the thermocline were consistently greater than in 1981, as would be expected from the entry and settling of greater amounts of inorganic particulate materials in 1982 with the greater runoff that year.

In both 1981 and 1982 there was a pronounced decline of total particulates in the surface water beginning in the last half of July. This was caused by a decline of phytoplankton biomass that time. This phenomenon will be discussed later at greater length. Fall mixing established relatively uniform distributions of particulates as expected.

Chlorophyll a

The distribution of chlorophyll a over time and depth is shown in Figures 28 and 29. There was a significant amount of chlorophyll under

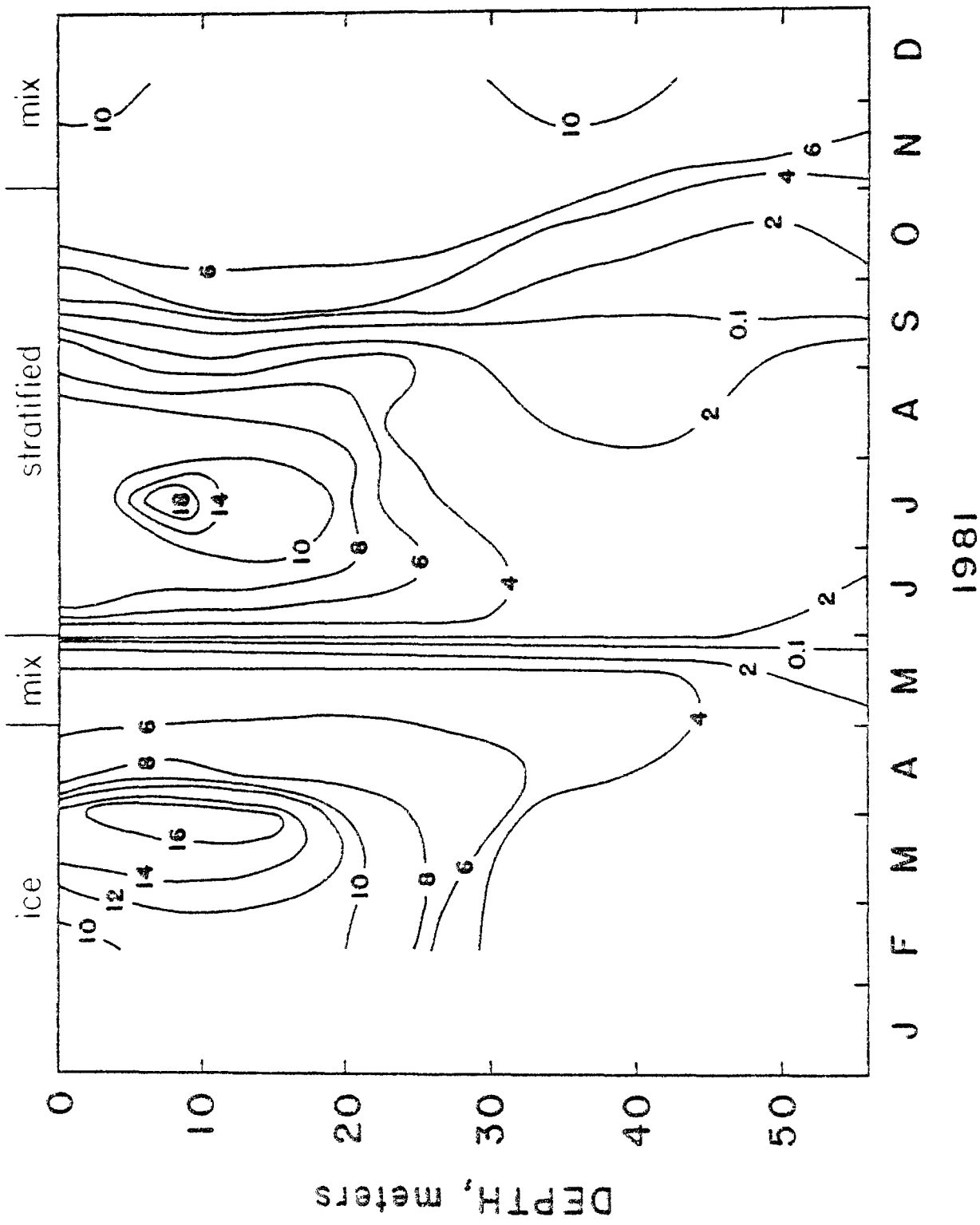


Figure 28. Time-depth diagram for chlorophyll a in Lake Dillon, 1981 (ug/l)

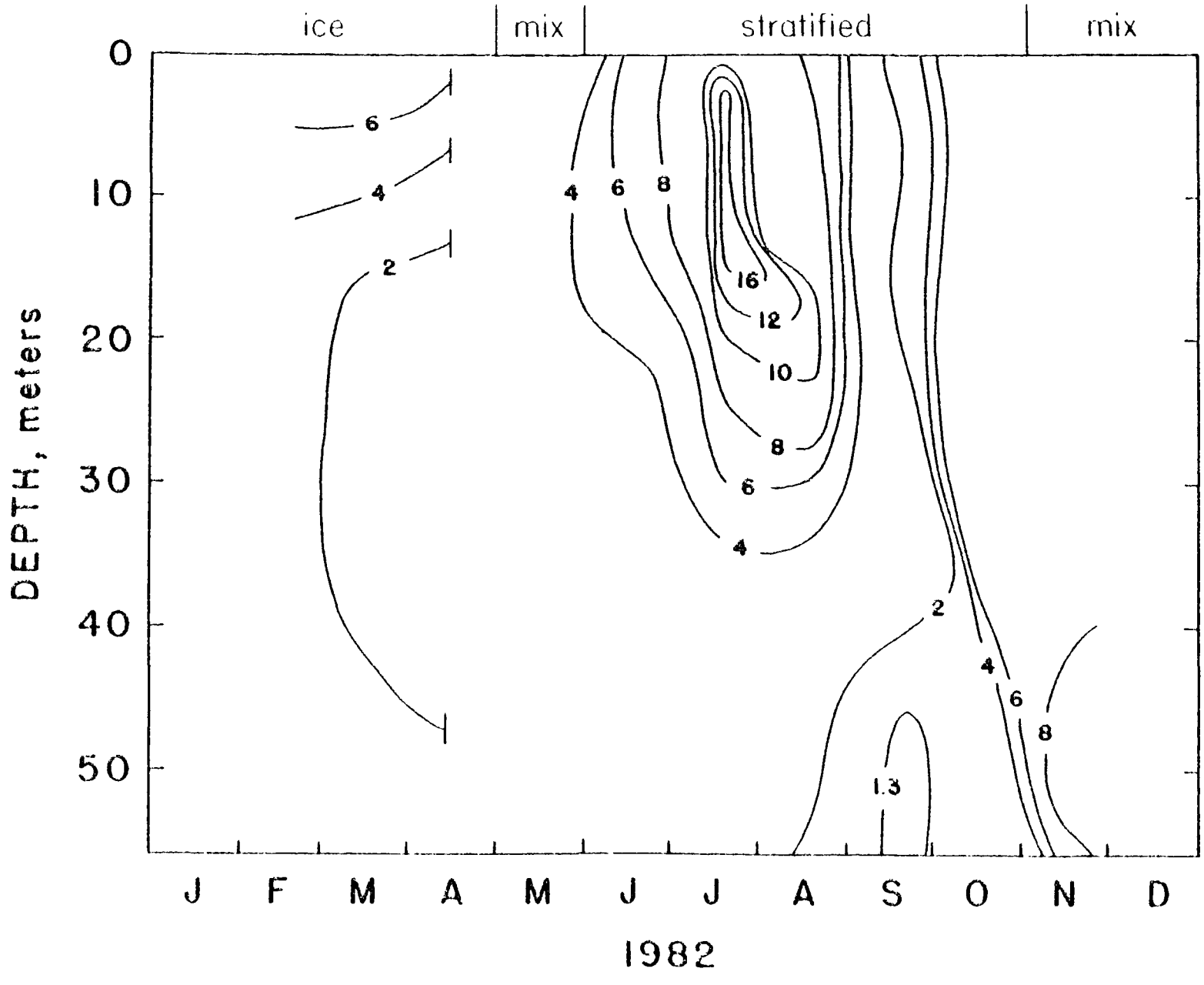


Figure 29. Time-depth diagram for chlorophyll a in Lake Tahoe, 1982 (ug/l)

ice in both years, but the amount was much higher in 1981 than in 1982 because of the lack of snow cover and transparency of the ice in January and February of 1981. In fact a pronounced chlorophyll peak developed under ice in 1981, as shown in the time-depth diagram. The chlorophyll under ice in 1982 should probably be regarded as more typical, since the large amount of light reaching the water in January and February of 1981 was only possible because of the very unusual weather conditions that year.

The spring mixing in May of 1981 drastically reduced the surface chlorophyll to a concentration well below 1 ug/l. Two factors contributed to this decline. First, there was a decline in surface chlorophyll that began well before the ice melted. This may have been caused either by light deprivation (due to light extinction by spring snow) or to nutrient depletion. An attempt to separate these effects will be made in a later chapter. Secondly, the decline was accentuated suddenly at the time of melt by the mixing of surface waters, already low in chlorophyll, with deeper water having essentially no chlorophyll. This was mainly a dilution effect. Dilution may have been accentuated, however, by light deprivation due to deep mixing, which moved the algal cells to such great depths that they spent much time in the dark. This effect would have been short-lived, however, since incipient layering quickly began to impede mixing.

The light deprivation phenomenon can be shown more clearly by use of the concept of effective light climate (Ramberg 1976, 1979):

$$\bar{I} = I_o' \frac{(1-e^{-\eta_t z_m})}{\eta_t z_m}$$

where $\bar{I}$ is effective light climate (mean exposure of the average cell in the mixed layer, expressed as langleys per day PAR), z_m is the thickness of the mixed layer (adjusted for morphometry), η_t is the extinction coefficient (m^{-1}), and I_o' is penetrating PAR (langleys per day, PAR, with surface correction). In May, during mixing on Dillon, z_m was about 25 m and η_t was about 0.5 m^{-1}. On a cloudy day I_o' would have been about 145 ly/day, and on a bright day it would have been about twice this amount. Thus for a cloudy day $\bar{I}$ would have been 11 ly/day and for a bright day it would have been 22 ly/day. The presumed threshold for phytoplankton growth is about 10-20 ly/day, so it is clear that really rapid growth could not have occurred until there had been a change in η_t or z_m. Spring changes in η_t were actually upward, reducing $\bar{I}$. Thus growth could only have occurred through a major change in z_m. Stratification accomplished the needed change in z_m, bringing $\bar{I}$ into the range 50-120 ly/day in early June.

The degree of decline at the time of spring mixing in 1982 was not evident because samples could not be taken during the month of May, but in all probability it was very similar to the decline observed in 1981. In both 1981 and 1982, as expected from effective light climate, there was a rapid resurgence of chlorophyll concentration as soon as the water column stabilized in June. As is typical in a dimictic lake, the conditions for phytoplankton growth just after establishment of stratification in the spring were very good because available nutrient concentrations were at a maximum as a result of recent complete mixing of the water column, while daylength and water temperature were

increasing and the phytoplankton cells were held within the lighted zone by the presence of a thermocline.

In both years the chlorophyll concentrations reached their peak in the first half of July. In 1981 the increase in chlorophyll was more or less uniform over the top 20 m until the first of July, and thus extended through the epilimion into the metalimnion. Subsequent to the first of July, increases continued in the 5-10 m stratum, but not in the upper 5 m. Since the 5-10 m stratum incorporated the lower half of the epilimnion and the upper meter or two of the metalimnion, the annual maximum chlorophyll may have been partially metalimnetic. Metalimnetic maxima are common in stratified lakes (e.g., Fee 1976, Heaney and Talling 1980, Moss 1972), since light often penetrates through the epilimnion and into the metalimnion in sufficient amounts to support phytoplankton growth. Often nutrients are depleted first from the epilimnion, but growth continues in the metalimnion for some time subsequent to the epilimnetic depletion, thus producing a metalimnetic maximum. The sunlight reaching Dillon on a clear day is about 50% greater than at the same latitude at sea level (Hutchinson 1957). This and the minimal cloud cover over the lake tend to promote metalimnetic growth.

In 1982 the summer chlorophyll peak coincided almost exactly with the timing of the similar peak in 1981, but the chlorophyll maximum was more evenly distributed over the top 15 m. As in 1981, phytoplankton were present in quantity well below the epilimnion, to a depth of about 20 m. Both the transparency data (Figure 16) and the primary production data (see below) indicate that growth was mainly

confined to the upper 10 m, however, so the cells below this had probably grown in the upper 10 m and had sunk to greater depths. The more even vertical distribution over the top 10 m in 1982 is probably explained by the larger runoff in 1982 and the tendency of runoff to enter higher in the water column in 1982. In 1982 significant amounts of nutrients continued to enter the mixed layer well after the first two weeks of June (Figure 12, Figure 19), and this may have retarded the occurrence of nutrient depletion in the uppermost portion of the water column.

The size of the chlorophyll maximum was very similar in the two years. The maximum chlorophyll per unit volume was slightly higher in 1981 than in 1982, but the maximum chlorophyll at the lake surface (0-5 m) and the distribution of the maximum chlorophyll levels over depth were both greater in 1982 than in 1981. Thus the surface transparency of the lake was considerably less during July of 1982 than in July of 1981, despite the similarity in maximum chlorophyll concentrations of the two years. Because 1981 was a much more unusual year hydrologically than 1982, we would view the 1982 condition as the more typical.

In both years a chlorophyll decline began in the middle of July. We attribute this decline to nutrient deficiency. In both years July was the month when available phosphorus and available nitrogen were first reduced to their summer minimum levels (Figures 22, 23). The disappearance of critical macronutrients is only circumstantial evidence of the nutrient status of phytoplankton, however, because of

	1981		1982	
	Amount	Time	Amount	Time
Maximum chlorophyll concentration on any day or depth (ug/l)	19.8	13 July	17.9	19 July
Maximum summer chlorophyll concentration on any day, surface (0-5 m, ug/l)	9.7	10 Aug	17.4	19 July
Maximum summer chlorophyll concentration monthly average, surface (0-5 m, ug/l)	9.3	June	11.7	July
Average chlorophyll, summer, 0-5 m (post runoff stratification)	6.7	Jul-Oct	7.3	Jul-Oct

Table 23. Information on chlorophyll in 1981 and 1982.

the ability of phytoplankton to store significant amounts of macronutrients (Lund 1965). The hypothesis that the July decline in phytoplankton biomass represents the effect of nutrient stress will be further supported by other types of evidence, especially enrichment studies, to be presented in later parts of this report.

The decline of chlorophyll beginning in mid-July was different in the two years. In 1981 the decline of chlorophyll concentrations was steady at all depths. In 1982, however, the decline was steady in the top 10 m but there was a very deep maximum of chlorophyll centered at about 25 m in the first half of August. The transparency data for the same time interval show that significant photosynthesis cannot have occurred so deep in the water column (1% light at 10-15 m: Figure 16). We therefore conclude that the 25-m maximum represents sinking of viable but nongrowing phytoplankton cells that formerly belonged to the surface maximum. In fact this sequence of events is suggested by the shape of the curves on the time-depth diagram in Figure 29.

In the first half of September 1981 and 1982, there was a chlorophyll minimum. As in the spring, the minimum had a dual cause. There was a steady decline in chlorophyll until the end of August prior to mixing. This was due to severe nutrient depletion, as will be shown in the analysis of production and enrichment studies. The quick thickening of the mixed layer in September accentuated this minimum by dilution. The dilution renewed the nutrient supply, and thus led to a quick resurgence of chlorophyll from the minimum. As the epilimnion continued to thicken toward complete mixing in November, growth was

sustained by continual incorporation of available nutrients from deep water, and this explains the steady increase in chlorophyll after mid-September.

The increase in chlorophyll after mid-September was caused by relief of nutrient deficiency, but this relief was accompanied by declining light availability. Available light ($\bar{I}$, effective light climate) would have reached a minimum with complete mixing in November. In fact there was a definite slowing of the chlorophyll increase after October. If the entire water column were mixing day after day, $\bar{I}$ could not have exceeded 30 ly/day, which is near the threshold of growth. Only strong winds will mix such a deep water column, especially in the daytime. Thus the mixing can be sporadic even when water column stability is essentially zero. This phenomenon probably allowed growth in the upper 15 m. Algae in deeper water did not grow but remained viable. The combination of growth in the upper water column and viability below under sporadic mixing allowed chlorophyll to increase even after the water column lost thermal structure, especially in 1982.

Phytoplankton Composition and Seasonality

The phytoplankton composition of Lake Dillon has never been thoroughly studied in the past. The present analysis of phytoplankton composition and seasonality in Lake Dillon is based on examination and quantitative counts from sedimented whole samples, in which phytoplankton cells of all sizes were visible.

Table 24 summarizes the composition of the phytoplankton, and Figure 30 shows some of the more common taxa. Rare species are omitted from the table, and the remaining species are divided into categories according to their abundance. The dominant phytoplankton taxa are from three groups: blue-green algae, diatoms, and microflagellates.

By far the most important blue-green, and possibly the most important alga in the lake on an annual basis, is Synechococcus lineare. This species is often referred to as Rhabdoderma lineare in the literature, but has recently been renamed by Komarek (1976). The cells of Synechococcus are very small; in Lake Dillon, their diameter is very uniform at about 1 um. The cell length is more variable, but averages about 15 um. The cells are solitary or joined in pairs or units of 3 or 4, particularly during rapid growth, probably as a result of recent cell division. Because of its small size, this alga is frequently overlooked completely in quantitative counts, even when it is quite abundant. As a result, very little is known of its biology.

Lyngbya limnetica, the second blue-green that ranks as a dominant for Lake Dillon, was considerably less abundant than Synechococcus on an annual basis but did reach the top two species in abundance at certain times. This species of Lyngbya is characterized by extremely tiny elongate cells having dimensions and shape very similar to those of Synechococcus. The cells are united in filaments, which typically range from 5 to 15 cells. The filaments are surrounded by a sheath, which is characteristic of the genus. This species is extremely

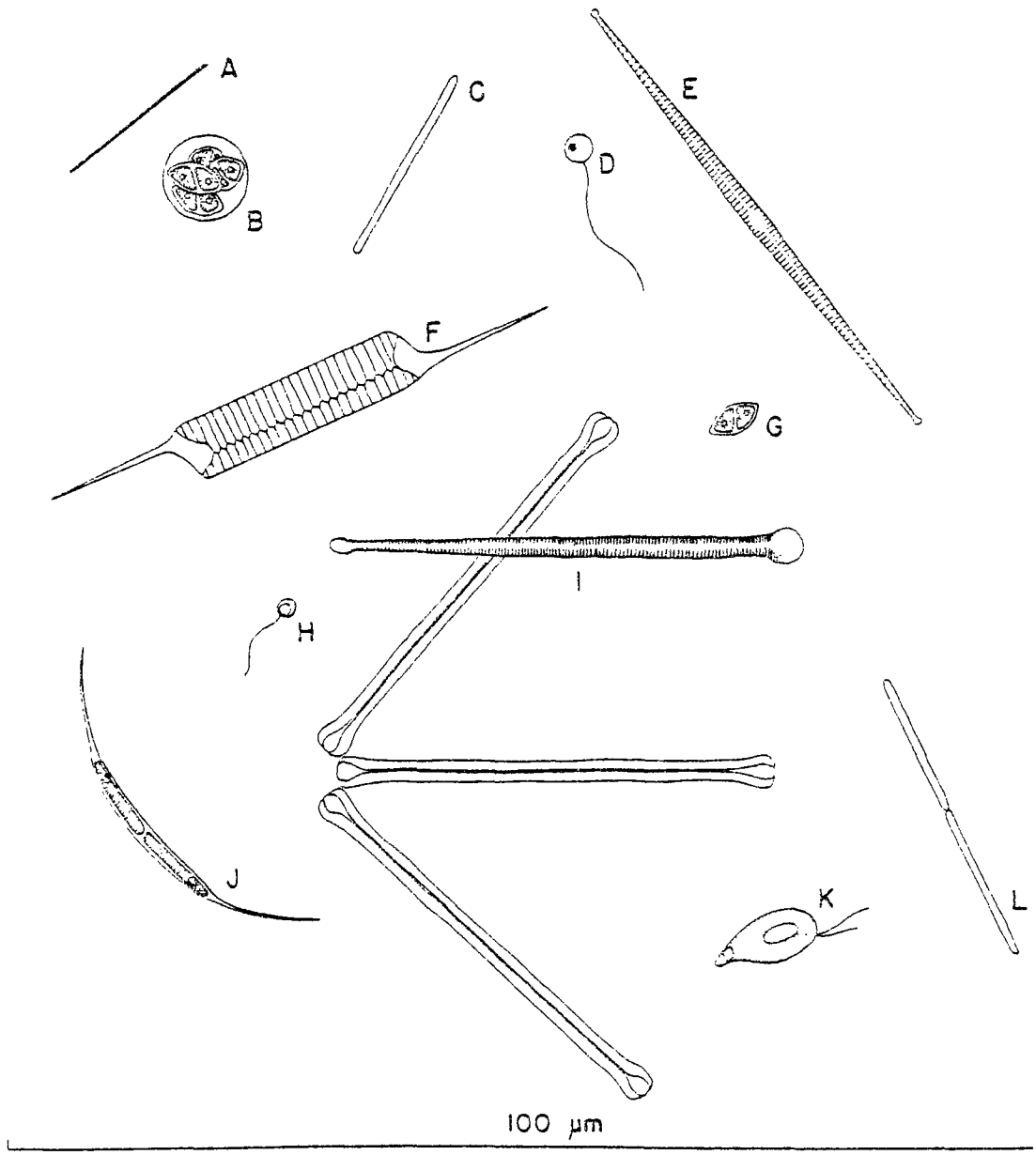


Figure 30. Scale drawings of Dillon phytoplankton: Lyngbya (A), Oocystis (B, G), Synechococcus (C, L), microflagellates (D, H), Synedra (E), Rhizosolenia (F), Asterionella (I), Monoraphidium (J), Rhodomonas (K).

Taxon	Abundance*
Cyanophyceae	
Synechococcus lineare	D
Aphanothece nidulans	C
Lyngbya limnetica	D
Lyngbya sp.	C
Chlorophyceae	
Monoraphidium contortum	C
Monoraphidium setiforme	S
Scenedesmus granulatus	C
Scenedesmus ecornis	S
Oocystis parva	C
Kirchneriella obesa	C
Cosmarium sp.	S
Coccomyxa	C
Chrysophyceae	
Dinobryon petiolatum	S
Dinobryon divergens	S
Bacillariophyceae	
Asterionella formosa	D
Nitzschia sp.	C
Synedra radians	D
Rhizosolenia eriensis	D
Stephanodiscus sp.	C
Cryptophyceae	
Rhodomonas minuta	C
Cryptomonas erosa	C
Unclassified	
Microflagellates	D

* D = Dominant, i.e., ranking the top one or two species in abundance at certain times of the year; C = Common, i.e., ranking in the top 5 species in abundance at some time of the year; S = Secondary, i.e., reaching significant numbers but never ranking high in abundance.

Table 24. Summary of the phytoplankton composition of Lake Dillon, 1981-1982.

widespread, spanning the tropics to far north temperate latitudes, and is found in waters of widely divergent trophic status.

Neither of the two dominant blue-greens nor any of the other blue-greens listed in Table 24 is characterized by gas vacuole formation, nor is any of these species capable of nitrogen fixation. Large-celled vacuolate forms such as Microcystis, Anabaena, and large Oscillatoria were entirely absent in 1981 and 1982.

Three diatom species are classified as dominant in Table 24. Asterionella formosa is a large-celled colonial diatom common in the spring growth of temperate lakes in all parts of the world. Synedra radians, a solitary species, is also a common contributor to the diatom growth of lakes, and is distributed from the tropics throughout the temperate zone. Neither Asterionella nor Synedra is easily identified with any particular trophic state. Rhizosolenia eriensis, a major contributor to diatom biomass in 1981, is a much more unusual diatom dominant, although it is distributed in a wide variety of lakes at all latitudes. It may often be overlooked because of its extremely delicate frustule, which is almost invisible even under phase optics. The frustule is subject to dissolution even by relatively mild preservatives. In the preserved Lake Dillon samples, the frustules disappeared within a period of three weeks. The presence of the diatom was detected by the examination of fresh samples and, once its presence was known, counts could be made at any later time from the protoplast, which resists dissolution.

The third group of species contributing to the dominants of Lake Dillon phytoplankton consists of unclassified microflagellates. These

are extremely small Chrysophyceae and Chlorophyta that are generally recognized as important under certain conditions in temperate lakes but are virtually impossible to deal with taxonomically. The largest of these taxa in Dillon was about 8 um in diameter and the smallest was just 1 um in diameter.

The Chlorophyceae, Chrysophyceae, and Cryptophyceae were represented in the 1981-1982 Lake Dillon phytoplankton but did not contribute any dominants. Among the chlorophytes, Monoraphidium (formerly called Ankistrodesmus) was a steady contributor, as were the genera Scenedesmus, Oocystis, and Kirchneriella. Desmids were in general not well represented, although one unidentified Cosmarium species did make a significant appearance. A tiny spherical chlorophyte, Coccomyxa, was present at times. This is a widely-distributed contributor to the so-called u-algae (Lund 1961), which in Dillon would also include the microflagellates. Two Dinobryon species, both common contributors to coldwater phytoplankton, were observed frequently but never became very common. The genera Rhodomonas and Cryptomonas, which are almost universal contributors to the phytoplankton of lakes and are extremely broad in their tolerance of trophic and thermal conditions, made significant contributions but never reached dominance.

It is typical of dimictic lakes to show major seasonal changes in composition of phytoplankton, and Lake Dillon is no exception in this respect. It is also common for dimictic lakes to show considerable variation in phytoplankton composition from one year to the next, especially if basic environmental conditions are markedly different.

At the same time, there is typically a degree of repeatability in the seasonal sequence of composition from one year to the next (Lund 1971, 1979). Lake Dillon shows some major differences between 1981 and 1982, yet certain features were common to the two years. 1981 and 1982 probably approximate the maximum expected difference between two years in sequence because of the very unusual hydrologic conditions in 1981 and the development of large amounts of phytoplankton under the ice in that year.

Figures 31 and 32 are a simplified representation of the major changes in composition that occurred in 1981 and 1982 in Lake Dillon. The major chlorophyll peaks are shown from the chlorophyll time-depth diagrams, and the major contributors to these peaks are indicated. Significant but subdominant contributors are also indicated in parentheses.

The phytoplankton under ice in 1981 included large contributions by the delicate diatom Rhizosolenia, whose average abundance in the top 25 m reached levels of about 16,000 cells/cc. The tiny blue-green Synechococcus was also a major contributor numerically, although it is small and thus contributed less per cell to biomass (or to chlorophyll) than a number of other species (Table 25). Synechococcus reached abundances of about 6,000 cells/cc under ice in 1981. Smaller but significant contributions were made by the chlorophyte Kirchneriella (about 1200 cells/cc) and the diatom Asterionella, whose peak numerical abundance was much lower (500-600 cells/cc), but which contributed substantially to the biomass because of its larger cell size. Although

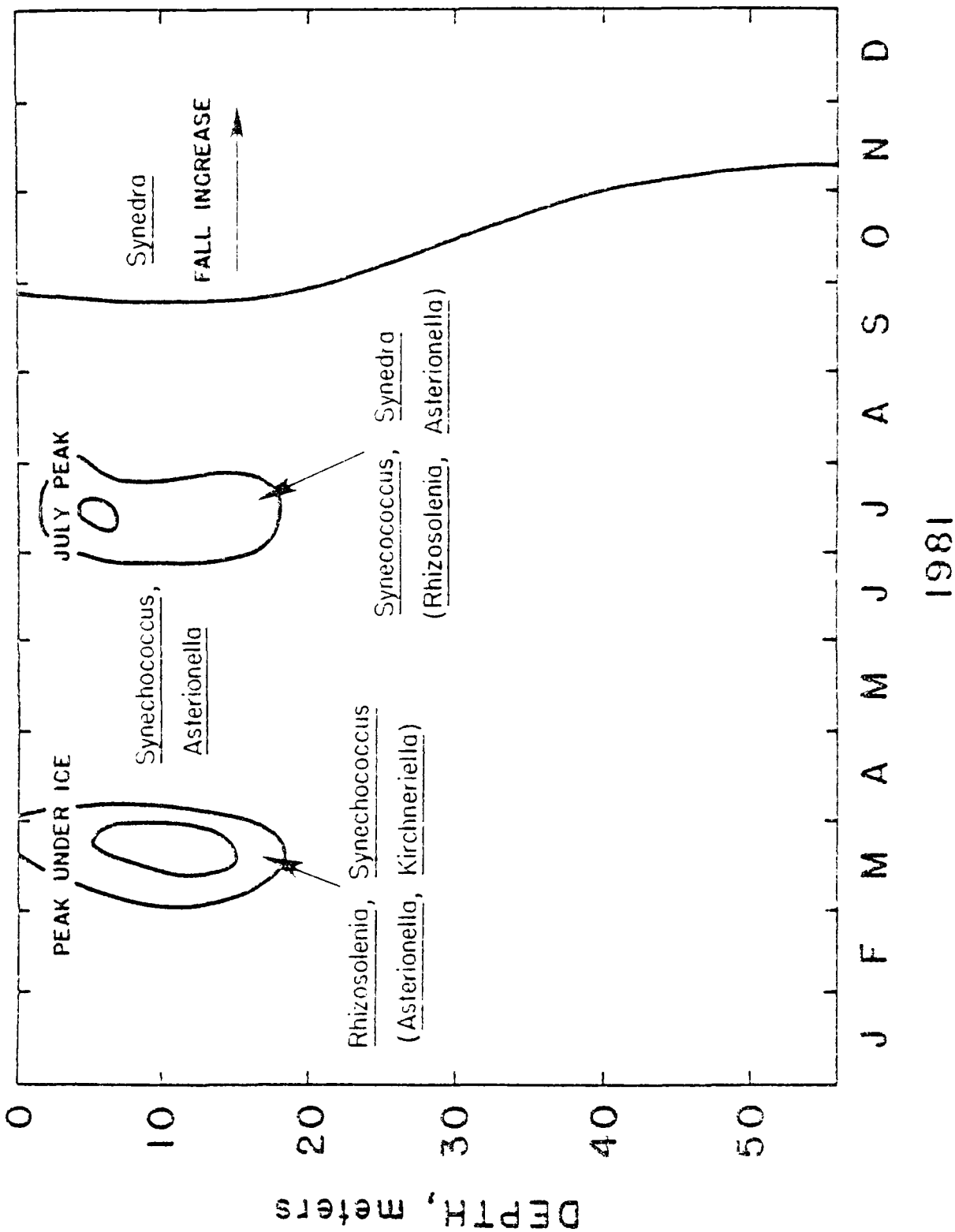


Figure 31. Simplified version of the chlorophyll time-depth diagram with the main contributions of algal taxa indicated. For peaks, the most abundant taxa are shown without parentheses and taxa of secondary rank are shown in parentheses.

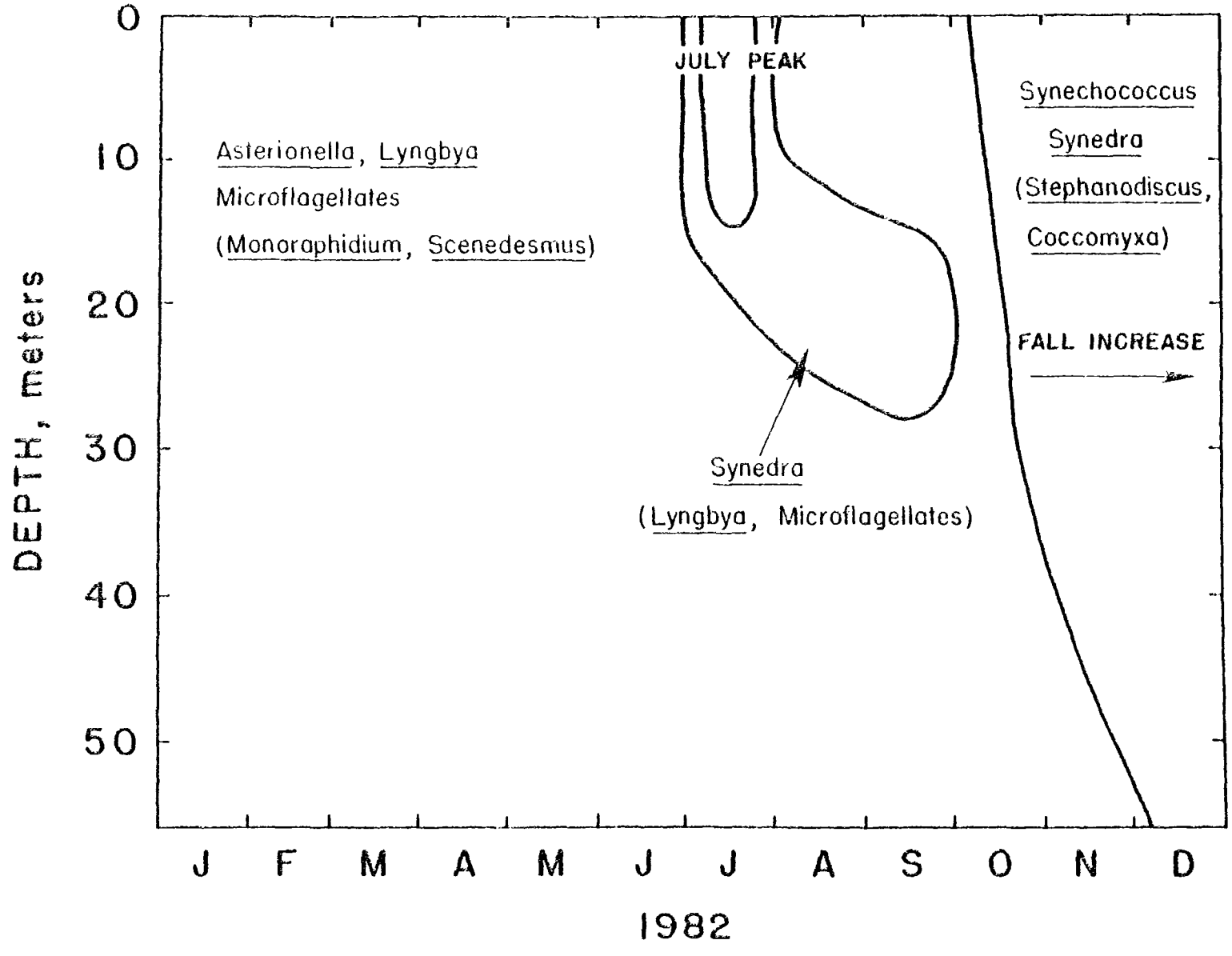


Figure 32. Algal abundance summarized as in Figure 31.

Taxon	Cell Volume um^3	Ratio to *Synechococcus*
Synechococcus	12	1.0
Lyngbya	7	0.6
Asterionella	400	33.
Synedra	300	25.
Stephanodiscus	150	13.
Rhizosolenia	25	2.1
Coccomyxa	0.2	0.017
Microflagellates	3	0.3

Table 25. Sizes of important Lake Dillon phytoplankton in absolute terms and relative to *Synechococcus*.

Asterionella is principally a spring diatom, significant growth under ice is well known (e.g., Moss 1972).

The disappearance of ice cover virtually eliminated the large Rhizosolenia population, but actually increased the Synechococcus population to about 12,000 cells/cc and brought up the Asterionella population to about 1700 cells/cc. The phytoplankton were of course more evenly distributed vertically at this time than they were under ice.

The next major event in the phytoplankton dynamics of 1981 was the July peak, at which time the maximum annual chlorophyll levels were observed. In terms of cell numbers, Synechococcus was the dominant at this time (175,000 cells/cc). Asterionella had also reached higher populations (about 3500 cells/cc). Synedra had overtaken Asterionella, however, with an abundance of about 9000 cells/cc. Rhizosolenia (about 8500 cells/cc) had rebounded to about half its density under ice.

The July peak was followed by a general decline and by a more even distribution of abundance over depth as nutrient depletion became pronounced. As indicated in the analysis of chlorophyll data, there was a resurgence of phytoplankton following the thickening of the mixed layer and renewal of nutrients. This resurgence was accounted for principally by increases in the abundance of Synedra.

In 1982 the large phytoplankton populations under ice did not appear as they did in 1981. The small populations that did develop under ice were dominated numerically by microflagellates, which reached abundances of about 2000 cells/cc, and by Lyngbya, which reached similar abundances. Asterionella reached abundances of about 1000

cells/cc, which was quite significant in terms of percent biomass because of the large cell size of Asterionella. The chlorophytes Monoraphidium and Scenedesmus also reached abundances of about 1000 cells/cc. The very large abundances of Rhizosolenia and Synechococcus that were observed under ice in 1981 did not make their appearance in 1982.

In 1982 the very large summer population of Synechococcus that was so evident in 1981 was not repeated, but the large summer growth of Synedra that appeared in 1981 was repeated in 1982 (18,000 cells/cc). Other diatoms made only minor contributions. Microflagellates and Lyngbya made contributions of about 4000 cells/cc each, and several species of chlorophytes made contributions of 1000 or 2000 cells/cc each. The fall growth was dominated by Synedra (6500 cell/cc) and Synechococcus (46,000 cells/cc), which appeared in large numbers after being rare throughout the earlier part of the year. Also appearing in the 1982 fall growth were large numbers of Stephanodiscus (3500 cells/cc) and Coccomyxa (12,000 cells/cc), neither of which had appeared prominently prior to fall 1982.

In summary, the stable elements of the annual composition cycle seem to be Asterionella growth under ice and in early spring, and a major Synedra peak in July. Sporadic but repeated compositional elements include Synechococcus and Rhizosolenia.

Elemental Composition of Phytoplankton Biomass

Information is available on the amounts of carbon, phosphorus, nitrogen, and chlorophyll a in the particulate material (seston) of the

top 5 m of Lake Dillon over the two years of study. Information of this type is sometimes useful in indicating the nutritional status of phytoplankton. There are a number of complicating factors that limit the usefulness of the data for nutritional interpretations in any particular situation, however. First, the seston is divided in an indeterminate way between living and nonliving fractions. The living fraction in Dillon is principally composed of phytoplankton (bacteria and protozoa are minor contributors on a weight basis). The nonliving fraction is composed of detrital organic matter and inorganic particles, principally silt or frustules from dead diatoms. When the phytoplankton contains a large contingent of diatoms, as is the case for Lake Dillon, the situation is further complicated by the presence of heavy and largely inorganic frustules around each living diatom cell. These frustules contribute 30 to 50% of the dry weight of diatom cells, depending on season and species composition (e.g., Bailey-Watts 1973).

The percentage of phosphorus and nitrogen in the ash-free dry weight of phytoplankton organic matter or, equivalently, the ratios of phosphorus and nitrogen to carbon in phytoplankton biomass, are useful indicators of the nutritional state of phytoplankton (Nalewajko and Lean 1980). Under certain conditions when the ratio of phytoplankton to nonliving particles is quite high, field data on these ratios are a good indication of the state of the phytoplankton in a lake. However, the presence of significant amounts of debris, particularly organic debris, confuses the interpretation because the ratio of elements in phytoplankton and in organic debris is likely to be very different,

thus distorting the apparent ratios for phytoplankton. Microscopic examination of the sedimented phytoplankton samples showed that Lake Dillon contained organic debris at virtually all times of the year, ranging from a minimum of 10 to 20% of total particulates under ice or at the height of stratification to 80 or 90% during spring mixing, spring runoff, and fall mixing. Thus the annual averages for element ratios would be virtually meaningless. It is worthwhile, however, to examine the element ratios for portions of the year when detritus contribution to filterable organic matter is known to be least. The appropriate times include the period of ice cover and the interval from the end of the first week in July, when runoff has ceased to affect surface water, to the middle of September.

The percentage of carbon in particulate materials filtered from the top 5 m of Lake Dillon varied from a sharp minimum of 5% at the peak of the 1982 runoff to a maximum of about 40% under ice during 1981 and 1982. The ratio of carbon to dry weight for phytoplankton, in the absence of diatom frustules, is about 36% (Stumm and Morgan 1981). Carbon percentages between 35 and 40% on four different occasions under ice coincided with the presence of large populations of nondiatomaceous algae, as expected from the literature values on % C. In all other seasons the percentage of carbon fell well below this, and was typically between 10 and 30%. In 1981, the percentage of carbon was typically higher than in 1982, probably because of the smaller relative contributions of diatoms in 1981 than in 1982. The sharp minimum of percentage carbon at runoff in 1982 is attributable to the inflow of significant amounts of silt with peak runoff. The coincidence of this

low percentage carbon with low transparency confirms the interpretation that has already been given concerning the control of transparency by inorganic particulate material at this time of year.

Table 26 shows phosphorus as a percentage of organic matter (computed from carbon) for the two time spans during each year when the influence of nonliving material on the chemistry of seston was likely to have been lowest. Percentage of phosphorus in phytoplankton protoplasm as reported in the literature generally falls between 0.5 and 3.0% of dry weight (Parsons et al. 1977), although lower concentrations are known and may sometimes be adequate to sustain high growth rates (Lund 1970). The observed ratios for Dillon were highest under ice, especially in 1982 when populations were small, and lowest during the post-runoff stratification. This is to be expected in view of the greater likelihood that nutrient depletion during summer stratification will be more extended and more severe than under ice cover.

	P as % organic dry weight	N as % organic dry weight	Chl. *a* as % organic dry weight
1981			
Ice Cover	1.0	13.8	0.87
Post-Runoff stratification	0.8	8.1	0.38
1982			
Ice Cover	1.4	-	0.73
Post-Runoff stratification	0.6	4.9*	0.42

* with one unrealistically low value excluded

Table 26. Composition of organic matter in the top 5 m of Dillon at times when the contribution of non-phytoplankton sources to organic matter was lowest.

Nitrogen percentages are also given in Table 26. The expected percentages are between 4 and 9% for healthy phytoplankton cells (Fogg et al. 1973). Three percent is considered the absolute minimum for phytoplankton growth (Fogg 1975). Under ice cover during 1981 the percentage was very high, perhaps unrealistically so, although phytoplankton are known to have significant storage capabilities for nitrogen (Fogg 1975) and are known to exceed 10% N on occasion (Lund 1965). Too many data are missing for the period of ice cover in 1982 to yield an estimate for that time. During post-runoff stratification both years the percentages were lower than under ice cover in 1981, and fell within the expected range for healthy cells. Significant changes in nitrogen stores may have occurred during the course of the post-runoff stratification, but these could not be deduced from individual determinations on field populations. Aside from the known presence of some non-living material, resolution of changes over short time intervals is not really possible from individual determinations of element ratios because the degree of uncertainty attached with any given ratio determination.

Chlorophyll *a* can be expected to account for 0.5 to 2% of phytoplankton dry weight in most instances. The median is in the vicinity of 1% (Wolk 1973, Taguchi 1976, Gibson 1978, Bailey-Watts 1978). As shown in Table 26, the values for Dillon conform very well to these expectations. Under ice cover the values are near the midrange and in the post-runoff stratification the values are toward the bottom of the range of expected values (0.4%). Low ratios of chlorophyll to organic matter are symptomatic of nitrogen deficiency

(Fogg et al. 1973). Nitrogen deficiency is more likely in post-runoff stratification than under ice, and this may explain the difference between the two seasons in the ratio of chlorophyll to carbon. Other factors, especially light adaptation, must also be considered, however.

The degree of certainty that can be attached to the ratio of chlorophyll a to carbon is greater than for any of the other ratios, since chlorophyll a is not likely to be affected significantly by nonphytoplankton organic matter. For this reason, it is worthwhile to examine trends in individual values over the last half of the stratification season. Figure 33 shows the ratios of chlorophyll a to carbon between the middle of July and the middle of September for both years. In both years there was a trend toward lower ratios as time progressed. This downward trend is suggestive of nutrient stress, specifically nitrogen deprivation, and is part of the circumstantial evidence that concurs with the evidence of enrichment studies for nitrogen deprivation provided by enrichment studies.

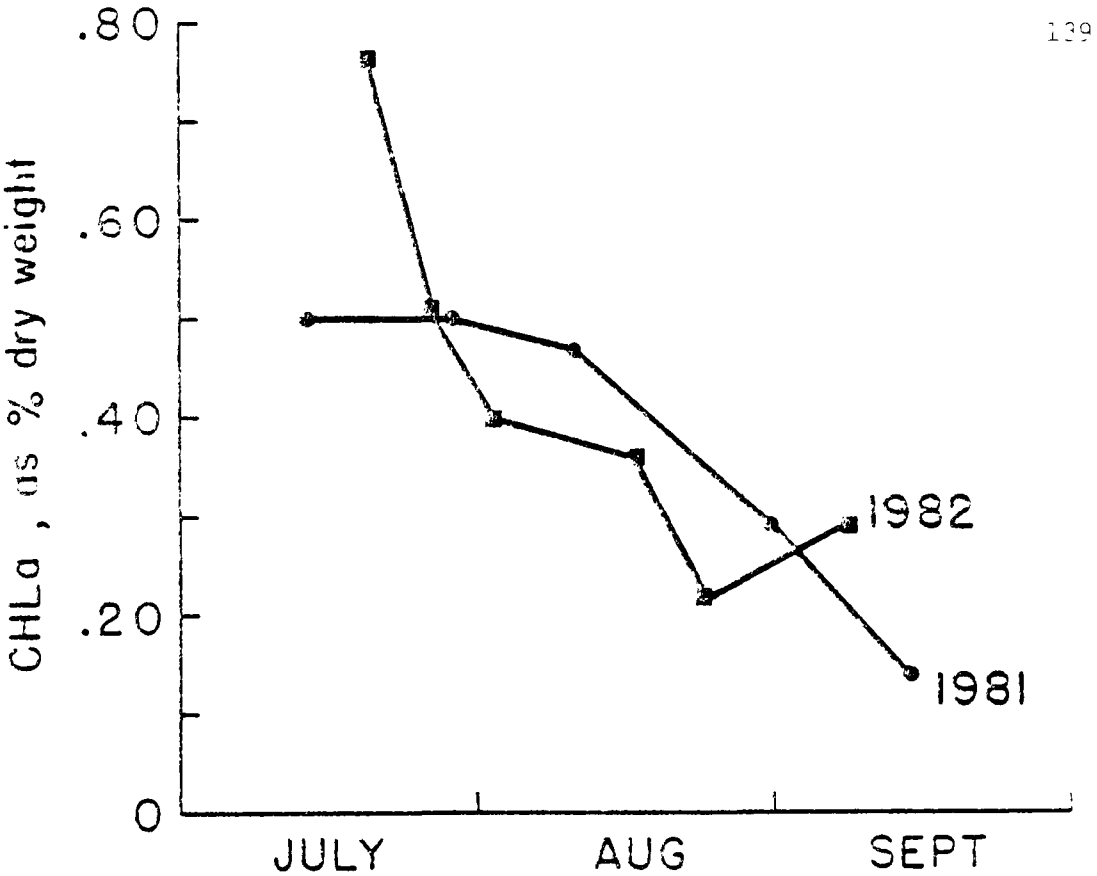


Figure 33. Chlorophyll a as a percentage of total particulate dry weight.

Photosynthesis and Oxygen Consumption in the Water Column

Photosynthesis

The rate of photosynthesis, especially in relation to light and chlorophyll, is the most direct indicator of the physiological condition of a phytoplankton community. It is well known that the standing stock of phytoplankton, as measured by chlorophyll concentration for example, is not always a good indicator of the vigor of a phytoplankton population or phytoplankton community. The ratio of photosynthesis to biomass tends to be highest when populations are first exposed to optimal growth conditions. By the time biomass has reached its peak, the conditions for growth are typically less optimal, and the rate of photosynthesis may have actually declined. Thus direct measures of photosynthesis facilitate the interpretation of events in the phytoplankton community by giving information that cannot be obtained from standing stock (chlorophyll) alone.

As described in the Methods Section, primary production measurements were made on Lake Dillon by the C-14 method. These data can be combined as needed with information on chlorophyll and phytoplankton composition on the same dates. Four indexes related to primary production will be of interest here.

(1) Primary production per unit area. This is a measure of the total photosynthesis occurring under a unit area of the lake over an entire day, and is expressed as $mgC/m^2/day$. Primary production per unit area is affected mainly by the amount of light reaching the surface on a given day, by the amount of nonphytoplankton materials

causing light extinction in the water column, by the size of phytoplankton populations, and by the physiological condition of phytoplankton populations (principally their internal nutrient inventories).

(2) Column efficiency of photosynthesis. This is the ratio of primary production per unit area to photosynthetically available radiation (PAR) penetrating the surface (Lewis 1974, Lewis and Weibezahn 1976). PAR is the proportion of total irradiance between 350-700 nm, estimated as 46% of total irradiance. Surface loss (reflection) is estimated as 10% (Talling 1971). Since phytoplankton biomass and sunlight can both be expressed as energy, it would be possible to express the column efficiency as a true dimensionless efficiency number, but, since there is no particular advantage in this, the units to be used here are mgC/m^2/langley. This computation removes most of the day-to-day variation caused by seasonal and irregular differences in daily insolation. Main causes of variation in the column efficiency include extinction of light in the water column by nonphytoplankton materials, size of phytoplankton populations, and physiological condition of the phytoplankton. The estimate of column efficiency is more subject to error under ice than at any other time of the year, since the effect of snow cover on light penetration is difficult to quantify. For present purposes we have used an absorption coefficient for snow of 0.24 cm^{-1}.

(3) Depth of the photosynthetic maximum. Since phytoplankton populations are typically not distributed evenly and can show inhibition of photosynthesis due to high light intensities near the

surface in the middle of the day, the photosynthetic maximum is usually found somewhat below the surface. The depth of the maximum on a given day can be approximated from the shape of the photosynthesis profile on that day (Figure 34). The depth of the photosynthetic maximum is affected mainly by the amount of surface light, its extinction with depth, the vertical distribution of phytoplankton, and the light history of the phytoplankton (acclimation to high or low intensities).

(4) Maximum photosynthesis per unit chlorophyll. Photosynthesis per unit time at the depth of maximum photosynthesis is divided by the amount of chlorophyll at the same depth, yielding a measure of the photosynthesis per unit chlorophyll at that depth (mgC/mg chlorophyll a/hour), often represented as P_{max}. P_{max} is affected principally by the physiological condition of the phytoplankton and by temperature (Taguchi 1976, Harris 1978). The Q_{10} (increase caused by 10°C temperature rise) of P_{max} is about 2.0 (Harris 1978), so a 10°C rise in temperature will double P_{max} if other factors are equal. If temperature is taken into account, P_{max} is the most direct indicator of the nutrient status of phytoplankton (Ganf 1975, Vollenweider 1965). P_{max} and the other three indexes for primary production are presented in Figures 35 and 36.

Production under ice during both winters was easily measurable, as would be expected from the chlorophyll levels. The absolute levels of production per unit surface averaged only about one-tenth the production levels observed in the summer, however. This is mainly explained by low light availability under ice, since the column efficiency was not low. Contrary to what would be expected from the

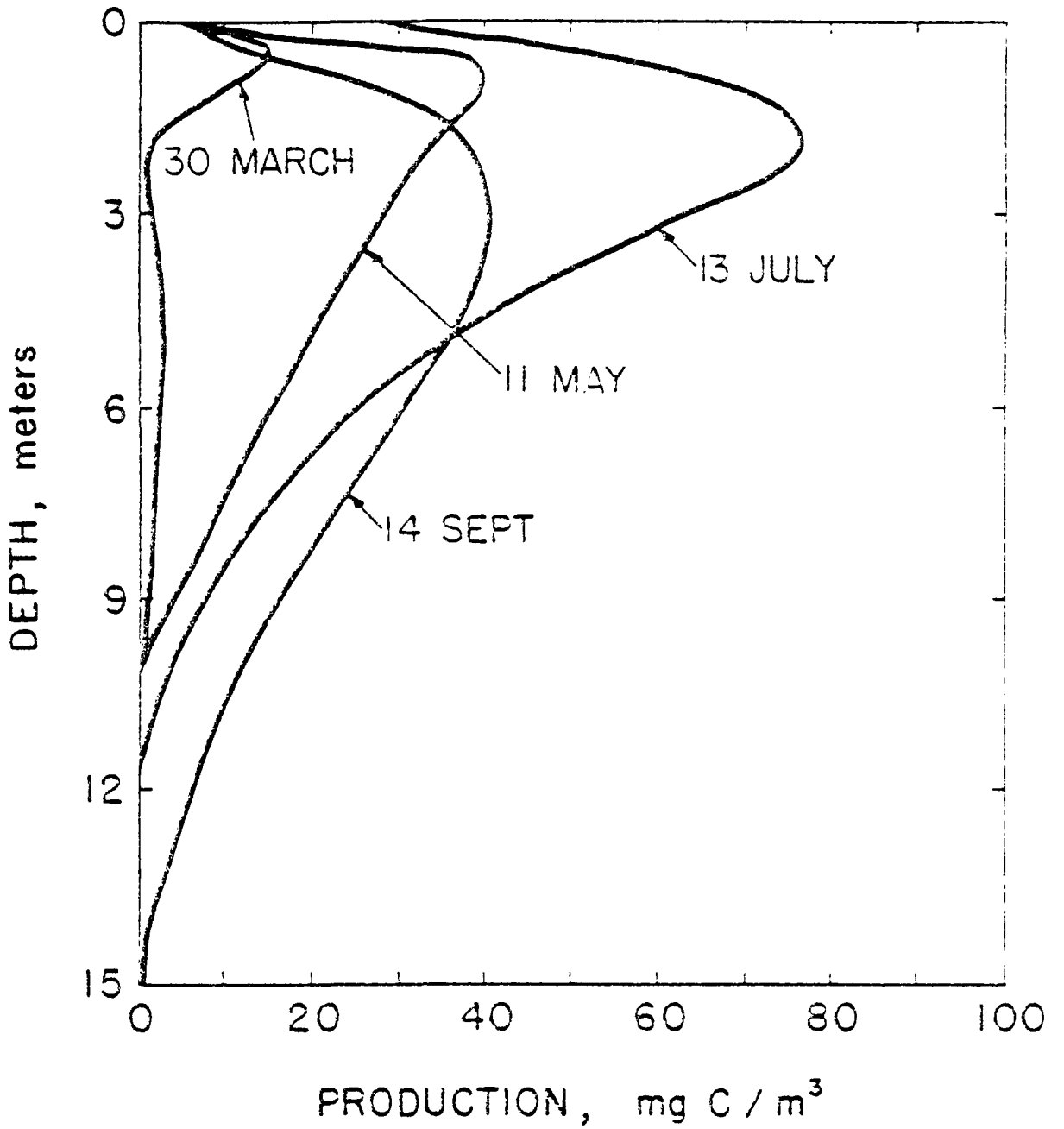


Figure 34. Production per unit volume versus depth for selected times of 1982 as measured by C-14 uptake, showing shapes of photosynthesis curves.

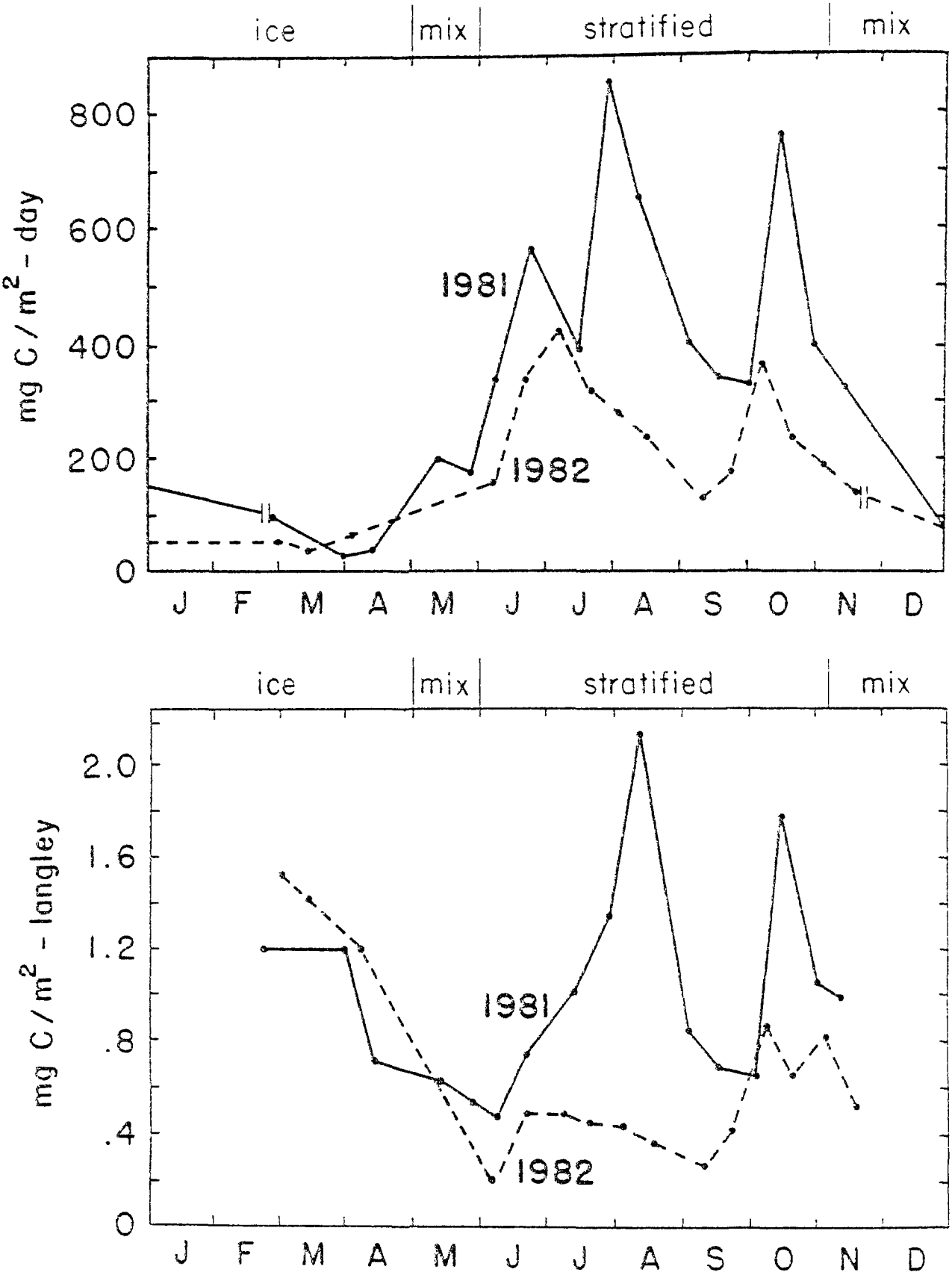


Figure 35. Primary production per unit area (above) and column efficiency of photosynthesis (below).

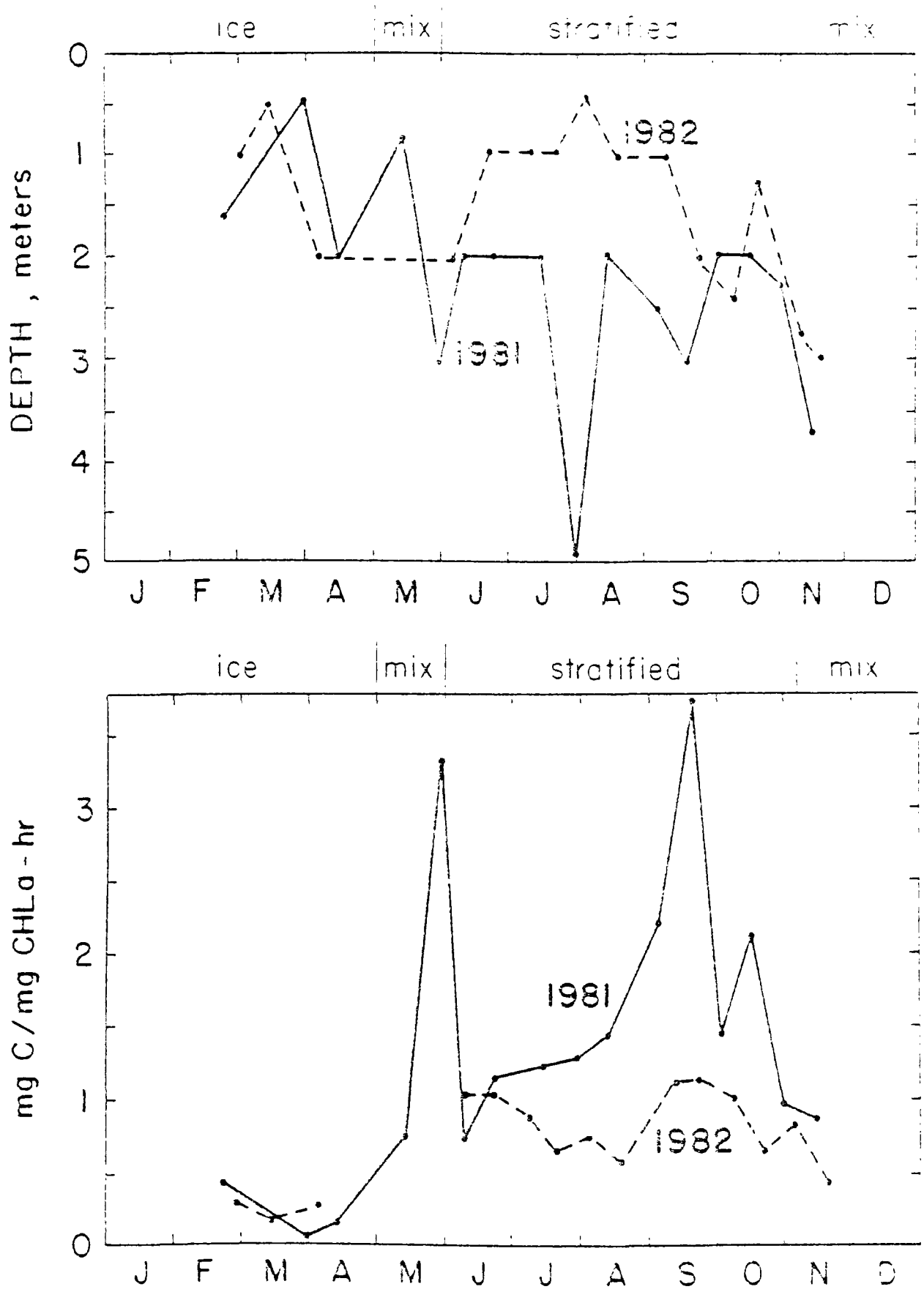


Figure 36. Depth of maximum photosynthesis (above) and maximum photosynthesis per unit chlorophyll (below).

much higher chlorophyll levels of 1981, the production under ice in 1981 was higher than in 1982 only on the first date of measure (February 23). This cannot be explained simply by the amounts of incident sunlight on the dates of incubation, as the column efficiencies of photosynthesis were similar under ice in 1981 and 1982. The higher chlorophyll levels of 1981 were thus not paralleled by higher production during the last half of the period of ice cover. Due to thin ice, no data are available for the first two months of ice cover, but it can be shown that high levels of chlorophyll under ice in 1981 must have been caused primarily by a burst of growth during January and February. The chlorophyll present at the time of first sampling thus would represent the accumulated biomass from this earlier growth which was declining by the time the first sample was taken. Consistent with this explanation, the highest production figure under ice was the first one in 1981 (23 February). The total accumulation of chlorophyll between January 1 and the middle of March 1981 was at least 250 mg/m^2. From the ratio of chlorophyll to organic dry weight under ice in 1981 (Table 26), we estimate that this corresponded to 28 g of phytoplankton dry weight per m^2, or 11 g of carbon per m^2. Since the period of synthesis extended over 85 days, the average net synthesis per day must have been about 120 mgC/m^2. In view of the observed decline by early March, synthesis during January and February must have exceeded 120 mgC/m^2/day on earlier dates.

The P_{max} values were low under ice both years, suggesting nutrient limitation. Nutrient limitation would almost certainly have to be due to low amounts of available P, since N was available in

quantity under ice (Figures 24, 25). When light is available in such low amounts, however, P_{max} must be interpreted with caution, since light may be insufficient to saturate photosynthesis. The consistency of apparent surface inhibition (Figure 36) indicates that saturating intensities were reached during all incubations, however, despite the low surface intensities in winter. Saturation at low intensities is not unusual for phytoplankton adapted to low light. Saturation at intensities as low as 30 uEinsteins/m^2/sec are known (Harris 1979), and the Dillon intensities even under snow exceeded this. We therefore conclude that the P_{max} values were truly low under ice, even in consideration of low temperature, and are thus indicative of P depletion.

During the spring mixing, production per unit area was intermediate between production of the period of ice cover and production of the period of stratification. Column efficiency of photosynthesis was also intermediate, but P_{max} was high. Low column efficiency is accounted for by extreme vertical dispersion of the chlorophyll, which caused most of the light to be taken up by nonchlorophyll absorbance (see Figure 17). At the same time, P_{max} was very high because the phytoplankton, although too dispersed to harvest the light efficiently, were in top physiological condition (no nutrient deprivation). This is expected due to nutrient redistribution with mixing.

Production per unit area rose shortly after the onset of stratification. This is explained by increasing amounts of light due to longer days, by a larger stock of chlorophyll, and by reduced

vertical dispersion of chlorophyll after the cessation of mixing. Column efficiency, however, decreased to a minimum in early June of both years. This was a result of runoff, which temporarily added large amounts of particulates capable of taking up most of the light before it reached phytoplankton chlorophyll. The effect was much more prolonged in 1982 because of higher runoff and shallower entry of runoff into the water column that year. P_{max} in 1981 declined after the onset of stratification from its spring peak, suggesting the onset of some nutrient limitation very shortly after stratification began. In 1982 the June P_{max} values were of similar levels, although the decline was not documented because the lake could not be sampled in May.

In 1981, photosynthesis per unit area and column efficiency climbed steadily until August, after which there was a precipitous decline. The situation was very different in 1982, when the rise of both variables, which began much the same way as in 1981, leveled off in June and began a slow decline in July. The depth of maximum fixation was much closer to the surface than in 1981 and P_{max} declined in parallel with total production. Two factors explain the difference between years: nutrient depletion and interception of light by inorganic particulates. Nutrient limitation began early in stratification both years, as shown by the decline in P_{max} from spring mixing. In 1981, inorganic particulate loading was small after June and passed into the water column below the producing zone. Low chlorophyll at the surface, caused by nutrient depletion there, allowed the chlorophyll peak to develop deeper in the water column where

nutrients were more available. This accounts for the increasing depth of maximum fixation. In 1982, production in deeper water was not possible because of lower transparency caused by much higher particulate input to the lake, and by the proximity of this input to the surface. Surface nutrient depletion was thus not compensated by downward shift in the depth of maximum production in 1982, and production was consequently lower.

In both years there was a major thickening of the epilimnion in mid-September. The immediate consequence of this was to raise P_{max}, because of the relief of nutrient depletion by deepwater nutrients. Production per unit area and column efficiency were slower to respond, since there was a dilution effect along with the improvement of nutrient supply. By the end of September, however, these variables also showed a peak reflecting better nutrient supply. Biomass buildup led to a return of lower P_{max} and a downturn of production in October. Since nitrate was abundant at this time, phosphorus depletion was the probable cause of declining P_{max}.

Among well-studied lakes exposed to similar climatic conditions, Lake Dillon has an annual production similar to that of Lake Erken, Sweden (Nauwerck 1963). Erken is considered mesotrophic. In general, oligotrophic lakes are considered to be typified by production falling below 300 $mgC/m^2/day$ during the growing season, and mesotrophic lakes are typically between this and 1000 $mgC/m^2/day$ (Likens 1975). Since the average production of Lake Dillon was well above 300 $mgC/m^2/day$ in 1981 and near 300 $mgC/m^2/day$ in 1982, the lake would be most reasonably classified as mesotrophic on the basis of production.

The P_{max} values of Dillon are comparable to those observed in other lakes. For example, Glooschenko (1973) found that P_{max} in Lake Huron varied between 0.5 and 3.5 in the ice-free season; this range is similar to Dillon's.

The pattern of production in 1981 and 1982 was essentially the same: spring maximum, early fall maximum, summer depression, and late fall depression. This is a common pattern for temperate lakes (diacmic: Hutchinson 1967). Despite the similarity in pattern, the two years differed considerably in total production. The lower production in 1982 was caused by earlier termination of the spring rise in production, which has already been explained in terms of nutrient depletion and inorganic turbidity. The difference between years can be traced back to hydrology. Since 1981 was exceptional hydrologically, 1982 should be viewed as the more typical of the two years.

Table 27 compares the primary production of Lake Dillon with other lakes of the Colorado Rockies. The comparisons cannot be considered very definitive, since complete annual data are available only for Dillon. The comparisons are based on single measurements for a given day within two days of 31 August on the year indicated in the table. Since the comparative data were taken in the mid-1960's, some of the production values may now be higher at the same time of year as a result of additional nutrient loading. The table indicates that the production of Dillon, at least in the early fall, is within the mid-range of values to be expected in lakes and reservoirs of comparable size.

Lake	Year	$mgC/m^2/day$
Green Mountain	1963	108
Carter	1964	153
Dillon	1982	279
Granby	1963	351
Grand	1963	387
Dillon	1981	402
Shadow Mountain	1963	459
Estes	1964	468
Horsetooth	1964	774

Table 27. Comparison of primary production, as measured by C-14, between Dillon and other Colorado lakes. All production figures are for single measurements made within two days of August 31 in the indicated year. All data except for Dillon are from Nelson (1971).

Oxygen Concentrations and Oxygen Consumption

In an oligotrophic lake of moderate to great depth, the oxygen concentrations are expected to remain near saturation throughout the year over the entire water column. Since temperature affects saturation, there will be some seasonal variation in oxygen concentration at the surface, even in the most oligotrophic lakes, but the concentration will always be near saturation at the prevailing temperature. In lakes of high trophic status, oxygen concentrations well above saturation occur in the euphotic zone due to the steady supply of oxygen from photosynthesis, and significant oxygen depletion develops in deep water due to the steady consumption of oxygen by respiration. Deviation of oxygen concentration between surface and deep water is greatest under ice or during summer stratification when vertical exchange is minimal. Mesotrophic lakes of comparable size and depth are intermediate in degree of surface supersaturation and deepwater depletion.

Oxygen depletion is of direct interest because of its significance to bottom fauna and deepwater organisms, including fishes, and because of the major change in internal nutrient flux that occurs when the oxidized microzone at the mud surface becomes reduced and thus much more permeable to the chemical constituents of the interstitial waters (Hutchinson 1957, Golterman 1976). The deepwater oxygen concentrations of a lake are also of indirect interest as an indicator of trophic status, provided that correct compensation is made for morphometric variation.

The oxygen concentrations of Lake Dillon in 1981 and 1982 are shown in the time-depth diagrams of Figure 37. Table 28 shows the oxygen concentrations corresponding to saturation at the elevation of Lake Dillon at various temperatures. It is evident from the table and from the time-depth diagrams that the surface waters of Lake Dillon were near saturation throughout 1981-1982. Production was never high enough to hold oxygen concentrations much above saturation. The most extended surface deviations from saturation occurred under ice, when atmospheric exchange was impeded. Surface oxygen concentration was slightly above saturation under ice in February of 1981. This was the result of the unusual winter burst of production in January and February, when the ice was clear and free of snow that year. Other instances of supersaturation were very minor. There were also a few instances of subsaturation oxygen concentrations near the surface. A significant depression of surface oxygen concentration occurred in September 1981. This was caused by thickening of the mixed layer and incorporation of deeper, less oxygenated water with surface water. September suppression of surface oxygen due to mixing in 1982 was also detectable, but was considerably smaller because the speed of incorporation of deeper water was slower in September of 1982 than in September of 1981. Complete mixing in the first week of November lowered the oxygen concentration at the surface also (not enough to be evident in Figure 37a, 37b).

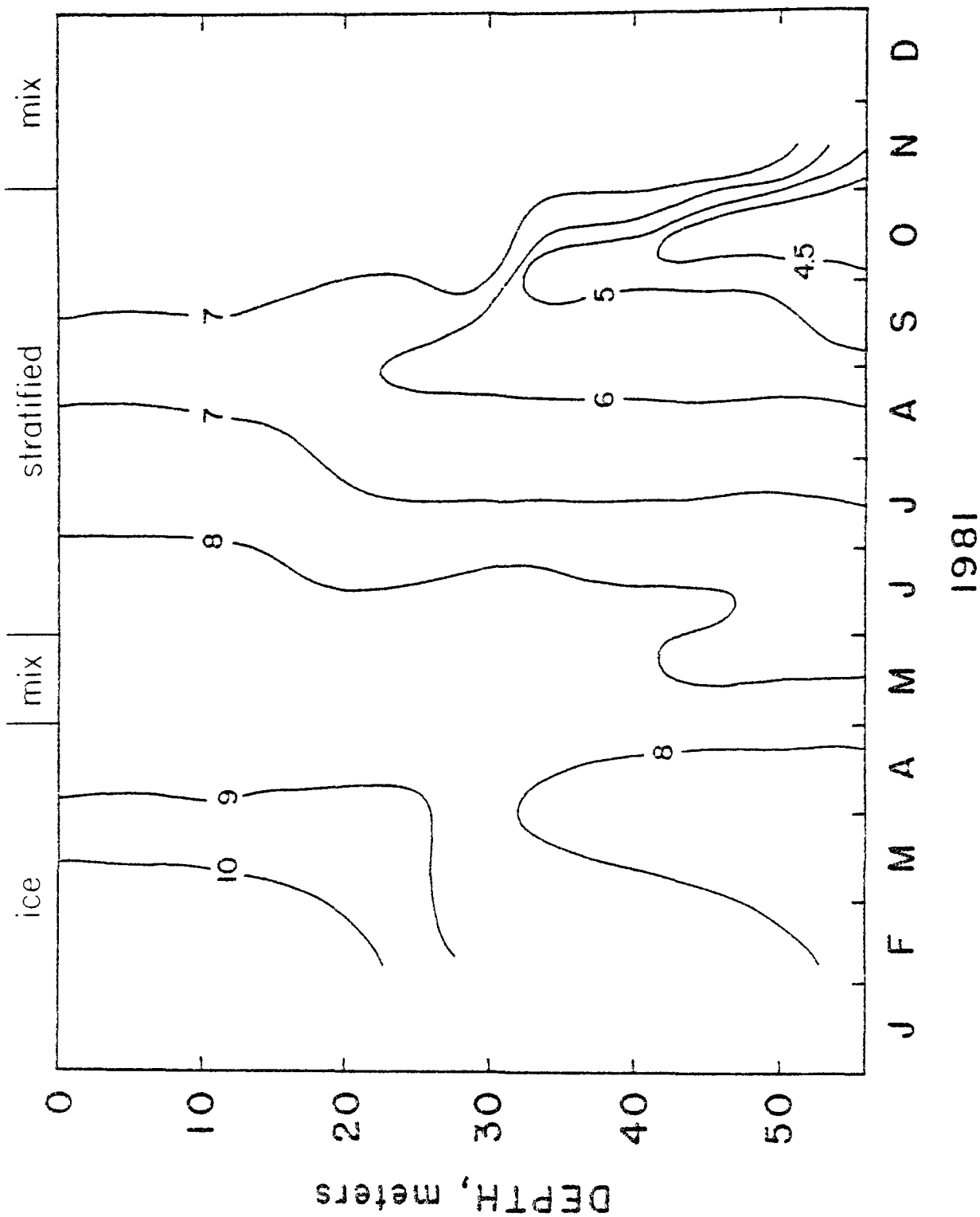


Figure 37a. Time-depth diagram for oxygen in Lake Dillon, 1981 (mg/l).

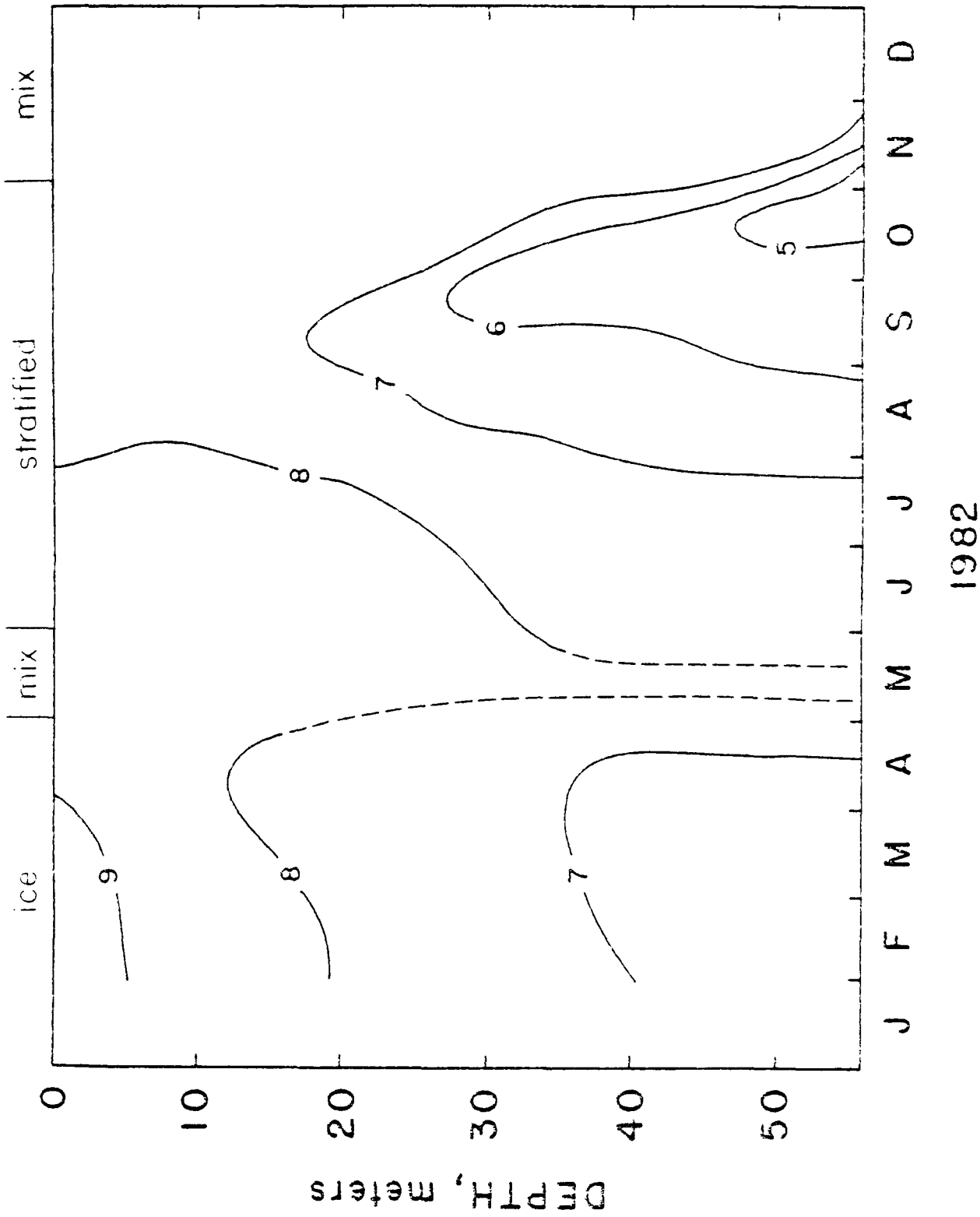


Figure 37b. Time-depth diagram for oxygen in Lake Dillon, 1982 (mg/l).

	Oxygen, ppm					
Temperature, °C	5	6	7	8	9	10
0	50	59	69	79	89	99
5	57	68	79	91	102	113
10	64	77	90	103	116	128
15	72	86	101	115	129	144

Table 28. Percent saturation corresponding to various temperatures and oxygen concentrations at the elevation of Lake Dillon.

Dissolved oxygen concentrations in deep water did not remain near saturation throughout the year. Saturation in deep water was achieved during spring and fall mixing, so that the water column entered the period of winter ice cover or the period of summer stratification with no oxygen deficit. There was significant depletion of oxygen in deep water both under ice and during summer stratification. The bottom 10 meters of the water column under ice declined to a winter minimum between 6 and 7 ppm, as compared with a saturation concentration of about 9 ppm at the prevailing temperatures. Deepwater oxygen depletion during summer stratification was considerably more severe, partly because of the greater length of time over which depletion could occur and partly because of more rapid supply of organic materials subject to decay during the summer months. Oxygen concentrations 5 m above the bottom declined to 4.4 ppm in 1981 and to 4.6 ppm in 1982. Perhaps even more significant than the minimum levels in the deepest water was the great vertical extent over which notable oxygen depletion occurred. The bottom half of the water column was markedly affected in both years. Since deepwater oxygen concentrations at saturation would have

been about 9 ppm, the observed oxygen levels were as low as 50% of saturation.

Salmonid fishes and many species of invertebrates common to coldwater oligotrophic lakes require high oxygen concentrations. Although the thresholds vary according to the circumstances and the species, notable stress begins to occur in some species in the vicinity of 5 ppm (Alabaster and Lloyd 1980). Thus Lake Dillon is experiencing oxygen depletion to a degree sufficient to begin having biological effects. Major chemical effects are typically not noticed until oxygen is completely depleted at the mud-water interface.

Because oxygen depletion is so important biologically, and because it is an indication of total biological activity in a lake, there have been a number of efforts to link oxygen depletion quantitatively to the productivity of lakes. Although it is generally accepted that the linkage exists, the quantitative complications are numerous. First and most obvious is the dependence of the degree of oxygen depletion on the volume of oxygen-rich water at the beginning of the season during which oxygen depletion occurs. This so-called "morphometric effect" (Hutchinson 1957) is dependent on the ratio of the epilimnion volume to hypolimnion volume in a given lake. To compensate for the different ratios of layer volumes in various lakes, and thus achieve a comparison between lakes, the concept of the areal hypolimnetic oxygen deficit (AHOD) was introduced (Strom 1931, Hutchinson 1938). The average amount of oxygen in the hypolimnion per unit surface of hypolimnion is computed by integration of oxygen concentration with depth from the top of the hypolimnion to the lake bottom. The amount of oxygen per unit

area is plotted against time and the slope of the decline is determined by linear regression (Lasenby 1975). The slope is the AHOD, expressed as $mgO_2/m^2/day$. The terminology, which is dictated by traditional usage, is somewhat confusing in that AHOD is not really a deficit, but rather a depletion rate.

Among different lakes, AHOD is significantly correlated with trophic indicators such as total phosphorus, primary production, and transparency (Cornett and Rigler 1980). These relationships are potentially useful in predicting the degree of oxygen deficiency to be expected in a given lake in response to changes in trophic status. However, despite the corrections for major differences in layer volumes and in the duration of stagnation that are inherent in the computation of AHOD, a significant amount of scatter does remain in all of the relationships between trophic indicators and AHOD. Some scatter is removed by use of depth in the equations, but as yet there seems to be no simple method for reducing scatter beyond this (Cornett and Rigler 1980).

The AHOD was calculated for Lake Dillon for the 1981 and 1982 summer stagnations. The decline of hypolimnetic O_2 per unit area was very linear both years, thus the AHOD could be estimated within narrow limits. In 1981 the AHOD was 710 $mgO_2/m^2/day$ (standard error, 40) and in 1982 it was 630 $mgO_2/m^2/day$ (standard error, 37). These are considerably higher than the values for oligotrophic lakes given by Lasenby (1975).

Three separate relationships were developed by Cornett and Rigler (1980) to predict AHOD from key trophic indicators. Table 29 shows the information required to make the predictions and Table 30 shows the predicted and observed AHOD values for Dillon. The phosphorus equation substantially underestimates AHOD for Dillon. Dillon produces about double the oxygen deficit that would be expected from P concentrations. This is due to the higher biological activity supported by a given P concentration in Lake Dillon than in most lakes; this phenomenon will be documented and explained in other sections of this report. The equation based on primary production is also a poor predictor, at least for 1981 when hypolimnetic temperatures were exceptionally warm, but this is not surprising since Cornett and Rigler found it to be the least reliable of the relationships they tested. Secchi depth produces the best predictions. 1981 is underestimated by about 20% and 1982 is estimated almost exactly. Clearly the secchi depth equation would be the best choice for predicting AHOD in Dillon.

Year	Mean Total P (0-5 m), ug/l	Mean Summer Secchi depth (m), 25 June - 30 Sept	Primary Production $gC/m^2/yr$	Mean Depth m	Mean Hypolimnetic Depth, m	Mean Hypolimnetic Temp., °C
1981	7.63	3.14	110	24.1	23.7	8.9
1982	9.15	2.78	57	24.1	23.7	6.9

Table 29. Information required for prediction of AHOD by the equations of Cornett and Rigler (1980).

Year	Observed AHOD $mgO_2/m^2/day$	Predicted AHOD, $mgO_2/m^2/day$ Using Equation with Indicated Variables*		
		P, $\bar{z}$	Secchi, $\bar{z}$	Production, $\bar{z}_H$, T_H
1981	710	295	580	4841
1982	630	323	636	636

* P = mean total P concentration, $\bar{z}$ = mean depth, $\bar{z}_H$ = mean hypolimnion depth, T_H = volume weighted mean temperature of hypolimnion.

Table 30. Observed areal hypolimnetic oxygen deficits in 1981 and 1982 compared with those predicted from three separate equations developed by Cornett and Rigler (1980).

Nutrient Enrichment Studies

The purpose of a nutrient enrichment study is to determine the actual or potential limitation of phytoplankton by specific nutrients. The literature on nutrient enrichment contains a wide variety of experimental designs and methods for such studies. This variation in methodology is in part explained by differences in the exact purpose of the studies. In general, nutrient enrichment studies fall into one of four categories:

(1) Enrichment of laboratory algal cultures with lakewater filtrate. For this type of enrichment, a water sample is removed from the lake under study and is passed through a filter sufficiently fine to remove the phytoplankton. The water is then added to standard algal cultures, along with a spike of phosphorus, nitrogen, or any other individual nutrient or nutrient combination that might be of interest. The response of the standard alga (often Selenastrum capricornutum in the United States) is measured in terms of cell number or chlorophyll. The responses in comparison to the control (no nutrient additions) are taken as an indication of the identity of the potentially limiting nutrient in the lake. The special advantages of this method are that it does not require extended field work and that it provides a common basis for the comparison of very different kinds of lakes. The disadvantage is that standard laboratory cultures differ considerably from a natural algal assemblage. For example, it well known that algal cultures, either through physiological acclimation or genetic adaptation, typically require much higher amounts of nutrients to

produce significant growth than do field populations (Nalewajko and Lean 1980, Lund 1965).

(2) Short-term in-situ enrichment. Short-term enrichment is probably the most common type of enrichment study when the emphasis is on the identity of actual (not potential) limiting nutrients at a given time in a particular lake. A sample is taken from the growth zone and placed in a number of bottles. Nothing is added to certain bottles; these serve as the control. Other bottles are enriched with individual nutrients or nutrient mixtures. All bottles are then inoculated with C-14 and resuspended in the euphotic zone. Production is estimated from C-14 uptake over a period of a few hours. In theory, algae that are limited by a given nutrient will show increased photosynthesis in response to the addition of that nutrient. Despite the extensive use of this method, a critical flaw in the underlying assumptions has recently been demonstrated by Lean and Pick (1981). For phosphorus, and possibly for other nutrients, the response of nutrient-starved cells over the first few hours after their exposure to new supplies of the nutrient is the diversion of cellular resources to uptake of the nutrient rather than growth. A few hours may therefore be simply too short a time over which to measure the growth response of phytoplankton to the addition of a limiting nutrient.

(3) In-situ enrichments lasting a few days. This type of enrichment is identical to the short-term enrichment except that the containers enclosing phytoplankton that have been subjected to various nutrient treatments are suspended at depth in the lake for one to several days instead of a few hours. The growth response is typically

measured by change in chlorophyll with respect to a control rather than by C-14, as the C-14 method is not valid for application over extended time intervals. The main disadvantage to this method is the artificiality of enclosure conditions as compared with open-water conditions. This disadvantage is combatted principally by the use of large containers with soft sides rather than small bottles.

(4) Long-term enclosure methods. Very large corrals or plastic spheres are used in some instances for determination of nutrient limitation. Although these are exceedingly useful and informative methods, they typically are far too expensive for routine application.

Enrichment studies of Lake Dillon in 1981 and 1982 were done by the third method cited above, in-situ enrichment lasting a few days. All enrichments were carried out at the index station. On the day of each enrichment study, integrated samples of the top 5 m of the water column were removed from the lake and placed in a large chamber where they could be homogenized. These were then siphoned into a number of soft plastic containers, each with a volume of 10 l. Certain of the containers were left unaltered as a control. Other containers were enriched with KH_2PO_4 in amounts sufficient to increase the original phosphorus by 100 ug/l in the container. Nitrate enrichment was by means of KNO_3 to the 200 ug/l level. The amount of enrichment was intended to exceed any possible requirement of either N or P over the interval of incubation but to be below the amount that might have toxic effects. The number of replicate containers for each treatment and for the control was typically three, and occasionally four. All except one enrichment study involved a nitrogen treatment, a phosphorus

treatment, a phosphorus plus nitrogen treatment, and a control. The first enrichment study omitted the nitrogen treatment.

After the containers were filled and spiked as needed, they were all resuspended at a depth of approximately 2 m in the water column. This depth corresponds very closely to the expected depth of maximum photosynthesis (Figure 36). Because the containers were hooked to a floating buoy, they moved up and down with the wave action at the lake surface. This and the incorporation of an air bubble inside each of the containers induced sufficient turbulence to keep the algae inside the containers in suspension. The containers were left in place for a period of three to seven days, depending on the time of year. When the containers were removed from the lake, they were taken immediately to the Snake River Wastewater Treatment Plant, where water from each container was filtered for chlorophyll analysis as described in the Methods section. Some analyses were also done for other variables.

Three enrichments were done in 1981. In 1982 the number of enrichments was increased to seven because the 1981 data indicated a shift in the limiting nutrient, and the timing of the shift could only be identified with certainty on the basis of more closely spaced enrichments.

The chlorophyll concentrations for each enrichment were subjected to analysis by one-way ANOVA in which the enrichment treatment served as the variable of classification and the dependent variable was the final chlorophyll concentration. The null hypothesis was that different enrichments were not significantly different in chlorophyll from each other or from the control.

Table 31 summarizes the results of the enrichment studies. The one-way ANOVA tests showed that the mean response to the four treatments was not homogeneous on any one of the ten dates ($P < 0.01$). In other words, the four treatments could be divided into two or more groups on any one of the ten dates. In order to determine which of the treatments on any given date were statistically distinct from each other, the Student-Newman-Keuls (SNK) Multiple Range Test was applied to the data. Table 32 separates the four treatments into statistically distinct groups according to the SNK Test. The groups are arranged in the table in order of ascending response to enrichment.

Since there was always a statistically significant heterogeneity among the four treatments, it is defensible to identify a limiting nutrient for each of the ten dates. This is accomplished by the combined use of Tables 31 and 32. The limiting nutrient is here operationally defined as the nutrient producing greatest growth response. It is taken for granted that multiplicative effects may exist by which one nutrient interacts with another (Droop 1976). The degree of response to enrichment, which can be expressed conveniently as the ratio of the phosphorus plus nitrogen treatment to the control, varied considerably between dates, as indicated in the last column of Table 31.

Certain features were common to all the enrichment studies. The P + N treatment always fell in the highest group of responses, as shown in Table 31. In six of the ten enrichments, the response to P + N was higher than the response to either nutrient alone. Given the

Date	Treatment* C	P	N	P + N	Dominant Limitation	Ratio P+N/C
1981						
20 July	4.4	5.7	-	20.5	N	4.7
24 August	2.1	2.2	5.7	20.6	N	9.8
28 September	5.3	5.8	5.0	11.1	P/N	2.1
1982						
24 May	2.7	3.9	3.2	3.8	P	1.4
14 June	6.8	11.7	6.8	11.1	P	1.6
10 July	8.3	17.1	8.9	20.5	P	2.5
26 July	7.9	7.1	11.9	39.0	N	4.9
23 August	5.0	4.9	12.0	25.4	N	5.1
7 September	1.8	2.0	2.3	2.6	P/N	1.4
18 October	6.5	13.4	6.8	14.5	P	2.2

* C = Control, P = Phosphorus only, N = Nitrogen only, P + N = Phosphorus plus Nitrogen.

Table 31. Chlorophyll concentrations (ug/l) for the ten enrichment studies.

Date	Group 1* (Lowest)	Group 2	Group 3 (Highest)
1981			
20 July	C, P	P + N	-
24 August	C, P, N	P + N	-
28 September	C, P, N	P + N	-
1982			
24 May	C, N	P, N, P + N	-
14 June	C, N	P, P + N	-
10 July	C, N	P	P + N
26 July	C, P	N	P + N
23 August	C, P	N	P + N
7 September	C, P	P, N	N, P + N
18 October	C, N	P, N + P	-

* C = Control, N = Nitrogen only, P = Phosphorus only, P + N = Phosphorus plus Nitrogen

Table 32. Enrichment treatments separated into statistically coherent groups based on the Student-Newman-Keuls Multiple Range Test ($p < 0.05$). Groups are arranged from left to right in order of increasing chlorophyll response. A given treatment can belong to more than one group if it is statistically inseparable from more than one group.

relatively short duration of the incubations, such a result suggests that phosphorus and nitrogen requirements are closely balanced to the supply of these nutrients, even though one or the other may be identifiable at a particular moment as the dominating control on phytoplankton growth. Another feature common to the enrichment studies is the appearance of the control (i.e., no enrichment) in the lowest statistical response group. Since this is expected given the assumptions of the studies, it suggests that the studies were methodologically sound.

The number of enrichments was much smaller in 1981 than in 1982. Where the enrichment studies overlapped for the two years, however, the results are so similar as to suggest a common pattern. In July and August of 1981 the response to phosphorus plus nitrogen was very strong but the response to phosphorus only was very low or nil. Unfortunately, a nitrogen-only enrichment is not available for 20 July 1981, but the low P response and the high P + N response that year suggest nitrogen limitation. A much stronger case can be made for nitrogen limitation over the comparable interval in 1982. The 26 July enrichment showed a definite nitrogen response, as did the 23 August enrichment. The 10 July enrichment, however, showed an unequivocal response to phosphorus. We therefore conclude that there was a switch from phosphorus to nitrogen limitation in the middle of July in 1982 and probably in 1981 also.

Phosphorus limitation was unequivocal in the June enrichment. The May enrichment, which coincided with spring mixing, also produced a definite phosphorus response, but the magnitude of the response was

much lower at this time, both in absolute and relative terms. We conclude that nutrient limitation was not pronounced in May, and was possibly induced by confinement of the samples in the zone of ideal light for growth. In both years, enrichment during the fall thickening of the mixed layer produced similar results to the enrichments at the time of spring mixing. The absolute and relative magnitudes of the response were small, and the evidence points toward phosphorus limitation more than nitrogen. Under field conditions, it is likely that neither element was immediately limiting at such times, but phosphorus was incipiently limiting and this was brought out by a few days of sample confinement. The October enrichment in 1982 shows a return to pronounced phosphorus control of phytoplankton growth.

The composite picture that emerges from Tables 31 and 32 is summarized in Figure 38. Although no enrichments were done through the ice, the presence of significant inorganic nitrogen under ice and the evidence of phosphorus limitation just before ice formation and just after disappearance of ice strongly support the case for phosphorus limitation under ice. Thus on a calendar year basis the period of phosphorus limitation extended from 1 January until the onset of nitrogen limitation, which appeared rather abruptly in the middle of July. The coincidence of this switch with the depletion of nitrate is shown in Figure 38. Late summer thickening of the mixed layer relieved nitrogen limitation (mid September). September can be regarded as a month of transition, during which the response to enrichment was less pronounced but indicative of phosphorus domination. Unequivocal return

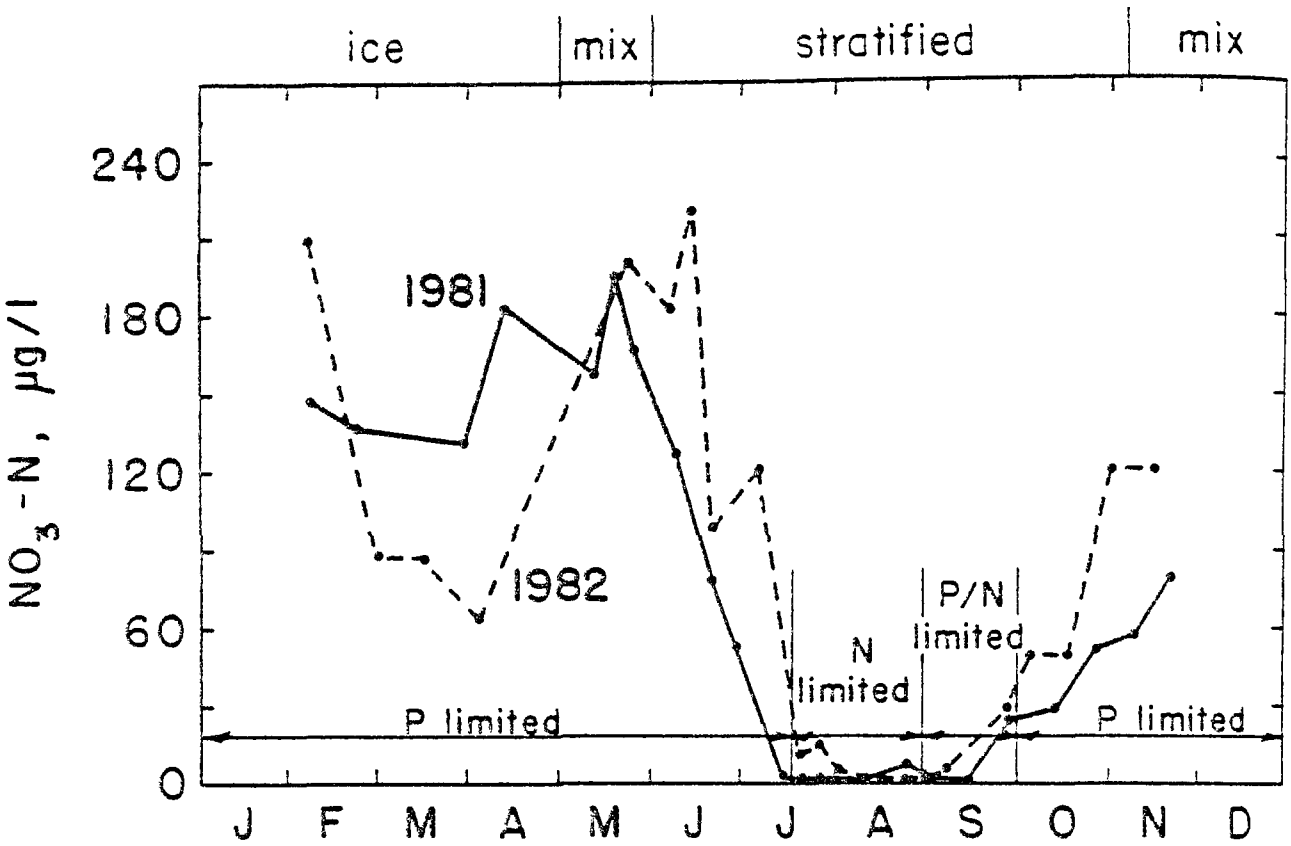


Figure 38. Nutrient limitations as shown by the enrichment experiments superimposed on the nitrate concentrations near the surface (0-5 m).

to phosphorus limitation occurred with increased thickening of the mixed layer in October, leading to complete mixing in November.

The interpretation given here of the enrichment studies is consistent with the inorganic chemistry and with the primary production data. As shown in the previous chapter, P_{max}, an indicator of the nutritional status of phytoplankton, was highest during May or early June and in September. These are periods when the plankton were abundantly supplied with nutrients, and they coincide with the minimal responses to artificial enrichments. P_{max} shows evidence of nutrient stress under ice cover (P depletion) and again in mid-summer (N depletion). In the fall there is again evidence of phosphorus deficiency in the P_{max} values.

The enrichment studies showed that phytoplankton biomass in Lake Dillon was alternately controlled by phosphorus and nitrogen. Switching of this type is known from certain other lakes (Lake Washington: Edmondson 1972). The switching occurred only during summer stratification and appeared to be predictable in its timing. At the beginning of stratification, the ratio of available nitrogen to available phosphorus was very high. This was determined principally by the very high ratio of nitrogen to phosphorus in the water entering the lake. After stratification began, however, the amount of incoming water declined, and much of it entered the middle and deeper layers of the lake, thus not greatly influencing the surface chemistry. Biological processes then gradually changed the ratio of available phosphorus to available nitrogen in the upper water column. From the organic nutrient chemistry, we believe that the mechanism for this

change has to do with the relative abilities of phytoplankton to use soluble organic nitrogen and soluble organic phosphorus. As stratification progressed, the relatively large amount of phosphorus present in the soluble organic pool gradually disappeared, presumably because it was taken up by the phytoplankton and microbes. This was not true of soluble organic nitrogen, which appeared to be less available to the plankton. Because the phytoplankton and microbe cells were able to divert more of the organic phosphorus to their metabolic needs, the ratio of phosphorus to nitrogen inside the cells changed to such an extent that the phytoplankton become nitrogen limited. Thickening of the mixed layer in September reintroduced abundant supplies of inorganic nitrogen, and thus returned the lake to the phosphorus control that was typical of early stratification.

The effectiveness of the phytoplankton in sequestering and storing phosphorus well beyond their needs is illustrated by the particulate phosphorus data that were taken as part of the enrichment studies. These data are summarized in Table 33. Particulate phosphorus differed significantly among treatments on all dates (one-way ANOVA, $P < 0.01$). Phosphorus and phosphorus plus nitrogen treatments were always lumped together by the SNK criteria, and control and nitrogen treatments were always lumped together by the same criteria. This was true whether or not the dominant limitation was at a given time principally associated with phosphorus or with nitrogen. Thus the phytoplankton, even when strongly nitrogen limited, took up substantial amounts of the added phosphorus. This lack of linkage between the uptake of phosphorus and the immediate nutritional requirement for phosphorus illustrates the

	Particulate P - ug/l	
Date (1982)	C, N*	P, P + N
24 May	5.0	17.1
14 June	11.9	33.2
10 July	6.2	30.2
26 July	3.9	29.1
23 August	4.5	28.4
7 September	3.2	9.8
18 October	3.4	22.8

* C = Control, N = N added, P = P added, P + N = P + N added.

Table 33. Particulate P concentrations in the enrichment treatments. C (control) and N (nitrogen treatment) are together because the SNK test shows them to be consistently indistinguishable for particulate P. The P (phosphorus) and P + N (phosphorus plus nitrogen) treatments are together for the same reason.

significance of luxury consumption of phosphorus in the phytoplankton, and the impossibility of evaluating phytoplankton nutrient status from external nutrient conditions.

Horizontal Spatial Variation in the Lake

Up to this point most of the analysis has dealt with samples taken from the index station in the middle of the lake. The question arises how typical the index station is of the lake as a whole. This question is answered here in two stages. First, an analysis is made of the five-station heterogeneity series. As described in the sections on study design methods, this series consisted of analyses made on a set of samples taken on 32 different dates at the four main stations and the index station. The top (0-5 m) and bottom of the water column were sampled at each station. Since each one of the four main stations was located at the mouth of one of the four main arms of the lake, this sample series gave information about the degree of variation to be expected over the deep-water section of the lake.

The second stage of the analysis is based on a treatment of the 14-station heterogeneity series. This series included surface samples (0-5 m) from 14 different sites taken on 10 different dates. Unlike the 5-station heterogeneity series, these 14 stations extended over the entire lake surface without regard to depth, and thus included shallow-water as well as deep-water areas.

The 5-Station Heterogeneity Study

Samples taken at the same depth at multiple stations will always show some degree of variation. This variation can be divided into three components (Lewis 1980): (1) error variance, (2) fixed horizontal spatial variation, and (3) ephemeral horizontal spatial variation. Error variance is that which would be observed in

replicates from the same site on a given date. In effect this is analytical error variance, although some handling variance could be included and, if the replicates were taken separately at the same site, some true microscale patchiness might also be contributory to the error variance defined in this way. In most cases, and certainly in the case of the variables of concern here, analytical error dominates the variance of replicates from a given site on a given date.

Fixed spatial variation is associated with average differences between stations. For example, if a nearby shoreline or point source of nutrients continuously influences the chemistry of a particular station, this would increase the variance of a set of samples including that station. In contrast, ephemeral spatial variation is due to horizontal variation that is not stable over time. There would be a considerable difference among stations on any given date, but still no difference in the average values for a given variable among the different stations, if ephemeral variation were high and fixed variation were nil. Typically lakes can be expected to show some combination of fixed and ephemeral variation, but among these two components ephemeral variation usually dominates except in the most extreme cases of physical or chemical gradients.

For the 5-station heterogeneity series, we first consider fixed horizontal spatial variation, i.e., the possibility that the five stations were significantly different when averaged through time with respect to their chemical or biological properties. The analysis is framed as a statistical null hypothesis, which is that all stations

have the same average values. The hypothesis is tested by one-way ANOVA.

Table 34 shows the mean value of 11 different variables for samples taken in the 0-5 m layer at each of the five stations. The table also shows the grand mean for each variable over the 32 different dates and five different stations and shows the outcome of the one-way ANOVA testing for significant differences among stations. A cursory examination of the means for the different stations shows that they are remarkably similar to each other. The results of the one-way ANOVA show that in all instances the small amount of variation that is observed from one station to the next is statistically insignificant at $P = 0.05$. We therefore conclude that, from the viewpoint of annual averages, horizontal spatial variation at the surface over the deep part of the lake, including the mouths of the arms of the lake, is trivial. This does not imply that there is no variation between stations on a given day (ephemeral variation), but it does indicate that one of these stations is as good as any other in representing the deep-water portion of the lake over the long run.

The same analysis was done for samples taken at the bottom of the water column at each of the five stations and the results are summarized in Table 35. The list of variables is slightly different than for the surface samples because some analyses that are meaningful at the surface of the water column are not meaningful at the bottom (e.g., secchi depth). The results for the bottom of the water column are tabulated separately because they are subject to a slightly different kind of interpretation. Because the bottom of the water

Variable	Station						
	Index (A)	Dillon Bay (B)	Snake Arm (C)	Blue Arm (D)	Tenmile Arm (E)	Grand Mean	Significant Differences
Temperature, °C	9.5	10.2	10.0	10.2	10.3	10.0	none
Secchi, m	3.0	2.6	2.6	2.7	2.7	2.7	none
Conductance, umho/cm	161	162	159	162	162	161	none
PO_4-P, ug/l	1.2	1.0	0.7	0.9	1.0	1.0	none
Total Soluble P, ug/l	3.4	3.2	3.2	2.9	4.0	3.3	none
Particulate P, ug/l	5.1	5.1	5.2	4.9	5.0	5.1	none
NO_3-N, ug/l	80	71	77	69	69	73	none
NH_4-N, ug/l	16	12	14	13	13	14	none
Total soluble N, ug/l	287	292	275	265	258	275	none
Total particulates, mg/l	2.9	3.1	3.2	3.0	2.8	3.0	none
Chlorophyll a, ug/l	8.3	8.0	8.2	8.6	8.0	8.2	none

Table 34. Means for variables at the index station and 4 main stations on 32 sampling dates spread over the two-year study period (0-5 m layer).

Variable	Station						
	Index (A)	Dillon Bay (B)	Snake Arm (C)	Blue Arm (D)	Tenmile Arm (E)	Grand Mean	Significant Differences
Temperature, °C	4.3	5.1	4.7	5.0	5.0	4.8	none
Conductance, umho/cm	179	167	166	168	171	170	none
PO_4-P, ug/l	1.0	1.3	1.0	1.0	1.2	1.10	none
Total Soluble P, ug/l	3.0	3.7	3.4	3.5	3.1	3.34	none
Particulate P, ug/l	3.1	7.8	9.4	6.3	8.6	7.0	yes
NO_3-N, ug/l	201	132	141	148	143	153	yes
NH_4-N, ug/l	18	25	25	22	25	23.0	none
Total Soluble N, ug/l	387	327	311	423	351	360	none
Total Particulates, mg/l	1.8	4.9	3.8	2.7	3.4	3.3	yes

Table 35. Means for variables at the index station and 4 main stations on 32 sampling dates spread over the two-year study period (bottom of water column).

column is found at different depths depending on the station, a certain amount of variation due to depth is blended into the true horizontal variation between stations. Thus while the bottom samples at the index station in the middle of summer were most likely to have been taken at about 40 m, the bottom samples in Dillon Bay were more likely to have been taken at 20 m. This potentially will cause some variation due to depth alone. No attempt is made here to separate depth variation from the variation due to horizontal patchiness, since the focus of attention is on the nature of the environment near the bottom of the lake.

Table 35 indicates that, despite the additional possibility for variation due to differences in depth between stations, only three variables show significant differences. These three variables are nitrate nitrogen, total particulates, and particulate phosphorus. In the case of nitrate nitrogen, the variation between stations is without question caused by the much greater depth of the water column at the index station. Since nitrate nitrogen increased with the depth during stratification, as shown in the seasonal analysis of data for the index station, this is not an unexpected result. The Student-Newman-Keuls Multiple Range Test segregates the index station from the other four stations at the 5% probability level. The results do indicate the significance of the rather large pool of nitrate nitrogen very deep in the water column, as shown also by the higher averages for nitrate near the bottom at the index station.

For total particulates and particulate phosphorus, the values for the index station are very much lower than for other stations.

However, examination of the raw data shows that the higher averages at the four main stations were due entirely to a handful of very high values clustered in the winter of 1981. Since there was very little water movement under ice in the winter, especially in the dry year of 1981, these high values were almost certainly produced by overly close approach of the sampler to the bottom, stirring up sediment and thus inflating the total particulate and particulate phosphorus values. This was not a problem in 1982. Thus for total particulates and for particulate phosphorus, we conclude that the average bottom values for the four main stations in this series are spuriously high.

Even though fixed horizontal variation is not significant either at the surface or at the bottom of the water column with the exception of nitrate in deep water and the spurious cases of total particulate and particulate phosphorus heterogeneity in deep water, ephemeral horizontal spatial heterogeneity may still be significant. A detailed look at the magnitude of ephemeral spatial variance is possible by separation of variance components. The separation will only be done for the surface sample series because of the much greater practical significance of heterogeneity in the growth zone of the lake.

Total variance of surface samples at five stations on a given date (s^2_t) can be broken down as follows:

$$s^2_t = s^2_e + s^2_s$$

where s^2_e is the error variance and s^2_s is the true variance attributable to stations. Since it has been shown that the true variance attributable to stations does not include any significant contribution resulting from fixed (temporally stable) differences

between stations, it follows that s^2_s is entirely accounted for by ephemeral spatial variation. Thus if separation of the variance components can be achieved, we will have a quantitative estimate of the magnitude of the ephemeral spatial variation.

The magnitude of error variance for the Dillon samples was obtained as part of the quality assurance program by routine analysis of replicates. For each date and each variable, the total variance among stations was obtained from the raw data and the replicate variance was subtracted from this total, leaving s^2_s. From s^2_s we obtained the standard deviation attributable to ephemeral horizontal spatial variation (s_s). We then calculated the coefficient of variation associated with this ephemeral spatial variation ($s_s/\bar{x} \cdot 100$). The mean values for these statistics are reported in Table 36.

The amount of ephemeral spatial variation, both in absolute terms (s_s) and in relative terms ($s/\bar{x} \cdot 100$) is exceedingly small for all variables. As might be expected, conductance, which is a biologically conservative property, shows the least ephemeral spatial variation, and some of the more biologically sensitive variables show greater ephemeral spatial variation. The overall conclusion is clear: on the average, the degree of horizontal spatial variation over the area covered by the main stations and the index station is very small in relation to the absolute magnitude of the variable that is being measured. A measurement at any one of these stations on a given day tells almost as much as there is to know about important chemical and

biological features of the main body of the lake and the mouths of the arms on that day.

When the values of s_s or the coefficients of variation corresponding to s_s are plotted against time, some trends are evident. For all of the variables, there were times of the year when the ephemeral spatial variation was essentially 0, and other times of the year when the ephemeral spatial variation was very definitely above 0, although still small in absolute terms. Table 36 incorporates a summary for each variable of the time spans when the ephemeral spatial variation was maximal in relative terms. Certain patterns are evident from the list of maxima and from the raw data themselves. First, all variables with the exception of total soluble N showed maxima in relative variation under ice or over a time period that overlaps with ice cover and early spring mixing. The nearly universal tendency toward maxima of relative heterogeneity under ice is explained by the physical condition of the lake at this time. Since the lake surface is not exposed to wind action, any heterogeneity that is induced by outside influences, or by biological phenomena, is much more likely to persist than it would be in the ice-free condition when the surface water can be homogenized by wind action. Many variables tended to show a period of higher heterogeneity only toward the end of ice cover, and this also makes sense in terms of the physical events that were occurring in the lake. Whereas watershed influence on the lake was absolutely minimal from the formation of ice cover in early January into March, spring weather was accompanied by rapidly increasing

Variable	Standard Deviation Among Stations*	Coefficient of Variation** among Stations (%)*	Time of Peak Variation
Conductance, umho/cm	1.9	1.2	April-May
PO_4-P, ug/l	0.20	8.3	Jan-April
Total Soluble P, ug/l	0.43	11.6	Jan-July
Particulate P, ug/l	0.26	4.5	Jan-April August
NO_3-N, ug/l	4.4	6.4	March-April, July
NH_4-N, ug/l	1.5	5.9	Jan-April
Total Soluble N, ug/l	33.	8.7	July
Total Particulates, mg/l	0.12	3.4	April-June
Chlorophyll *a*, ug/l	0.31	3.9	Jan-April, July-August

* Error variance has been removed.
**$(s_s/\bar{x}) \cdot 100$.

Table 36. Summary of variation among the 5 main stations (0-5 m), after removal of error (= analytical) variance, and the months of peak variation for each variable.

external influence in the form of snowmelt, which at first passed under the ice cover, thus magnifying the sources of heterogeneity.

Another common tendency was for relative ephemeral spatial variation to increase during late summer. This was true especially of inorganic nitrogen and phosphorus and to some degree also of chlorophyll *a*. At this time of year the phytoplankton were under extreme nutrient stress so that minor physical phenomena could induce measurable heterogeneity either in the amounts of soluble inorganic nutrients or in the amount of phytoplankton chlorophyll. It is important to note that, while these peaks of heterogeneity were undoubtedly real, their absolute magnitude was exceedingly small. By way of illustration, Figure 39 shows the mean chlorophyll values for the 5 stations over the two-year period of study and the accompanying estimates of standard deviation (s_s).

The general conclusion of the 5-station heterogeneity study is that the main body of the lake, as encompassed by the index station and the 5 main stations, can be treated as a unit, and that any one station on a given day is a good indicator of the condition of the entire central region of the lake.

The 14-Station Heterogeneity Study

The 14-station heterogeneity study included not only stations over deep water but also stations near the head of each one of the bays, and a number of stations in the broad Tenmile Arm. The locations of these stations have already been shown (Figure 2). The stations are organized in Table 37 according to location. Table 37 also summarizes

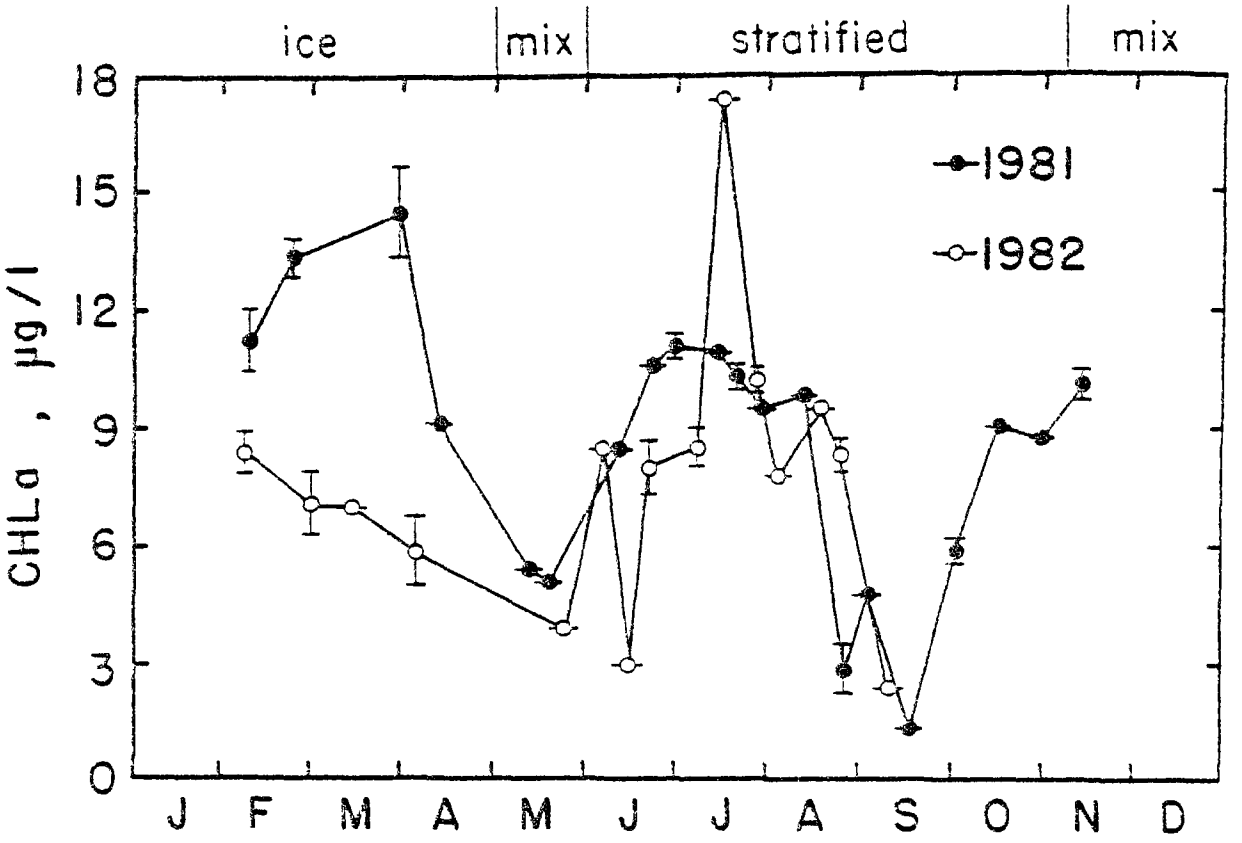


Figure 39. Means on various dates for chlorophyll for the 5-station heterogeneity study. For each mean, the vertical bar shows the standard deviation caused by true variation between stations (error variance removed).

Station location	Secchi (m)	Conductance (umho/cm)	PO_4-P (ug/l)	Total Soluble P (ug/l)	Particulate P (ug/l)	NO_3-N (ug/l)	NH_4-N (ug/l)	Total Soluble N (ug/l)	Particulates (mg/l)	Chlorophyll a (ug/l)
Lake Center										
Index Station (A)	2.8	164	1.6	3.6	5.2	105	12	227	3.3	7.3
2 km S. of Index (LS3)	2.5	158	0.6	4.3	5.4	93	11	225	3.5	8.1
Dillon Bay										
Near Dillon (LS1)	2.1	163	0.9	4.0	6.1	76	15	348	5.0	6.7
Mouth of Bay (B)	2.4	167	1.2	3.8	5.2	75	15	251	3.5	7.4
Snake River Arm										
Head of arm (LS2)	2.2	163	0.8	3.2	5.7	86	14	235	4.7	7.3
Mouth of arm (C)	2.4	158	0.6	3.8	5.3	83	12	252	4.3	7.3
Blue River Arm										
Head of arm (LS4)	2.8	164	0.8	3.6	6.8	78	34	256	4.1	8.6
Mouth of arm (D)	2.6	163	0.8	3.2	5.4	89	13	231	3.4	8.0
Tenmile Arm										
Head of arm east (LS9)	2.0	177	1.7	4.6	7.5	53	82	434	3.5	7.1
Head of arm west (LS8)	2.1	201	0.7	3.5	7.5	132	30	361	4.2	6.2
Middle of arm south (LS7)	2.4	175	0.8	3.7	6.0	83	18	215	3.1	7.4
Middle of arm north (LS5)	2.6	172	0.8	3.8	6.2	83	12	256	3.4	7.2
Gilbertson Bay (LS6)	2.5	171	0.9	4.1	6.4	78	15	243	3.1	7.6
Mouth of arm (E)	2.6	164	0.7	3.9	5.1	82	11	260	2.9	6.8
Grand Mean	2.4	168	0.9	3.8	6.0	86	20	267	3.7	7.3
Significant Differences	None	None	None	None	None	None	None	None	None	None

Table 17. Results of 14-station heterogeneity study, showing the means over the 14 stations (0-5 m) for the 10 sampling dates spanning the two years of study.

the mean values of each one of the variables for each of the 14 stations over the 10 sampling dates covered by the 14-station heterogeneity study. The data for each variable were subjected to analysis by one-way ANOVA according to a rationale identical to that used in the 5-station heterogeneity study. Once again, the null hypothesis was that the 14 stations are not significantly different from each other in the mean values for any given variable. In no case could the null hypothesis be rejected at the 5% probability level. We thus conclude that there are no significant differences in the mean values for any of these variables among the 14 stations, despite the coverage of all the different arms and of both shallow-water and deep-water portions of the lake.

Since the ability of a statistical test to discriminate differences is directly related to the amount of data available, extension of the 14-station heterogeneity series to cover 30 or 40 different dates instead of 10 different dates might have shown certain significant differences between stations. The important point, however, is that any such differences were so minor that they could not be detected at all with a 10-date series, which would have been sensitive to consistent differences of any great magnitude between stations.

The raw data and the table of means are suggestive of certain patterns that may be real, although they could not be verified statistically. For example, it would appear from Table 37 that the Tenmile Arm averaged slightly higher in conductance than any of the other arms of the lake or the main body of the lake. The higher

conductance of incoming water is largely offset by efficient mixing, however. It is likely that a more extensive data set would be able to show statistically that the Tenmile Arm has a slightly higher conductance on the average than the rest of the lake, but the degree of difference is sufficiently small that biological consequences would be minimal.

Examination of the raw data also indicates the tendency of shallow-water stations to show higher values of chlorophyll and lower transparencies during late summer at the time of most aggravated nutrient depletion (July-August). However, since this pattern is sustained for a very short time, it is difficult to demonstrate statistically and must be considered as suggestive rather than definitive. The pattern does make sense biologically insofar as there will be slightly richer supplies of inorganic nutrients where there is more intimate mud-water contact, as in shallow-water areas, and this may induce a noticeable difference at times when nutrient depletion is extreme.

Another pattern that is highly suggestive but cannot be proven statistically is the tendency of the stations nearest to wastewater treatment plant outfalls to show higher average values for inorganic nutrients, and especially for ammonia. In fact it is remarkable that these stations, because of their proximity to the outfalls, did not show significant differences from the other stations of the lake. Examination of the raw data shows, however, that the sewage effect is, although noticeable, relatively small in magnitude and sporadic in nature. Vigorous wind-driven circulation and variations in the

effluents themselves caused the effect to be undetectable more often than not even at the stations closest to the discharges.

The 14-station heterogeneity study argues very strongly that the lake, in terms of general chemistry, nutrient chemistry, and biological variables, can be treated as a unit, and not as a collection of semi-isolated arms and bays that react at grossly different rates to nutrient loading or seasonal events. Although this is counterintuitive to the apparent semi-isolated condition of the arms and the continuous entry of water of different quality into each of the arms, it is obvious that mixing of the lake is sufficiently pronounced under the influence of the wind to homogenize the chemistry and biological variables very efficiently. Perhaps this is not so surprising, in view of the relatively small size and high wind exposure of the lake. Both of these factors act against the development of pronounced or persistent patchiness that would produce significant differences in the arms or between stations.

Overview of Limnology and Trophic Status

The Annual Cycle

Figure 40 is a general synopsis of the annual cycle based on the foregoing chapters that have dealt with physical, chemical, and biological variables in Lake Dillon. The cycle is generalized for the two years of record. Layering and mixing patterns are the key to understanding chemical and biological changes through the year. Since layering and mixing are principally under the control of seasonal changes in air temperature and solar radiation, the general pattern will be very much the same from one year to the next. This is well illustrated by the similarity of layering and mixing phenomena in 1981 and 1982, which were meteorologically quite different in amount of runoff.

The predictable duration and timing of ice cover, spring mixing, fall mixing, and summer stratification enforce a certain amount of order in the other events in the lake. We thus expect the synopsis shown in Figure 40 to have general validity even if the trophic status of the lake should change in the future. Chlorophyll maxima will consistently occur in the month of July, and the months of May and September will show surface chlorophyll minima. The nutritional status of phytoplankton will be most optimal in May and September and least optimal in August. Transparencies will consistently be low in June due to runoff and in July due to phytoplankton, and will be highest in August because of low chlorophyll concentrations resulting from nutrient depletion at that time.

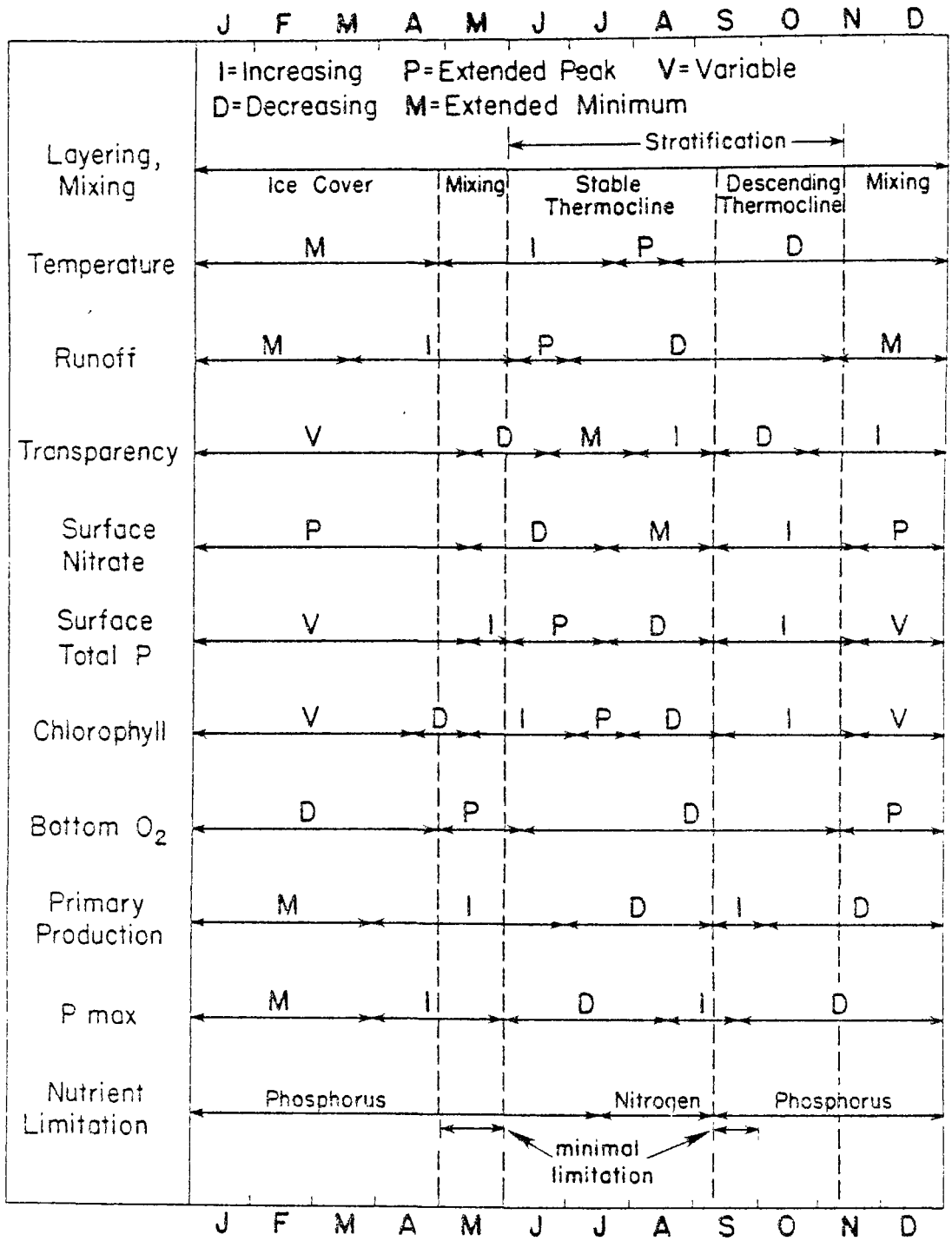


Figure 40. Summary of seasonal events, generalized as a composite of 1981 and 1982.

Despite the persistent annual pattern, certain features of the annual cycle are subject to influence by irregular variations, and others are subject to change if the nutrient supply to the lake should change. Runoff is exceedingly variable from one year to the next and it has certain direct influences on the chemistry and biology of the lake. It is runoff, not phytoplankton, that determines the transparency of the lake in June. Higher runoff will thus mean a lower transparency in June. In fact, under present conditions, the annual minimum transparency in an average year is as likely to occur in June in connection with particulate loading as it is to occur in July in connection with the chlorophyll maximum.

Higher runoff will also raise the phosphorus peak that occurs in June and July, and this will in turn raise the peak and average annual total phosphorus, at least in the upper water column. Counter to intuition, the present study shows that higher runoff than average is likely to suppress primary production even though it raises the peak total phosphorus concentration. Inorganic particulates connected with high amounts of runoff act against higher production, as they interfere with use of light by the algae. Higher runoff also maintains a thinner epilimnion and causes injection of the particulates closer to the lake surface. These factors enhance the extinction of light by nonliving particulates, thus discouraging significant production below about 5 m. This accounts for the suppression of production per unit area by particulates.

Biomass accumulation, as measured by chlorophyll concentrations, is likely to be very similar under high and low runoff conditions,

despite the difference in production. The chlorophyll maximum in a year of low runoff is likely to be situated well below the surface, while the maximum in a year of normal or above-normal runoff will probably include the upper 5 m. On the other hand, the total amount of chlorophyll under a unit of lake surface will be pretty much the same under the two conditions, if the two years of record are accurate indicators.

Under the present trophic condition, the phytoplankton of Lake Dillon respond to enrichment with phosphorus plus nitrogen at any time of the year, although the degree of response is minimal in May and September when the upper water column has been freshly enriched with deep water containing supplies of available nutrients. In 1981 and 1982, there was a critical switch in limiting nutrients from phosphorus to nitrogen in the middle of July, and this is probably a general phenomenon for the lake in its present condition. The phytoplankton appear to use up the orthophosphate almost immediately after the onset of stratification. Growth continues nevertheless and is accompanied by steady decline in soluble organic phosphorus. The algae at this time are phosphorus limited, and respond strongly to orthophosphate addition. However, they have not exhausted the phosphorus supply because they are able to use soluble organic phosphorus, although they obtain it more slowly and less efficiently than they would orthophosphate and are thus somewhat suppressed in their growth rates by the unavailability of orthophosphate. In the middle of July, growth is essentially halted not by phosphorus but rather by inorganic nitrogen, which is present in abundance at the beginning of

stratification but is finally used up by the middle of July. The effect of this nitrogen exhaustion is more drastic than the exhaustion of orthophosphate, implying that the phytoplankton are less able to use the soluble organic nitrogen pool than to use the soluble organic phosphorus pool.

Although phosphorus is presently limiting over a much greater segment of the year than is nitrogen, it is inorganic nitrogen that halts the progress of the summer growth and thus determines the maximum amount of chlorophyll in the epilimnion. Since the summer minimum of transparency is a point of major practical concern resulting from eutrophication, it is fair to ask whether the growth control strategy for algae should be based on nitrogen rather than phosphorus. Aside from economic considerations, two biological factors should be weighed. First, under the present conditions, the lake is very close to a balance between phosphorus and nitrogen limitation. By the time inorganic nitrogen limitation occurs, phytoplankton have removed not only all the orthophosphate but virtually all of the soluble organic phosphorus as well. Large responses to enrichment can be obtained only with a combination of phosphorus and nitrogen. Since the balance is so close, the choice for management is essentially arbitrary and can be based on economic considerations or convenience (cf. OECD 1982).

Another biological consideration is the possible encouragement of nitrogen fixers if the present nitrogen limitation were exacerbated by control of nitrogen input with the same phosphorus loading or greater phosphorus loading. As the balance is shifted more and more strongly toward nitrogen limitation, there is an increasing ecological incentive

for the development of algal taxa that do not have high nitrogen requirements. The algae that have lowest nitrogen requirements are those that can fix nitrogen, specifically the heterocystous blue-greens such as Anabaena and Aphanizomenon. Any shift toward the dominance of such algae is usually considered undesirable, as these algae also have gas vacuoles which cause them to rise to the surface and form scums. Thus while the July chlorophyll maximum of 1982 gradually sank out of sight as shown in Figure 29, a comparable amount of Anabaena biomass might well have come to the surface, where it would have affected the appearance and surface transparency of the lake. This and other considerations will be taken up again later in connection with predictions and modelling.

Trophic Status of the Lake

The present trophic status of the lake is a matter of concern. The trophic status of the lake can be evaluated on the basis of any one or a combination of several trophic indicators. The most useful indicators are those which can combine the practical concerns of eutrophication and biological fidelity. Table 38 lists four such indicators available for Lake Dillon.

Lakes are traditionally classified as oligotrophic, mesotrophic, or eutrophic. The assignment of a given lake to one of these categories is to some degree arbitrary. Using any single trophic indicator, lakes could be arranged in a continuous spectrum that lacks sharp boundaries. Furthermore, the use of several different trophic indicators will typically provide slightly different perspectives on

Indicator	Magnitude	Trophic Indication
Total Phosphorus (ug/l)		
Spring surface maximum day	12-16	Lower mesotrophic or
Spring surface maximum month	11-13	upper oligotrophic
Annual average surface	7-9	
July-October average 0-15 m	7-8	
Transparency (m)		
July secchi minimum day	1.5-1.9	Mesotrophic
July secchi average	1.8-2.3	
July-October average	2.5-3.5	
Chlorophyll a (ug/l)		
Summer surface maximum day	10-17	Mesotrophic
Summer surface maximum month	9-12	
July-October average	6-8	
Primary Production		
Annual ($gC/m^2/yr$)	50-100	Mesotrophic
Maximum daily ($mgC/m^2/day$)	500-900	
Summer oxygen		
AHOD ($mgO_2/m^2/day$)	600-700	Lower eutrophic
5 m above bottom minimum (mg/l)	4-5	

Table 38. Summary of trophic indicators for Dillon under conditions at the time of the study. Range of magnitudes shows, in round figures, probable span with varying weather in different years.

the placement of a particular lake in a trophic sequence. Thus while Table 38 attempts to assign a trophic status on the basis of each of the trophic indicators, principally on the basis of the literature survey by Welch (1980). The categories are of course to some degree simply a reflection of tradition in classifying lakes.

Phosphorus is a widely used criterion for classifying lakes according to trophic status. Although it has been recognized for decades that phosphorus is of great importance in determining trophic status, an attempt to make a quantitative linkage between phosphorus and trophic status dates back principally to Vollenweider (1965). Vollenweider, following Sawyer (1947), specified that oligotrophic lakes have total phosphorus concentrations below 10 ug/1, that mesotrophic lakes have total phosphorus concentrations between 10 and 20 ug/1, and that eutrophic lakes have total phosphorus concentrations above 20 ug/1. No simple rules are available according to which one can select appropriate sampling times and the appropriate number of samples for determination of total phosphorus, however. Depending on whether one uses annual or seasonal averages or seasonal peaks, Dillon might currently be classified as lower mesotrophic or upper oligotrophic according to Vollenweider's criteria. Furthermore, it should be remembered that Vollenweider's classification has presently been modified to take flushing rate into account, and this has not been done for Table 38. This matter will be taken up more fully along with the model of Lake Dillon.

Transparency, chlorophyll and primary production are also possible indicators of trophic status. If phosphorus is a major determinant of

trophic status, then transparency and chlorophyll are major symptoms. Secchi depths in Dillon have low values both in June and July. Table 38 gives the July secchi depth as the basis for judging trophic status in this lake simply because the low secchi depth values in June are due to inorganic particulates rather than chlorophyll. Secchi values in July are low enough to bring the lake into the mesotrophic category by traditional criteria (Likens 1975, Welch 1980). Similarly, chlorophyll a maxima and growing season averages indicate mesotrophic status. In fact the chlorophyll a values are higher than in most lakes that have comparable total phosphorus values (e.g., Kalff and Knoechel 1975). The reasons for this will be explored in connection with the development of the model. Primary production, considering the short growing season, is also in the mesotrophic range.

Areal hypolimnetic oxygen deficit is also a potential indicator but is less often used than the other indicators of Table 38. Mortimer (1941) set boundaries of 250 and 550 $mgO_2/m^2/day$ dividing the three trophic categories. Thus according to this indicator, Lake Dillon is just entering the eutrophic category.

In summary, the best characterization of Lake Dillon at the present time seems to be mesotrophic. The lower mesotrophic boundary is an important one, as the visible changes in the lake are most notable to the casual observer over the mesotrophic range. Lake Dillon has passed an important transition between trophic categories and can no longer be considered oligotrophic.

Chemistry of Nutrient Sources as They Enter the Lake

Pathways followed by water and nutrients entering Lake Dillon can be divided for present purposes into six categories: (1) major rivers, (2) streams not joining a major river prior to reaching the lake, (3) sewage effluents, (4) miscellaneous surface drainage not accounted for in other categories, (5) precipitation, and (6) groundwater. All of these categories except miscellaneous surface drainage and groundwater, which made minor contributions, were studied chemically at or very near their point of entry to the lake. In the first category are the Snake River, the Blue River, and Tenmile Creek. These will take up most of the attention of the section, since they account for the bulk of surface loading to the lake. In the second category are Soda Creek and Miner's Creek, both of which were sampled above the point sources that enter them near their mouths. In the third category are the effluents of the Frisco WWTP (which enters Miner's Creek near the lake) and Snake River WWTP (which enters Soda Creek near the lake). Both of these effluents were sampled prior to entering streams. Breckenridge effluent was part of the Blue River sample, since the effluent enters above the river mouth, and Copper Mountain effluent was part of the Tenmile Creek sample for the same reason. Discharge data for rivers were obtained from U.S.G.S. data records. For the effluents, discharge data were obtained from plant operators, and for Miner's Creek and Soda Creek the data were based on our own measurements. All chemistry data were from our own analyses. The precipitation chemistry data were based on the collector near the Snake River WWTP.

The chemistry of the nutrient sources that were sampled routinely will be considered here. Contributions of groundwater and miscellaneous surface drainage will be approximated in the next section, which gives total transport, and the breakdown of surface sources above their points of entry to the lake will be made in a subsequent section on land use in relation to nutrient yield.

24-hour Variation in River Chemistry

The mouths of the major rivers were sampled on 54 different dates over the two-year period. Since the samples were taken mostly during the daylight hours, a question arises as to whether there is any regular periodicity in stream chemistry that might cause this type of sampling to introduce bias. Although no major periodicity was expected *a priori*, two 24-hour studies were undertaken in order to show whether periodicities were of any concern. A secondary purpose of these studies was to show the degree of short-term irregular variation, which would contribute to random noise in the data but would not bias the results of a large data set.

On 14 January 1981, samples were taken at each of the three river mouths at three-hour intervals beginning at 1600 and ending at the same time the next day. These samples were analyzed for total particulates, ammonium, nitrite plus nitrate, and orthophosphate. The study was repeated in November 1981 with a slightly different sampling interval (four hours). In the November sampling, analyses were made for total soluble phosphorus, total soluble nitrogen, particulate phosphorus, and total particulates. In November 1981 the samples of the Blue River

were taken above the Breckenridge Wastewater Treatment Plant, and in January 1981 they were taken below it. In November 1981 the Tenmile Creek samples were taken above Frisco and the January samples were taken below Frisco.

The results of the two 24-hour studies are analyzed here in two stages. First, the data series for each variable at each river mouth on each date is tested for the existence of significant additional variance above and beyond that which would be expected from analytical variance alone. The presence of significant additional variance indicates either irregular or patterned variation in the stream, or some combination of these. If additional variation is present, the distinction between irregular and patterned variation will be made on the basis of subjective evaluation of pattern in the data, since the time series is too short to allow statistical time series analyses.

Table 39 gives the mean for the variables over the 24-hour sampling interval at each of the three sites on each of the two dates. In addition, the standard deviation of each series is shown. The ratio of the square of the standard deviation to the error (= analytical) variance is the F statistic, which is also given. The F statistics for each set of analyses were compared with critical values for F at the 0.05 probability level (Rohlf and Sokal 1969). On this basis, F values indicating sample variance significantly in excess of the analytical error variance were identified and have been marked with asterisks in the table. In instances where the F statistic gives evidence of significant true variation in stream chemistry over the 24-hour cycle, the variation may have been either patterned or irregular.

	Snake River			Blue River			Tenmile Creek		
	Mean	S.D.	F	Mean	S.D.	F	Mean	S.D.	F
January 1981									
Particulates, mg/l	1.46	1.40	6.32*	0.98	0.44	0.62	1.11	1.84	10.7*
NH_4-N, ug/l	10.3	6.4	1.30	434	223	1570.*	495	81	209.*
NO_2 + NO_3-N, ug/l	124	8.0	1.83	577	417	4995.*	1006	85	207.*
PO_4-P, ug/l	.33	.61	2.32	1.45	1.51	14.*	.77	.68	2.89*
Particulate P, ug/l	1.71	1.13	3.79*	3.6	0.53	0.83	2.63	1.65	8.09*
November 1981									
Particulates, mg/l	1.62	.61	1.19	1.84	0.53	0.90	.79	.65	1.34
Total Soluble N, ug/l	263	36.	0.21	118.	39.	0.25	117.	20.	0.06
Particulate P, ug/l	1.1	.70	1.45	2.9	.2	0.11	1.8	.6	1.07

* sample variance exceeds error variance significantly ($p < 0.05$)

Table 39. Means and standard deviation for the two 24-hour sampling series, and the F statistic for each series.

Significant variation in stream chemistry above the error variance did not occur for any of the analyses at any of the three rivers in the November sample series, so we conclude without further analysis that on this date a sample taken at any time of day would have represented without significant bias the chemistry of the river from which it was taken. In January, however, there was significant variation in stream chemistry over the 24-hour cycle for about two-thirds of the combinations of variables and collection sites. Tenmile Creek showed significant variation for all variables. The Blue River showed significant variation for three of the four variables and the Snake River showed significant variation for only one variable.

Figure 41 depicts the stream chemistry values for each combination of sites and variables in the January series where there was evidence of a significant variation in stream chemistry over the 24-hour sampling period. The data do not suggest any kind of repeated pattern. For example, there was a definite minimum of ammonium on the Blue River at 4 p.m. one day, but the next day the ammonium concentration was high at 4 p.m. Although the data set is not large enough to allow rigorous statistical analysis for pattern, we conclude with reasonable certainty that the added variance in each case is due to randomly timed irregularities and not to predictable patterns in stream chemistry. This being the case, samples taken at any time of the day or night will represent the stream without bias, although a larger sample series is required to yield a given confidence interval for transport when added variance is present.

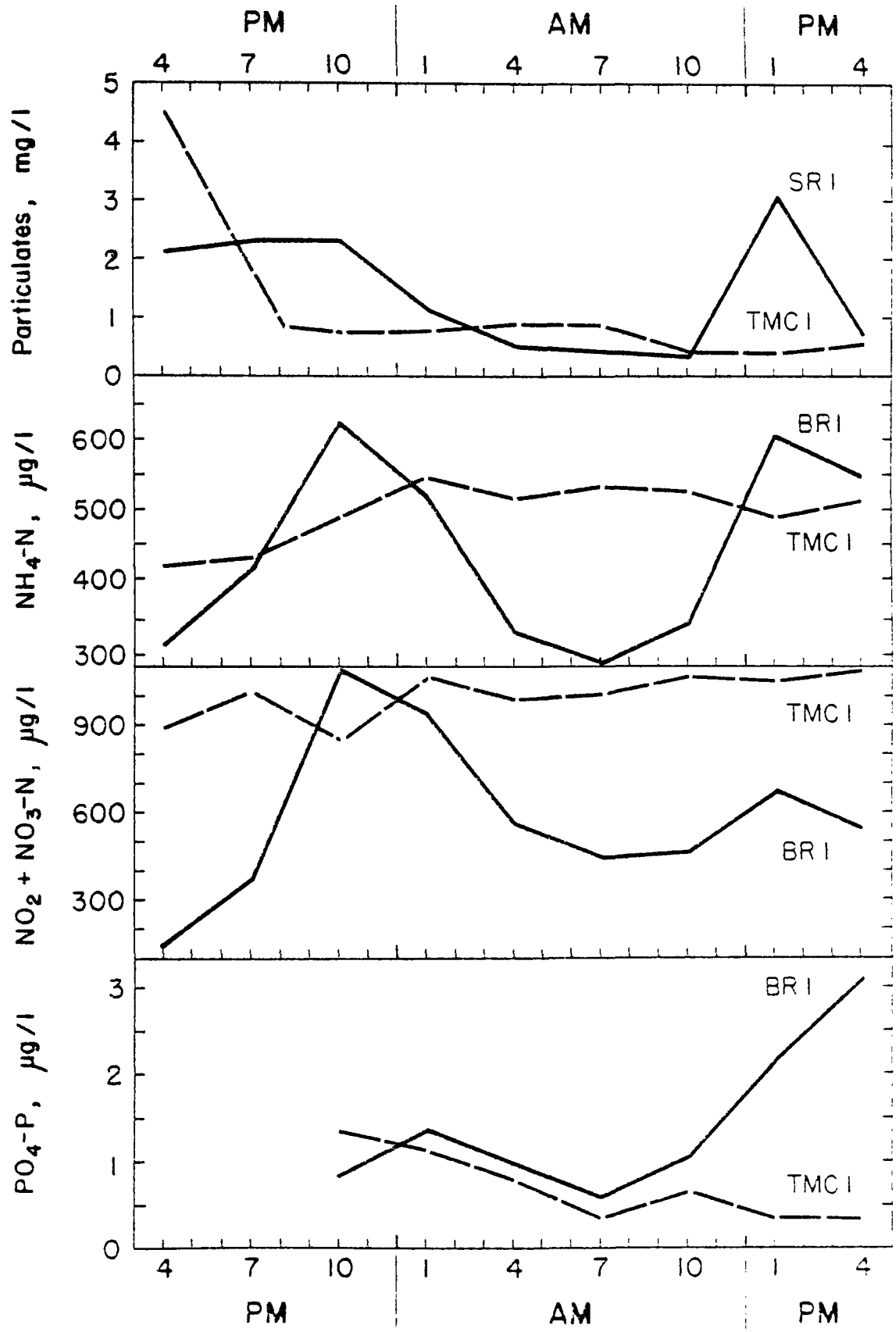


Figure 41. Data that showed statistical evidence of significant diel variation during the study of January 1981 (SR1 = Snake River TMC1 = Tenmile Creek, BR1 = Blue River).

Chemistry of the Three Rivers as They Enter the Lake

Table 40 summarizes the average concentrations of dissolved and suspended materials in the three rivers as they enter the lake. Averages can be obtained by unweighted averaging, time-weighted averaging, or discharge-weighted averaging:

$$\text{Unweighted averaging} \quad \bar{C} = \sum_{i=1}^{n} C_i / n$$

$$\text{Time weighted averaging} \quad \bar{C} = \sum_{i=1}^{n} C_i t_i / \sum_{i=1}^{n} t_i$$

$$\text{Discharge weighted averaging} \quad \bar{C} = \sum_{i=1}^{n} C_i D_i / \sum_{i=1}^{n} D_i$$

where $\bar{C}$ is mean concentration, C_i is the concentration at the ith time interval, t_i is the length of the interval, and D_i is the discharge over the interval.

In the present case unweighted averaging is not very defensible because the samples were taken more frequently at times of high discharge when concentrations of dissolved and suspended constituents were likely to be changing most rapidly. A time-weighted average would give the best idea of the chemical composition of the rivers without regard to seasonal changes in discharge, but the discharge-weighted average is the best representation of the chemistry of the entire annual flow accumulated in proportion to discharge, as it actually is in the lake. Thus Table 40 shows the discharge-weighted averages. Time-weighted averages might be useful for certain purposes, but are not tabulated because they are in most instances very close to the

	1981						1982					
	Snake River		Blue River		Tenmile Creek		Snake River		Blue River		Tenmile Creek	
	Mean	S.E.	Mean	S.E.	Mean	S.E.	Mean	S.E.	Mean	S.E.	Mean	S.E.
NO_2 N, ug/l	0.9	0.12	10.	1.92	3.5	2.4	1.4	0.58	3.2	0.79	11.	1.5
NO_3-N, ug/l	95.	6.7	227.	43.7	212.	30.	97.	10.6	131.	17.0	188.	25.1
NH_4-N, ug/l	11.	1.9	187.	36.0	26.	13.	12.	1.6	107.	32.5	94.	13.9
Total Soluble N, ug/l	183.	32.3	551.	104.	357.	78.	234.	30.6	382.	74.7	476.	69.1
Particulate N, ug/l	33.	6.8	28.	5.4	66.	16.	107.	34.9	73.	17.9	89.	17.9
PO_4-P, ug/l	1.2	0.15	14.	2.55	1.1	0.14	2.7	0.50	4.3	0.68	2.1	0.37
Total Soluble P, ug/l	2.0	0.21	14.	2.89	3.6	0.44	6.3	1.2	6.7	0.69	5.5	0.73
Particulate P, ug/l	4.2	0.53	5.5	0.64	7.1	1.7	6.9	1.0	5.6	0.54	5.9	0.63
Total Particulates, mg/l	4.0	0.79	2.6	0.48	10.	3.0	6.1	.81	3.8	0.37	4.5	0.58
Particulate C, ug/l	252.	38.7	194.	21.4	691.	195.	451.	91.0	184.	24.3	304.	44.7
Alkalinity, mg/l	17.	1.1	44.	0.94	35.	2.18	23.	1.9	40.	1.3	39.	1.9
pH	7.0	0.05	7.6	0.09	7.3	0.07	7.4	.03	7.8	0.05	7.8	0.05
Conductance, umho/cm	101.	2.55	168.	3.14	190.	25.3	84.	4.2	134.	3.5	236.	18.0
Discharge, l/sec	1145.	*	1585.	*	1839.	*	2334.	*	3113.	*	3299.	*
Discharge, cfs	44.	*	56.	*	65.	*	92.	*	110.	*	117.	*

* Continuous record from U.S.G.S.

Table 40. Discharge weighted means and standard errors of means for water chemistry variables in the 3 rivers at their point of entry to the lake.

discharge-weighted averages. There are a few exceptions to this. For example, total soluble nitrogen can be as much as 25-50% higher when expressed as a time-weighted average than it is when expressed as a discharge-weighted average. Total particulates and all particulate constituents are consistently higher when expressed as discharge-weighted averages than when expressed as time-weighted averages. This is explained by a general tendency of particulate loads of running waters to increase as a power function of discharge (Leopold et al. 1964, Bormann et al. 1969). In contrast, some dissolved constituents show a tendency toward higher means for time-weighted averages than for discharge-weighted averages because of dilution effects that operate especially on point sources, but also on non-point sources and of certain constituents of natural soil-water systems (Lewis and Grant 1979).

A detailed consideration will be presented later of the relative contributions of point and non-point sources to the total transport of the three rivers. For the time being, however, it suffices to note that the distribution of land uses and point sources differs among the three rivers. The Snake River is not influenced by any major point source, although it does reflect the influence of non-point source nutrient loading from ski slopes and residential development. The Blue River is influenced by a variety of non-point sources and also by a major point source, the Breckenridge Wastewater Treatment Plant. Tenmile Creek is influenced by mining operations at Climax Molybdenum Mine in the headwaters, plus a variety of non-point sources between the headwaters and the mouth. In addition, Tenmile Creek is influenced by

the Copper Mountain Wastewater Treatment Plant, which is situated about 10 km upstream from the sampling point at the mouth (Figure 1).

Table 40 shows, as might be expected from the foregoing synopsis of the three watersheds, that the soluble inorganic nitrogen and total soluble nitrogen concentrations for the Snake River in 1981 and 1982 were considerably lower than for the other two rivers. For all three rivers, half to three-quarters of the soluble nitrogen was inorganic. Particulate nitrogen was generally present in concentrations much lower than soluble nitrogen. Only in the Snake River in 1982 did the particulate nitrogen approach one-third of the total nitrogen.

Contrasts among the three watersheds were not so extreme for phosphorus compounds as they were for nitrogen compounds. Orthophosphate was considerably higher in the Blue River than in either of the other two rivers. This was largely due to the proximity of the treatment plant to the collection point. The 1981 concentration was exceptionally high, and requires special explanation. A scatterplot of the concentration of orthophosphate in water of the Blue River shows that the very high average for 1981 was caused by exceptionally high values of orthophosphate for the interval September through November of 1981 (Figure 42). These high concentrations were not repeated in the following year, nor did they appear at the next station upstream on the Blue River between September and November of 1981. It is thus clear that some special phenomenon occurred at the treatment plant between September and November of 1981 that was responsible for the exceptionally high average of orthophosphate in 1981. We postulated a treatment plant malfunction, and this was subsequently confirmed by

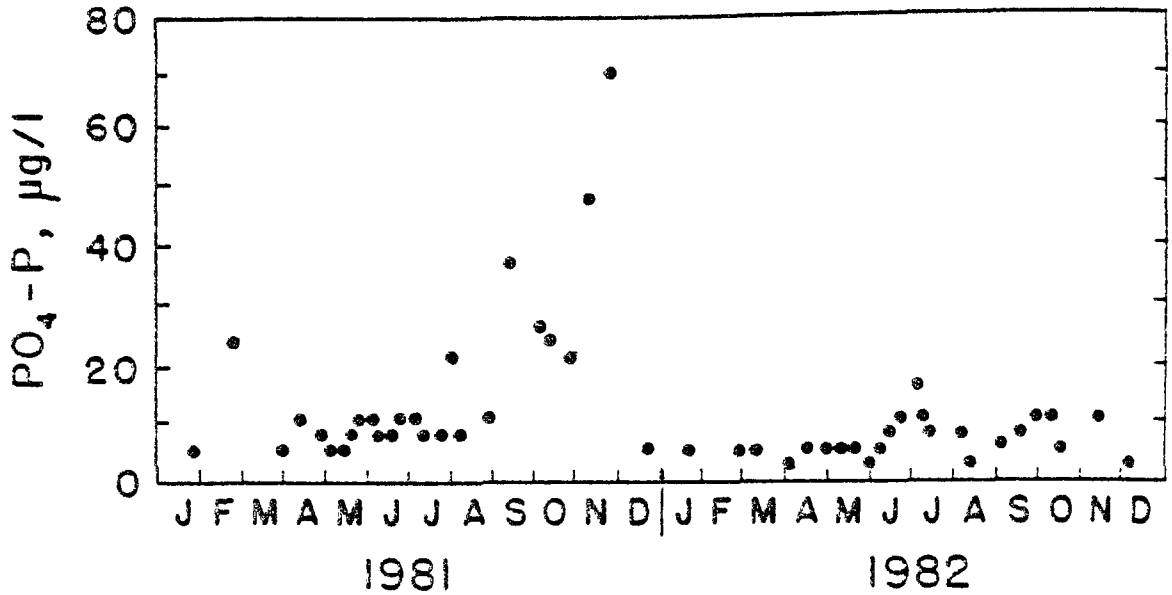


Figure 42. Orthophosphate data for the Blue River at its point of entry into Lake Dillon, showing the effect of malfunctions in tertiary treatment at the Breckenridge WWTP, September-November 1981.

contact with the plant operators. Although the plant had returned to normal function by November when values were still high, there was evidently some residual effect. Figure 42 is a dramatic illustration of the effectiveness of present treatment practices.

The concentrations of total soluble phosphorus and orthophosphate both tended to be higher in 1982, a wet year, than in 1981, a dry year (discounting the high values for Blue River 1981). A similar trend has been shown for an undisturbed watershed near the Continental Divide (Lewis and Grant 1979). Higher streamwater concentrations in 1982 are consistent with the higher total soluble phosphorus concentrations in the lake in 1982. The higher concentration of soluble phosphorus in wet years is postulated by Lewis and Grant (1979) to be caused by a critical shift in the balance of biological phosphorus demand, which sequesters P, and the influence of physical forces, which remove soluble P from the zone of demand.

The particulate phosphorus and total particulate concentrations of Tenmile Creek were higher in 1981 than in 1982, contrary to what might have been expected. Examination of the data shows that this was due mostly to high particulate concentrations observed at peak runoff in 1981 but not in 1982. This phenomenon was observed not just at the mouth of Tenmile Creek but at other points on the Creek as well. Its explanation will be considered along with the segment-by-segment analysis of Tenmile Creek to be given in another section of this report.

Conductance, representing total soluble ionic materials, shows the expected dilution effect between years of high discharge and years of

low discharge for the Snake River and Blue River, but not for Tenmile Creek. In addition, the data show clearly that the conductance of Tenmile Creek averaged considerably higher than that of either of the other two rivers. Major sources of ionic dissolved solids associated with mining are located near the head of Tenmile. These raise the average conductance for Tenmile and can produce irregularities that are not part of the natural seasonal cycle.

Most of the dissolved and suspended materials showed some kind of seasonal pattern in all three rivers. The natural seasonal pattern is more difficult to pick out for the Blue River and Tenmile Creek than it is for the Snake River because of the influence of point sources. We therefore use the Snake River to illustrate the background seasonal patterns, and then describe how the other two rivers differ from the Snake River.

Figures 43-44 show the seasonal patterns of selected variables over the two years of study for the Snake River. Some variables have been omitted because their indications are redundant or because they are not of special interest here. Discharge is a key variable, not only because it embodies the seasonal changes that affect concentration, but also because it is multiplied by concentrations in the process of obtaining discharge-weighted concentration averages. The pattern shown by discharge in Figure 43 for the Snake River is reflected very closely in the discharge patterns of the other two rivers. The discharge patterns were strongly dominated by seasonal runoff associated with snowmelt. In 1981, when total annual discharge was low, discharge peaked in the first week of June. The peak was

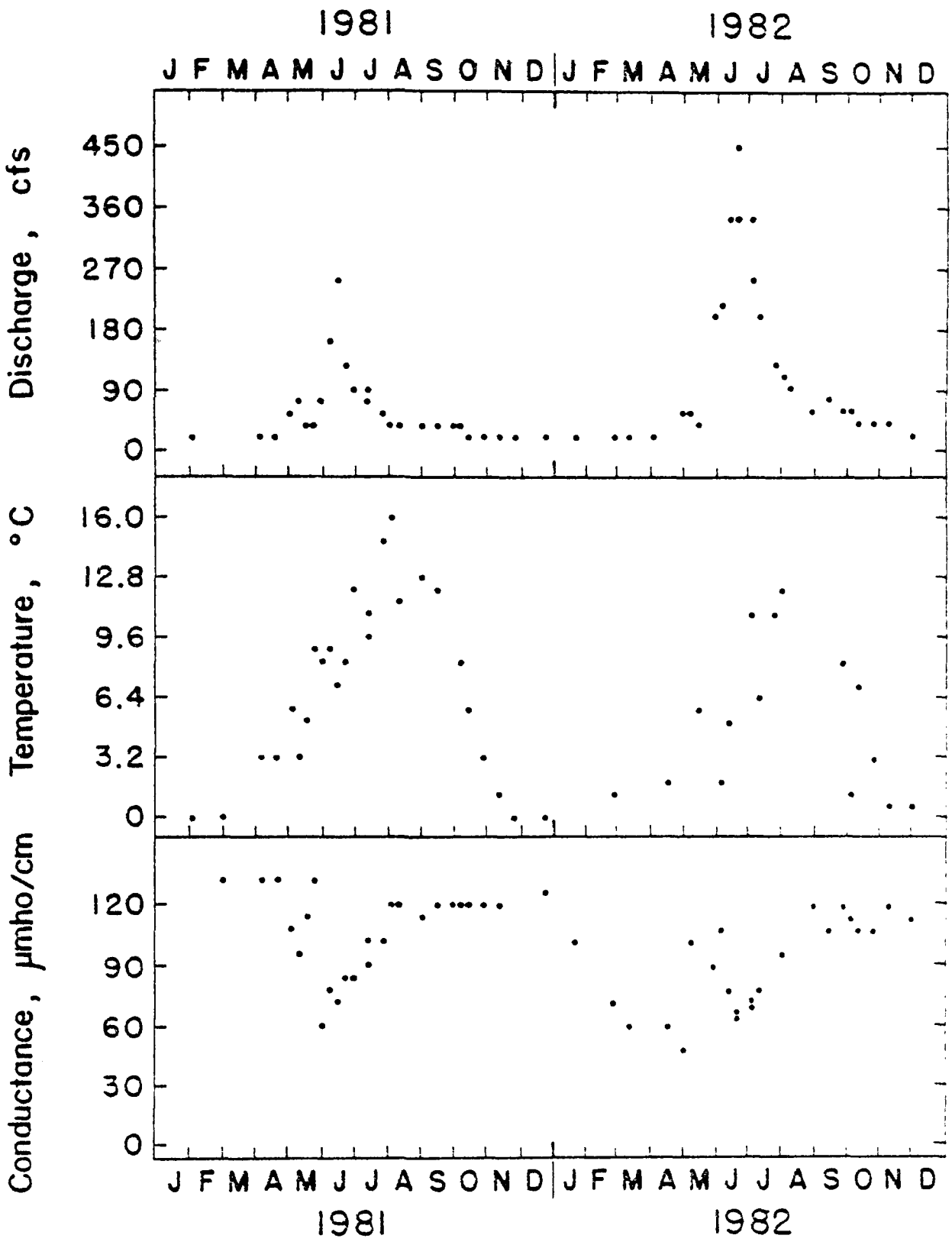


Figure 43. Values of selected variables for the Snake River as it enters Lake Dillon.

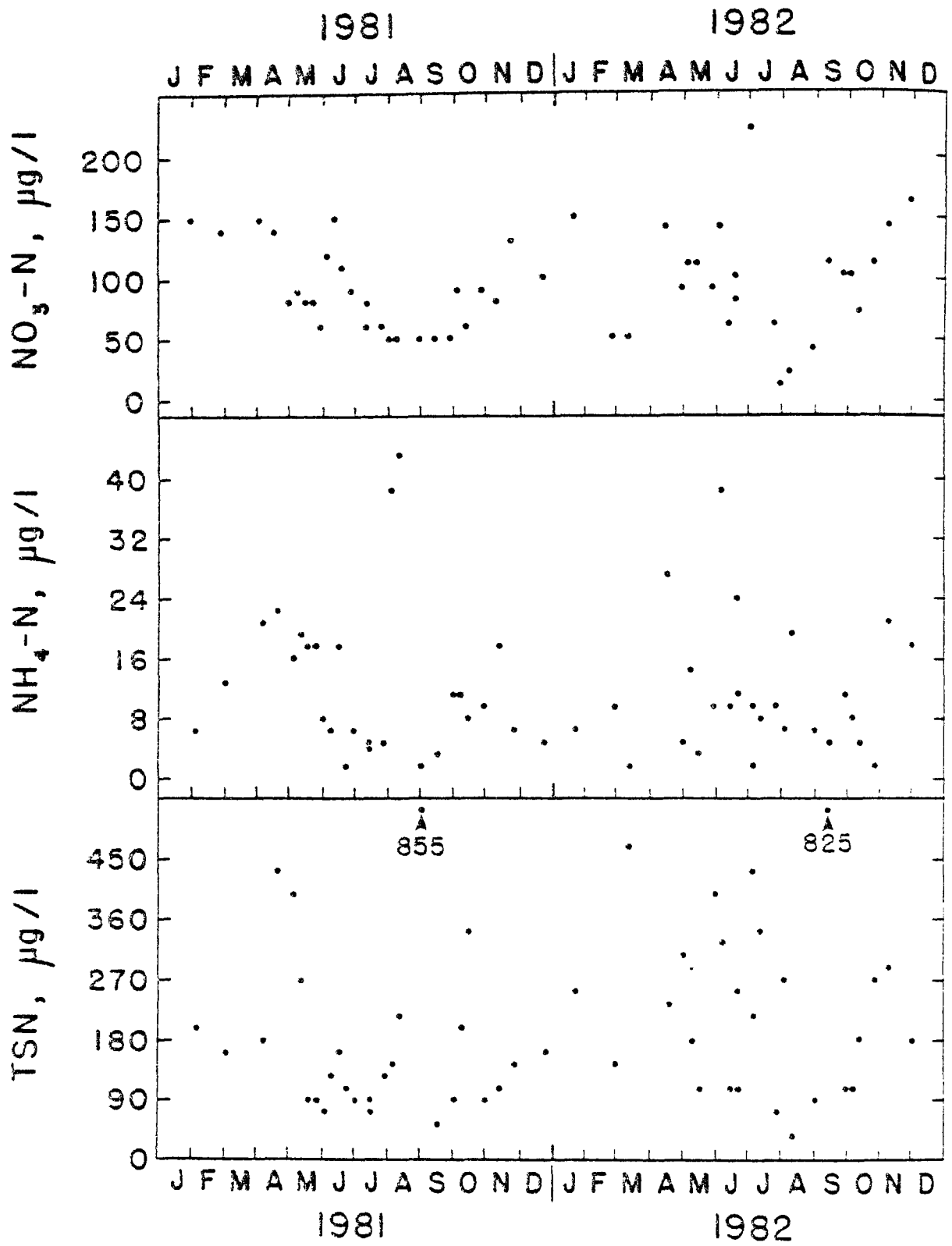


Figure 44a. Values of selected variables for the Snake River as it enters Lake Dillon.

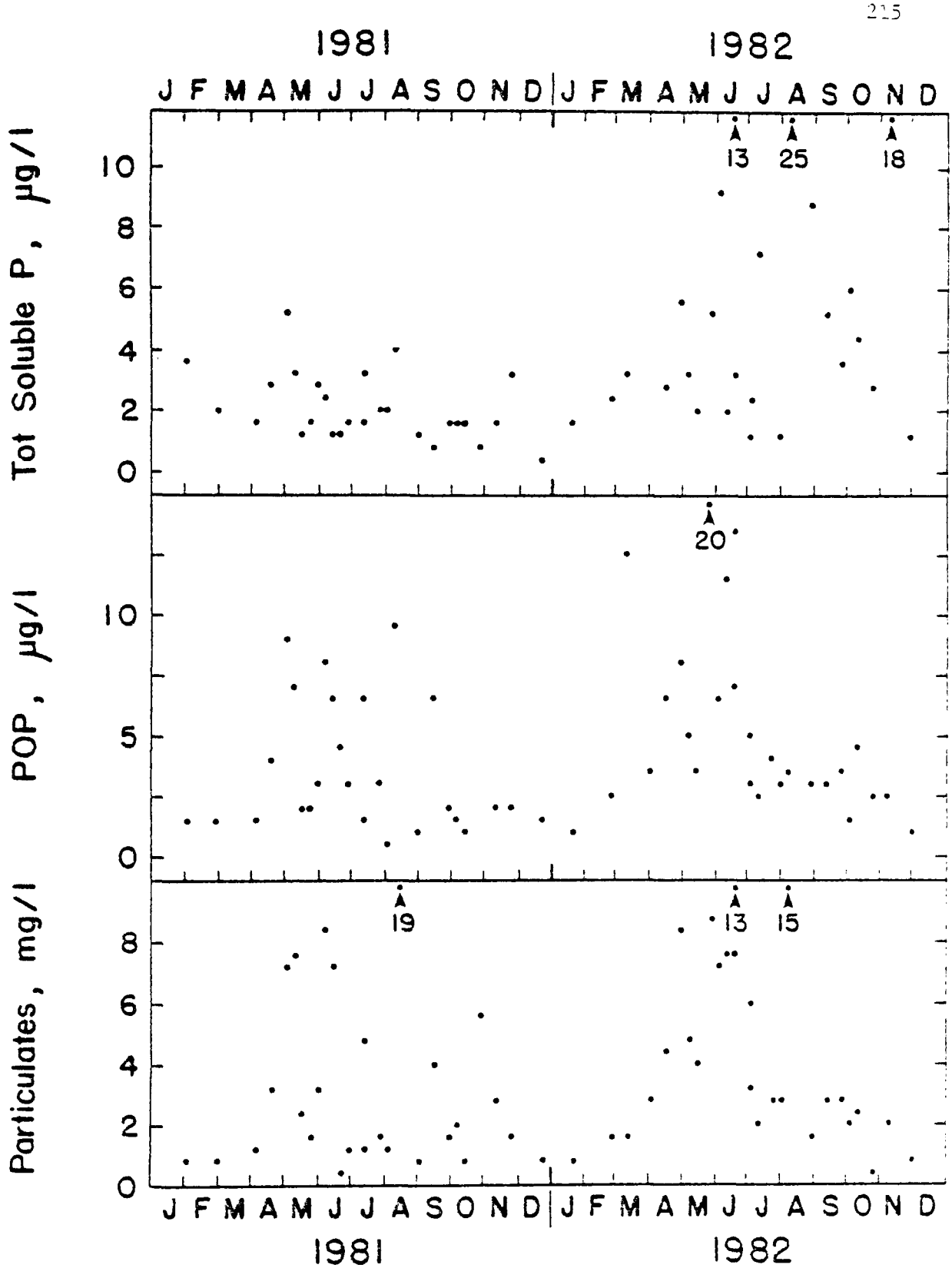


Figure 44b. Values of selected variables for the Snake River as it enters Lake Dillon.

broader in 1982 and the highest values did not occur until the end of June. In both years there was a small early peak in May, probably due to early melt in the valley bottoms, and a shoulder or low peak in August or September. The amount of variation around the seasonal pattern was small, which provides an ideal situation for the estimation of weighted mean concentrations and transport.

Stream temperatures varied between 0 and 16°C. The highest temperatures occurred during the last half of July. Maximum temperatures were lower in 1982 than in 1981 by 2-4°C. The seasonal change in temperatures was relatively smooth, although some irregularities were caused by variations in time of day when temperature was taken. Variation of temperature with season and between years affected the depth of penetration of water entering the lake, as already explained.

Conductance showed the dilution effects that one would expect to be associated with changes in discharge for materials that are soluble and little affected by biological activity. For the Snake River, the conductance faithfully reflected the early small peak in discharge (May), the subsequent brief decline in discharge, and the major peak of discharge caused by spring runoff. This sequence of events was visible both years for the Snake River and was also characteristic of the other two rivers. However, the winter conductances of Tenmile Creek were especially high because of the presence of especially potent sources of soluble materials in the upper part of this watershed, at Climax Molybdenum tailings ponds.

Nitrate concentrations of the Snake River were high at the time of runoff and then decreased steadily until August, after which they increased steadily until December or January. Midwinter concentrations were typically high. The low nitrate concentrations in the late summer months were almost certainly due to high summer biological demand for inorganic nitrogen, which became all the more effective in reducing nitrate levels as the amount of water movement declined at the end of summer. With the onset of cool weather, and reduced biological demand, nitrate concentrations again rose to higher winter levels. The winter nitrate levels of the Snake were somewhat higher than would be expected for unaltered watersheds in this region (cf. Lewis and Grant 1979, 1980b). They probably reflect the influence of dispersed nutrient sources, especially from residential areas, which have an especially noticeable influence on winter concentrations because of the very low discharges in winter and the low biological demand for nitrogen at this time. The seasonal pattern of nitrate concentrations observed in the Snake River is, however, essentially what would be observed in an unaltered watershed (Lewis and Grant 1980b). The same pattern is visible in the data for Blue River and Tenmile Creek, but with considerable scatter due to the influence of point sources. The late summer minimum and winter maximum are evident, however.

Tenmile Creek was unusual in that nitrate concentrations were higher at all times of the year than in other rivers. In 1981 nitrate nitrogen approached 1,000 ug/l in winter (Figure 45). The high nitrate concentrations are explained by the large amounts of nitrate exported from the site of Climax Molybdenum at the head of Tenmile Creek. This

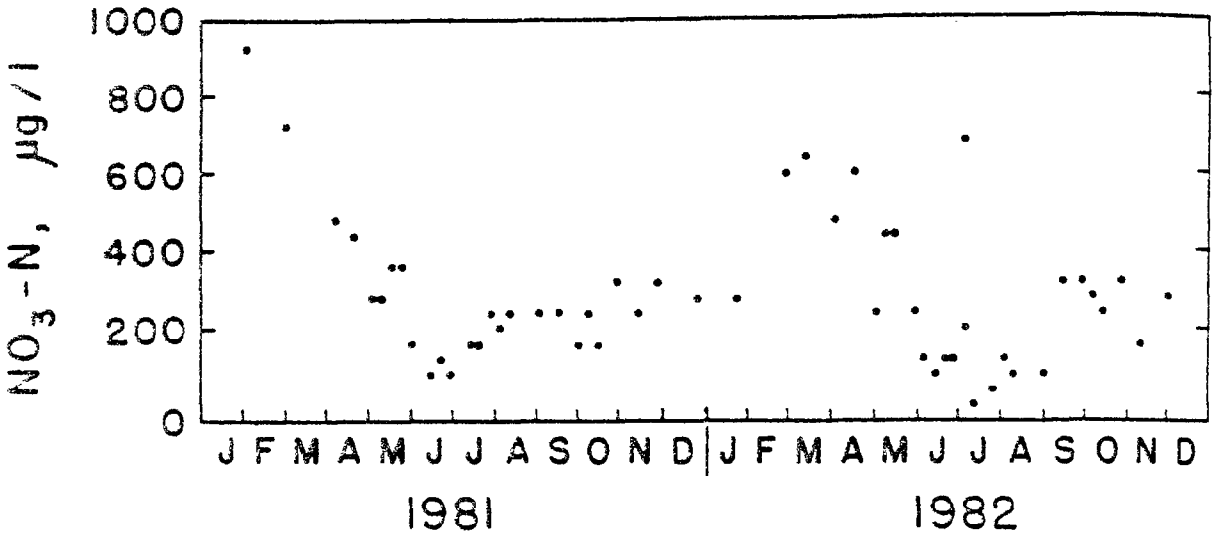


Figure 45. Nitrate values for Tenmile Creek at its point of entry to Lake Dillon, illustrating the very high nitrate levels at certain times of the year.

special nitrogen source will be considered further in the segment-by-segment analysis of Tenmile Creek.

For ammonium, there was also a seasonal pattern of concentrations in 1981 and 1982, but it was not so clear as the pattern for nitrate. In part this was due to the much lower concentrations of ammonium than of nitrate, especially in the Snake River. The Snake River data do show indications of an early spring maximum in concentration before discharge had begun to increase drastically, a late summer minimum corresponding to the postulated peak of biological demand for inorganic nitrogen, and a fall increase after the stream water had begun to cool. Two high values in early August of 1981 for the Snake River departed from the seasonal pattern and were not repeated in 1982. These high values were confirmed, however, by other samples upstream and were almost certainly caused by some temporary or sporadic man-made source.

Patterns in the ammonium data for the Blue River and for Tenmile Creek are scarcely discernible because of the large amount of variability introduced by point sources. Ammonium concentrations were affected simultaneously not only by seasonal factors but by variations in release from point sources and variations in the conversion rate from ammonium to nitrate.

In all three rivers, total soluble nitrogen was accounted for approximately half by organic nitrogen and half by ammonium plus nitrate. Winter values of total soluble N tended to be higher than summer values, in parallel with nitrate, a major contributor to total soluble N. Seasonal pattern was weak, however, especially for the Blue River and Tenmile Creek.

Total soluble phosphorus concentrations showed no strong seasonal pattern in the Snake River, nor did orthophosphate concentrations. The average concentrations were often high at times of high discharge, however. The implication is that dilution effects associated with major seasonal increases in discharge were effectively cancelled by increased transport at such times (cf. Lewis and Grant 1979). There were no clear seasonal patterns in concentration for the Blue River or Tenmile Creek, where the point sources introduced considerable additional random variation. For the Blue River, the increase of orthophosphate and total soluble phosphorus in September, October, and November of 1981 caused by treatment plant malfunctions has already been mentioned. A similar but less exaggerated phenomenon was observed in Tenmile Creek in the first half of 1981, but was not repeated in 1982 and was of obscure origin.

Total particulates, particulate phosphorus, particulate nitrogen, and particulate carbon are united by a tendency to show patterns similar to each other. In the Snake River there was a trend toward high concentrations around the time of maximum discharge. Superimposed on this was an occasional exceptionally high value associated with some kind of short-lived earth disturbance in the watershed, as shown for the Snake River in late August of 1981 and in the summer of 1982. Such short-term spikes obviously can be extremely important in total annual transport of particulate materials.

Chemistry of Miner's Creek and Soda Creek

Because of the relatively small proportion of total land area drained by Miner's Creek and Soda Creek, the potential contribution of these two watersheds to total loading of the lake with nutrients is small. Their chemistry is considered here mainly for the sake of completeness, since these two watersheds are separate from the three major river drainages. Table 41 gives the discharge-weighted averages for the chemistry of the two streams and the standard error for each variable. As in the case of the three rivers, the time-weighted means were calculated but are not shown in the table. For Miner's Creek and Soda Creek, the time-weighted means are even closer to the discharge-weighted means than for the three major rivers.

The small size of contributions of these two creeks to total loading of the lake is evident from the discharge figures in Table 41. The chemical data are unexceptional. Soluble nitrogen concentrations were in the range of those observed in the Snake River, and considerably below those of the Blue River and Tenmile Creek as reported in Table 40. This is not unexpected, since Miner's Creek and Soda Creek at the sampling points were not influenced by point sources of nutrients, but were influenced by some relatively dispersed human influences analogous to those characteristic of the Snake River drainage.

For Miner's Creek, the phosphorus concentrations, including both soluble and particulate fractions, were relatively low, and were comparable to the values observed in the Snake River. For Soda Creek, however, both particulate and total soluble phosphorus concentrations

	1981				1982			
	Miner's Creek		Soda Creek		Miner's Creek		Soda Creek	
	Mean	S.E.	Mean	S.E.	Mean	S.E.	Mean	S.E.
NO_2-N, ug/l	0.8	.11	1.2	.32	1.4	.20	2.3	.23
NO_3-N, ug/l	32.2	4.3	12.7	4.5	20.2	4.1	63.	14.
NH_4-N, ug/l	14.9	3.0	12.2	2.8	10.0	1.9	11.5	1.4
Total Soluble N, ug/l	141.	34.	90.0	21.	102.	26.	202.	41.
Particulate N, ug/l	34.	13.	56.	17.	83.	16.	100.	29.
PO_4-P, ug/l	1.9	0.23	8.7	1.2	2.9	0.47	13.4	2.7
Total Soluble P, ug/l	3.2	0.34	18.3	3.3	5.1	0.60	20.5	3.0
Particulate P, ug/l	6.4	1.2	11.0	1.5	5.8	0.76	21.6	4.0
Total Particulates, mg/l	6.9	1.9	6.1	1.2	7.7	1.7	16.3	3.1
Particulate Carbon, mg/l	320.	70.0	351.	63.2	531.	116	501.	103.
Alkalinity, mg/l	28.9	1.7	59.0	3.2	26.0	1.2	62.3	2.4
pH	7.4	0.08	8.0	0.15	7.5	0.05	8.0	0.08
Conductance, umho/cm	74.	3.4	168.	9.8	58.	2.6	142	8.5
Discharge, l/sec	88.	10.	12.	3.1	246.	47.	54.	15.
Discharge, cfs	3.1	0.37	0.42	0.11	8.7	1.7	1.9	0.52

Table 41. Discharge-weighted means for the two small streams entering the lake (sampling stations above WWTP effluents).

were higher than for any of the three major rivers, and the discharge was especially low in relation to drainage area. Conductance was also notably higher for Soda Creek than for Miner's Creek. Water is evidently used within the drainage, and this results in reduced discharge. Water use and possibly springs may account for high conductance.

Frisco and Snake River Treatment Plant Effluents

Table 42 summarizes the chemistry of the effluents from the Frisco and Snake River wastewater treatment plants. Total soluble nitrogen concentrations were 50 to 100 times higher than the stream concentrations and were relatively similar between plants and between years. The phosphorus values present quite a different picture. First, the effluents of the two plants were rather different in the partitioning of total phosphorus. In the Snake River Wastewater Treatment Plant effluent, the largest amount of phosphorus was connected with the particulate fraction, while in the Frisco effluent the largest fraction was connected with the soluble components. For the Snake River Plant, the phosphorus concentrations were about 10 times above those of the receiving stream. A few exceptionally high concentrations were checked against plant records. High values typically coincided with construction or equipment problems. The average total P concentrations for the Frisco Plant were considerably higher than for the Snake River Plant, and there was a major difference between 1981 and 1982 for the Frisco Plant. In 1982 the discharge-weighted averages were exceptionally high at the Frisco Plant. Examination of the raw data shows that the high averages were

	1981				1982			
	Frisco WWTP Effluent		Snake WWTP Effluent		Frisco WWTP Effluent		Snake WWTP Effluent	
	Mean	S.E.	Mean	S.E.	Mean	S.E.	Mean	S.E.
NO_2-N, ug/l	135.	15.8	116	17.	113.	25.	268.	50.
NO_3-N, ug/l	1579.	407.	3527.	786.	1228.	281.	2251.	910.
NH_4-N, ug/l	2787.	670.	1825.	620.	2868.	543.	1957.	453.
Total Soluble N, ug/l	8911.	1168.	6798.	774.	7096.	867.	8398.	1627.
Particulate N, ug/l	167.	23.	496.	160.	-	-	-	-
PO_4-P, ug/l	53.6	18.	4.5	2.4	496.	169.	24.4	12.
Total Soluble P, ug/l	76.7	19.	22.	3.4	556.	177.	38.6	13.
Particulate P, ug/l	17.6	2.2	45.9	11.2	57.0	16.	120.	32.
Total Particulates, mg/l	1.3	0.28	6.0	2.4	5.1	1.2	9.3	2.2
Particulate Carbon, mg/l	741.	163.	4710.	2384.	-	-	-	-
Alkalinity, mg/l	42.2	2.9	50.5	7.9	65.8	7.7	68.5	6.9
pH	6.8	0.08	6.5	0.16	7.2	0.10	7.0	0.09
Conductance, umho/cm	417	29.	450.	18.	449.	21.	429.	20.
Discharge, l/s	12.8	0.89	13.4	0.92	14.5	0.50	14.6	1.2
Discharge, cfs	0.5	0.02	0.5	0.04	0.5	0.02	0.5	0.04

Table 42. Discharge-weighted means for the two effluents discharging near the mouths of Miner's and Soda Creeks.

caused by discontinuous but repeated occurrence of very high concentrations (1000 to 3500 ugP/l). Checks of the dates of high values with plant operators showed that, for the most part, the dates of high P values coincided with known equipment problems.

Precipitation Chemistry at the Main Station

The precipitation chemistry data consist of the continuous record from the collector situated near the Snake River Wastewater Treatment Plant and shorter records from two other locations. We will consider the more extensive record first and then compare it with the other two records.

All of the precipitation chemistry data to be dealt with here refer to bulk precipitation, which is the total of all wet and dry deposition on a collecting surface. A collecting surface accumulates dry deposition when there is no rain or snow. The occurrence of wet precipitation augments the delivery of materials onto the collector surface, especially for certain substances that are scrubbed out of the atmosphere effectively by water droplets (e.g., nitrate). There are typically changes in the concentration of dissolved materials in wet precipitation during the course of a rainstorm, principally as a result of the cleansing of the air by rain. Because of the continuous deposition of materials on a collector surface during dry weather and the continuous change of concentrations during a wet precipitation event, an average concentration is not very meaningful for precipitation chemistry. For this reason, we express all the results

for precipitation chemistry in terms of loading rates, i.e., the amount of material delivered to a unit surface over a specified time.

Table 43 summarizes the means and standard errors for the loading of chemical constituents analyzed in the samples from the main precipitation collector at the Snake River Wastewater Treatment Plant. The means are derived from 61 separate collections spread over the two-year study period. Since the collection intervals were not all of identical length, the means are time-weighted averages.

The hydrogen-ion loading was lower (4x) than loading reported for the Como Creek watershed at similar elevation near Ward, Colorado (Grant and Lewis 1982), but precipitation was frequently below pH 5.65, indicating the presence of strong mineral acids. Because of the dominant influence of stream chemistry on the chemistry of a lake such as Dillon, however, direct hydrogen ion loading from precipitation is of minimal significance to the biology of the lake. As expected, nitrite was a minor contributor among the inorganic soluble nitrogen species, but nitrate nitrogen was present in substantial amounts. As at Como Creek, which has been studied thoroughly in this regard, the precipitation at Dillon appears to be substantially augmented beyond background in nitric acid as a result of the presence of substantial amounts of NO_X in the atmosphere from fossil fuel combustion. The ammonium loading was relatively high but secondary to the nitrate loading as a nitrogen source.

The loading rates for both PO_4-P and total soluble phosphorus were considerably higher than at Como Creek (3-5x). Since the mechanisms of phosphorus transport through the air are poorly known, no

	1981 - mg/m^2/wk		1982 - mg/m^2/wk	
	Mean	S.E.	Mean	S.E.
NO_2-N	0.079	0.008	0.125	0.011
NO_3-N	2.616	0.284	3.724	0.351
NH_4-N	3.479	0.353	1.627	0.098
Total Soluble N	18.380	1.776	6.002	0.364
PO_4-P	0.732	0.087	0.630	0.123
Total Soluble P	1.012	0.143	0.695	0.134
Total Particulates	187.	8.01	152.	7.54
Particulate P	0.219	0.015	0.178	0.009
HCO_3^-	19.8	8.60	23.4	1.56
H^+	0.055	0.006	0.040	0.006
$SO_4^=$	12.8	0.86	46.7	5.7

Table 43. Loading rates for bulk precipitation at the main collecting station, 1981 and 1982.

explanation can be offered. However, the observed phosphorus transport values are within the range of values reported for a variety of sites in the literature (Likens et al. 1977). Particulate phosphorus transport was significantly lower than at Como Creek, and this partially offset the higher loading for soluble fractions. Sulfate was a major ion, as would be expected by analogy with the Como Creek studies.

Figure 46 shows time plots of loading rates for some of the nutrient fractions that have a significant amount of seasonal pattern. The transport of both nitrate and ammonium is facilitated by moisture, as is evident from a comparison of the amount of rain or snow and the loading rates for these ions. Total soluble phosphorus showed a pronounced maximum in June of both years. This early summer maximum, although quite high, appeared both years and at more than one station and was thus most definitely not an error.

Precipitation Chemistry at Other Stations

Phosphorus loading rates by bulk precipitation were determined at two additional stations in 1982 as a comparison with the main station located near the Snake River Wastewater Treatment Plant. The first of the comparison stations was located near Frisco. A different type of collector (tube collector, diameter 25 cm) was used at this station; the dates of comparison were June 1 to November 17, 1982. The means and standard errors for the main station and the Frisco station over this interval are summarized in Table 44. There was no significant difference in the average loading rates for the two stations ($p >$

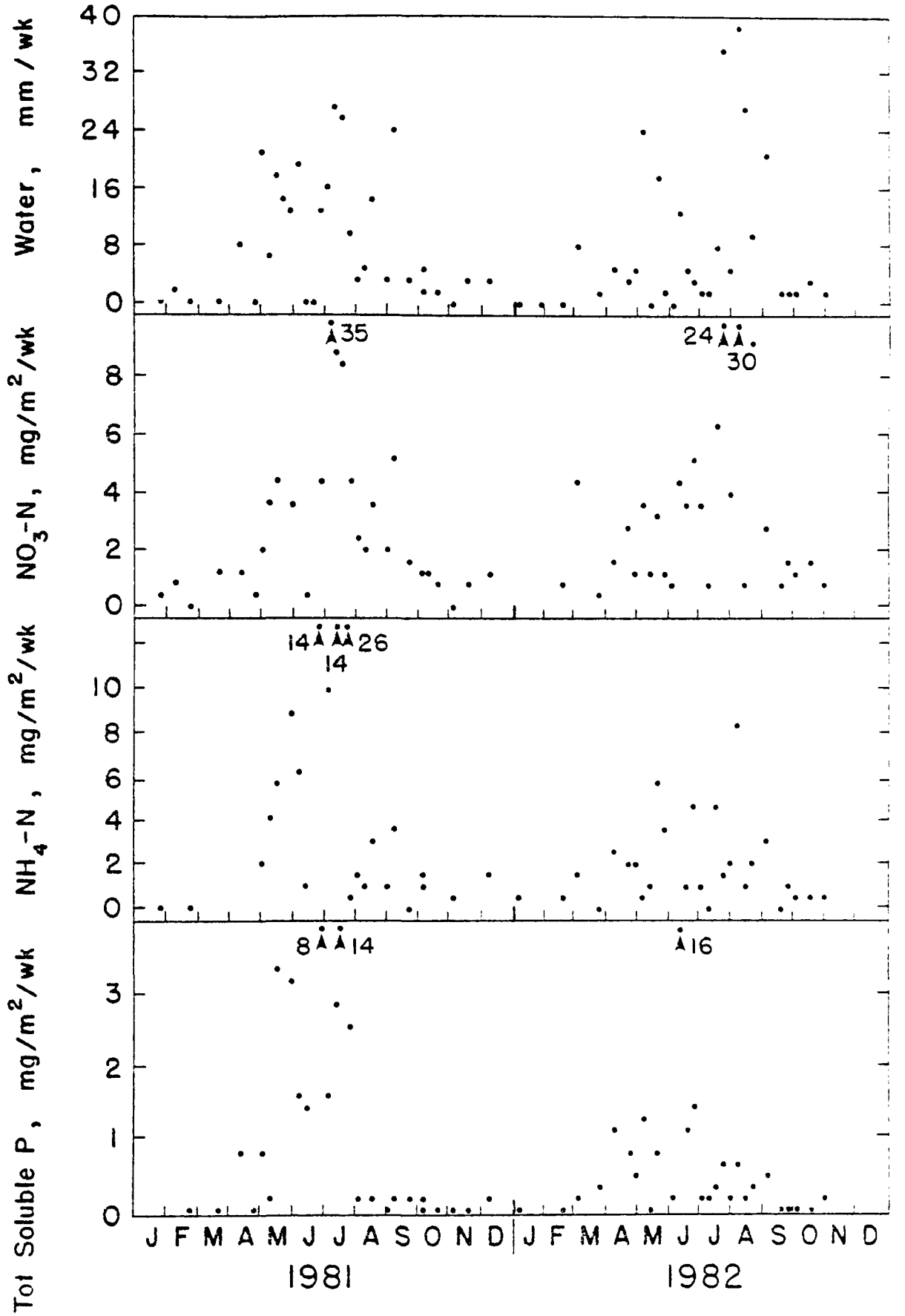


Figure 46. Selected variables for bulk precipitation near the Snake River WWTP.

0.05). The period of collection incorporated the high phosphorus loading pulse in June, which results in relatively high standard errors for both stations. However, the pulse appeared simultaneously at the two stations, verifying our conclusion that it was not an artifact.

	Total P Main Station			Total P Comparison Station	
	$mg/m^2/wk$		Comparison Station	$mg/m^2/wk$	
Dates of Comparison	Mean	S.E.		Mean	S.E.
1 June - 17 November	1.354	.969	Frisco	1.561	.688
1 July - 1 November	.427	.065	Raft	.533	0.099

Table 44. Summary of comparison for phosphorus loading by bulk precipitation at the main station and two other stations.

The second comparison station was situated on the Denver Water Department's raft, which was anchored on the water surface. This comparison station is of special interest because we wanted to rule out the possibility that a significant amount of phosphorus transport observed at the Snake River station could be accounted for by localized dust in that area. The dates of comparison were July 1 through November 1. From July 1 through mid-September the raft was anchored in midlake, and from mid-September through October it was anchored at the shoreline. The means for the main station and the station on the raft over these dates are summarized in Table 44 along with their standard errors. The standard errors are much lower in this case because the interval of sampling did not include the period of high phosphorus

loading in June. There was no significant difference in the loading rates between the two stations.

From the data at the two comparison stations, we have every reason to believe that the main precipitation collection station at the Snake River Wastewater Treatment Plant gave an accurate impression of the bulk precipitation loading to the surface of Lake Dillon.

Total Nutrient Loading of the Lake

Water Budget

Reliable estimations of total nutrient loading of the lake over the period of study presume the availability of sound information on the amounts of water reaching the lake by various pathways. If the flow measurements summarized in the previous chapter are accurate, their summation plus the unmeasured flows should be close to the calculated input determined by the Denver Water Department, which is obtained by an indirect method based on measured outflow and change in lake volume. Table 45 summarizes the flows from various sources in 1981 and 1982. The sources of the estimates for the rivers, streams and effluents are as given in the previous chapter. Two sources of water were estimated without direct measurement: miscellaneous surface runoff and groundwater.

Miscellaneous surface runoff was the contribution from a number of small areas near the lake that did not drain through watersheds where there was gauging or discharge measurement. These areas total 4800 ha, or 5.8% of the total land area of the Lake Dillon watershed. Their unmeasured contribution to runoff was estimated in two parts. The first of these is the Meadow Creek drainage, which includes some high-elevation areas. This drainage was considered to be hydrologically similar to Miner's Creek. The runoff was thus set at an amount equal to that of Miner's Creek, with appropriate adjustment for area. The second portion consists of smaller drainages that are low-lying and without steep contours. Because of low accumulation of

	1981 - Thousands		1982 - Thousands	
	m^3/yr	Acre-ft/yr	m^3/yr	Acre-ft/yr
Rivers				
Snake	40,100	32.54	81,800	66.35
Blue	49,900	40.45	98,200	79.61
Tenmile	58,000	47.02	104,000	84.31
Subtotal	148,000	120.0	284,000	230.3
Streams not on Rivers				
Miner's	2770	2.25	7760	6.29
Soda	380	0.31	1700	1.38
Subtotal	3150	2.55	9460	7.67
Effluents				
Frisco	403	0.33	457	0.37
Snake	423	0.33	460	0.37
Subtotal	826	0.66	917	0.74
Diffuse Surface Drainage	3366	2.73	10,385	8.42
Precipitation	4336	3.52	4057	3.29
Groundwater	8441	6.90	8441	6.90
Grand Total	168,119	136.4	317,269	257.4

Table 45. Summary of water flow into the lake in 1981 and 1982.

snowpack in such areas, the water yield was expected to be small. This second portion was assumed to have the water yield of Soda Creek drainage, with appropriate adjustment for area. Any errors in the approximation procedure will obviously not have a very great effect on the water budget of the lake because of the relatively small proportion of the watershed in the miscellaneous surface drainage category.

Groundwater also potentially supplies water, and thus nutrients, to the lake. A complete water budget was constructed on a monthly basis for 1976-1979 from the outflow figures of the Denver Water Department, the inflow data of U.S.G.S gauges, measures of precipitation, estimates of evaporation, and an allowance for miscellaneous surface drainage. This water budget should sum to zero if all estimates were made with good accuracy and if there was no groundwater entry into the lake. Some variance around the zero sum is expected due to inaccuracies of the various estimates, but a trend toward greater outflow than inflow would suggest the presence of an unmeasured term, which would have to be groundwater. The residuals over the three years had an average of 575 acre-feet per month. There was no strong seasonal or annual pattern in the size of the residual, and the standard error was about 300 acre-feet per month. This residual may have been affected by the amount of drawdown as well as other factors. Since the irregularities in the residual cannot be pinned to a specific pattern, we treat the groundwater inflow to the lake as constant at 575 acre-feet per month for both years of the study. Since this amount of flow is very small by comparison with the total flow to the lake, any errors in the estimate within the range of

one or two standard errors would not be significant to the overall conclusions of the study.

When all sources of water for the lake were summed (Table 45), the estimated inflow was 136,000 acre-feet for 1981 and 257,000 acre-feet in 1982. This figure can be compared with the independently derived estimate of the Denver Water Department for total input over the same interval. First, however, evaporation must be added to the Denver Water Department estimate, which does not include evaporation. From the averages of several years of study by the Denver Water Department, we estimate evaporation as 5900 acre-feet per year, which gives a corrected Denver Water Department input of 111,000 acre-feet per year for 1981 and 223,000 acre-feet per year for 1982. The deviation of this from the totals based on U.S.G.S. data as described above is 10-20%, which is more than anticipated. Reexamination of the data did not reveal any potential errors of this magnitude. For nutrient loading computations, we assume that the U.S.G.S. data for input are correct, although this assumption is to some degree arbitrary.

Phosphorus Loading

Table 46 summarizes the phosphorus loading of Lake Dillon according to the various pathways of nutrient flow that were recognized in the previous chapter and in the section on water balance. For rivers, streams, and effluents, the numbers in Table 46 were obtained by multiplying the discharge-weighted average concentrations as reported previously (Tables 40, 41) times the time-weighted average discharge and dividing the product by lake area. The estimate of

	Phosphorus - kg/ha/yr - 1981					Phosphorus - kg/ha/yr - 1982				
	PO_4-P	Total Soluble P	Part. P	Total P	%	PO_4-P	Total Soluble P	Part. P	Total P	%
Rivers										
Snake	.036	.060	.127	.187	8.7	.166	.386	.423	.808	22.0
Blue	.524	.524	.206	.729	33.8	.316	.492	.411	.903	24.5
Tennile	.048	.156	.308	.464	21.5	.164	.428	.459	.887	24.1
Subtotal	.608	.740	.641	1.380	64.0	.646	1.306	1.293	2.598	70.6
Streams not on Rivers										
Miner's	.004	.007	.013	.020	.9	.017	.030	.034	.064	1.7
Soda	.002	.005	.003	.008	.4	.017	.026	.028	.054	1.5
Subtotal	.006	.012	.016	.028	1.3	.034	.056	.061	.117	3.2
Effluents										
Frisco	.016	.023	.005	.029	1.3	.170	.190	.020	.210	5.7
Snake	.001	.007	.015	.022	1.0	.008	.013	.041	.054	1.5
Subtotal	.018	.030	.020	.051	2.4	.178	.204	.061	.265	7.2
Miscellaneous Surface Drainage	.011	.019	.035	.054	2.5	.045	.080	.091	.171	4.7
Precipitation	.381	.520	.114	.634	29.4	.328	.361	.093	.454	12.4
Groundwater	.006	.010	.000	.010	.5	.028	.066	.000	.066	1.8
GRAND TOTAL	1.030	1.331	.826	2.157	100.1%	1.259	2.073	1.599	3.671	99.9%

Table 46. Summary of phosphorus loading of the lake in 1981 and 1982.

loading for precipitation was taken directly from Table 43 of the section on precipitation chemistry. For miscellaneous surface drainage, the water contribution has already been obtained by approximation as reported in Table 45. The concentration of phosphorus fractions in this water must also be approximated before an estimate of loading can be made. Phosphorus concentrations in the miscellaneous surface drainage category were assumed equal to those observed in Miner's Creek because of parallel land usage, and these concentrations were applied to the discharge approximations made in connection with Table 45 to produce the estimate of loading shown in Table 46. A similar approach was taken for groundwater. The groundwater phosphorus concentrations were assumed to be equal to those of the Snake River drainage, a large watershed with mixed land use but lacking point sources. It was also assumed, however, that particulate phosphorus would not be transported through groundwater and this fraction was therefore set to zero. The concentrations were then multiplied by the amount of groundwater as reported in Table 45 to obtain the loading of phosphorus via groundwater. Both the miscellaneous surface drainage and groundwater categories were minor contributors, so the details of these approximations are of no great concern.

More than half of the total phosphorus from all sources entered the lake in soluble form. Table 46 shows that the bulk of total soluble phosphorus was in the orthophosphate fraction rather than the organic phosphorus fraction. The particulate fraction was significant both years and was slightly higher proportionally in 1982 than in 1981, probably because of higher discharge in 1982. The total phosphorus

transport in 1981 was about 60% of the total phosphorus transport in 1982.

The rivers contributed about two-thirds of the total phosphorus reaching the lake. The individual rivers made very different relative contributions in the two years, however. All were affected by higher discharge in 1982, but not identically. Loading from the Snake River was drastically affected by construction in the river bottom in the second year. Loading from Tenmile Creek was approximately proportional to the difference in flows between the two years. The Blue River loading was exaggerated in 1981 by problems at the Breckenridge WWTP. Other differences in sensitivity of the three rivers to changes in the hydrologic conditions between years can be traced to their differing land use patterns; these will be considered further when land use is analyzed.

Precipitation accounted for a large proportion of the residual loading beyond the contributions of the three major rivers. The proportion of total phosphorus loading attributable to precipitation in 1981 was considerably larger than in 1982. This is principally explained by the much greater contributions from surface runoff in 1982, which reduced the overall influence of precipitation loading.

The combined contribution of the Frisco and Snake River wastewater treatment plants was only 2.4% in 1981 and 7.2% in 1982. The contribution in 1982 should actually have been lower in view of the much higher phosphorus loading from non-point sources that year, but treatment plant malfunctions produced an exceptionally high effluent contribution on certain dates in 1982. There are two other major

wastewater treatment plants in the watershed: Breckenridge and Copper Mountain. The contributions of all four plants will be specified more exactly when the nutrient loadings are broken down more completely according to sources and land uses. It is obvious from Table 46 that the bulk of phosphorus entering Lake Dillon does not originate from point sources, however.

Nitrogen Loading

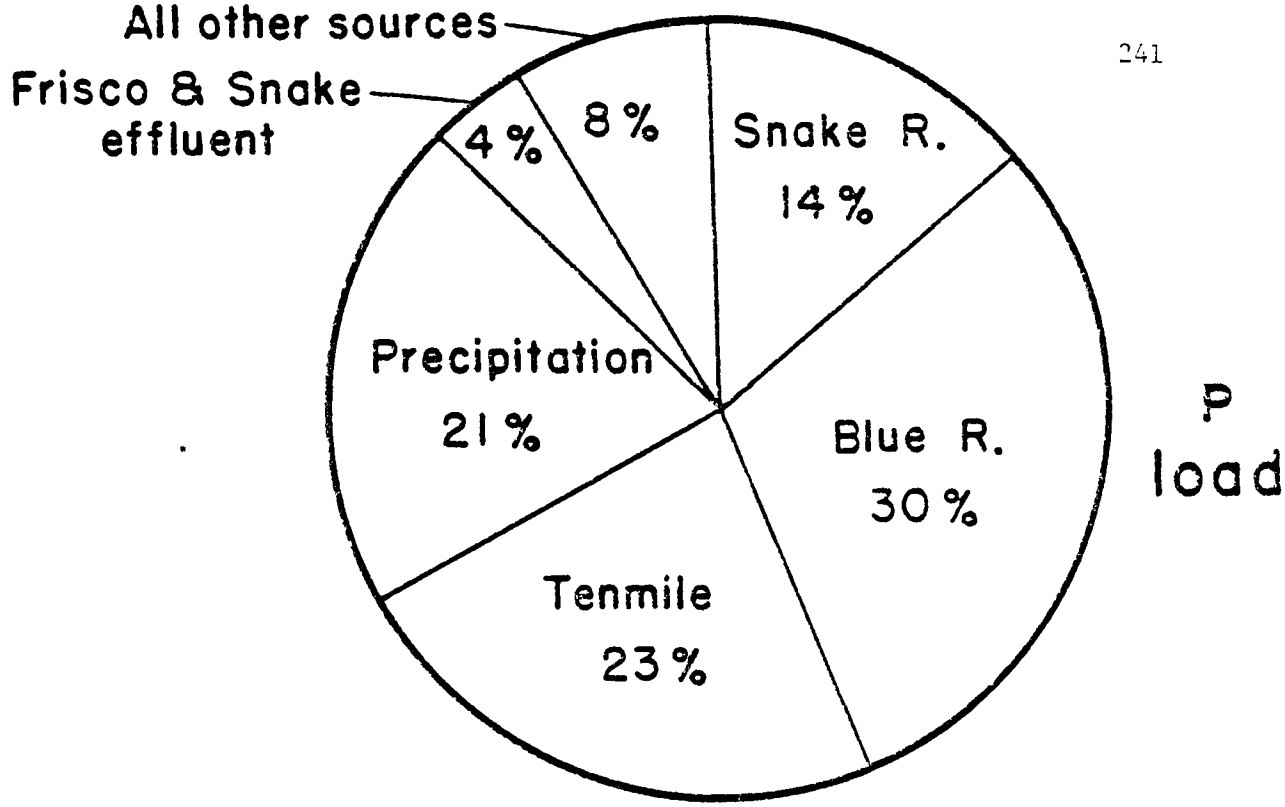
Table 47 summarizes the Dillon nitrogen loading. The methods of computation were similar to those used for phosphorus. Particulate nitrogen was not measured in the effluents in 1982 because it was such a small contributor to effluent N. The particulate N column in the table was filled in for 1982 by use of the 1981 data.

The inorganic fraction of nitrogen was slightly larger than the organic fraction for all sources combined. Unlike phosphorus, nitrogen loading was mainly associated with the soluble fractions; particulate nitrogen made a relatively small contribution. The difference between years in total nitrogen loading was very similar on a proportional basis to that observed for phosphorus: the loading for 1982 was almost double that of 1981. For nitrogen, the dominating effect of the three rivers on total loading was even more pronounced than it was for phosphorus. The influence of precipitation on nitrogen loading was relatively small by comparison with phosphorus. The contributions of the two effluents on a percentage basis were slightly larger than for phosphorus, but still relatively minor by comparison with total loading (Figure 47).

	Nitrogen - kg/ha/yr - 1981							Nitrogen - kg/ha/yr - 1982						
	NO_2-N	NO_3-N	NH_4-N	Total Soluble N	Part. N	Total N	%	NO_2-N	NO_3-N	NH_4-N	Total Soluble N	Part. N	Total N	%
Rivers														
Snake	.02	2.86	.33	5.49	.99	6.49	10.1	.09	7.16	.73	14.33	6.55	20.88	18.4
Blue	.37	8.49	6.99	20.61	1.05	21.65	33.8	.24	7.98	7.86	28.06	5.36	33.42	29.5
Tenmile	.15	9.20	1.13	15.49	2.86	18.36	28.6	.86	13.81	7.32	37.05	6.93	43.98	38.8
Subtotal	.55	20.55	8.45	41.59	4.90	46.50	72.5	1.17	28.95	15.91	79.44	18.84	98.28	86.7
Streams														
Miner's	.002	.07	.03	.29	.07	.36	0.6	.008	.12	.06	.59	.48	1.07	.9
Soda	.00	.004	.003	.03	.02	.04	0.1	.003	.08	.02	.26	.13	.38	0.3
Subtotal	.002	.07	.03	.32	.09	.41	0.7	.01	.20	.07	.85	.61	1.46	1.2
Effluents														
Frisco	.04	.48	.84	2.69	.05	2.74	4.3	.04	.42	.98	2.43	.05*	2.48	2.2
Snake	.04	1.12	.56	2.15	.16	2.31	3.6	.09	.78	.67	2.89	.16*	3.05	2.7
Subtotal	.08	1.59	1.40	4.84	.21	5.05	7.9	.13	1.20	1.66	5.32	.21*	5.53	4.9
Miscellaneous Surface Drainage*	.005	.18	.08	.79	.19	.99	1.5	.02	.32	.16	1.59	1.30	2.89	2.5
Precipitation	.004	1.36	1.81	9.56	1.17*	10.72	16.7	.07	1.140	.85	3.12	.95*	4.07	3.6
Groundwater*	.002	.23	.03	.44	.000	.44	.7	.01	.58	.06	1.16	.000	1.16	1.0
GRAND TOTAL	.64	23.98	11.79	57.54	6.56	64.11	100.0%	1.41	32.39	18.71	91.48	21.91	113.39	99.9%

estimated

Table 47. Summary of nitrogen loading of the lake in 1981 and 1982.



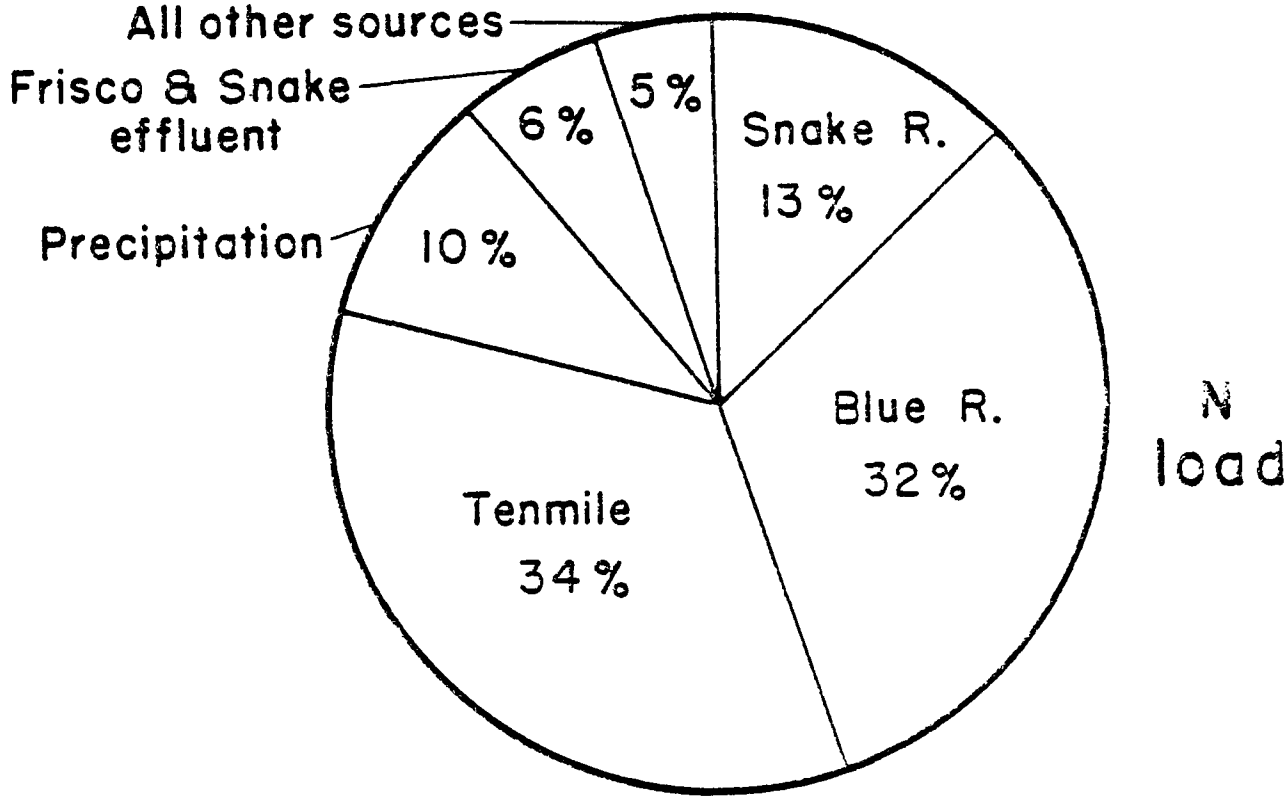


Figure 47. Itemization of sources of P and N loading for Lake Dillon at the points of entry to the lake (average 1981, 1982).

Overview of Phosphorus and Nitrogen Loading

Table 48 expresses the phosphorus and nitrogen loading in several different ways. Because of phosphorus removal at the point sources and low background phosphorus in the watersheds, the ratio of nitrogen to phosphorus in the loading was very high. The molar ratios of nitrogen to phosphorus were 69 in 1981 and 66 in 1982.

	Per unit lake area*		Entire lake	
	kg/ha/yr	$g/m^2/yr$	kg/yr	lb/yr
1981				
N	85.3	8.5	85,600	189,000
P	2.9	0.29	2,900	6,300
1982				
N	134.0	13.4	151,400	333,800
P	4.2	0.42	4,800	10,700

*Based on mean lake areas of 1004 ha in 1981 and 1130 ha in 1982.

Table 48. Summary of N and P loading.

Figure 48 puts the phosphorus loadings into perspective by use of the widely-used Vollenweider diagram (Vollenweider 1968), which predicts the trophic status of a lake on the basis of its mean depth and its phosphorus loading. The position of Lake Dillon on the diagram is shown with relation to the position of a small selection of other lakes. The diagram is presently considered to provide only a crude approximation of trophic status because it does not allow the phosphorus loading to be discounted for varying degrees of phosphorus sedimentation and loss through outflow. However, the diagram is in general agreement with the lake trophic indicators. A more detailed treatment of this matter will be given in connection with modelling.

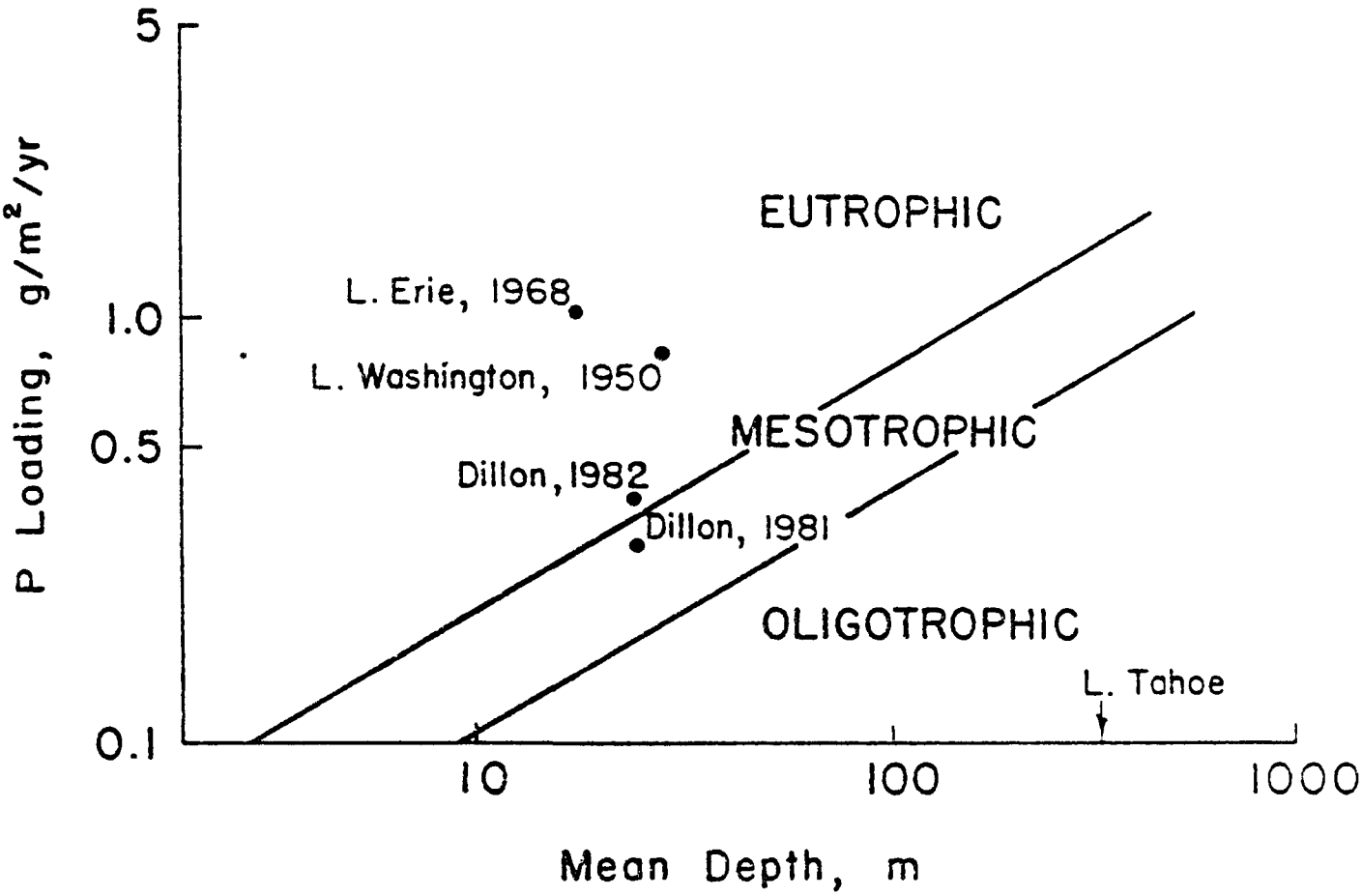


Figure 48. Position of Lake Dillon on the original Vollenweider diagram (Vollenweider 1968).

Nutrient Export in Relation to Land Use

Nutrient export must be considered in relation to land use. The central information base for this purpose is the representative watershed program, in which nutrient loss was measured for individual small watersheds dominated by particular land uses. The names and locations of the representative watersheds are shown in Figure 49. The watersheds represented eight different land uses: undisturbed (background), roads, interstate highways, residential development on sewer (nonpoint component), urban development on sewer (nonpoint component), residential development on septic, ski areas, and Climax Molybdenum operations. The nutrient yields from these representative watersheds are considered in sequence below. For each of the watershed types except the undisturbed type, the background nutrient yield is subtracted from the total yield and the residual is related quantitatively to the intensity of land use by some index such as the number of persons per unit area or the percentage of affected area. After the nonpoint sources have been treated in this way, consideration is given to the contributions per capita from small and large point sources.

Water yield enters into many of the nutrient yield relationships developed below for the representative watersheds. The water yields, given as mm/year of runoff, are in all instances based on discharge measurements. Water yields vary a great deal; the causes of this variation are numerous. There is a pronounced increase in water yield with elevation. Long-term records of snow accumulation also show that, for a given elevation, the western portion of the catchment tends to

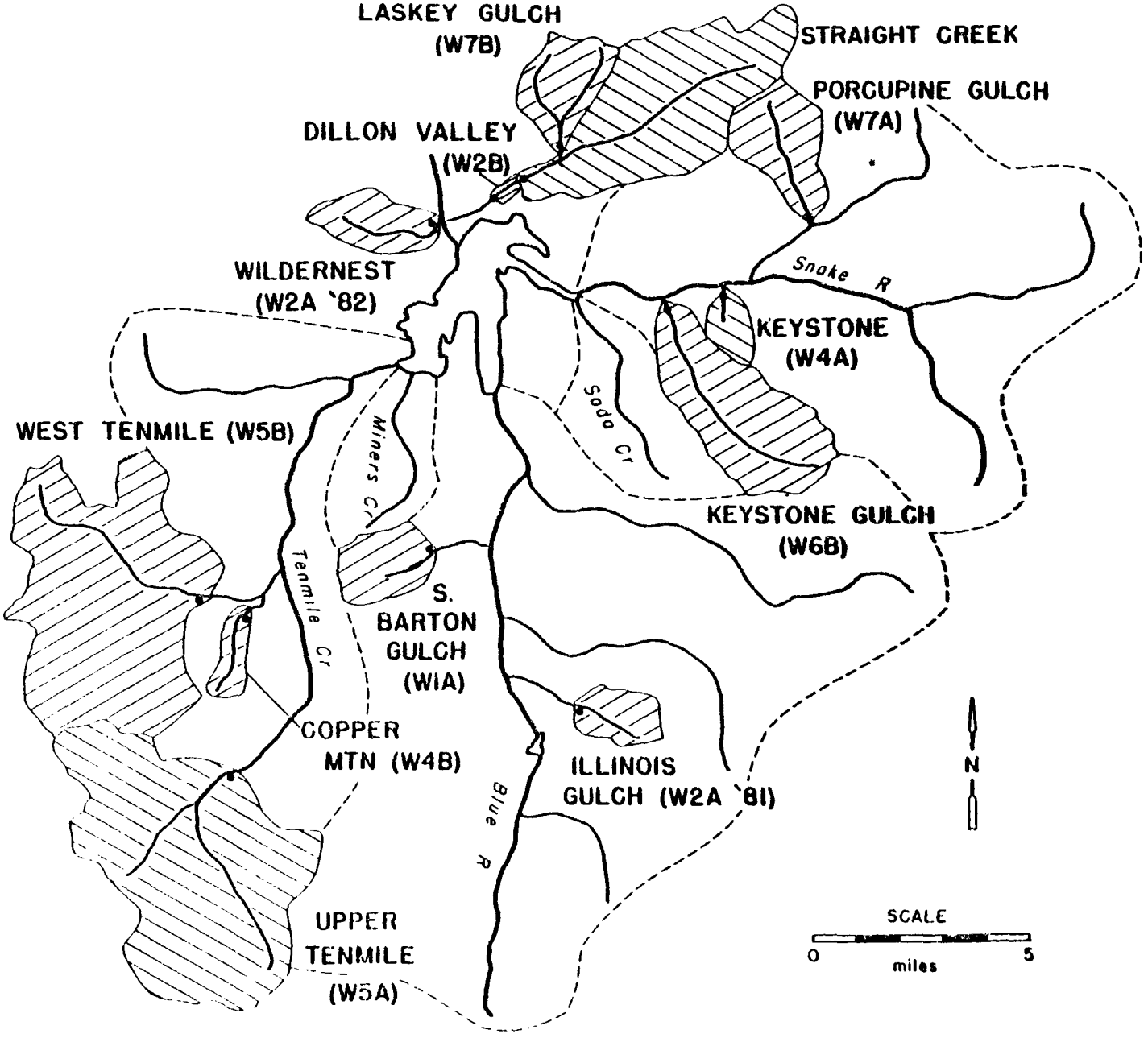


Figure 49. Representative watersheds.

receive more winter moisture than the rest. Slope, vegetation, and land use also have complex effects. A certain amount of water is also pumped or diverted in various places (e.g., local irrigation). For 1981, the catchment-wide average runoff at lakeside was 180 mm and in 1982 it was 340 mm.

Undisturbed Watersheds

Porcupine Gulch and Laskey Gulch provided information about nutrient yield from undisturbed areas. Table 49 summarizes the amount of runoff from these watersheds and the phosphorus and nitrogen yield in each of the two years. In these and a number of other representative watersheds the particulate nitrogen contribution was computed from the percent carbon and total particulates of the stream by use of a common C:N ratio equal to that of the Snake River, since the amount of particulate N was often below detection. The standard error for phosphorus is reported on the basis of weighted averaging. For nitrogen, the estimation of particulate nitrogen from carbon precluded estimation of the standard error, but the ratio of S.E. to mean would be roughly the same as for phosphorus.

The phosphorus yields of Porcupine Gulch and Laskey Gulch were close to those reported in the literature for cold temperate areas of similar soil characteristics. For example, Wright (1974) reported 1.5 $mg/m^2/yr$ P export for the Boundary Waters Canoe area of Minnesota, and Schindler et al. (1976) reported 5.0 $mg/m^2/yr$ for the Experimental Lakes Area, Ontario. The data also show great similarity

	Water Yield mm/yr	P yield - $mg/m^2/yr$ Total	S.E.	% Part.	N yield - $mg/m^2/yr$ Total	% Part.
Porcupine Gulch (W7A)						
1981	315	1.55	0.34	17	75	7
1982	541	3.52	0.43	32	126	35
Laskey Gulch (W7B)						
1981	129	0.50	0.10	33	29	12
1982	306	2.77	0.55	33	62	58
Keystone Gulch (W6B)						
1981	108	0.57	0.08	33	14	19
1982	328	3.19	0.44	38	46	41
Upper Snake (SR3)						
1981	240	1.14	0.15	61	64	15
1982	500	4.37	0.55	54	103	31

Table 49. Yield of water, total P, and total N from two watersheds representing background (undeveloped) conditions (W7A, W7B), and from two watersheds with roads but otherwise essentially undeveloped (W6B, SR3).

to data for the Como Creek watershed at a similar elevation in Boulder County (Lewis and Grant 1979).

Both the total phosphorus and nitrogen export showed considerable variation between years and between watersheds. Table 49 suggests, however, that this variation was not random; it was associated with the amount of water yield. As shown by the study of Lewis and Grant (1979) for the Como Creek watershed, the increase in soluble phosphorus yield between a dry year and a wet one is expected to be slightly higher than the increase in water yield. A quantitative relationship was sought between the water yield and the nutrient yields reported in Table 49. Since the Como Creek studies suggest that the relationships are likely to be slightly curvilinear for undisturbed watersheds, the following equation was used:

$$Y_n = aY_w^b$$

where Y_w equals the annual water yield, Y_n equals the annual nutrient yield (P or N), and a and b are the critical parameters of the relationship between nutrient yield and water yield. The equation was log transformed and then tested by linear regression for fit to the data. The fit was excellent ($P < 0.01$). A similar procedure for nitrogen also showed a good fit ($P < 0.01$).

Casual examination of data from watersheds with only gravel roads or dirt roads suggested that the effect of these roads might be sufficiently small that they would not raise the nutrient export much above background. Since this would be an advantage in that it would offer a larger number of watersheds from which the terms relating water and nutrient yield could be determined for background conditions, two

watersheds differing from background only by the presence of dirt or gravel roads were examined. These included Keystone Gulch, which was a representative watershed (W6B), and a segment of the upper Snake (SR3). The data for these watersheds are shown in Table 49. Keystone Gulch actually has near the watershed crest a small septic field that serves the upper lodge at Keystone ski area, but it is of sufficiently small size that it cannot account for more than 2% of N or P export from Keystone Gulch, so the septic effect was ignored.

The data analysis showed that the two background watersheds can be lumped with the two watersheds containing roads. When the equations for the two undisturbed watersheds only were used to project the yield from the two watersheds with roads, and the projections were subtracted from the observed yields, two of the nitrogen residuals and one of the phosphorus residuals were slightly negative, suggesting that the yields of watersheds with roads were very near background. The exponent (b) derived for phosphorus yield on the basis of the two undisturbed watersheds was 1.39, and with all four watersheds it was 1.37. The coefficient (a) was higher by 20% for all four watersheds than for the two undisturbed watersheds, but this is a small amount in view of the certainty of variation in background caused by differences in vegetation, exposure, elevation, and slope. We therefore conclude that all four watersheds should be treated together, and that the effect of small roads in undeveloped areas is minor enough to merge with background for present purposes. Thus eight watershed years are available for determination of background yield equations for N and P.

Table 50 summarizes the results of the log-transform regressions used to determine the coefficient and exponent of the background yield equations for the eight watershed years. The goodness of fit is excellent both for phosphorus and for nitrogen. For phosphorus the exponent is slightly greater than 1, as expected from the Como Creek work. For nitrogen the exponent is lower than for phosphorus. Once again, this is expected for undisturbed watersheds of this type from the Como Creek studies (Lewis and Grant 1979). The Como Creek work shows that nitrate yield typically increases more slowly than water yield, but that particulate N yield increases faster than water yield. The two effects partially cancel, producing an exponent in this instance only slightly greater than 1.0.

	Exponent (b)	Std. Error (b)	Coefficient (a)	Equation
Phosphorus*	1.372	0.20	0.000782	$Y_n = 0.000782\ Y_w^{1.372}$
Nitrogen*	1.154	0.17	0.0842	$Y_n = 0.0842\ Y_w^{1.154}$

* $mg/m^2/yr$

Table 50. Summary of statistical information on empirical establishment of the relation $Y_n = aY_w^b$ for background yield (Y_w = mm/yr water yield; Y_n = $mg/m^2/yr$ nutrient yield). Both equations are highly significant ($P<0.001$).

Residential Area on Sewer

A number of developments and subdivisions in the Lake Dillon watershed consist of clusters of homes or condominiums served by sewer.

These make a point-source contribution, which will be considered at the end of this chapter, and they also potentially make a nonpoint contribution to nutrient loading. The nonpoint contribution is associated with the presence of pavement, with changes in vegetation and soil cover within the development, with fertilizer application, with more or less continuous minor earth disturbance, and with other miscellaneous factors arising from human habitation. We consider this classification to apply to settlements or developments that have enough vegetative cover to break up runoff and thus slow the transport of materials to streams. This classification does not apply to areas that are very densely settled (more than 80% land coverage by concrete or dwellings). Such densely settled areas are treated below as urban area on sewer.

Several representative watersheds were initially chosen for quantification of the effect of residential development served by sewer. All of these except one proved impossible to use. Some were settled too sparsely and others lacked sufficiently coherent drainage to give good quantitative information. The information is thus based on a single watershed containing the developments of Wildernest and Mesa Cortina, the composite of which we shall call Wildernest. Data are available only for 1982. The data are summarized in Table 51. This watershed has an area of 246 ha, of which 126 ha is taken up by the residential development on sewer. The watershed is occupied by an annual average of 662 persons per day. It is ideal for present purposes in that it is settled densely enough to show any effects rather clearly, but is not so densely settled as to constitute an urban

	Residential on Sewer Wildernest (W2A)	Urban on Sewer Dillon Valley (W2B)
Water yield, mm/yr	246	326
Use Intensity, persons/ha	2.69	6.83
Phosphorus		
Total yield $mg/m^2/yr$	3.29	14.7
Std. Error	0.40	4.2
% Particulate	29	100
Background, $mg/m^2/yr$	1.49	2.19
Net yield, $mg/m^2/yr$	1.80	2.89*
Net yield per use unit: g/person/yr	5.63	4.22
Nitrogen		
Total yield $mg/m^2/yr$	131	456
% Particulate	39	56
Background, $mg/m^2/yr$	48	67
Net yield, $mg/m^2/yr$	83	218*
Net yield per use unit: g/person/yr	308	318

* Net yield after subtraction of interstate highway effect as well as background.

Table 51. Yield of water and nutrients from a watershed supporting residential area on sewer (1982 data, nonpoint contribution only), and a watershed supporting urban development on sewer (1982 data, nonpoint only).

area. Most of the dwellings or groups of dwellings are separated by trees and most areas are unpaved.

Table 51 gives the total water yield and total phosphorus and nitrogen yields for the watershed. The background, as computed from the background equations, has been subtracted from these to give the net yield attributable to the land use under consideration here. Table 51 expresses this net yield in terms of the use intensity index, which in this case is the number of persons. The net yields per use unit in 1982 were 5.63 g of phosphorus per person per year and 308 g of nitrogen per person per year. These are relatively small in relation to the sewage export per capita, which would be in the vicinity of 400-600 g per year for phosphorus and about five times this amount for nitrogen.

The question arises whether or not the yields shown in Table 51 are dependent on water yield. It seems certain that there is a positive relationship between nutrient yield and water yield, but this relationship cannot be quantified without more data. It will be assumed hereafter that total phosphorus and total nitrogen yield have a dependence on water yield of the same degree as documented for background yields. With the exponent fixed by this means, the coefficient can be determined algebraically from Table 51. The coefficient for P is 0.00295 and for N is 0.536 to give g/person/yr.

Urban Area on Sewer

Although we were not able to sample in such a way as to segregate cleanly any of the cities within the watershed, we did find one small

watershed segment almost exclusively on sewer that could be treated as an urban area. This watershed segment, which is referred to here as Dillon Valley, is shown in Figure 49. Development is much more dense in Dillon Valley than in Wildernest, which accounts for the difference in classification. The watershed segment has an area of 115 ha. The settled area is 65 ha, of which 62 ha is occupied by dwellings or roads. The area is occupied by a time-weighted average of 786 persons. The segment is sufficiently small and complex hydrologically that discharge could not be estimated easily. We therefore assumed runoff per unit area equal to Straight Creek. Computation of the urban effect also required subtraction of interstate highway effects. The basis of this correction is given in the section on yield from interstate highways. The results are summarized in Table 51.

The total yield of nitrogen and phosphorus for Dillon Valley was much higher per unit area than for residential area on sewer, but this was expected because of the higher density of development. On a per capita basis, the yield above background was similar to the yield for the residential area on sewer. As with the sewered residential area, we assume that the yield of both nitrogen and phosphorus is related to the water yield by an equation incorporating a common exponent (1.372 for P, 1.154 for N). Once the exponents are thus set, the coefficients of the equations can be determined algebraically from the data in Table 51, are 0.00150 for P and 0.410 for N to give the yield in units of g/person/yr.

Residential Areas on Septic Systems

Representative watersheds for residential areas on septic systems were South Barton Gulch and Illinois Gulch. Data are available both years for South Barton Gulch and in 1981 only for Illinois Gulch. The data are summarized in Table 52.

South Barton Gulch has a moderate density of development; 194 persons are distributed over 817 ha of watershed. Illinois Gulch is much more sparsely settled; it has a time-weighted average of 25 persons distributed over a watershed of 496 ha. The three watershed years provided a good range of water yields. Table 52 shows the net nutrient yield from each of the watersheds after subtraction of the background. The net yield is then expressed as per-capita phosphorus and nitrogen contribution. The data for nitrogen and for phosphorus suggest that water yield greatly affected nutrient export from these watersheds. This is to be expected, since nutrients stored by septic systems in the soil are carried out in proportion to the amount of water percolating through the soil.

The per capita phosphorus yields were plotted against water yields for each of the three years, and showed evidence of a strongly curvilinear relationship. By regression following log transformation it was established that the equation of best fit is $Y_n = 3.44Y_w^{0.759}$ where Y_n is yield above background expressed as g/person/yr and Y_w is water yield (mm/yr). The fit is good ($P = 0.04$, S.E. of exponent is 0.10). The exponent is less than 1.0, unlike the exponent for background P yield. The lower exponent implies that higher water yields were accompanied by P yields that were higher by a

	South Barton Gulch (W1A)		Illinois Gulch (W2A/81)
	1981	1982	1981
Water yield, mm/yr	190	522	72
Use intensity, persons/ha	0.147	0.147	0.050
Phosphorus			
Total yield $mg/m^2/yr$	4.14	9.83	0.70
Std. error	1.61	1.13	0.16
% Particulate	74	61	38
Background, $mg/m^2/yr$	1.04	4.19	0.28
Net yield, $mg/m^2/yr$	3.06*	5.50*	0.42
Net yield per use unit: g/person/yr	208	374	83
Nitrogen			
Total yield $mg/m^2/yr$	40	397	13
% Particulate	56	72	50
Background, $mg/m^2/yr$	36	115	11.7
Net yield, $mg/m^2/yr$	1.9*	275*	1.3
Net yield per use unit: g/person/yr	129	18722	257

* Contribution of 74 persons on residential sewer has been subtracted as well as background.

Table 52. Yield of water, nitrogen, and phosphorus for watersheds containing residential septic systems.

less than equal proportion. Individual septic fields thus appear to respond in a manner intermediate between that of background sources, whose yield increases faster than water flow, and a true point source, whose yield would not be affected by water flow.

The regression procedure was repeated for nitrogen. The fit with a curvilinear model was poor; fit with a linear model was much better. The resulting equation is $Y_n = 44.3Y_w - 5205$, where Y_n is yield of N as g/person/yr and Y_w is mm/yr water yield. The standard error for the slope is 11.8 and the significance of the relation is $P = 0.08$. The cause of the negative intercept is clear from Table 52: N yield above background becomes vanishingly small at water yields well above 0. This is explained by biological uptake and retention up to a certain flushing threshold, by denitrification, or by a combination of these.

Ski Slopes

The water and nutrient yields from Keystone and Copper Mountain ski slopes (including lodges, trails, and lifts) were quantified in 1981 and 1982. Contributions from the associated residential areas and point sources are not considered here. Table 53 summarizes the data. The yield of phosphorus was well above background. The nitrogen yield was also above background, but less markedly so than phosphorus. The percentage of each watershed accounted for by cleared areas was used as a measure of use intensity; for both resorts about 40% of the area is cleared. The number of skiers would probably serve equally well as an

	Keystone (W4A)		Copper Mountain (W4B)	
	1981	1982	1981	1982
Water yield, mm/yr	98	195	348	329
Use intensity, % open	40	40	42	42
Phosphorus				
Total yield, $mg/m^2/yr$	1.94	5.51	3.56	6.29
Std. Error	0.41	1.30	0.99	1.49
% Particulates	49	67	44	47
Background, $mg/m^2/yr$	0.42	1.08	2.40	2.22
Net yield, $mg/m^2/yr$	1.52	4.43	1.16	4.07
Net yield per use unit - mg/m^2used/yr	3.84	11.20	2.40	8.43
Nitrogen				
Total yield, $mg/m^2/yr$	27	75	74	88
% Particulate	59	53	27	37
Background, $mg/m^2/yr$	17	37	72	67
Net yield, $mg/m^2/yr$	10	38	2	21
Net yield per use unit, mg/m^2used/yr	25	96	4	44

Table 53. Yield of water and nutrients from watersheds supporting ski slopes (nonpoint contributions only).

intensity measure. In 1981 and 1982 Copper Mountain and Keystone had between 500,000 and 700,000 skier days per year.

The water yields from Copper Mountain were higher than for Keystone; this is explained partly by elevation and partly by location along the east-west moisture gradient. There is no indication in Table 53 of the close relation between water and nutrient yield documented for most other land uses. The explanation is not obvious, since the mechanisms of nutrient release are complex. Presence of the trails and slopes undoubtedly facilitates transport, even without human activity, especially since ski trails and slopes must be cleared parallel to the gradient. Possibly more important, however, is mechanical damage to the ground surface when snowpack is minimal. There is also a certain amount of spring fertilizer application on the slopes. Yields at either Keystone or Copper Mountain could have been augmented by construction on or near the slopes.

Since the yield from ski slopes does not show any indication of sensitivity to water yield, we use a mean. The mean for the four watershed years in Table 53 is 6.5 mg/m^2 of cleared area/year above background for P and 42 for N.

Interstate Highways

The effects of interstate highways were isolated in the upper segments of Straight Creek and West Tenmile Creek. Table 54 summarizes the data and expresses yield above background in relation to area of roadway plus right of way. For phosphorus, the yield was unexpectedly high in both watersheds. The relation of P yield to water yield

	Straight Creek (W2B2)	West Tenmile (W5B)	
	1982	1981	1982
Water yield - mm/yr	326	522	662
Use intensity, % roads	2.97	0.78	0.78
Phosphorus			
Total yield - $mg/m^2/yr$	4.09	5.38	7.19
Std. Error	0.84	1.76	0.67
% Particulate	60	47	50
Background - $mg/m^2/yr$	2.19	3.80	5.51
Net yield - $mg/m^2/yr$	1.90	1.57*	1.67*
Net yield per use unit - mg/m^2used/yr	64	200	213
Nitrogen			
Total yield - $mg/m^2/yr$	102	181	123
% Particulate	61	18	36
Background - $mg/m^2/yr$	68	115	151
Net yield - $mg/m^2/yr$	34	66*	-28*
Net yield per use, intensity - mg/m^2used/yr	1146	8426	-3574

* a small contribution from Vail pass public toilet septic field has been subtracted.

Table 54. Yield of water, N, and P from watersheds with interstate highways but little else.

was tested for fit to a function of the form already used for other nutrient sources. The fit was reasonably good, and resulted in the equation $Y_n = 0.00209Y_w^{1.799}$ where Y_n is nutrient yield per m^2 of highway plus right of way and Y_w is water yield (mm/yr). The slope has a standard error of 0.53 and P = 0.09 (ideally P should be below 0.05, but the power of the test is low with N=3). High yield is probably due in part to major disturbance of vegetative cover. Also the materials added to roads in the winter may contribute, as does the fertilization associated with revegetation programs now in progress.

The nitrogen yield associated with roads was low by comparison with other sources and less consistent than P, as shown by Table 54. The N yield above background for West Tenmile was negative one year. This is quite possible due to excess of uptake or denitrification over yield, but presents some difficulties in prediction. For purposes of curve fitting, the two yields from West Tenmile were averaged, giving a low positive yield. The fit to an exponential equation was then reasonably good: $Y_n = 1.71Y_w^{1.13}$, S.E. slope = 0.40, P = 0.11, where the units are the same as for the phosphorus equation.

Mining

The Climax Molybdenum mining operation is a unique mixture of nutrient sources and must therefore be treated separately from other land uses. Water from Climax Molybdenum has a high suspended and dissolved solids content. The water is retained in tailings ponds, where sedimentation occurs, and the ponds are purged during high water. The ponds receive not only industrial wastewater but also domestic

effluent derived from the toilets and kitchens of the plant. The domestic effluent is not treated, except by sedimentation in the ponds.

Table 55 summarizes the water and nutrient yields from upper Tenmile Creek where the active mines are located. There is little else in this watershed except old mines and State Highway 91. Roads, tailings, and bare areas around the plant account for 753 ha of the total 6447 ha. Virtually all of the nutrient yield above background can thus probably be attributed to Climax Molybdenum.

The P yields were above background and the N yields were even more so in 1981 and 1982. Neither seems related to water yield. Unlike most watersheds, the total nutrient yields for upper Tenmile were higher in 1981 than 1982. In 1981 the time-averaged work force at Climax Molybdenum was 2650 persons and in 1982 it was 1100 persons. The yield of both N and P thus seems related to size of work force. Table 55 expresses the yields both years on a per capita basis; the yield per capita was almost identical for the two years. We will thus treat N and P yield at Climax Molybdenum as a function of work force size.

The P yield per capita at Climax Molybdenum suggests that the P output above background is almost entirely a sewage contribution and not an industrial effect, since the yield per capita is in the range of what would be expected from the domestic output of the workforce. The ponds appear to be effective in reducing particulate P losses that would result from earth disturbance. N export is very high, suggesting major augmentation by industrial activity. Ammonium nitrate is used in

	Upper Tenmile - W5A	
	1981	1982
Water yield - mm/yr	435	513
Use intensity - persons/ha	0.41	0.17
Phosphorus		
Total yield - $mg/m^2/yr$	7.07	5.84
Std. Error	1.55	0.64
% Particulate	20	58
Background - $mg/m^2/yr$	3.26	4.09
Net yield - $mg/m^2/yr$	3.81	1.75
Net yield per use unit - g/person/yr	93	103
Nitrogen		
Total yield - $mg/m^2/yr$	821	491
% Particulate	5	15
Background - $mg/m^2/yr$	93	113
Net yield - $mg/m^2/yr$	728	378
Net yield per use unit - g/person/yr	17710	22150

Table 55. Yield of water, N, and P for the mining area on upper Tenmile Creek. Use intensity is based on number of persons working at Climax Molybdenum.

blasting and may be the source of this augmentation. Revegetation may also contribute.

Point Sources

There are two types of point sources in the Lake Dillon catchment: large sources with tertiary treatment and small sources with secondary treatment. We will establish here a contribution per capita for each category. Large point sources with tertiary treatment include the Snake River WWTP and Frisco WWTP, whose effluents were sampled directly. The results for these two have already been reported in the chapter on total loading of the lake, but have not yet been converted to a per capita basis. In addition, data can be obtained for the Breckenridge WWTP as the difference between stations BR1 and BR2 on the Blue River (just downstream and just upstream of the Breckenridge WWTP outfall). These data are summarized in Table 56. The Copper Mountain WWTP, designed as a tertiary plant, was in a transition from secondary to tertiary during the course of the study and will therefore not be considered here as a representative of tertiary treatment.

	Persons Served*	P - g/person/yr		N - g/person/yr	
		1981	1982	1981	1982
Frisco WWTP	3300	11.7	84.8	1108	955
Snake River WWTP	3230	9.1	22.3	1003	1260
Breckenridge WWTP	6700	113.	44.0	1378	1479

* Annual average persons per day

Table 56. Yield of P and N from three WWTP plants on a per capita basis. All three plants practice tertiary P removal.

Table 56 shows the effectiveness of the tertiary treatment, since all of the per capita P yields are well below the raw per capita

contributions coming to point sources (400-600 g/person/yr). At the same time, there was considerable variation in P yield per capita between years, even at the same plant. This is explained entirely by breakdown or shutdown of tertiary treatment, which occurred at all plants on one or more occasions over the course of the study. Although it could be argued that tertiary sources should be represented by their yields when treatment facilities are operating properly, which would give yields about half as great as the observed average (Table 57), it is probably realistic to assume that the plants will be impaired for a certain amount of time each year.

	g/person/yr
Phosphorus	
Secondary treatment	270
Tertiary treatment	47
Nitrogen	
All treatment	1197

Table 57. Average point source yields on a per capita basis.

Nitrogen yields from the point sources were much more uniform, since they were not affected as much by variations in plant operation. The Breckenridge plant practices ammonium removal, but this does not affect total N yield per capita (Tables 56, 57).

The P contribution from small point sources (package plants) subject to secondary treatment was estimated from data taken upstream and downstream of such sources along the Blue River above Goose Pasture Tarn. The results are reported in Table 57.

Overview of Nutrient Yields

Export of N and P from various sources is expressed in Figure 50 on a per capita basis wherever this is meaningful. In Figure 51 the export rates are recast on an area basis, assuming runoff is near that of an average year. Among the point sources, the potency of sources without tertiary treatment is striking. For nonpoint sources, septic systems stand out, as expected, as do interstate highways, which might not have been expected.

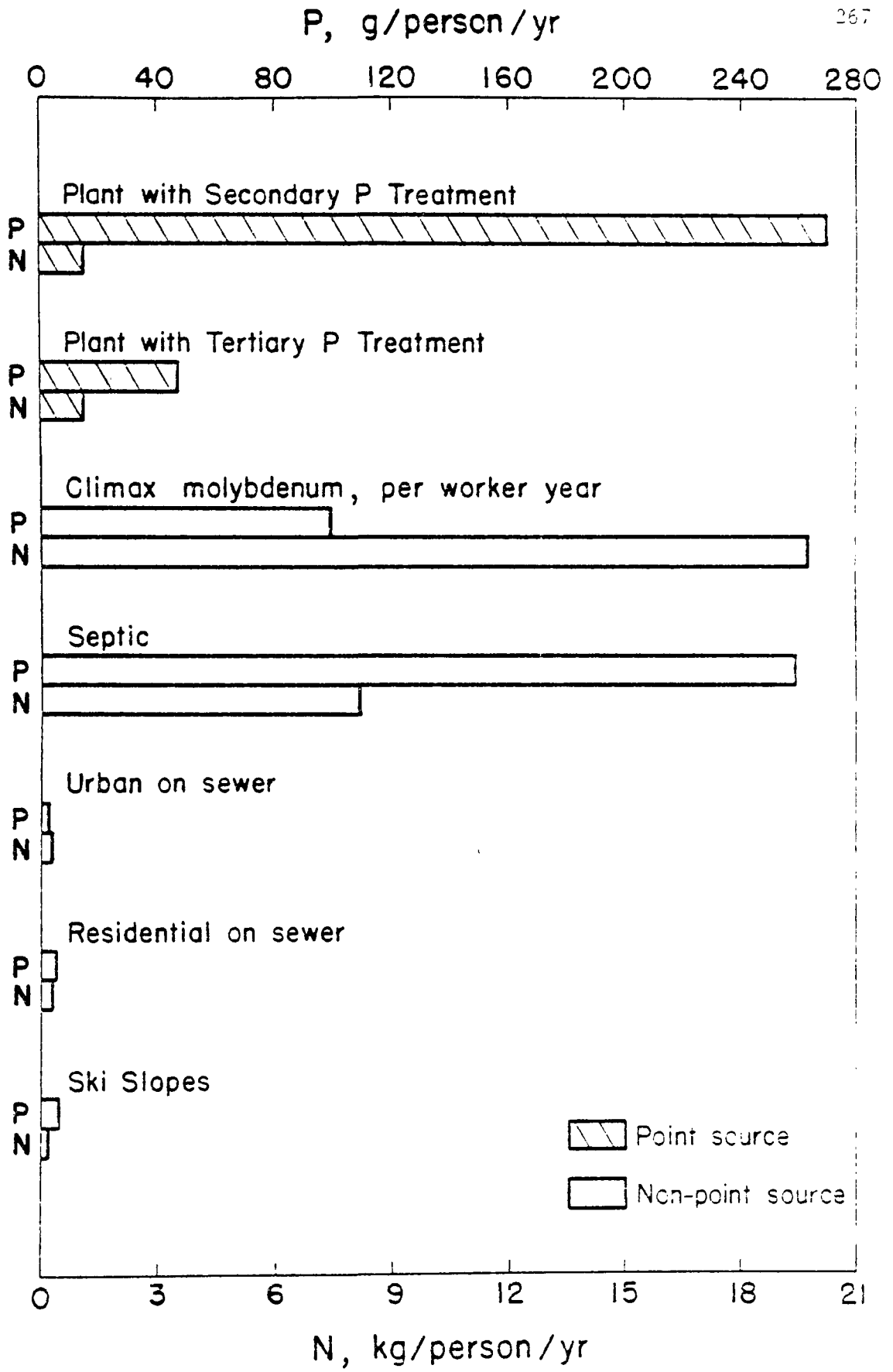


Figure 50. Nutrient yield from various sources expressed on a per capita basis, assuming 300 mm runoff. Sources that cannot be expressed on a per capita basis are not shown.

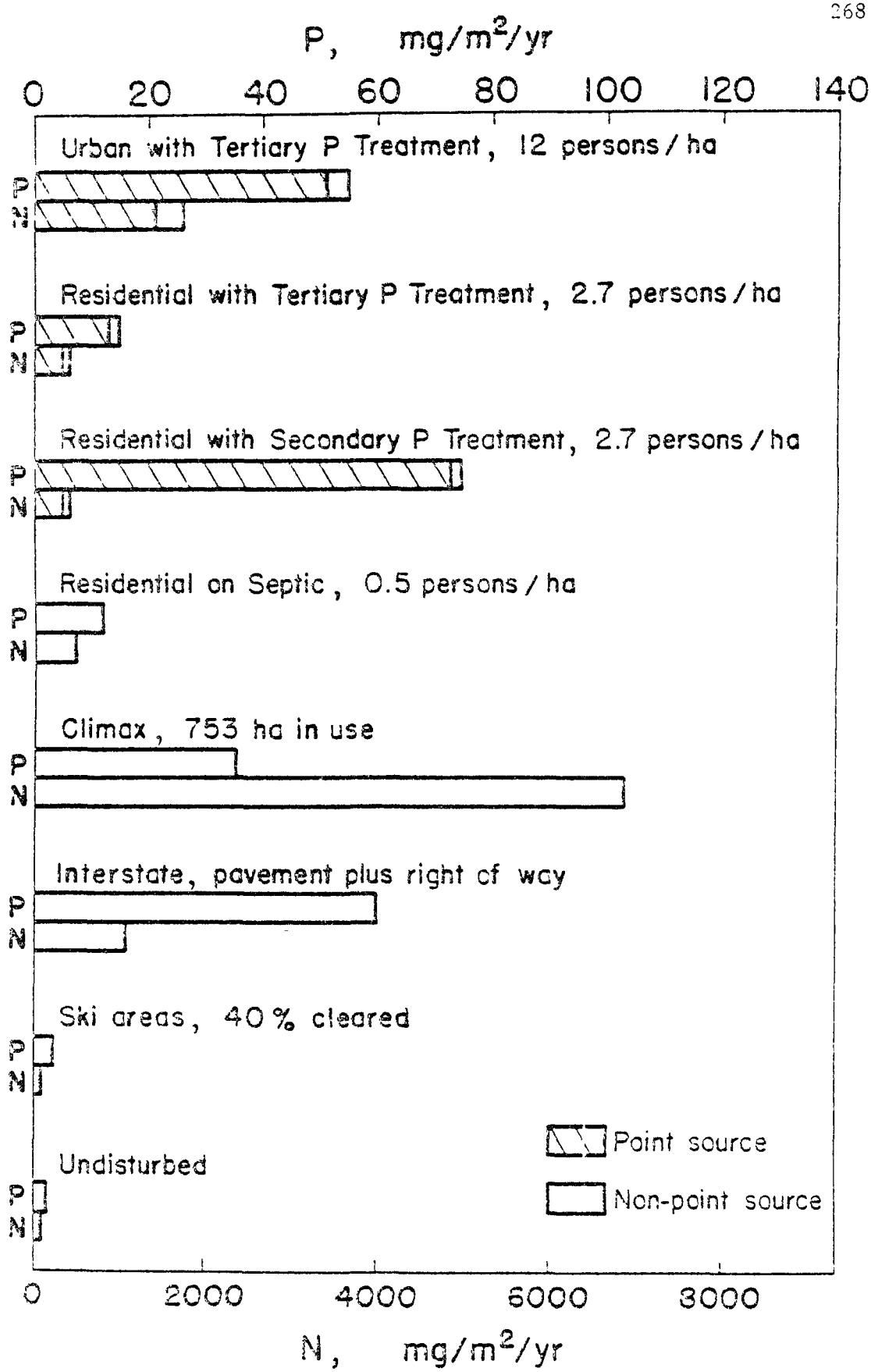


Figure 51. Nutrient yield from various sources expressed on an areal basis, assuming runoff of 300 mm.

Separation of Nutrient Sources within the Watershed

Chemistry and discharge measurements were taken at a number of points along each one of the three major rivers (Figure 3). Each of the river drainages was thus divided into a number of watershed segments. If the land use within any given segment is known quantitatively, the equations that have been developed in the previous chapter for nutrient yield associated with various land uses should provide a prediction of the nutrient yield from that segment. Three important factors may cause deviations between the observed and predicted nutrient yields from any given watershed segment.

(1) Estimation errors. The observed yields for any given segment are based on chemistry and discharge measurements, both of which are subject to analytical error. We may also include under the heading of error variance the confidence limits around any one of the curves used to predict nutrient yield from land use. These confidence limits reflect not only error variance in the original data from which the curves were developed but also a certain amount of individuality in watersheds with respect to soils, vegetation, exposure, slope, and elevation. If these sources of error were the only cause of deviations between predicted and observed, we would expect to see a random assortment of positive and negative deviations between the observed and predicted values.

(2) Unexpected sources. The second cause of deviation between observed and expected yields for individual segments is the presence of nutrient sources that are either unquantifiable or unknown. In the one

instance where such a source proved to be important, it was possible to determine the source of unusual and unexpected nutrient yield.

(3) Storage. The third factor affecting the agreement between observed and predicted yields is a storage effect associated with areas of low relief in the river bottoms. This has not been incorporated into the equations used for prediction. The equations were developed on the basis of small watersheds off the main channel where individual land uses could be isolated. When nutrients flow to lower elevation, and particularly into a much broader stream channel incorporating low-lying areas and wetlands, significant amounts of nutrient can be stored in years of low flow and purged in years of high flow. Mechanisms of phosphorus storage include sedimentation of particulate phosphorus, biological uptake of soluble phosphorus, and ligation or inorganic precipitation of soluble phosphorus. For nitrogen, the possibilities are more complex, including all of the possibilities for phosphorus plus denitrification, by which oxidized nitrogen species are converted to nitrogen gas, which is subsequently released into the atmosphere. All of these phenomena are most likely in the river bottoms and associated flood plains and wetlands.

Nutrients lost by sedimentation and ligation or uptake in years of low water may be returned to the open channel and thus enter the lake during a year of high water when the physical forces at work in the river bottoms are sufficient to move large amounts of accumulated materials. Thus the river bottoms can act as a storage site that retains a portion of the nutrient yield in dry years and gives back much or all of this retained portion in wet years. These storage and

release phenomena are most correctly regarded as temporary changes in the watershed nutrient inventory from one year to the next, and will be referred to below as the "inventory change." If inventory change is important in determining the yield of nutrients to Lake Dillon, we should see in dry years a consistently lower ratio of observed to predicted yield in river segments incorporating areas where the river water moves through wetlands, gravel beds, or ponds. If much or all of the storage (increase in inventory) is given up in a wet year (decrease in inventory), we would expect consistent increase in the ratio of observed to predicted yields from these segments in a wet year. If nutrient inventory change is shown by the data from individual stream segments, then additional equations must be developed to account for year-to-year inventory change of nutrients due to their accumulation in low-lying areas in dry years and their release from these areas in wet years.

Snake River Drainage

The Snake River drainage was divided into four segments as shown in Figure 52. Histograms of the observed and predicted yields are superimposed on each one of the stream segments. Phosphorus yields are presented on the left side of the river channels and nitrogen yields on the right side of the river channels. For each year and for each segment, one histogram bar shows the observed yield from the segment, and the adjacent bar shows the predicted yield. The scales for the yields are all the same for a given nutrient, but differ between the two nutrients because of the larger yields of nitrogen.

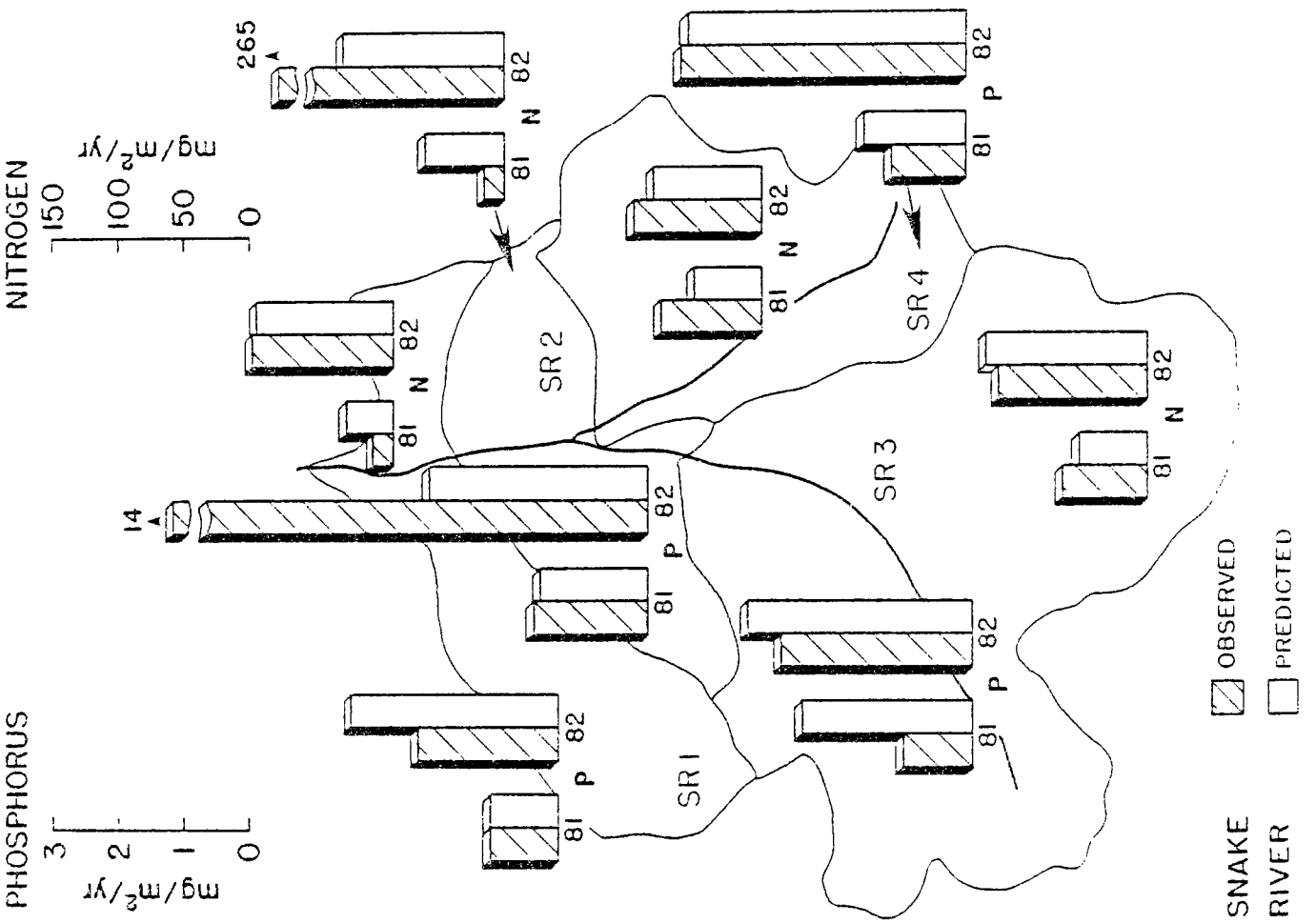


Figure 52. Observed and predicted yields from segments of the Snake River watershed. Phosphorus is on the left of the stream channel, and nitrogen is on the right of the channel in any given segment.

For phosphorus the agreement between predicted and observed yields from individual stream segments is generally very good. One notable exception is the phosphorus yield in 1982 from segment SR2, which incorporates Keystone Gulch, Keystone Ski Area, and Keystone Development. The observed phosphorus yield from this segment was extraordinarily high in 1982; it was far above any expected random deviations from the predicted. This exceptionally high observed phosphorus yield is explained by construction activities. Keystone Ski Area installed six 300-hp pumps in the river bed during 1982 and constructed an associated pipeline to carry the water from these pumps to ski slopes, where it was to be used for snowmaking. Construction was carried out at several other locations in various watersheds during the course of the study but did not produce such extraordinary yields of phosphorus. The especially high yield in this instance is undoubtedly accounted for by work in or near the river bed, where water movement greatly facilitated the transport of phosphorus-bearing materials loosened by the construction activities. Since the prediction equations do not take into account any such special activities, the gap between observed and predicted P of 11 $mg/m^2/yr$ can be accounted for specifically by the construction. Total 1982 transport due to the construction in this segment was 295 kg of phosphorus. This was approximately 7% of the runoff yield of phosphorus for the entire Lake Dillon watershed during 1982. Expressed in another way, it was equivalent to the annual yield of phosphorus from a tertiary point source serving 6300 persons, assuming efficiency of point source operation as shown in Table 57.

There is little evidence for significant valley bottom inventory changes affecting phosphorus yield in the four Snake River segments. The inventory change effect is unlikely in SR3 and SR4 because of the steep relief in these segments. The effect is most likely in segment SR2, where it would be obscured by the large construction effect in 1982.

Predictions for nitrogen are also in good agreement with the observed values with the exception of the 1982 value for segment SR2. Once again, we associate the extraordinarily high yield in 1982 for SR2 with the Keystone construction activities.

Blue River Drainage

The Blue River drainage was originally divided into seven segments (BR1-BR7). Figure 53 shows the segment yields based on a total of four segments. Segment BR1 has been omitted because this small segment isolates only the Breckenridge Wastewater Treatment Plant, whose yield has already been discussed in the previous chapter. Some of the smaller segments between BR2 and BR7 have been combined for present purposes because a considerable land area is required to show a quantifiable effect on the chemistry and discharge of a river as large as the Blue River (BR2 in Figure 53 includes the original BR2 and BR3; BR4 in Figure 53 includes original BR4 and BR5). In addition, diffuse drainage in the vicinity of Breckenridge made impossible any reliable determinations of discharge there.

There were no extraordinary departures of observed from predicted phosphorus yield in the Blue River drainage. The Blue River drainage

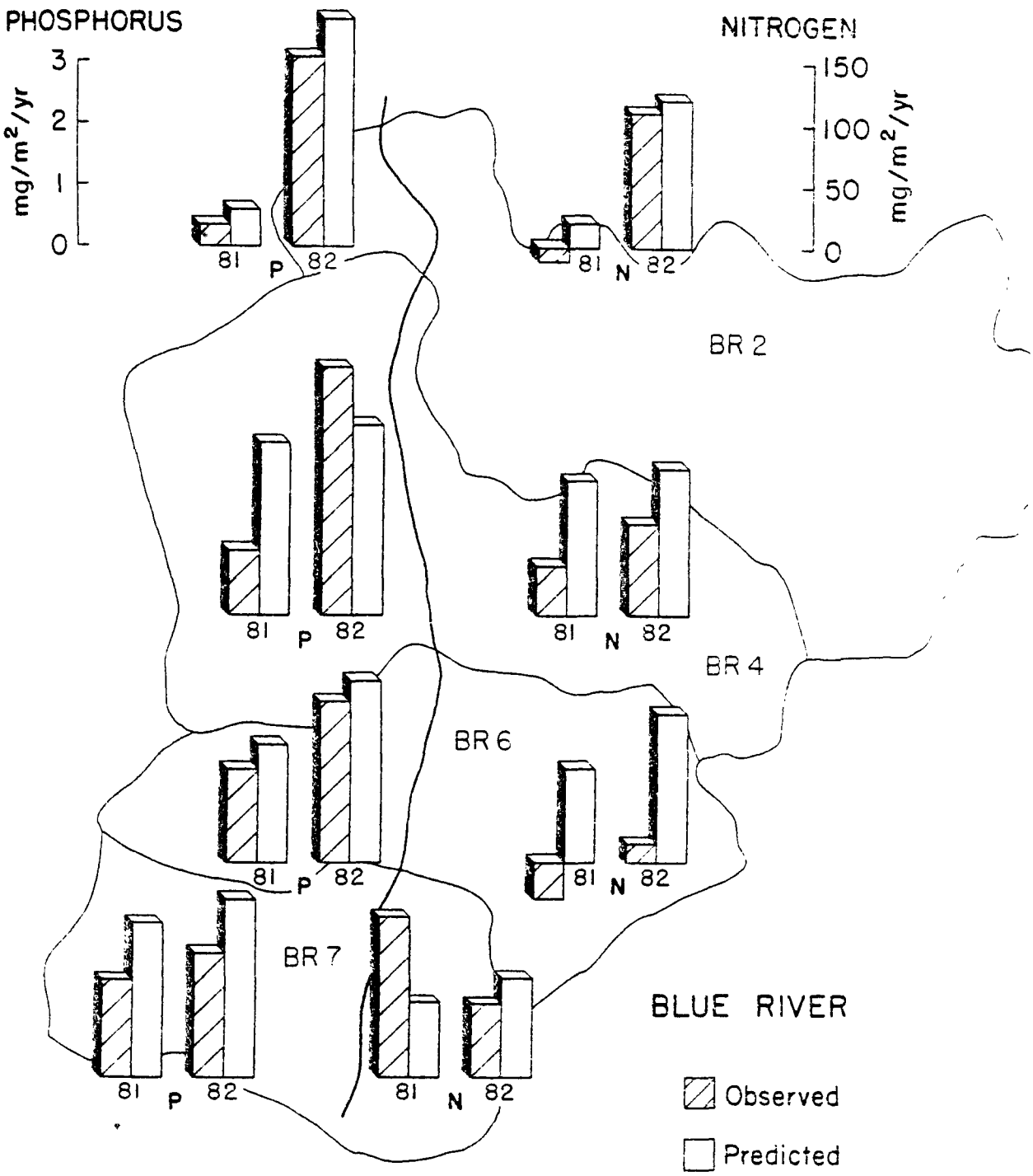


Figure 53. Observed and predicted yields from segments of the Blue River watershed. Phosphorus is on the left of the stream channels, nitrogen on the right.

is a good place to look for valley bottom inventory change because the main channel, even as high as BR7, incorporates extensive wetlands, small ponds, and accumulated old tailings in the river bed. In every segment of the Blue River drainage, the ratio of observed to predicted phosphorus was lower in 1981 than in 1982. This is explained by the inventory change effect, which leads to the entrapment of a certain amount of the yield in dry years, and a release of a portion of this material during wet years.

In BR7, the nutrient sources drain through extensive areas of low relief. The predictions were consistently above the observed yields in this portion of the watershed for phosphorus, and the ratio of observed to predicted was higher in 1982 than in 1981. The data thus suggest that significant retention occurred even in the wetter year of 1982, although it was especially pronounced in 1981. The same was true of BR6, which incorporates Goose Pasture Tarn as well as an extensive diffusely drained area above the Tarn.

Segment BR4 has a much different character from that of BR7 and BR6. The river bottom has been extensively modified, both by mining and by settlement around the town of Breckenridge. The phosphorus data show phosphorus accumulation in low-lying areas in 1981 and a strong purging of the stored material in 1982. The behavior of Segment BR2 was more similar to that of BR6 than BR4.

There was considerable difference between the yield patterns of nitrogen and phosphorus in the Blue River drainage. For several reasons, phosphorus and nitrogen cannot be assumed to behave similarly in any given watershed segment. First, a great deal of phosphorus is

transported in particulate form, whereas the portion of nitrogen transported in particulate form is generally much smaller. Thus the factors that lead to entrapment of significant amounts of particulate phosphorus in a given watershed segment will not necessarily have a similarly strong effect on total nitrogen yield from that segment. Furthermore, nitrogen yields can be affected by denitrification losses. Denitrification is especially likely in low-lying areas with diffuse drainage where anoxia can occur. Finally, the biological uptake of phosphorus and nitrogen is likely to differ.

The ratio of observed to predicted nitrogen yield for individual segments was higher in 1982 than in 1981 for all segments except BR7. The data thus indicate operation of the inventory change effect that was observed for phosphorus. However, it is clear that other factors complicated the yield of nitrogen from most of the segments. Negative yields (i.e., N sources plus incoming N greater than outgoing N) were observed for segments BR2 and BR6 in 1981. Yields in segment BR6 were exceptionally low even in 1982 by comparison with the prediction. The major deviations of segment BR6 from the predictions are almost certainly connected with losses of nitrogen associated with Goose Pasture Tarn. Concentrations of nitrogen above and below the Tarn indicate that it is a major nitrogen sink. Nitrogen loss in the Tarn is probably due to a combination of uptake by weedbeds, sedimentation, and denitrification in the mud.

Tenmile Creek Drainage

The data for Tenmile Creek are shown in Figure 54. Three segments are shown. Smaller segments have been combined for reasons similar to those mentioned for the Blue River. The upper segment is the site of Climax Molybdenum, for which predictions are good. The middle segment (TM4) has numerous dispersed sources of nutrients, and is also the discharge point for the Copper Mountain Wastewater Treatment Plant. The presence of the wastewater treatment plant complicates the interpretation of the data. The Copper Mountain plant was doing secondary treatment during almost all of 1981, and the effluent was for the most part pumped into a pond instead of into the river directly. Tertiary treatment was in force, but was considered not fully effective according to plant operators, during the first ten months of 1982, after which secondary treatment was resumed for the months of November and December. The presence of the wastewater treatment plant, and the problems associated with its new treatment practices, make the estimates of other factors difficult because the plant introduced much more irregular variation than would be expected in other watershed segments. There was major overestimation of phosphorus for TM4 in 1981 but not in 1982. The 1981 overestimate is probably due to storage in the pond and in the river bottom in the lower half of the segment. Storage would be very likely in the poorly drained areas known as Curtain Ponds and Wheeler Flats. However, segregation of this storage and purging effect is more difficult in TM4 than in any other location because of the changing operations of the point source.

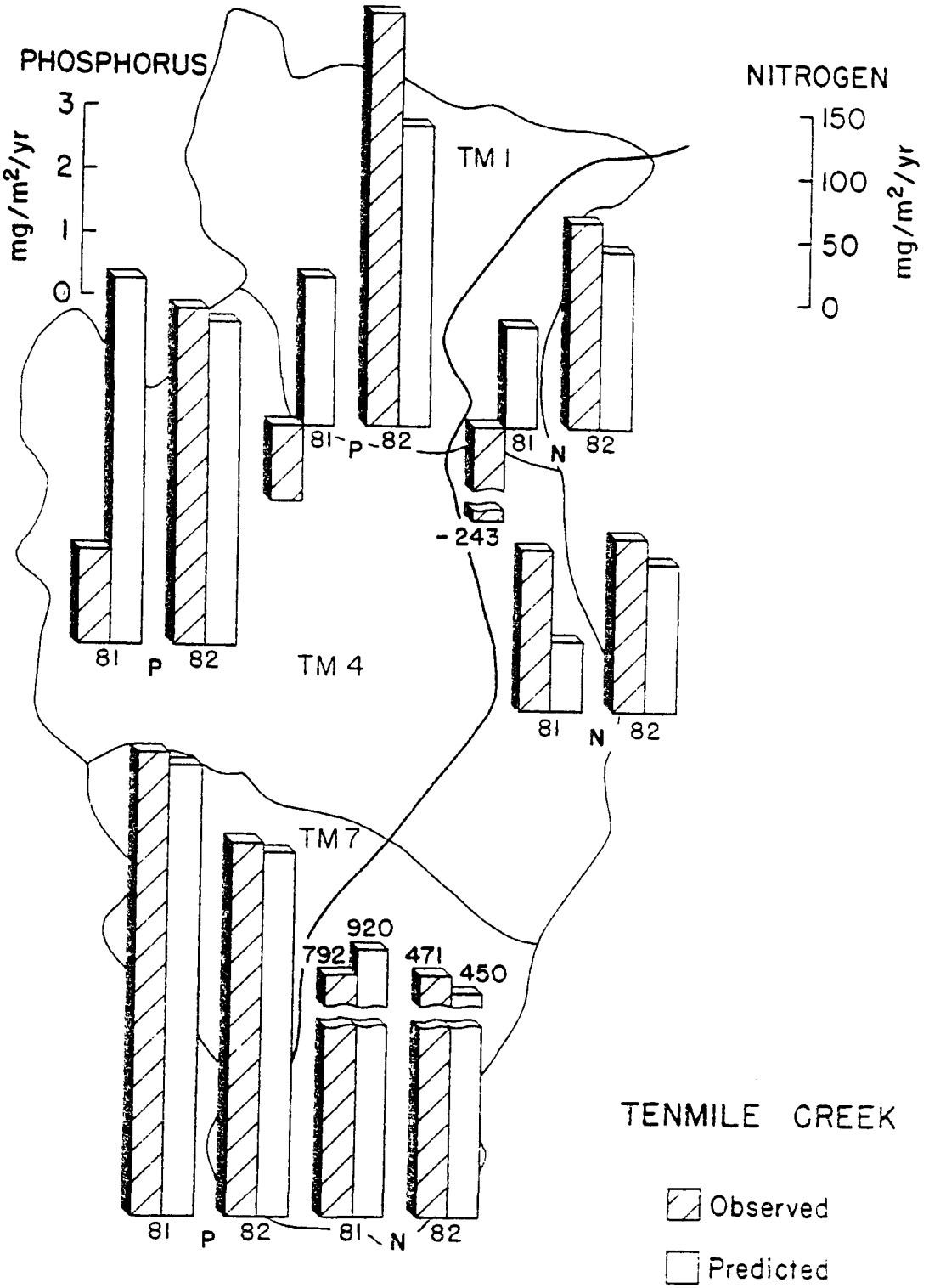


Figure 54. Observed and predicted yields from segments of the Tenmile Creek watershed. Phosphorus is on the left of the stream channels, nitrogen on the right.

Segment TM1 incorporates a large portion of the town of Frisco. Since the town is located just adjacent to the lake, the drainage of Tenmile Creek near its mouth is modified in a major way by the presence of the city. There is a large area of flat relief in the vicinity of Frisco. Probably because of this flat area, the phosphorus and nitrogen yield data for TM1 show evidence of inventory change between the dry and wet years. Both phosphorus and nitrogen yields for 1981 were negative, indicating that the areas of low relief within this drainage stored not only some of the yield from segment TM1 but also some of the yield of upstream segments. However, the observed storage in 1981 was almost exactly compensated by excess of observed over predicted yield in 1982, indicating a purging effect. The purging effect was less pronounced for nitrogen, possibly because of permanent nitrogen losses to denitrification or biological uptake.

Relationship between Runoff and Nutrient Storage

Since the data show that nutrients originating at various points in the watershed are stored in the river valleys during years of low runoff and purged from these areas during years of high runoff, the yield equations cannot predict total output from the three rivers until they have been coupled with an inventory change function. An estimate can be made of the function from the data at hand. Inventory change will be positive in those years when part of the yield from areas of steeper relief accumulates in areas of low relief and will be negative when part of this stored inventory is released during wetter years.

For nitrogen, the function will be considered to include any effects of denitrification.

The first step in estimating the inventory change function is to sum up the observed and predicted total yields from runoff for each of the two years of the study. The lakeside point sources (Frisco, Snake River, and Breckenridge WWTP) are excluded from the summation, since they are so close to the lake that the nutrients they give off are not subject to storage. The summations are corrected for major discrepancies from the predicted values that are known not to be connected with storage. In the two years of record, the only example of this is the boost of yield in the Snake River watershed during 1982 caused by construction in the vicinity of Keystone. After this correction, the remaining discrepancy between observed and predicted yields from all sources combined is assumed to be due to inventory change. Since inventory change is dependent on amount of runoff, we postulate a decline from positive to negative inventory change as runoff goes from the lowest to the highest annual values. A corollary is that the average change in inventory over a number of years is not very great. Even if the watershed is aggrading, as it may be with net accumulation of vegetation and filling in of excavated areas, the resulting departure from equilibrium over a number of years would amount to a relatively small percentage of nitrogen and phosphorus flux (cf. Bormann and Likens 1979).

Figure 55 shows the apparent change in nutrient inventory for the two years of record for both nitrogen and phosphorus. The nutrient inventory change is expressed as a percent of watershed overland

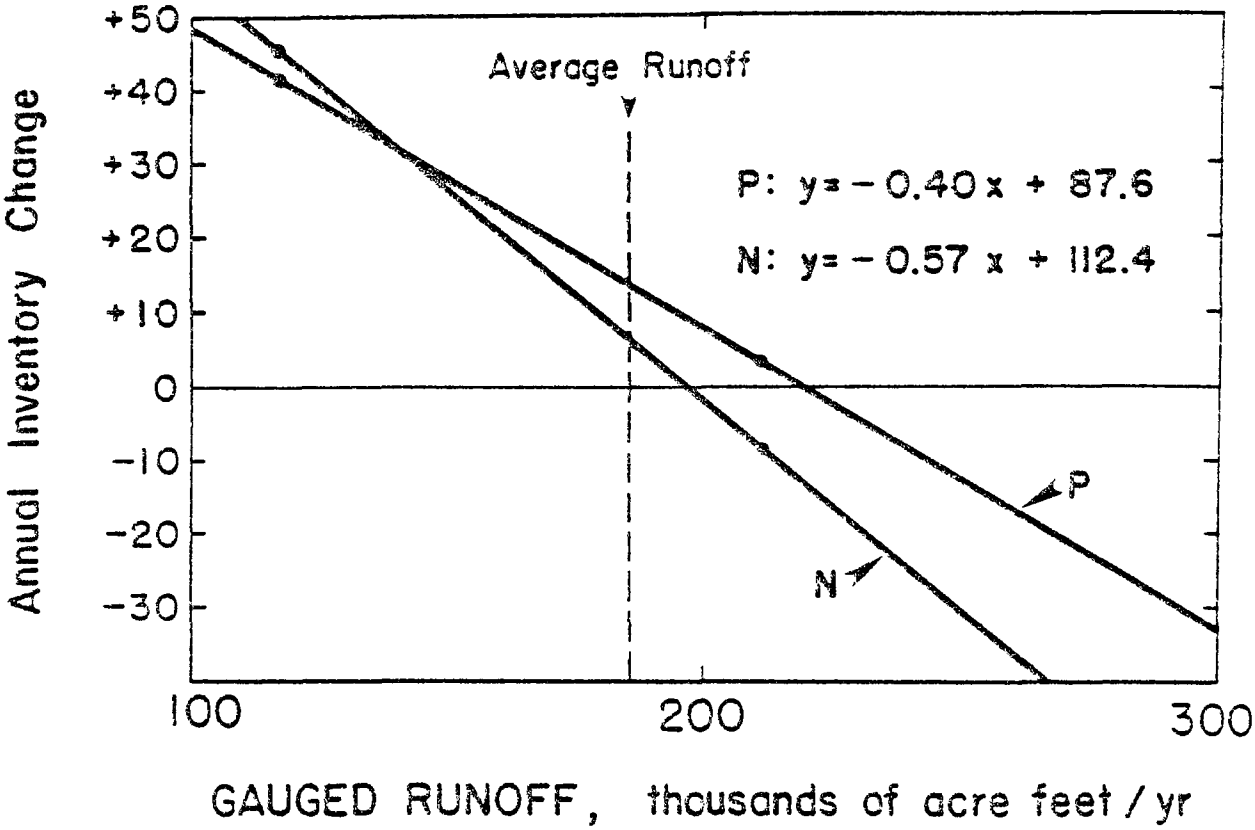


Figure 55. Linear plot of the inventory change functions for nitrogen and phosphorus. Positive inventory change indicates storage. Inventory change is expressed as a percentage of the summed runoff nutrient sources.

nutrient yield, excluding lakeside WWTP sources. The percentages were calculated as follows:

$$W_n = \frac{(Y_{n,o} - Y_{n,e})}{Y_{n,o}} \times 100$$

where W_n is nutrient inventory change (%), $Y_{n,o}$ is observed overland yield, and $Y_{n,e}$ is the expected overland yield from summation of all sources with the assumption that inventory change is zero. Figure 55 shows the relationship of W_n to discharge. The two points (1981, 1982) for each of the two nutrients are joined by straight lines. Although the form of the relationship is not necessarily linear, numerous additional points would be required to show curvature. Unless there is very marked curvature, the line will provide reasonable approximations.

The watershed appears to be aggrading with respect to phosphorus, since an average water year is accompanied by a 10-15% retention of yield in valley bottoms. Nitrogen also shows some net retention in an average year, but less than phosphorus. The equations for the lines, as indicated in Figure 55, will be used to represent the expected change in inventory in any particular year.

Comparison of Observed and Predicted Total P and N Loading

The land-use equations, point source equations, and inventory-change function lead to predictions of total runoff yield. All identifiable sources are dealt with by these equations except major construction in or near stream beds, such as that observed near

Keystone in 1982. Although the frequency of this activity is largely unpredictable, it will surely be above zero. As an approximation of the effect, we use the average increase in yield over the two years of record expressed as percent of other runoff. This boosts the P prediction by 3.5% and the N prediction by 2.2%.

When the equations for yield from all watershed sources are used, and a mean figure is used for contributions from precipitation and groundwater, the result is an overall predicted loading of N and P for the lake in 1981 and 1982. These predicted loadings are shown in Table 58, which also shows the observed loading for comparison. The observed and predicted values agree very well (<10% difference). Deviations of predicted from observed are caused mostly by the irregular nature of failures in tertiary treatment plants, and to some extent by irregular variation in other sources.

	P load, kg/yr		N load, kg/yr	
Year	Observed	Predicted	Observed	Predicted
1981	2900	2700	85,600	83,500
1982	4800	4600	151,400	154,300

Table 58. Comparison of predicted and observed N and P loads for the two years of study.

Itemization of Lake Nutrient Sources for 1981 and 1982

In the chapter dealing with total nutrient loading of the lake, nutrient sources were itemized according to the major pathways by which they entered the lake (Tables 46, 47). On the basis of the nutrient yield equations, nutrient sources can be further broken down according to land use. In making this breakdown, we use the values

reported in Tables 46 and 47 for precipitation, groundwater, and effluent contributions from the Frisco and Snake River plants. The contributions of the Breckenridge Wastewater Treatment Plant are taken from the analysis of the watershed segment BR1 as reported in Table 56. The contributions of all other sources are obtained by application of the yield equations and the inventory change equations to the land use data and water yield data for 1981 and 1982. Since the ability of these equations to predict observed yield accurately for 1981 and 1982 has already been demonstrated, it is expected that the sum of individual sources obtained by this means will be very close to, but not exactly equal to, the observed loading.

Table 59 shows the complete breakdown of sources. The table gives the kilograms of nitrogen and phosphorus from each source reaching the lake in 1981 and 1982, and also expresses these as a fraction of total annual loading. Although data for both years are given, it should be noted that the distribution of contributions for 1982 is much more typical of the median or average situation under present land use conditions than is the distribution for 1981. There are several reasons for this. First, the runoff for 1981 was exceptionally low. The sum of flows for the 22 years of record at the four U.S.G.S. gauges averages 185,600 acre feet per year. The gauged flow for 1981 was 116,500, or 63% of the average. Although the runoff for 1982 was above the average (210,900), it was only slightly so (14%) and thus represents more accurately the median distribution of various contributions to the total nutrient loading of the lake. In addition, the wastewater treatment plant at Copper Mountain was operating under

	Phosphorus					Nitrogen				
	1981		1982			1981		1982		
	kg	%	kg	%	Mean %	kg	%	kg	%	Mean %
Tertiary Outfalls										
Frisco	39	1.3	280	6.1	3.7	3656	4.2	3152	2.1	3.2
Snake River	29	1.0	72	1.6	1.3	3240	3.8	4070	2.7	3.2
Breckenridge	757	25.3	295	6.4	15.9	9233	10.7	9909	6.5	8.6
Copper Mountain	-	-	41	0.9	0.5	-	-	1893	1.2	0.6
Subtotal	825	27.6	688	14.9	21.3	16129	18.7	19024	12.5	15.6
Secondary Outfalls										
Package plants	49	1.6	49	1.1	1.3	198	0.2	198	0.1	0.2
Copper Mountain	322	10.8	47	1.0	5.9	1305	1.5	378	0.2	0.8
Subtotal	371	12.4	96	2.1	7.2	1503	1.7	576	0.4	1.0
Climax Molybdenum	152	5.1	104	2.2	3.4	28583	33.0	23736	15.6	24.3
Keystone Construction	-	-	295	6.4	3.2	-	-	3883	2.6	1.3
Dispersed Nonpoint Sources										
Residential areas (sewered)	9	0.3	32	.7	.5	487	.6	1901	1.3	1.0
Urban areas (sewered)	9	0.3	31	.7	.5	761	.9	2702	1.8	1.4
Septic	161	5.4	383	8.3	6.9	6713	7.8	21750	14.3	11.1
Interstate highway	21	0.7	132	2.9	1.8	461	0.5	2198	1.4	1.0
Ski slopes	36	1.2	59	1.3	1.3	214	0.2	428	0.3	0.2
Background runoff	545	18.2	2105	45.6	31.8	16577	19.2	69015	45.4	32.3
Subtotal	781	26.1	2742	59.4	42.8	25213	29.2	97994	64.4	46.8
Precipitation	846	28.2	606	13.1	20.7	14311	16.6	5433	3.6	10.0
Groundwater	13	0.4	88	1.9	1.1	587	0.7	1549	1.0	0.8
GRAND TOTAL	2988	99.8	4619	100.0	99.7	86326	99.9	152196	100.1	100.0

Table 59. Breakdown of nutrient contributions to Dillon in 1981 and 1982.

secondary treatment during 1981, whereas the standard of operation in the future, and for most of 1982, is tertiary treatment. Finally, the performance of the largest wastewater plant (Breckenridge) in 1981 was unusually poor due to equipment shutdown and malfunction.

The various nutrient sources have been grouped in a number of ways in Table 60 to emphasize different aspects of the distribution of the load among sources. In 1981 the contribution of wastewater treatment plants to the total loading was exaggerated by low runoff, which increased the relative role of these sources, and by the difficulties at the Breckenridge Wastewater Treatment Plant. The 1982 situation, which was more characteristic, showed about 16% of the phosphorus and about 13% of the nitrogen coming from the four major wastewater treatment plants combined. Package plants made only a minor addition to this total. Thus it is clear that well over three-quarters of phosphorus and nitrogen loading under the present land use conditions comes from nonpoint sources of natural and man-made origin.

Climax Molybdenum contributed about five percent of the total phosphorus loading in 1981 and about half as much in 1982. The decline is explained by the reduced pace of operations, and finally the shutdown in 1982, which drastically reduced the nutrient output from that source. Thus for Climax the 1982 data probably underestimate the equilibrium contribution, assuming that the mines ultimately resume operation. An equilibrium number would thus be more in the vicinity of five percent than two percent. Although Climax has not traditionally been thought of as a point source for nutrients, it actually behaves much the way a point source would, especially in the apparent direct

Categories	P load, %			N load, %		
	1981	1982	Mean	1981	1982	Mean
Natural vs. Man-made						
Natural	47	54	50	36	50	43
Man-made	53	46	50	64	50	57
Point vs. Non-point						
Wastewater plants	38	16	27	20	13	16
Package Plants and Climax	7	3	5	33	16	24
All other sources	55	81	68	47	71	60
Human Waste (WWTP's, Package Plants and Septic) vs. all others						
Human waste	45	26	36	28	28	28
Other sources	55	74	64	72	72	72

Table 60. Percentage contributions to loading of the lake aggregated in three different ways. The 1982 data are most representative of a median year.

dependency of the nutrient yield from the source on the size of the workforce. It has already been noted that the nitrogen yield from Climax is exceptional, and this is shown very clearly in Tables 59 and 60. The nitrogen yield of Climax Molybdenum is greater than all four wastewater treatment plants combined.

Among the dispersed nonpoint sources, the background contribution, including everything that would be expected in the absence of human habitation, is considerably greater than all others combined. Under present conditions, it accounts for about half the total loading of nitrogen and phosphorus. Among the dispersed nonpoint sources related to human activity, septic systems stand out clearly as the highest contributor. All other sources account for contributions of a fraction of a percent to a few percent each. Although no single dispersed source related to human activity is of outstanding importance except for septic systems, the aggregate effect of the dispersed sources is significant.

Table 60 shows that the natural background and the sources associated with human presence share the phosphorus loading almost equally for phosphorus, and for nitrogen are only slightly skewed toward the man-made sources. In general Table 60 shows that, under the present land-use conditions, in a median year hydrologically, both the phosphorus and nitrogen loading of the lake can be considered to be divided approximately into four quarters. The first two of these quarters are taken up by the natural sources. The third quarter is taken up by sources associated directly with human waste (wastewater treatment plants, package plants, septic systems), and the fourth

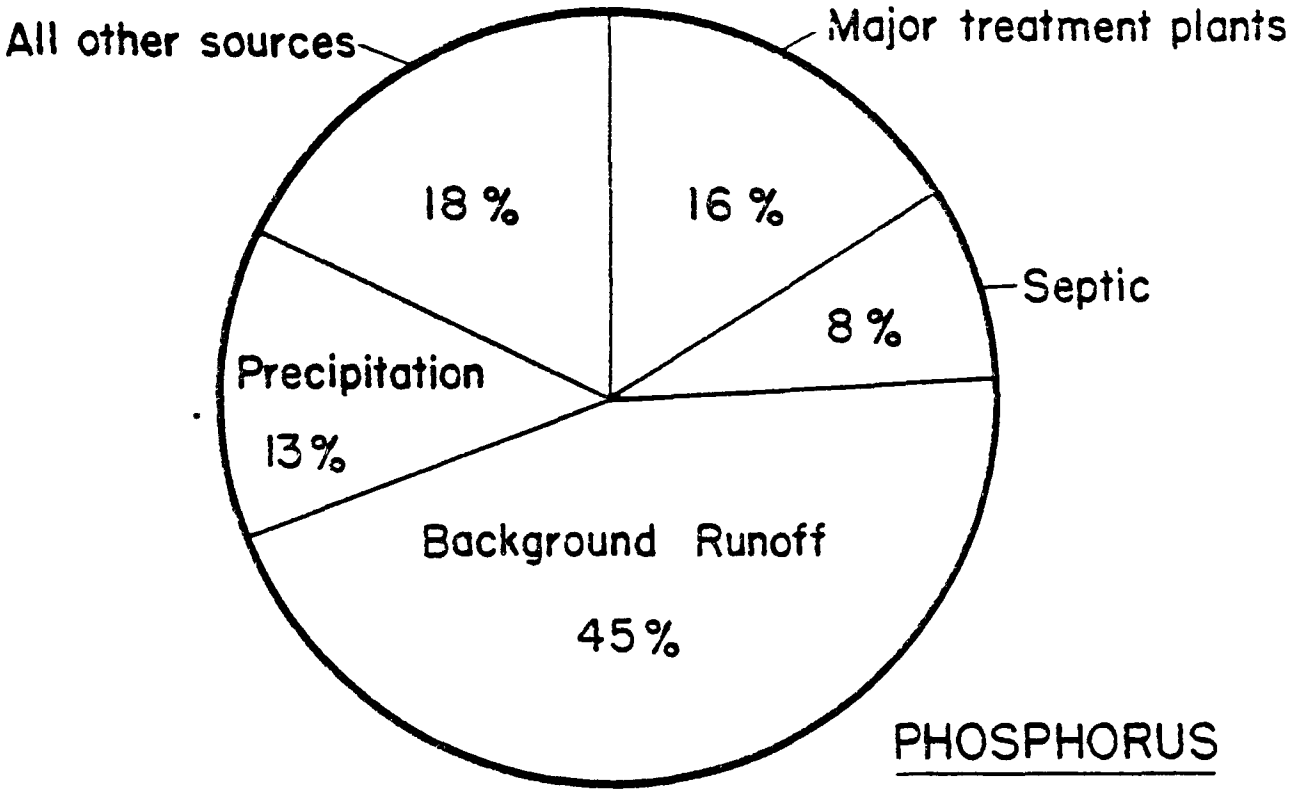
quarter is accounted for by human activity not related to the processing of human waste. In this last category we have a wide assortment of mechanisms including earth disturbance, change in vegetative cover, fertilization, and others.

Figure 56 gives a visual impression of the nutrient contributions from various sources. The 1982 data are shown in the figure because these are most representative of the median condition at the present time.

Possibilities for the Reduction of Nutrient Loading

The foregoing analysis allows us to suggest a number of ways the nutrient load reaching the lake might be reduced. The maximum imaginable reduction under the current land use conditions would be about 25%, assuming virtually no limit on financial investment and that no water is diverted away from the lake. Approximately 50% of the present loading (i.e., the natural background) is completely uncontrollable and roughly half of the remaining portion would probably be uncontrollable even with maximum investment. Thus the following alternatives in aggregate probably represent no more than 25% of the total present loading.

A. Major point sources. There are four major wastewater treatment plants in the drainage. We have assumed up to this point that all four plants will operate much as did the Breckenridge, Frisco, and Snake River plants in 1981 and 1982. Since the Copper Mountain Plant was changing over to tertiary treatment during this interval, we take the other three plants as a better indication of the immediate



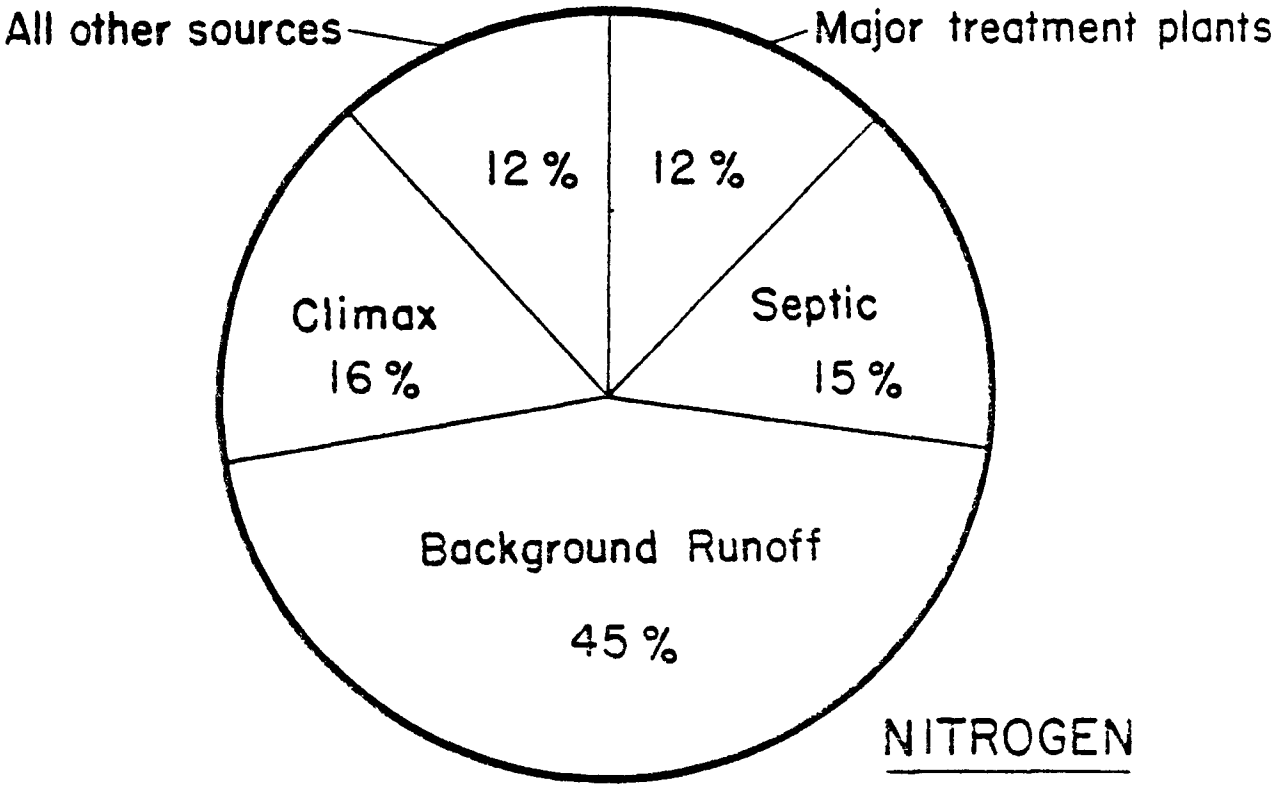


Figure 56. Percentage of loading due to various sources in 1982. All sources over 5% are shown separately.

future situation for all four wastewater treatment plants. The three operating tertiary plants added at least twice as much phosphorus to the lake in 1981 and 1982 as they might have if they had been able to operate continuously without equipment failures. Thus a reduction of something of the order of 8% in total phosphorus loading to the lake might be achieved if tertiary plant failures could be held at a level well below the level observed in 1981 and 1982, or if there were some kind of fail-safe system for each plant. Perhaps the disruptions of operation will be far less numerous when all of the plants have reached their capacity and thus are no longer under construction or expansion. If so, then a significant reduction of nutrient loading from this source may occur. At any rate, it may be important to pay closer attention to the occurrence and duration of plant failures and to embark upon a more vigorous program by which these can be minimized or avoided.

B. Climax Molybdenum. Climax Molybdenum is a significant source of phosphorus and behaves essentially like a point source. With the workforce at 1981 levels, the phosphorus contribution from this source will be roughly five percent. If tertiary treatment standards were applied to this source for phosphorus, the contribution could be reduced considerably, perhaps to as little as one percent of the total phosphorus loading of the lake.

C. Construction. The 1982 data on the Keystone construction in the Snake River bottom illustrates the potential exacerbation of normal phosphorus loading by heavy construction in or very near the river beds. Although such construction is not routine, it will probably

continue to occur at irregular intervals. Precautions to prevent the dispersion of disturbed earth and sediments from entering the stream channel might easily reduce this source by half.

D. Sewered residential areas. Runoff control in sewered residential areas, including especially the densely settled urbanized zones, could potentially reduce nutrient input to the lake by one percent if applied to all sewered areas. Although reductions of this magnitude may seem scarcely worth bothering with, it is obvious that any and all sources of nutrient reduction should be considered if additional development of the watershed is to be expected. The data show that nutrients associated with sewered residential areas, and particularly with urban areas, are transported specifically in the wettest years, and presumably at the time of greatest runoff. Technology for the ponding of runoff of this type is available and would be of interest in reduction of the nutrients from this source. However, removal of the trapped nutrients to prevent later transport to the lake would be essential if the control program were to be really successful.

E. Septic Systems. Septic systems are a major contributor of phosphorus. If septic systems could be replaced by sewer, the yield from these areas could probably be reduced by half (from about eight percent to about four percent of the total). Greater reduction than this is probably unrealistic for several reasons. First, a certain fraction of the contribution is associated with the presence of a residences and not with the presence of septic fields. Secondly, abandoned septic fields will presumably continue to yield phosphorus

and nitrogen for some time to come, and the decline in yield may not be immediate. Finally, tertiary treatment following the addition of sewer cannot be expected to remove all of the phosphorus.

Present and future yield of septic systems could probably be reduced significantly by proper maintenance of septic fields. This has not been a local government priority in the past but perhaps could be in the future as part of a nonpoint control program.

F. Interstate highway. Interstate highway contributed 2.8 percent of the total phosphorus yield in 1982. This is relatively high in view of the fact that interstate highway does not cover a very large percentage of the watershed and does not require the disposal of any human waste. In other words, interstate highway is a surprisingly rich source of phosphorus on a unit area basis. It is difficult to prescribe control measures for this source, since the mechanism by which phosphorus yield comes from interstate highways is not clear. The application of phosphorus for revegetation should be computed to see if it is potentially significant. Possibly some very localized sampling would be of use in showing more precisely the nature of the source connected with interstate highway. Containment systems may be feasible, particularly if the source is related to the transport of particulates from eroding areas or with heavy use of phosphorus-containing fertilizers (this may decline with time). Quite possibly this source could be cut in half, although the expense may be considerable if erosion control is involved.

G. Ski slopes. The yields from ski slopes accounted for slightly more than one percent of the phosphorus load in 1982. This is

obviously not a major increment, but it is still vulnerable to some control because of its localized nature. As with interstate highway, the prescription is difficult because exact mechanisms of increased nutrient yield are difficult to pinpoint and in an all probability are diverse. Fertilizer use, mechanical action of skiers on the slopes, and erosion aggravated by construction may all play a part. Ponding of runoff by a rationale similar to that which is used in urban areas is one possibility and might conceivably reduce the phosphorus yield to half its present amount.

One final possibility is the addition to Dillon of significant amounts of water lower in phosphorus than the average runoff currently entering the lake. This would in effect lower the mean phosphorus concentration of entering water and thus lower the equilibrium phosphorus concentrations in the lake. Although this alternative is almost certainly not feasible solely as a mechanism for the control of trophic status of Lake Dillon, it may be considered as a potential side benefit to various water diversion projects, but only if those projects involve the use of water having very low phosphorus concentrations.

The Dillon Clean Lakes Model

A major purpose of the present study was to develop a model that could assimilate information on changes in land use and from this predict changes in the lake, and especially those changes of aesthetic or economic importance. We have developed such a model from the information on the lake and watershed nutrient yields. We will refer to this model as the Dillon Clean Lakes Model.

The Dillon Clean Lakes Model consists of three parts, which are shown in diagrammatic form in Figure 57. These are the land use component, the trophic status component, and the effects component.

The Land-Use Component of the Model

The land-use component requires as input a matrix of land use information. The land-use matrix specifies the intensity of different types of use (on an area or population basis) for each of 19 watershed segments (Figure 58). The land-use component also requires the amount of gauged runoff (sum of the four U.S.G.S. gauges) for the year that is being modelled, the amount of water to be pumped or diverted from other watersheds, and the mean concentration of this pumped or diverted water on a month-by-month basis. The land-use component accepts as input the water yield from all watershed segments, but will compute these yields from the total gauged runoff if the yields are not supplied.

The land use computations are organized on a segment-by-segment basis. The nutrient export from each segment is computed from the water yield and the land-use matrix by use of the equations that were developed in the analysis of nutrient yields. The yields are summed

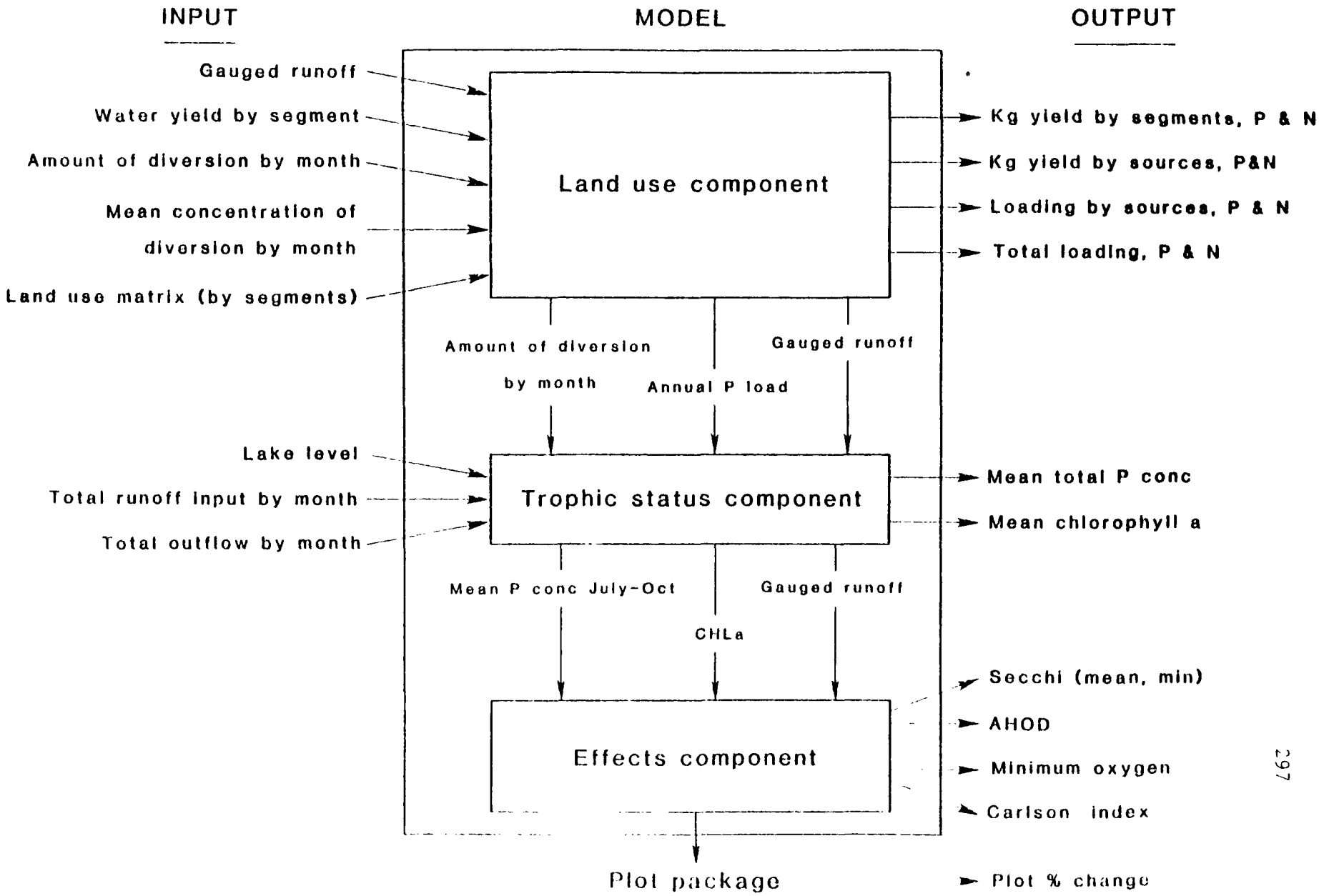


Figure 57. Structure of the Dillon Clean Lakes Model.

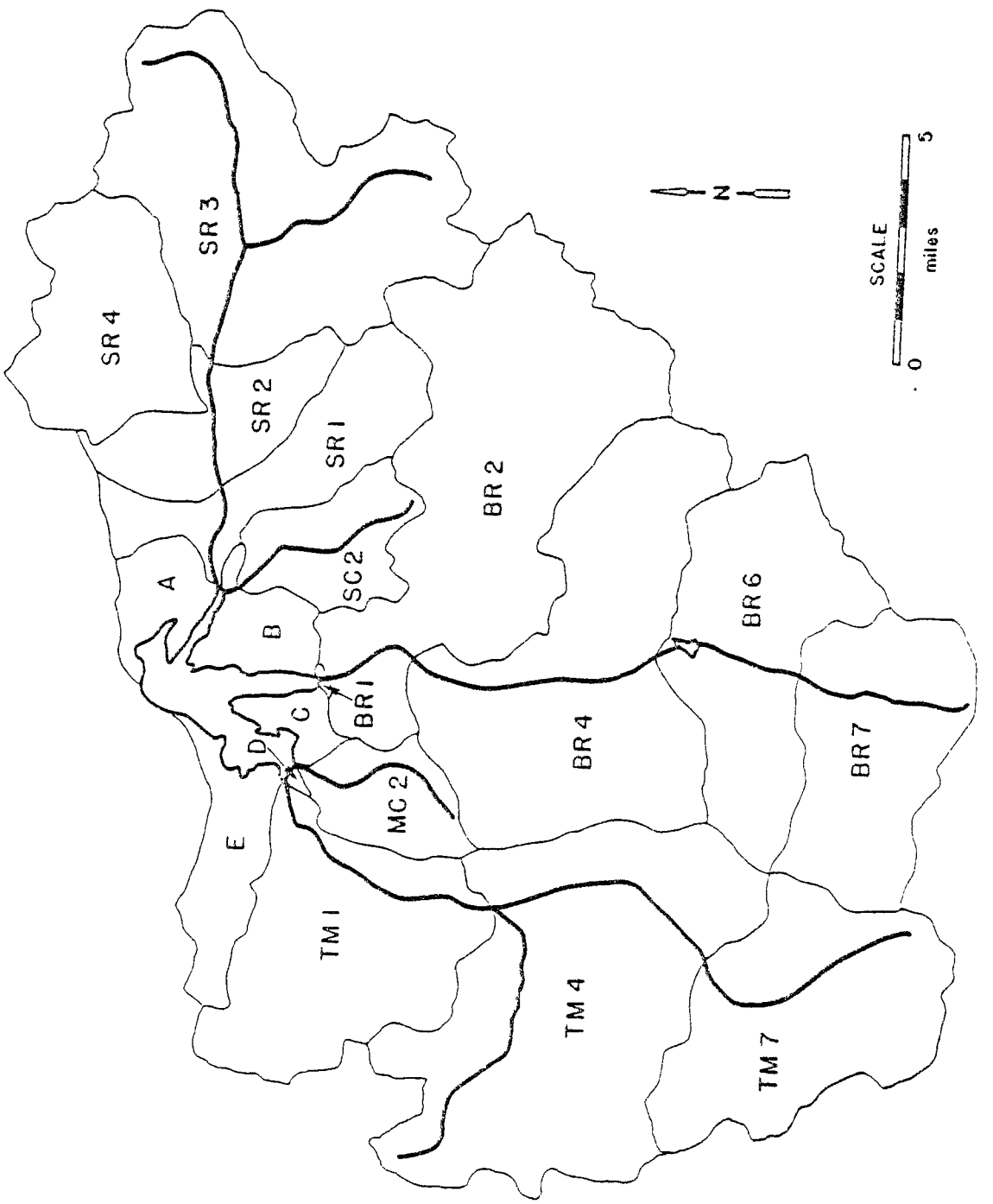


Figure 58. The 19 segments used in modelling.

for all segments and the inventory change function is applied in order to correct for changes in the phosphorus or nitrogen inventories in the valley bottoms according to the amount of runoff. Other sources are then added, including the three lakeside WWTP's, groundwater, precipitation, diversions, and irregular construction activities.

Output from the land use component of the Dillon Clean Lakes Model includes the nutrient yield by watershed segments and by source categories for both P and N. The total annual P and N reaching the lake is also part of the output.

The Trophic-Status Component of the Model

The land-use component of the Dillon Clean Lakes Model passes certain critical information to the trophic status component of the model. This information includes the monthly schedule of diversions or pumping, the concentrations of phosphorus and nitrogen on a monthly basis for pumped or diverted water, the annual phosphorus load by runoff and precipitation, and the amount of gauged runoff (i.e., sum of the four U.S.G.S. gauges). Additional input required at this point but not supplied by the land use component of the model includes the total surface water and ground water entering the lake on a monthly basis (this will be slightly larger than the gauged runoff), the monthly average lake level, and the amount of water leaving the lake on a monthly basis. The output includes the mean total phosphorus concentration of the top 15 m during the period July through October and the mean chlorophyll *a* in the top 5 m over the same interval.

Because the trophic status of lakes has been the subject of much

modelling effort, there is considerably more latitude in the choice of approaches for the trophic-status component of the Dillon Clean Lakes Model than for the land-use component. Whereas with the land-use component there is little basis for anything other than a simple empirical approach to modelling, the trophic-status component could be based in a more complex and fundamental way on current knowledge of biological, physical, and chemical processes in lakes. In discussing the alternative approaches, we can recognize two different classes of models: process models and empirical models.

A process model assumes that the critical biological, chemical, and physical processes that have an important bearing on the variable of interest (in this case, trophic status and its derivatives) are sufficiently well known to be represented by equations. For example, in a process model dealing with algal growth, logical functions to be incorporated in the model would include nutrient uptake, sedimentation of algae through the metalimnion, and decomposition of algae to release nutrients to the water. Since even the simplest natural planktonic systems are exceedingly complex at the elementary functional level, any such approach inevitably incorporates very large systems of coupled equations. An example is the Lake George Model, CLEANER (Park et al. 1979).

Process models have value as exploratory tools for basic ecosystem research, but their use for prediction, and thus for problem solving, is probably unwarranted and ill-advised at present. Opinions are by no means undivided on this subject, but a consensus seems to be developing about the practical restrictions on process models (e.g., Talling 1979,

Harris 1980, Hobbie and Tiwari 1977). The main reasons for the limitations on usefulness of process models is that the critical phenomena controlling biological variables in ecosystems are too poorly understood, too labile, and too complex to be treated in this manner. It is sometimes stated that the use of process models on biological systems only awaits the accumulation of further basic information. The implication is that the ecology of biological systems is in a relatively primitive state and, as it matures, will logically support complex models that produce satisfyingly exact answers to practical problems. This line of reasoning is probably misleading in several ways. First, the fund of information presently available is quite vast, especially for planktonic systems. It seems unlikely that any short-term improvement in this fund of information will drastically alter the feasibility of process models. Secondly, other disciplines in which the use of models to deal with large and complex systems has a long history have encountered much the same problems in the use of process models as ecosystem biology. Economics and meteorology are two such disciplines. The motivations for predicting either the economy or the weather well in advance are obviously enormous, and yet the success of models to accomplish this is very modest; available models deal best with short-range predictions that assume nothing unusual happens.

There exists in the study of ecosystems, and particularly of lakes, an alternative to the process model. Lakes are so numerous that empirical experience can be accumulated on their responses to widely varying conditions. This experience can serve as the basis for models. The philosophy of such models is entirely different from that of

process models. The dependent variable of interest is examined from the viewpoint of a very small number of master variables. While these master variables are chosen for their known direct and indirect relationships to the dependent variable, no attempt is made to dissect the numerous separate functions that contribute to the overall significance of such a master variable. This approach was first popularized for lakes by Vollenweider (1968), who related trophic status to phosphorus concentration and phosphorus concentration to phosphorus loading and mean depth.

Empirical models inevitably evolve, taking on more complexity. While the general validity of such a model can be established by sampling large numbers of systems and comparing the observed to the predicted values for the variables of interest, there will always be a certain amount of scatter in the agreement of observations and predictions. An obvious challenge then is to reduce the scatter by introduction of new concepts involving the same master variables or addition of other master variables. The model thus tends to become more complex, and in principle converges with the process model by building backward from the level of master variables to the level of processes. It is doubtful that this convergence will ever be completed, however, as there are severe practical limits to the reduction of scatter in the relationship between observed and predicted values. Even in their most complex forms, present modifications of the Vollenweider model are very distant in rationale and complexity from a process model such as that developed for Lake George.

For reasons outlined above, we have accepted as a philosophical basis for the modelling of the trophic status of Lake Dillon an empirical modelling approach. We have resolved to use a Vollenweider type of model, but to consider modifications as appropriate in the particular circumstances of Lake Dillon.

The first and most obvious improvement in the original Vollenweider model, which was based solely on phosphorus loading and mean depth, involved corrections for the flow of water through lakes. Obviously two lakes with equal phosphorus loadings and mean depths but very different flushing rates might easily differ in trophic status. It seems intuitively obvious that a certain amount of the phosphorus income should be discounted for losses through the outlet. This discounting concept has been considered by a large number of students of eutrophication (Welch 1980).

It is a short step from corrections based on flushing rate to corrections based on measured phosphorus retention coefficients. This approach has been taken by Larsen and Mercier (1976) and by Dillon and Rigler (1974). Since phosphorus retention and flushing rate interact, a number of formulations are possible, as described by Vollenweider et al. (1980). Underlying all of these approaches is a general mass balance equation by which a steady-state phosphorus concentration is defined in terms of phosphorus gains and phosphorus losses. In a lake such as Dillon, where flowthrough is substantial, there is little doubt of the need to incorporate some kind of correction for the hydraulic residence time, or the sedimentation coefficient for phosphorus, or both of these.

The approach to be taken here can be credited to Vollenweider (1969), although it is a specific derivative of general mass balance equations. Since this formulation was proposed by Vollenweider subsequent to his well-known original formulation, which did not take into account hydraulic residence time, we refer to it as the "modified" Vollenweider model. The critical equation is as follows:

$$C_p = L_p/\bar{z}(1/t_w + s) \qquad (16\text{-}1)$$

where C_p equals the phosphorus concentration of the lake, L_p is phosphorus loading of the lake, $\bar{z}$ is the mean lake depth, s is the phosphorus sedimentation coefficient and t_w is the residence time for water (Vollenweider et al. 1980). Vollenweider was able to reduce the number of parameters in the equation by assuming a sedimentation rate of 10-20 m per year. The simplified equation thus became

$$C_p = L_p t_w/\bar{z}(1 + t_w^{0.5}). \qquad (16\text{-}2)$$

Exact meaning of the model parameters is to some degree open to definition by the user and is not fully standardized. For Lake Dillon, we defined C_p as the mean phosphorus concentration of the upper 15 m of the water column from the interval between July 1 and October 30. The depth zone 0-15 m corresponds to the maximum extent of the growth zone. Since changes in transparency and chlorophyll a in the upper water column are ultimately of interest, this was considered to be the most advisable choice for the definition of C_p. However, had C_p been defined in any other reasonable way, major differences would not have resulted. It was considered advisable to define the time interval for determination of C_p in such a way that the spring runoff, which incorporates an exceptionally large fraction of heavy particulate

material that settles out rapidly, would not unduly influence C_p. This prevents the model from being unacceptably sensitive to variations between years in the amount of runoff.

The hydraulic residence time, t_w, can also be defined in a number of ways. In a lake that remains at steady state, all definitions are essentially equivalent. In the case of a lake such as Dillon, outflow and inflow are not always equal for a given year. For present purposes we use as our definition of t_w the ratio of lake volume to total inflow. This is partly necessitated by discrepancies between the U.S.G.S. and Denver Water Department figures on inflow and outflow, which have forced us to choose one or the other of these data sources. For consistency, we use the U.S.G.S. inflow figures.

Use of the simplified formula (eq. 2) for the modified Vollenweider model predicts P values that are reasonable but are consistently too high for the two years of record. Since overestimates occurred both in a low-water and a high-water year, the implication is that the simplified formula underestimates the sedimentation coefficients for Dillon, and that the performance of the model could be improved by use of a more correct sedimentation coefficient, which in turn necessitates use of the more complex version of the relationship (equation 16-1). For this reason, sedimentation velocities (v, where $v = \bar{z} \cdot s$) were approximated according to the procedure outlined by Higgins and Kim (1981). This procedure relies on knowledge of the inflow and outflow concentrations. The resulting estimates of sedimentation velocities for the two years were between 13 and 15 m per year. These sedimentation velocities were used to compute values of s

($\bar{z}$ was 23 m both years), which were in turn used in the Vollenweider equation (16-1). The results were better but still consistently somewhat high for both years, implying that the actual sedimentation coefficient was still being underestimated. Since the estimation procedure from Higgins and Kim assumes a steady-state condition, which is not ever literally applicable, the estimate is subject to unknown error. Furthermore, the outflow concentrations could not be determined directly for the water going through the Roberts Tunnel, which was not available for sampling, and this may have resulted in some estimation error.

Since the sedimentation coefficients were underestimated even after correction by the procedure outlined by Higgins and Kim, another approach was taken. Each year was treated independently and the equation was solved for s using the observed values of the other parameters for the year in question. The resulting estimates of sedimentation velocity were 24 m/yr for 1981 and 32 m/yr for 1982. The larger coefficient for 1982 is quite reasonable in view of the fact that heavy particulates are moved in much larger quantity in years of high runoff than in years of low runoff (see Figure 22a in the section on lake phosphorus). The question then arises how the sedimentation coefficients should be set for purposes of prediction in years of different runoff. The positive relationship between sedimentation coefficient and amount of runoff seems justified from the information at hand but the exact form of the relationship cannot be determined from two points. It is therefore assumed that the relationship is linear. Nonlinearity will result in a certain amount of error, but the

assumption of linearity is much better than assuming constancy of sedimentation, which is clearly incorrect.

A second element of the trophic status component of the model is the relationship between phosphorus and chlorophyll. A number of statistically derived relationships are presented in the literature. These show consistently a statistically significant trend toward higher summer chlorophyll a with higher total phosphorus in the mixed layer. There is, however, quite a bit of scatter and this has been the subject of considerable discussion and analysis.

The general equations that are available in the literature (e.g., Dillon and Rigler 1974, Carlson 1977, Jones and Bachmann 1976, Jones and Lee 1982, Lambou et al. 1982) do a uniformly poor job of predicting Lake Dillon chlorophyll a. Observations are 3-6 times higher than predicted values; Dillon produces considerably more chlorophyll per unit of P than most lakes. This is unusual but not unique, as shown by the scatter of points in the above-cited publications, but the explanation is not immediately obvious.

A recent study by Smith (1982) appears to explain the unusual phosphorus-chlorophyll relationship of Lake Dillon. Smith assembled data from several surveys in which both the nitrogen and phosphorus concentrations were known. He treated nitrogen as an added variable in studying the relationship between phosphorus and chlorophyll. He showed that nitrogen is a very important covariate, greatly increasing the amount of variance that can be explained even when phosphorus clearly limits growth rates. His conclusion is that increases in the nitrogen to phosphorus ratio boost the amount of chlorophyll that will

be produced for a given amount of phosphorus, even when phosphorus is limiting. The mechanism by which this occurs is unknown, but the reduction of variance by incorporation of nitrogen into the equations is impressive. It is worth noting that OECD (1982) found no added variance component explained by nitrogen. Smith and OECD did use different data bases; possibly Smith's was better suited for statistical illustration of the role of nitrogen. For present purposes, we accept Smith's conclusions as valid and attempt to apply them to Lake Dillon.

The lakes studied by Smith varied widely in the ratio of nitrogen to phosphorus. Smith developed equations for predicting chlorophyll in lakes of differing nitrogen to phosphorus ratios. His equations show that the effect of increasing the nitrogen to phosphorus ratio is to shift the line relating phosphorus to chlorophyll upward (Figure 59). At low to moderate TN:TP ratios, the upward shift occurs without changes in slope; the relationship thus takes the form of a family of parallel lines. In the highest category used by Smith, however (TN/TP > 35 by weight), the slope increases, and simultaneously becomes insensitive to further change in the TN/TP ratio. For lakes with low TN:TP ratios, it is essential to use N as a dependent variable as well as P. For lakes with a high TN:TP ratio, N can be dropped from the equation, but the coefficient on P will differ from that of lakes having low TN:TP ratios.

Lake Dillon, with TN/TP = 54 (weight basis), falls in the very highest category of nitrogen to phosphorus ratios among the lakes studied by Smith. In this highest category, as the influence of

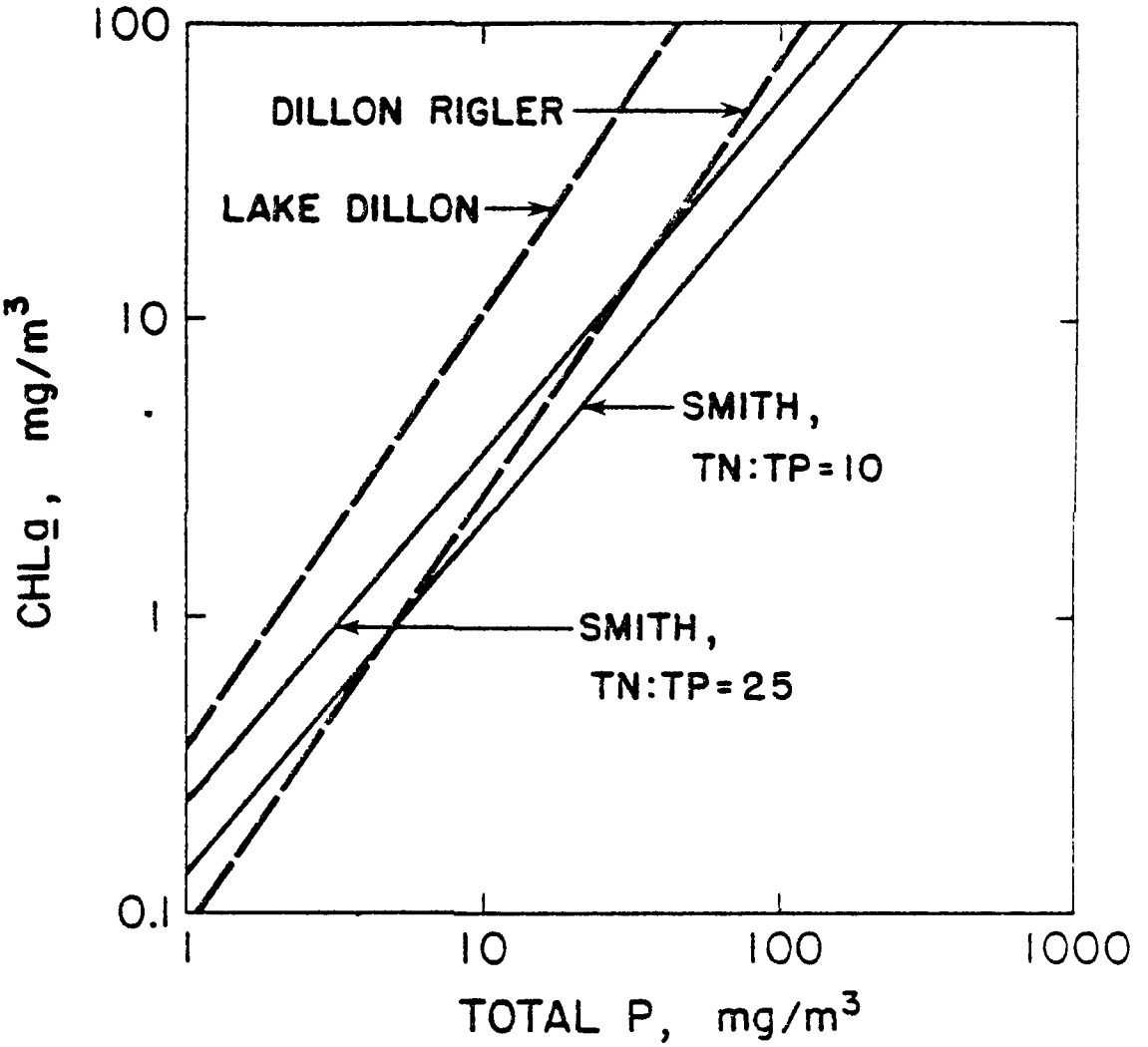


Figure 59. Log-log plot of total P and chlorophyll a, showing lines for two of Smith's (1982) lower TN:TP categories, the Dillon-Rigler line, and the Lake Dillon line.

further change in TN:TP becomes negligible, the coefficient on P is high and convergent with that of the Dillon-Rigler equation (1974). This convergence is expected, since the Dillon-Rigler equation was developed without inclusion of N. However, Smith's study shows that the intercept may continue to be affected as TN:TP is raised to high levels, even though the slope stabilizes. We thus approach the Dillon data with the following points in mind: (1) the TN:TP ratio is so high in Dillon that N need not be used in the equation, (2) the appropriate slope can be taken from the Dillon-Rigler equation, (3) the intercept should be determined uniquely for Dillon. We therefore set the slope of the phosphorus-chlorophyll equation equal to that of the Dillon-Rigler equation, but assume that the intercept will be determined by the nitrogen to phosphorus ratio. We determine the intercept empirically by solving for the intercept using the slope from the Dillon-Rigler equation and the observed chlorophyll a values. The resulting equation is as follows:

$$\log B = 1.449 \log(Cp) - 0.398 \qquad (3)$$

where B is chlorophyll a (July-October, ug/l, 0-5 m) and Cp is total P (0-15 m, July-October, ug/l). This equation performs well for both years (predictions will be given below).

Effects Component of the Model

Total phosphorus and chlorophyll a, as predicted by the trophic status component of the Lake Dillon Clean Lakes Model, have a number of correlates of economic and aesthetic importance. Since public attention and the attention of lake managers and users is likely to be

focussed on some of these correlates, it is useful to be able to make predictions for them.

Transparency is perhaps of greatest concern as a correlate of eutrophication. This is the aspect of eutrophication most obvious to the casual observer and most intimately connected with the aesthetic appeal of a lake. The most common measure of transparency is the secchi depth. Since a great deal of information is available from the literature on the relationship between secchi depth and chlorophyll a, and since there is a complete set of secchi depth measurements for Lake Dillon in 1981-1982, we have designed the effects component of the model to predict secchi depth. The prediction is based on chlorophyll a, which is the variable most directly responsible for changes in transparency as eutrophication occurs. The interval of interest is July 1 through October, corresponding with the post-runoff stratification season. As in other instances, we have excluded the month of June from the predictions because all predictions are complicated in this month by the dominance of inorganic particulate matter in the water column as a result of runoff. The observed mean secchi depth over the July-October interval in 1981 was 3.15 m and in 1982 it was 2.76 m.

The literature offers a number of equations relating secchi depth to chlorophyll a. There are two drawbacks to the use of these very general equations on Lake Dillon. First, there is a great deal of scatter around any of the lines taken from the literature, as would be expected for the comparison of very different kinds of lakes. Secondly, Lake Dillon is not located in the densest cluster of points

used for any of these equations, since its chlorophyll a values are still relatively low. Thus the general equations, which for transparency are particularly likely to go astray when extrapolated from lakes of low transparency to those of high transparency, may not be especially appropriate for Lake Dillon.

The relationship between secchi depth and chlorophyll a is always nonlinear. Transparency as measured by secchi depth decreases at a slower rate than chlorophyll increases. The fit of points to an empirical transparency/chlorophyll a relationship is typically rendered linear by log transformation. The log-transformed relationship thus has a slope and an intercept. Problems related to the determination of the slope and the intercept are different. The slope of the relationship should be much more universal than the intercept. The slope shows the increment of change in secchi depth for an increment of change in chlorophyll a, and lakes cannot be expected to differ widely with respect to this relationship. It is thought that the package size of chlorophyll a (i.e., the average cell size for phytoplankton) results in different values of chlorophyll-specific extinction (Harris 1978), and this may be responsible for a certain amount of variation in the slope. However, we have already shown that Lake Dillon is very average with respect to its chlorophyll-specific extinction (e_s = 0.015 m^2/mg chlorophyll a). We therefore believe it is justified to use a slope from one of the equations in the literature. We choose for this purpose the slope of the equation of Carlson (0.68), which is based on 147 lakes and has an r value of 0.93 (Carlson 1977). The National Eutrophication Survey data have a somewhat higher slope (0.86:

Lambou et al. 1982), but the observed values for Lake Dillon for the two years of record suggested much closer agreement with the Carlson coefficient than with the National Eutrophication Survey coefficient. We therefore adopt the Carlson coefficient.

The intercept of the log-transformed relationship between secchi depth and chlorophyll a is likely to vary much more extensively between lakes than the slope, particularly if one considers reservoirs and natural lakes together. The intercept for a given lake will be determined largely by a combination of nonchlorophyll factors contributing to the extinction of light. In very transparent lakes, this includes principally the extinction effect of pure water and of dissolved substances. In lakes that receive substantial amounts of inorganic particulates, extinction by particulates becomes a major determinant of the intercept.

It is feasible to customize the intercept for a prediction equation to a given lake by using a slope from a series of lakes and the observed values for the lake in question to solve for the intercept. We did this separately for 1981 and 1982 on the Lake Dillon data using the Carlson slope. The intercept for 1981 was 2.44 and the intercept for 1982 was 2.37. These intercepts were so similar that a common intercept of 2.4 was adopted for use with the Dillon data. This intercept is higher than the intercept of the Carlson equation (2.04), probably because the Carlson equation is based on natural lakes lacking the inorganic particulate input that one sees in Lake Dillon. The intercept is much closer to but slightly lower than the intercept for

the National Eutrophication Survey (2.56), which includes a large number of reservoirs.

The equation resulting from the above analyses is as follows:

$$\ln SD = 2.4 - 0.68 \ln B$$

where SD is secchi depth in meters and B is chlorophyll a in ug/l. The prediction of 1981 and 1982 chlorophyll from this equation is excellent (see below), suggesting that use of the Carlson slope is realistic.

A rough approximation can also be made of the minimum secchi depth from the average secchi depth. The minimum secchi depth in this case applies to the time of maximum phytoplankton standing stock. There is another seasonal minimum associated with runoff, which is not of concern here. In 1981 the ratio of minimum secchi depth to average secchi depth was 0.76 and in 1982 it was 0.65. As an approximation we take the ratio to be 0.70. There is no extensive literature on this subject and the prediction is therefore less secure than the other predictions mentioned up to this point. However, the use of this ratio will give some indication of the transparency of the lake at that particular time of the year when transparency problems may be most aggravated (July).

Oxygen depletion in deep water during the summer is also of concern in connection with eutrophication. In the section on oxygen, we presented an analysis of the areal hypolimnetic oxygen deficit and showed good agreement between the predicted AHOD and an equation developed by Cornett and Rigler (1980) based on secchi depth and mean depth. We use this equation in the effects component of the Dillon Clean Lakes Model, but we believe it is feasible to improve the

prediction by making a correction for the amount of water entering the lake in a given year. In 1981 the equation underpredicted the AHOD by 130 $mg/m^2/day$ (20%) and in 1982 it overpredicted by 6 $mg/m^2/day$ (1%). The difference in the two years is almost certainly due to the different amounts of oxygen transported by advection to deep water in the years of high runoff and low runoff. In a wet year, advective transport is greater and the areal hypolimnetic oxygen deficit is thus lower. We correct by making the assumption that runoff is proportional to the differential between the observed and actual hypolimnetic oxygen deficits. This results in a small correction that is added to the model.

Although the AHOD is of some direct interest, the minimum oxygen level in deep water is of more specific concern. There are no general equations for prediction of minimum oxygen content, since this prediction would involve the shape of the lake and the duration of the stratification period. However, for a given lake, there is obviously a very close relationship between the AHOD and the amount of oxygen depletion. The relationship should be very close to linear if the duration of the stratification period is constant. For Lake Dillon the initial oxygen can safely be assumed equal to 9.0 mg/l, the saturation concentration at spring overturn. The oxygen depletion is equal to the difference between 9.0 mg/l and the minimum oxygen observed toward the end of stratification. We deal specifically with a point 5 m over the bottom of the index station, and use the degree of depletion (mg/l) in each of the two years as the dependent variable and the AHOD ($mg/m^2/day$) as the independent variable. This yields a slope of

0.00698. This slope, when multiplied times the AHOD, gives the predicted amount of depletion. The amount of oxygen remaining is then the difference between the depletion and 9.0. The lower bound of remaining oxygen is set to zero, even for depletion more than severe enough to remove all oxygen.

Comparing Predictions and Observations for 1981 and 1982

The complete Dillon Clean Lakes Model was run on land use and runoff data for 1981 and for 1982. In these runs, the unusual features of 1981 or 1982 that would not be obvious from runoff or from land use data of the type accepted by the model for prediction purposes were ignored. For example, the output of wastewater treatment plants was computed on the basis of the number of persons served and the generalized equation described in the section on land use rather than the known yields from these plants. Similarly, the construction in the Snake River bottom near Keystone in 1982 was not treated in any special way in the model for 1982. Thus the agreement between predictions and observations for the two model runs gives an idea of the deviation between observed and expected that could be produced by randomly occurring events whose timing cannot be anticipated.

The predictions and observations are summarized in Table 61. For all variables the agreement is within 5%, except for the minimum deep-water oxygen, which is within 10%. The performance of the model on these years of known characteristics is thus excellent, despite the occurrence of a number of events whose timing cannot be anticipated by such a model.

	1981		1982	
	Predicted	Observed	Predicted	Observed
Total P, ug/l*	6.7	7.0	7.1	7.4
Chlorophyll *a*, ug/l*	6.3	6.7	6.9	7.3
Mean secchi, m*	3.2	3.2	3.0	2.8
Minimum secchi, m	2.2	2.4	2.1	1.8
AHOD, $mg/m^2/day$	706	710	590	630
Minimum deepwater O_2	4.1	4.4	4.9	4.6

* Mean, July-October

Table 61. Comparison of predictions from the Dillon Clean Lakes Model with observations for 1981 and 1982.

Predictions for Development Scenarios

The Dillon Clean Lakes Model was applied to 10 different scenarios provided by the Dillon Clean Lakes Steering Committee. We report below the results of modelling for each scenario. The 10 scenarios comprise five sets of assumptions for development and land use, each of which was applied to hydrologic assumptions for a dry year and for a wet year. Predictions in each case include the total loading of the lake, the source distribution of the loading, and the response of the lake to the predicted amount of loading.

The dry-year hydrologic conditions were always set identical to 1981. Since the runoff for 1981 was among the lowest for the period of record, 1981 provides a good estimate of the limiting hydrologic conditions for a dry year. For the wet year, we used the hydrologic conditions of 1982. 1982, although definitely above average, was not so extreme hydrologically as was 1981. Use of the hydrologic conditions of 1981 and 1982 is advantageous since the behavior of the lake in these two years under present land use is very well documented and can thus be compared easily to future years of similar hydrologic conditions but very different land uses. We have also used the observed lake level changes for 1981 and 1982 along with the hydrologic data for these two years. Thus we have assumed that the schedule of water release from the lake remains more or less as it is presently. Predictions might be different if the Denver Water Department began to release water in a very different way, e.g., by more excessive summer drawdown or more extended periods of drawdown.

The predictions of trophic indicators are translated into a trophic status category. For this, certain conventions are necessary. We use chlorophyll as the key trophic indicator, since algal biomass is the central concern of changing trophic status. We set limits on the trophic categories from a midrange of boundaries accepted by various sources. According to the review by Welch (1980), the oligotrophic boundary will be somewhere between 2 and 4 ug/l and the eutrophic boundary will be between 6-10 ug/l. We therefore set the mesotrophic range for present purposes at 3-8 ug/l chlorophyll a (this is also the range used by OECD 1982). One other convention is the adoption of an annual time frame for trophic classification. Trophic classes make no sense at intervals less than one year, so this is the maximum resolution possible. There is some merit to the argument that status should be based on a run of years, or on a year of median conditions. However, since we are interested in variation between years, we adopt for modelling purposes an annual interval.

Subsequent to the presentation of model runs is a more generalized graphical presentation based on the logic of the model.

Model Runs for Present Land Use Patterns

Before any of the scenario data were used, the model was run twice (wet year/dry year) with a land use input matrix representing present conditions. The purpose of these model runs was not to check the behavior of the model, which has already been discussed, but rather to define the loading and trophic condition of the lake under standardized conditions subsequent to 1982. These model runs differ

from the predictions discussed in previous sections in that Copper Mountain is assumed to have transferred completely to tertiary treatment, and to serve 2500 persons, slightly more than 1981-1982. Climax Molybdenum is set at the 1981 employment level (2650 persons). Also, no allowances are made for any of the special events that occurred in 1981 and 1982. Thus these two model runs for present conditions give for comparative purposes the output of the model under the assumption that present land use stabilizes (by resumption of activities at Climax and completion of the transfer of Copper Mountain to tertiary) but that no growth occurs. The results of the two model runs on stabilized present conditions are shown in Tables 62 and 63. As expected, the predicted loading and response of the lake are very similar to those actually observed in 1981 and 1982.

Scenarios 1A, 1B: Low Growth

These scenarios specify that wasteload allocations for point sources will be met, and that septic areas will be built out to 70%. This means that 70% of the lots will be occupied by dwellings used in the same way as present dwellings.

The point sources are at present meeting their wasteload allocations, although there is some random variation from one year to the next. We therefore leave constant at the 1981-1982 levels the number of persons served by the Breckenridge, Snake River, and Frisco wastewater treatment plants. We assume that Copper Mountain serves 2500 persons, which is slightly above the 1981-1982 level, and that complete tertiary treatment for Copper Mountain is put into effect. We

Scenario Conditions	Total P Load, kg	Percent WWTP*	Percent Septic	Percent Diversion	Percent Background**	Percent Other
Wet Year (similar to 1982)						
Current conditions***	4678	17.4	8.2	0	60.5	13.9
Low Growth (1A)	5049	16.2	14.7	0	56.1	13.0
Low Growth with Diversions (2A)	50048	1.7	1.5	89.9	6.7	0.2
High Growth (3A)	6389	12.8	31.9	0	44.4	10.9
High Growth, Best Controls (4A)	3657	22.3	0	0	77.5	0.2
High Growth, Cost-effective Controls (5A)	5001	16.3	20.4	0	56.6	6.7
Dry Year (similar to 1981)						
Current conditions***	2459	30.0	6.6	0	51.7	11.7
Low Growth (1B)	2618	28.3	12.0	0	48.5	11.2
Low Growth with Diversions (2B)	47618	1.6	0.7	94.5	2.6	0.6
High Growth (3B)	3190	23.2	27.2	0	39.9	9.7
High Growth, Best Controls (4B)	2021	36.6	0	0	62.9	0.5
High Growth, Cost-effective Controls (5B)	2596	28.5	16.7	0	49.0	5.8

* Includes package plants

** Includes background runoff and precipitation

*** Assumes Copper Mountain on tertiary, Climax at full employment 1981 levels.

Table 62. Results of the application of the Dillon Clean Lakes Model to scenario data.

Scenario Conditions	Lake Total P, ug/l	Chlorophyll ug/l	Mean Secchi, m	Minimum Secchi, m	Minimum O_2, mg/l	Trophic Status
Wet Year (similar to 1982)						
Current conditions***	7.3	7.1	2.9	2.0	4.8	Mesotrophic
Low Growth (1A)	7.9	8.0	2.7	1.9	4.6	Mesotrophic
Low Growth with Diversions (2A)	57.	141.	0.38	0.27	0.0	Eutrophic
High Growth (3A)	10.0	11.2	2.1	1.5	3.8	Eutrophic
High Growth, Best Controls (4A)	5.7	5.0	3.7	2.6	5.4	Mesotrophic
High Growth, Cost-effective Controls (5A)	7.8	7.9	2.7	1.9	4.6	Mesotrophic
Dry Year (similar to 1981)						
Current conditions***	6.1	5.4	3.5	2.4	4.3	Mesotrophic
Low Growth (1B)	6.4	6.0	3.3	2.3	4.2	Mesotrophic
Low Growth with Diversions (2B)	74.	206.	0.29	0.21	0.0	Eutrophic
High Growth (3B)	7.9	7.9	2.7	1.9	3.6	Mesotrophic
High Growth, Best Controls (4B)	5.0	4.1	4.2	4.8	4.8	Mesotrophic
High Growth, Cost-effective Controls (5B)	6.4	5.9	3.3	3.0	4.2	Mesotrophic

* Includes package plants
** Includes background runoff and precipitation
*** Assumes Copper Mountain on tertiary, Climax at full employment 1981 levels.

Table 63. Results of the application of the Dillon Clean Lakes Model to scenario data.

assume that Climax Molybdenum is operational with 2650 persons in the work force. The number of persons on septic systems represented by 70% buildout would be 3052 (full-time equivalent residents). These would be distributed geographically by proportions identical to septic distributions for 1981-1982. All other land uses remain unchanged.

The model was run first for a wet year (1A) and then for a dry year (1B). The results are reported in Tables 62 and 63. Figures 60 and 61 give a graphical representation of the degree of change as compared with present conditions. In the wet year, the predicted loading is higher by slightly less than 10% as a result of low growth. This causes an increase of chlorophyll to 8 ug/l. As explained previously, we have taken the range of chlorophyll values between 3 ug/l and 8 ug/l as indicating a mesotrophic condition. Thus in a wet year the low growth scenario pushes the chlorophyll a exactly to the upper limit of the mesotrophic span in a wet year. There is an accompanying decrease in transparency and a decrease in the minimum oxygen at the bottom of the lake.

For the dry year, the degree of change in total loading is smaller. Furthermore, the lake is much more toward the middle of the mesotrophic range for chlorophyll and other trophic indicators during a dry year, except for deepwater oxygen. Thus increasing the loading slightly to the degree specified by the low-growth scenarios does not bring the chlorophyll level so close to the upper boundary of the mesotrophic range. For a dry year, the lake remains solidly mesotrophic.

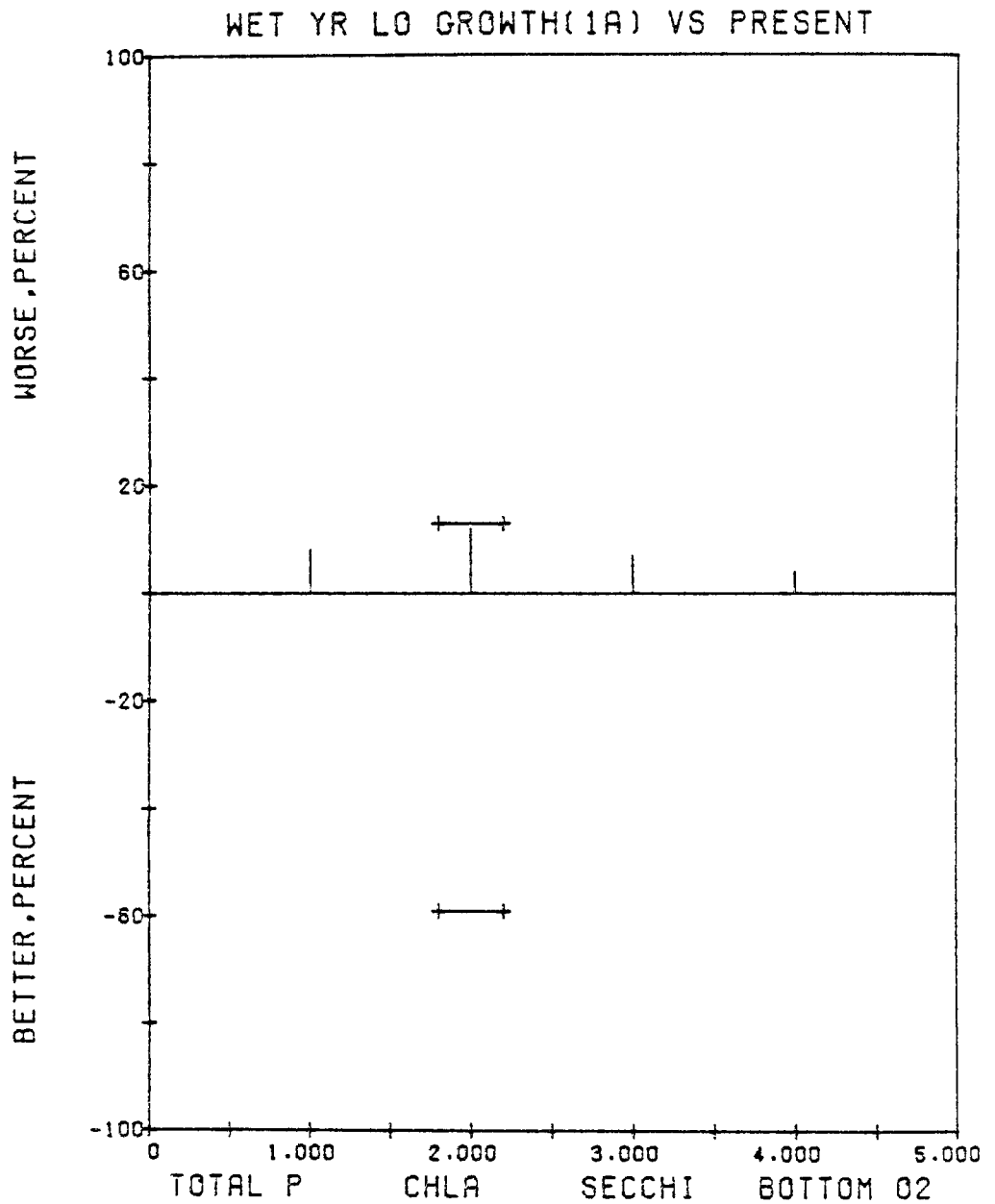


Figure 60. Graphical summary of Dillon Clean Lakes Model outputs for four key lake characteristics. Vertical lines show deviation from present conditions; horizontal lines with "+" mark the oligotrophic and eutrophic boundaries for chlorophyll a.

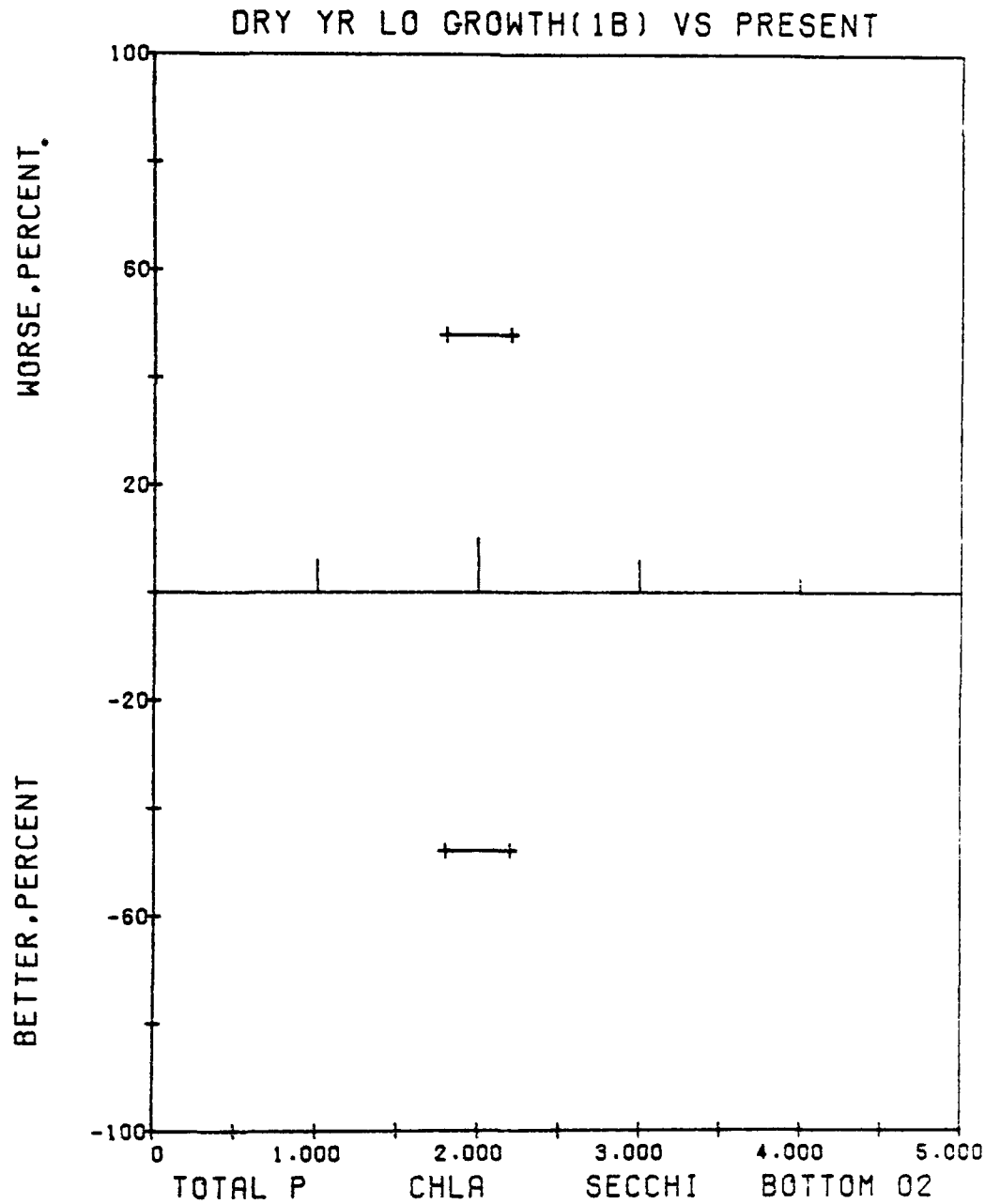


Figure 61. Graphical summary of Dillon Clean Lakes Model outputs for four key lake characteristics. Vertical lines show deviation from present conditions; horizontal lines with "+" mark the oligotrophic and eutrophic boundaries for chlorophyll a.

Scenarios 2A, 2B: Low Growth with Diversions

In these scenarios it is assumed that 10,000 acre-feet/year is diverted into Lake Dillon from Straight Creek and 183,000 acre-feet/year is diverted into Dillon by the Eagle-Colorado Project. The total phosphorus concentrations are set at 30 ug/l for Straight Creek and to 200 ug/l for Eagle-Colorado. The amounts of water diverted are assumed to apply to years of average or above-average moisture. For the dry-year run of the model (2B), the amounts of diversion are scaled down in proportion to the gauged runoff (sum of the four U.S.G.S. gauges in the watershed). The diverted water is assumed to enter the lake on a schedule identical to the Tenmile Creek hydrograph. All other conditions are set identical to scenarios 1A and 1B.

The results of the two model runs (2A, 2B) are summarized in Tables 62 and 63. Figures 62 and 63 give a graphical representation. The phosphorus loading coming in by way of diversion is roughly ten times the loading under present conditions without diversions. The lake shows drastic increases in total phosphorus concentrations and in chlorophyll levels. Complete oxygen depletion is predicted for the hypolimnion, and predicted secchi depth values are extremely low. The lake under these circumstances would be unequivocally upper eutrophic (sometimes called hypereutrophic). Nuisance blooms would be almost a certainty under these conditions.

The chlorophyll levels predicted for the diversion scenarios are so high that they probably exceed the theoretical maximum chlorophyll possible in freshwaters. About 300 mg/m^2 of chlorophyll *a* absorbs

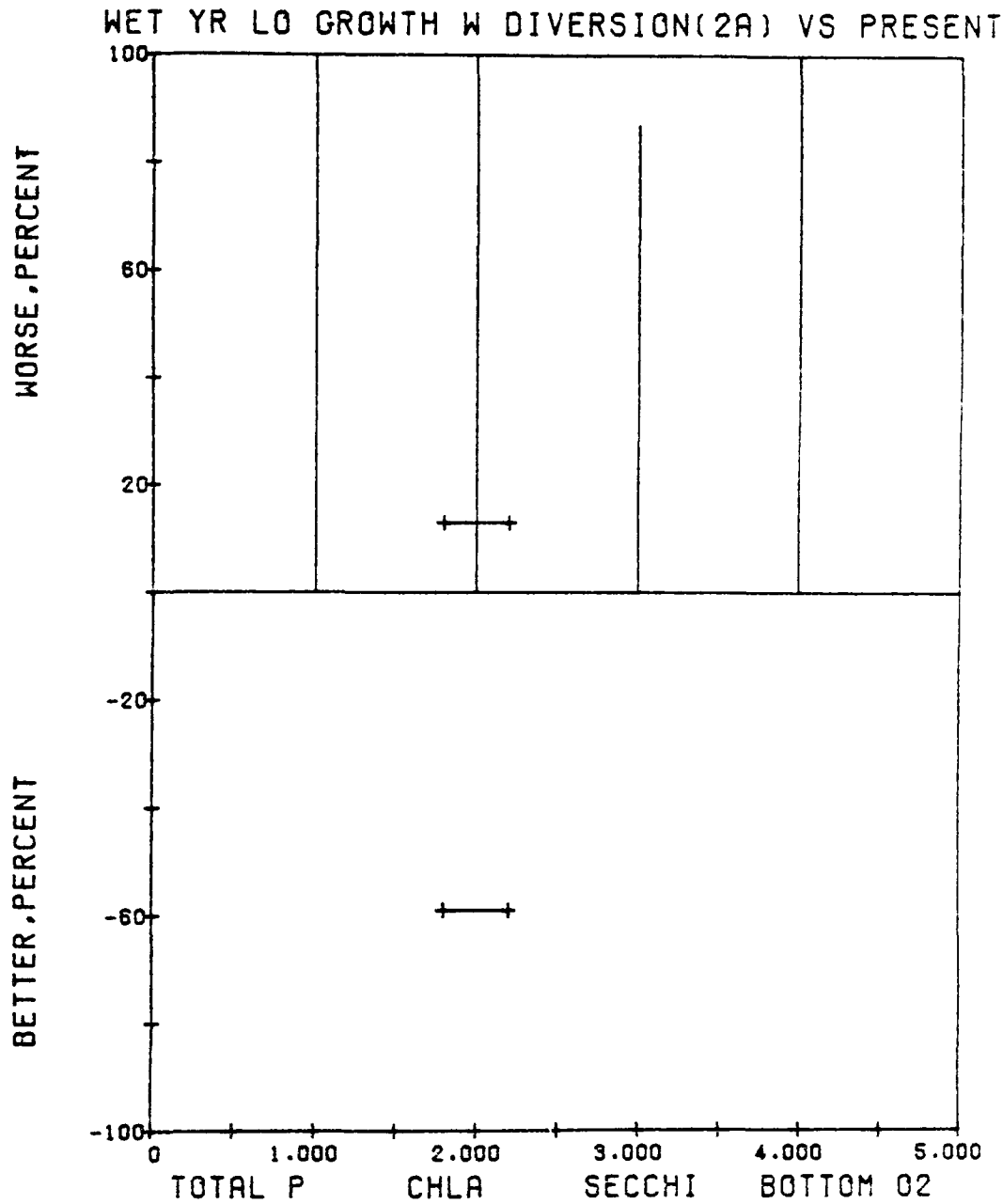


Figure 62. Graphical summary of Dillon Clean Lakes Model outputs for four key lake characteristics. Vertical lines show deviation from present conditions; horizontal lines with "+" mark the oligotrophic and eutrophic boundaries for chlorophyll a.

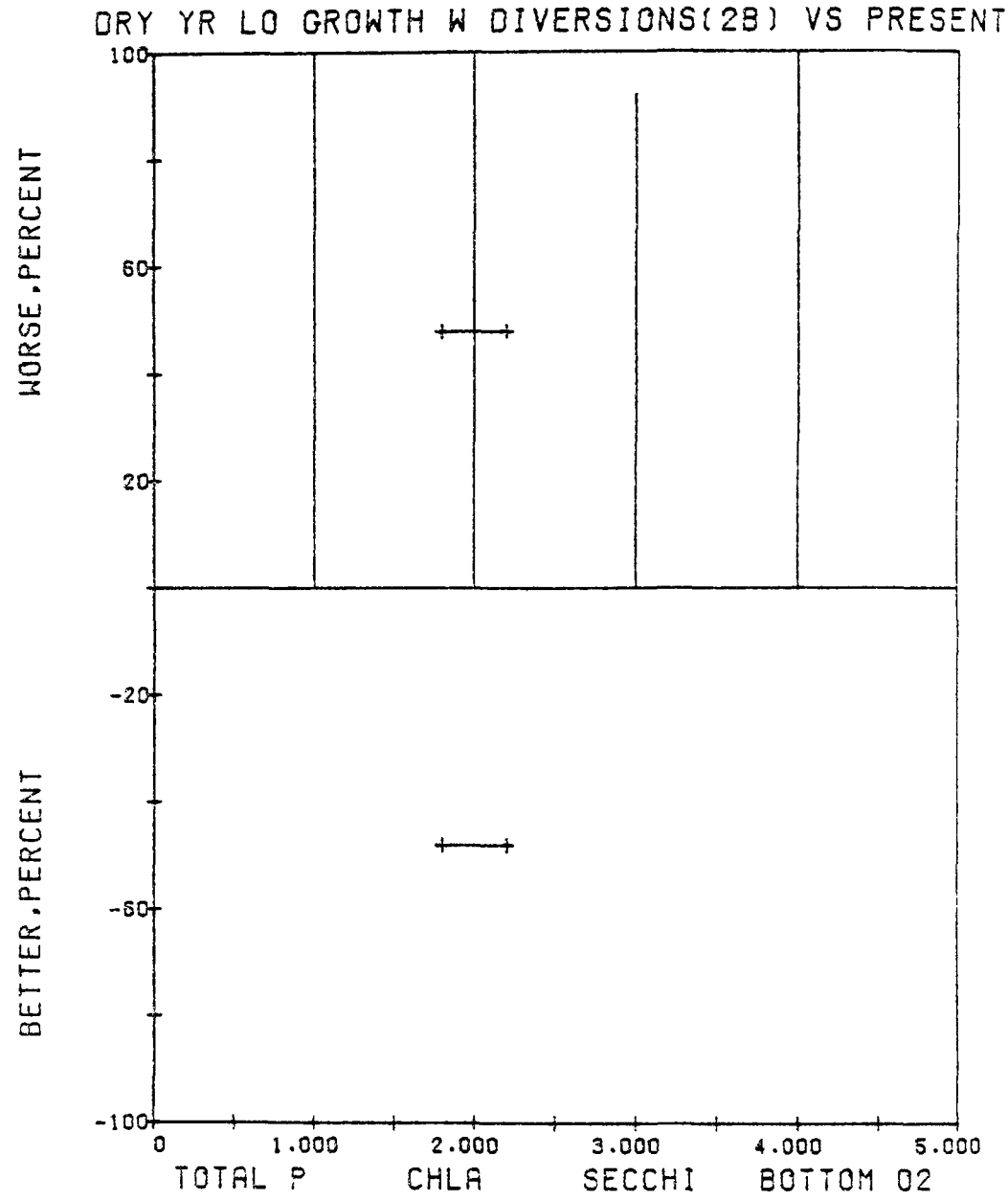


Figure 63. Graphical summary of Dillon Clean Lakes Model outputs for four key lake characteristics. Vertical lines show deviation from present conditions; horizontal lines with "+" mark the oligotrophic and eutrophic boundaries for chlorophyll a.

99% of the light (Margalef 1978, Talling 1982). At this point light, rather than nutrients, becomes limiting and further addition of nutrients does not boost algal biomass. In the upper 5 m of Lake Dillon, to which algal growth would be essentially confined under the low transparency conditions of these scenarios, we could expect to find at a maximum about 60 ug/l of chlorophyll a, which would account for 300 mg/m^2 summed up over the 5-m mixed layer. Thus the chlorophyll predictions, which are in excess of 100 ug/l, should not be taken too literally. They merely indicate that the lake reaches the theoretical maximum biomass possible under the light limitation conditions prevailing in Lake Dillon. The exact concentrations are of course of little interest once they reach such high levels.

The lake under the conditions of scenarios 2A and 2B would have the qualities of a sewage lagoon. The lake would thus differ drastically from its present appearance and would be very different biologically because of the extreme depletion of oxygen in deep water and very large crops of algae.

The phosphorus concentrations in the diversion water are the cause of the extreme trophic response of the lake to diversion. Accurate knowledge of the total phosphorus concentrations in the diversion water is absolutely essential if accurate predictions are to be made. For scenarios 2A and 2B, by far the greatest uncertainty of the prediction has to do with the phosphorus values that are fed into the model for the diversion water. The concentrations were supplied by Tom Elmore and were derived by him from existing measurements not taken in connection with the Dillon Clean Lakes Study. Although the present

modelling exercise certainly shows that the worst-case situation for phosphorus in diversion water would result in a very undesirable degradation of the lake, sound predictions beyond this will require extensive and careful study of total phosphorus in diversion water.

Scenarios 3A, 3B: High Growth

These scenarios assume that current wasteload allocations are met for the wastewater treatment plants. This implies essentially no expansion beyond scenarios 1A, 1B of the number of persons served by any one of the four wastewater treatment plants. The number of persons served by these plants is therefore allowed to remain as for scenarios 1A, 1B and Climax Molybdenum is assumed to operate with a workforce of 2650 persons. The high-growth scenarios assume that septic systems will serve a peak population of 37,000 persons. This corresponds to a time-weighted average of 8409 full-time equivalent residents. It is assumed that this number of persons is distributed within the watershed according to the same pattern of present septic system users. However, it should be noted that this number of persons cannot be housed on the existing developments served by septic, even if all of these developments are built out to 100%. Thus these scenarios assume that additional developments will be built and served by septic systems, or that the present developments will have a higher density of housing, or some combination of these possibilities

Tables 62 and 63 and Figures 64 and 65 summarize the results of the model runs for high growth in a wet year (3A) and in a dry year (3B). For the wet year, the scenarios indicate an increase in total

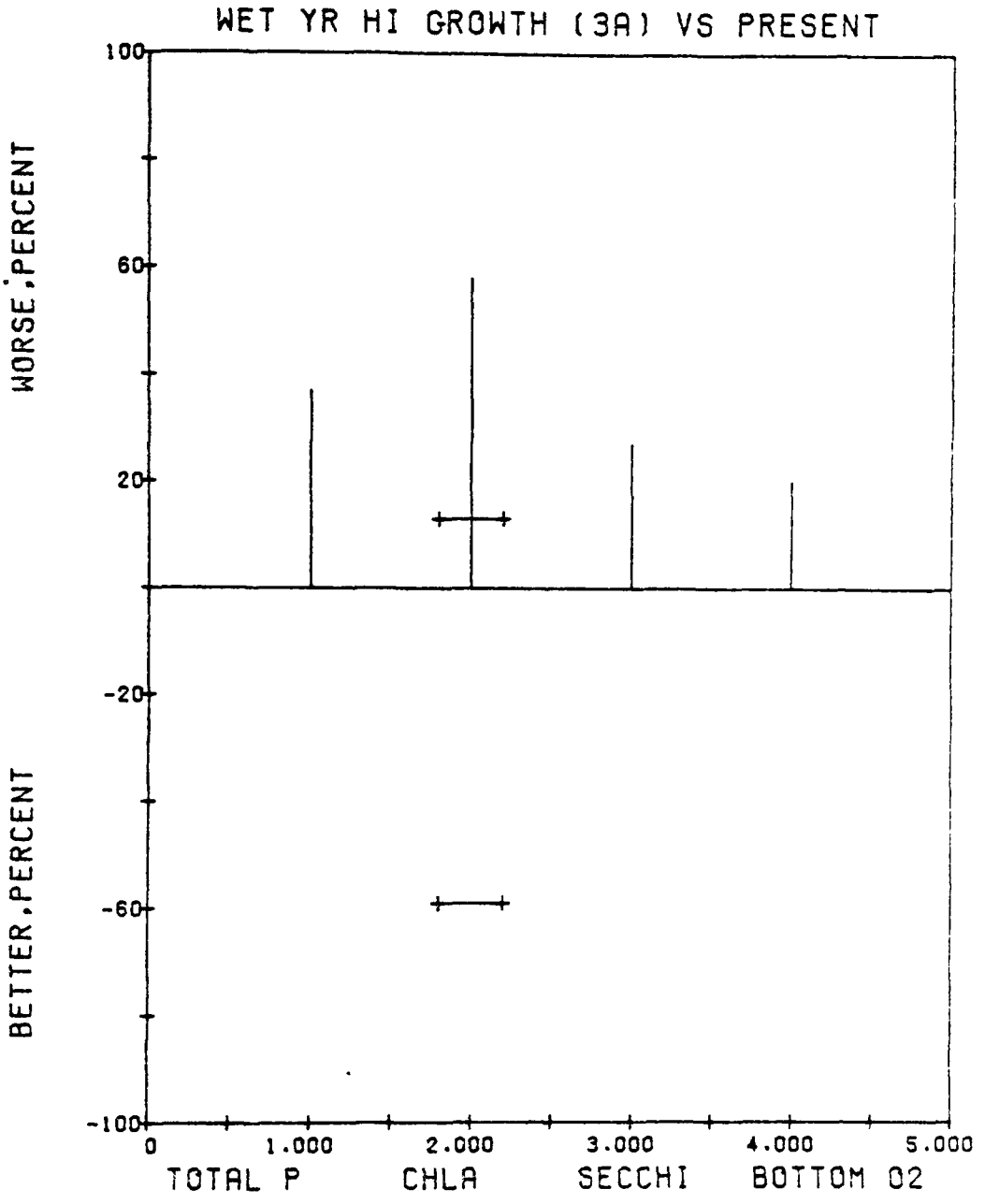


Figure 64. Graphical summary of Dillon Clean Lakes Model outputs for four key lake characteristics. Vertical lines show deviation from present conditions; horizontal lines with "+" mark the oligotrophic and eutrophic boundaries for chlorophyll a.

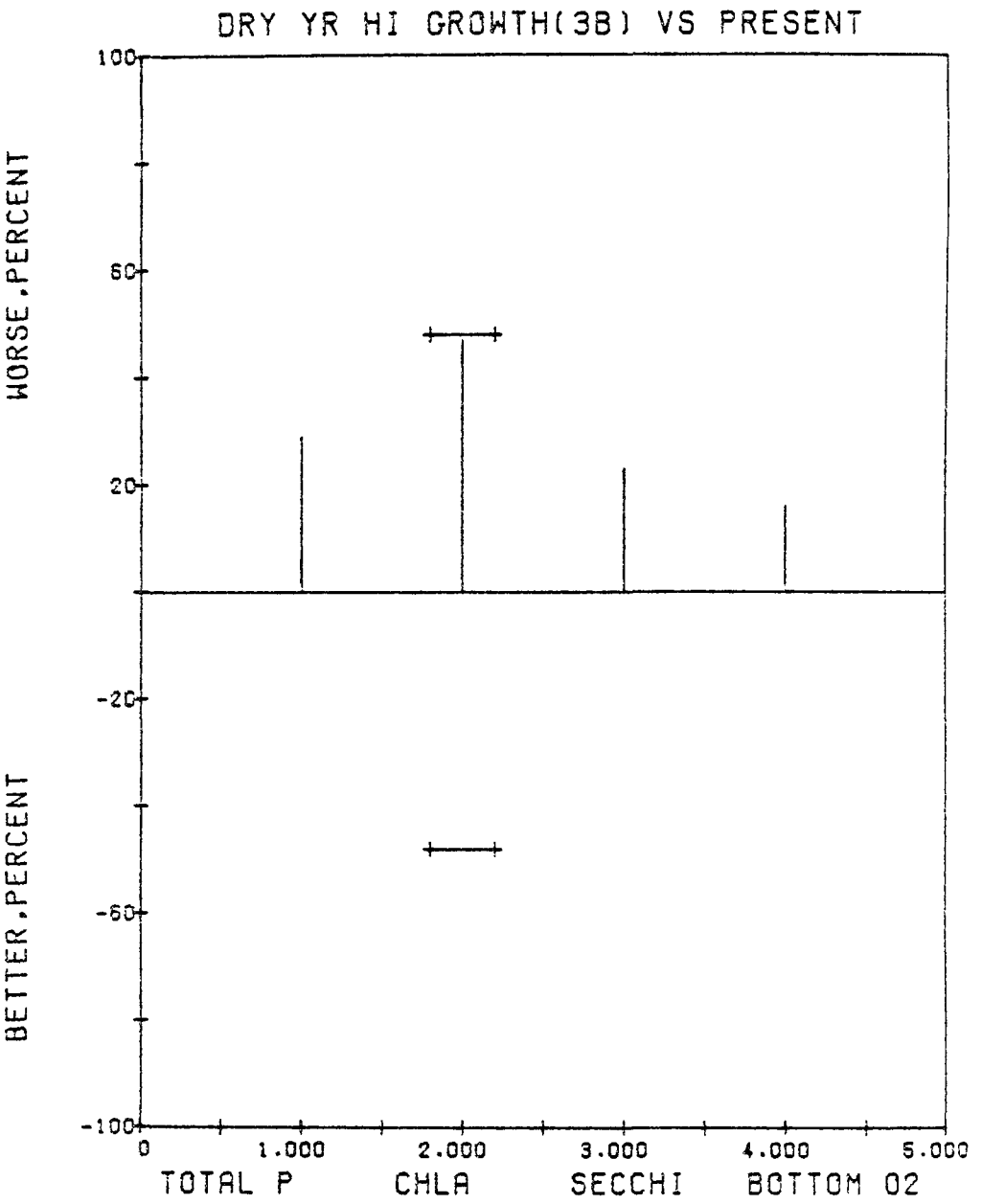


Figure 65. Graphical summary of Dillon Clean Lakes Model outputs for four key lake characteristics. Vertical lines show deviation from present conditions; horizontal lines with "+" mark the oligotrophic and eutrophic boundaries for chlorophyll a.

wet-year loading of approximately 50% over present conditions. There is a drastic increase in the percentage of loading caused by septic systems, as would be expected. Chlorophyll is predicted at 11.2 ug/l, which is well over the boundary from mesotrophic to eutrophic. Transparencies and minimum oxygen in deep water similarly reflect a higher trophic status. Although not nearly so drastic as the changes of scenarios 2A and 2B, the changes of scenario 3A with respect to the present condition would almost certainly be significant enough for a casual observer to notice.

For the dry year, the effects of high growth are of smaller magnitude but still significant. Chlorophyll is predicted to fall at 7.9 ug/l, just below the upper mesotrophic boundary at 8 ug/l. Thus the lake could probably be classified in the very dryest of years as mesotrophic, but at all other times would be eutrophic, and in wet years would be especially so.

Scenarios 4A, 4B: High Growth with State of the Art Controls

This scenario specifies that wasteload allocations are met by point sources, and that no diversions enter the lake. All sources except the wastewater treatment plants, package plants, groundwater, precipitation, and background are set to 0. Thus it is assumed that all nonpoint sources are completely eliminated.

The results of the model runs are shown in Tables 62 and 63 for a wet year (4A) and for a dry year (4B). Figures 66 and 67 show the corresponding graphs. In either a wet or a dry year, the model predicts significant improvement of the trophic status of the lake,

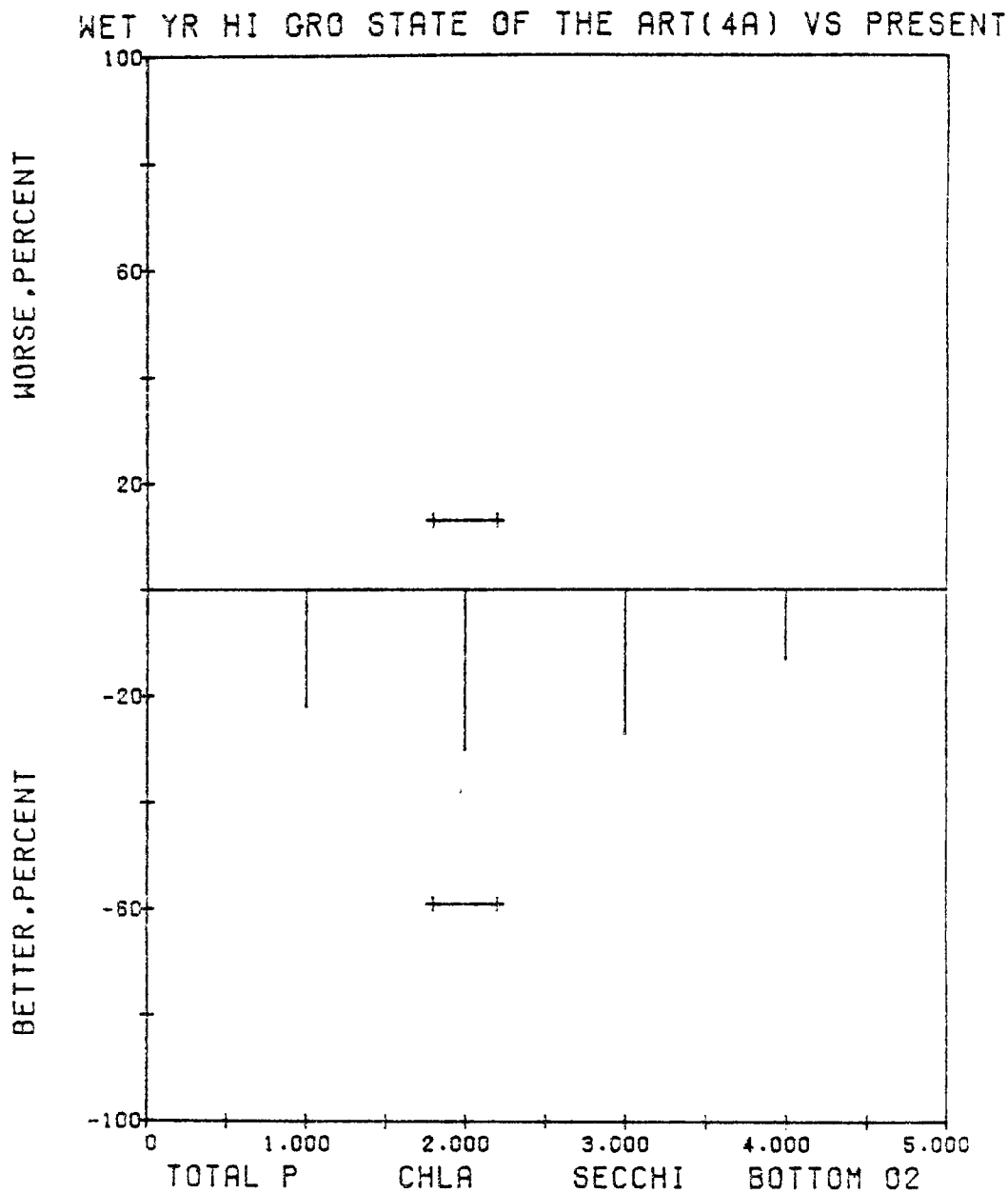


Figure 66. Graphical summary of Dillon Clean Lakes Model outputs for four key lake characteristics. Vertical lines show deviation from present conditions; horizontal lines with "+" mark the oligotrophic and eutrophic boundaries for chlorophyll a.

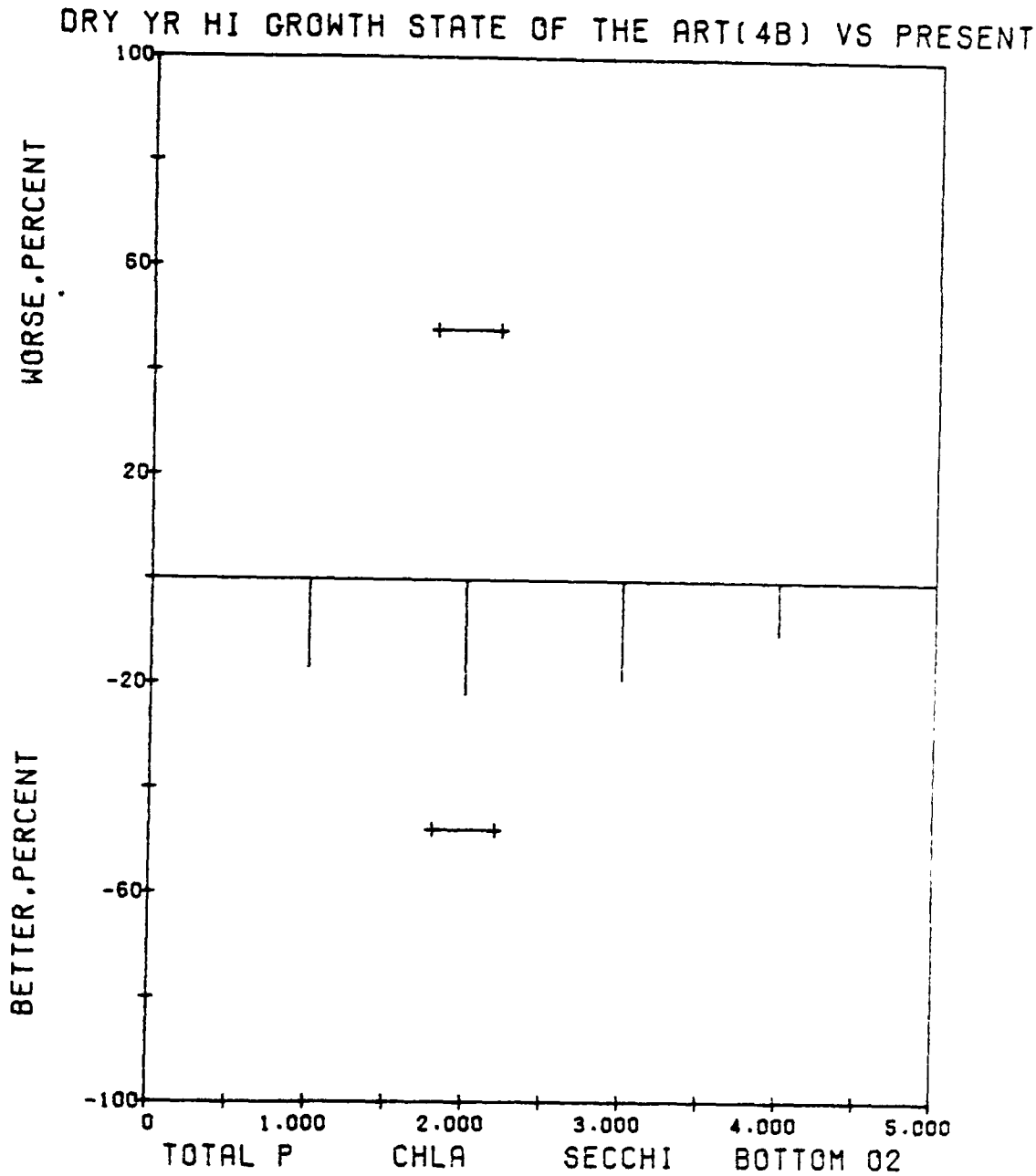


Figure 67. Graphical summary of Dillon Clean Lakes Model outputs for four key lake characteristics. Vertical lines show deviation from present conditions; horizontal lines with "+" mark the oligotrophic and eutrophic boundaries for chlorophyll a.

despite high growth. Chlorophyll values show lower mesotrophic status for the lake under this scenario. Transparency, bottom oxygen, and other indicators are correspondingly improved.

As attractive as this scenario seems, it is almost certainly unrealistic. It implies that there would be no yield whatever above background for septic systems, for Climax Molybdenum, and for the many distributed sources of nutrients associated with human activity. The high-growth conditions imply that about 8000 persons (annual average) will be served by septic systems. This is essential because the wasteload allocations cannot be held constant if additional population is to be served on sewer. Thus a substantial septic contribution or a change in the wasteload allocation seems inevitable as an accompaniment to high growth, but scenarios 4A and 4B allow neither. The results should therefore be viewed as representing an unrealistically high degree of control over phosphorus loading, regardless of financial considerations.

Scenarios 5A, 5B: High Growth with Cost-Effective Controls

Scenarios 5A and 5B are intended as counterparts to scenarios 4A and 4B, but with controls set at levels that are more likely to be realized. The assumptions are identical to those of scenario 4A and 4B, except that all nonpoint sources, which were set to 0 in 4A and 4B, are allowed to assume half their uncontrolled value. The results appear in Table 62 and 63 and Figures 68 and 69.

For a wet year, the model predicts loading that is slightly more than 5% above present conditions. This causes a slight increase in the

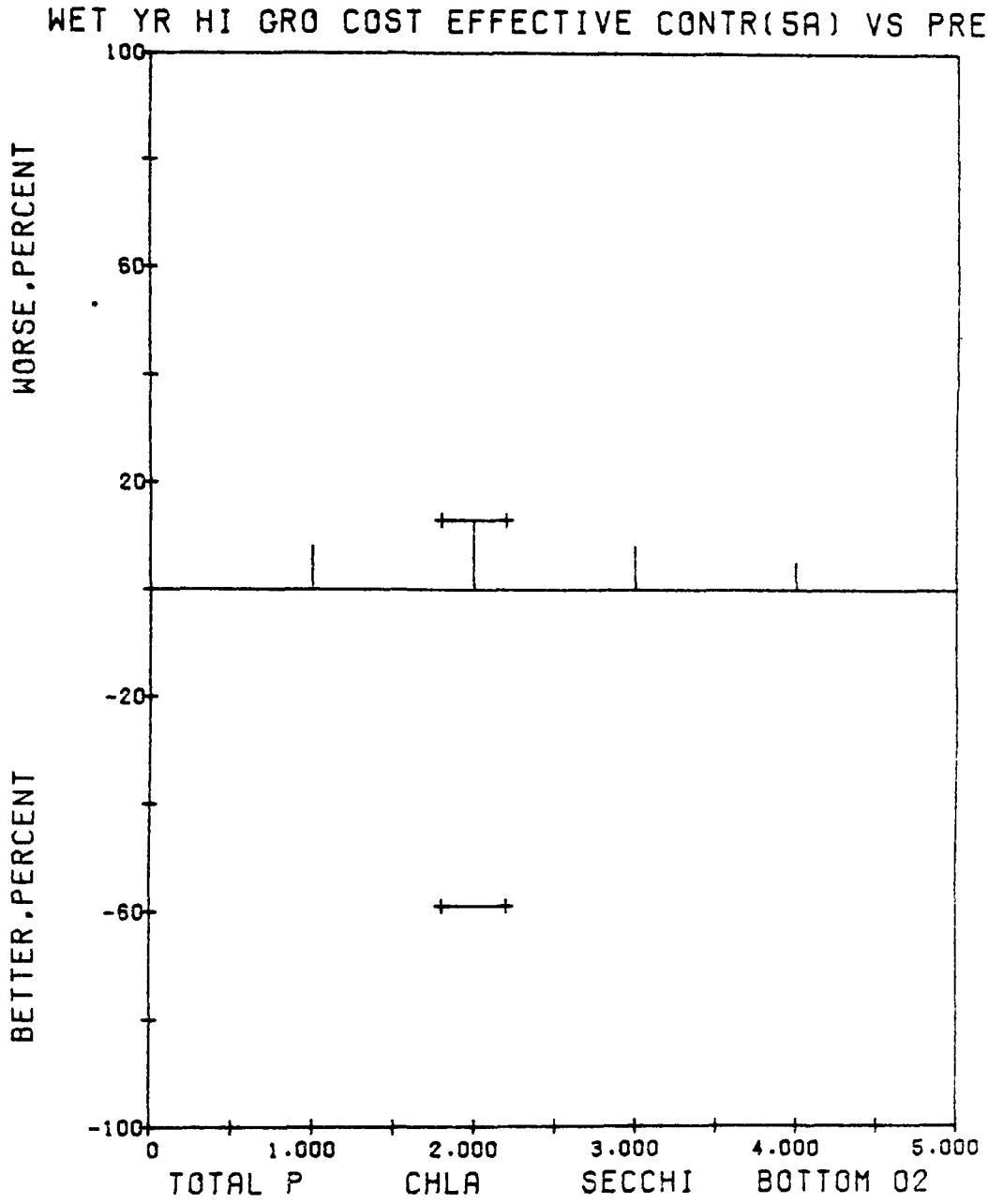


Figure 68. Graphical summary of Dillon Clean Lakes Model outputs for four key lake characteristics. Vertical lines show deviation from present conditions; horizontal lines with "+" mark the oligotrophic and eutrophic boundaries for chlorophyll a.

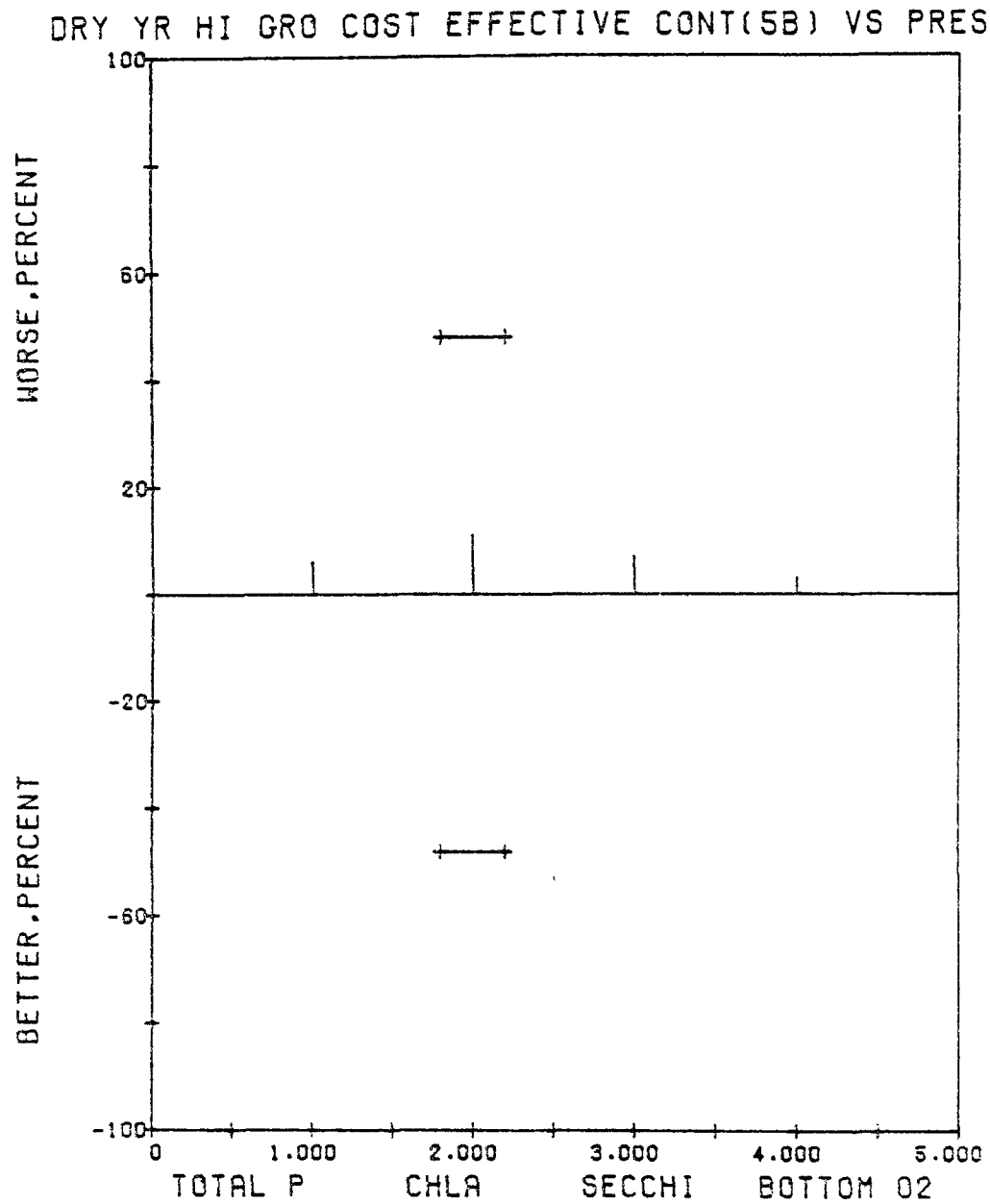


Figure 69. Graphical summary of Dillon Clean Lakes Model outputs for four key lake characteristics. Vertical lines show deviation from present conditions; horizontal lines with "+" mark the oligotrophic and eutrophic boundaries for chlorophyll a.

expected chlorophyll level, which reaches 7.9, very close to the upper mesotrophic boundary at 8.0. For a dry year, there would also be a slight increase in chlorophyll above the present conditions, but the dry-year increase would not bring the chlorophyll so close to the upper mesotrophic boundary. Thus the model shows that, if the controls could be accomplished, the lake would in most years fall in the mesotrophic range.

Although scenarios 5A and 5B are labelled as cost-effective, they may actually imply intolerable expenditures. For example, the scenarios assume that phosphorus yield from the expected 8000 persons on septic systems is somehow reduced by 50%, and they assume the same for Climax Molybdenum. They also assume that such diffuse sources as the interstate highway and ski slopes can be treated in such a manner as to reduce yields by one-half. Thus the realism of this scenario is open to some skepticism, but it certainly approaches the reality of maximum-investment controls much more closely than scenarios 4A and 4B.

Generalized Graphical Predictions

A large number of relationships can be developed in graphical form using the logic that is inherent in the model. These graphical relationships are necessarily less specific in their predictive capabilities than actual model runs, since simplifying assumptions are required for a generalized graph that would not be required for a model run. Nevertheless, generalized graphs give an overall impression of the sensitivity of the lake and its watershed to development of

different kinds. We have developed for a graphical presentation three kinds of relationships: 1) the relationship between total phosphorus loading and chlorophyll, 2) the relationship between number of persons on sewer and total phosphorus loading, and 3) the relationship between the number of persons on septic system and total phosphorus loading.

Figure 70 shows the general relationship between total phosphorus loading and chlorophyll concentration. As in the modelling exercises, the chlorophyll concentration referred to in the graph is the average for the 0-5 meter layer over the post-runoff stratification interval. The relationship of chlorophyll to phosphorus loading is depicted for two different sets of conditions. The first set of conditions is based on the 1981 study year. In the preparation of this curve it was assumed that the runoff, water level, and lake volume changes were identical to those of 1981. The second curve is based on 1982 and involves similar assumptions for that year. Since 1981 was a very dry year and 1982 was a year of above-average but not extreme wetness, a median year would fall between the two curves but nearer to the 1982 curve than the 1981 curve. Changes in lake operation (water levels) for either of these conditions would change the positions of the curves, but the curves would not move dramatically unless there were very significant departures from the present mode of lake operation.

Figure 70 indicates along with the two curves the lower limit of loading for the two hydrologic conditions. These represent the natural background for the two years depicted in the graph. Also indicated in Figure 70 are the observed loadings in 1981 and 1982 and the trophic

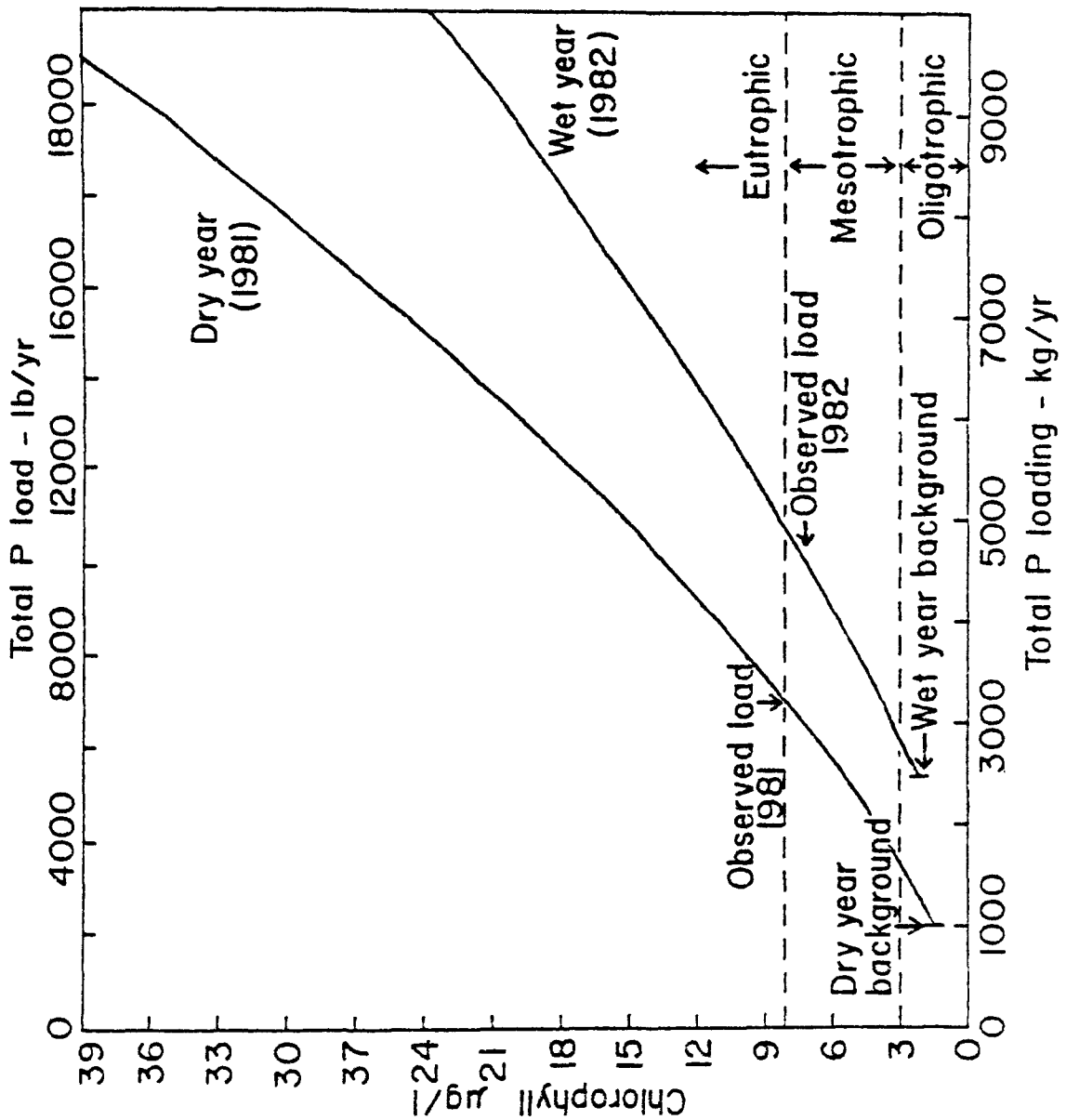


Figure 70. Average chlorophyll for the post-runoff stratification season (0-5 m) in Lake Dillon as a function of total phosphorus loading under hydrologic conditions similar to 1981 and 1982.

classifications that correspond to various chlorophyll levels. It is clear from the graph, as it was from the modelling exercises, that minor increases in loading either in a dry or wet year will push the lake from the upper mesotrophic category where it presently lies into the eutrophic category.

Figure 71 shows the relationship of number of persons on sewer to the phosphorus loading in kilograms per year. Number of persons is annual full-time equivalent (1 full time equivalent equals ca. 100 gallons per day effluent). As with Figure 70, the graph includes two curves, one of which is based on conditions identical to 1981 and the second on conditions identical to 1982. The intersection of each of these lines with the y-axis represents the phosphorus loading to the lake if there were no contributions whatever from sewered areas. This can be considered a sort of non-sewer background for the watershed. Contributions of sewered areas above this background are then added on the basis of contributions of sewered areas through point and nonpoint sources. Point sources are assumed to consist of waste water treatment plants utilizing tertiary treatment. Tertiary treatment is assumed to function at the same overall efficiency as observed in 1981-1982. In addition to this point-source contribution, the non-point source contribution corresponding to urban area on sewer is also added to get the total per capita contribution from sewered areas. Since the nonpoint contribution is dependent on amount of runoff, the nonpoint contributions for years of different wetness are different, but in both cases nonpoint contributions were below 10% of the point source contributions.

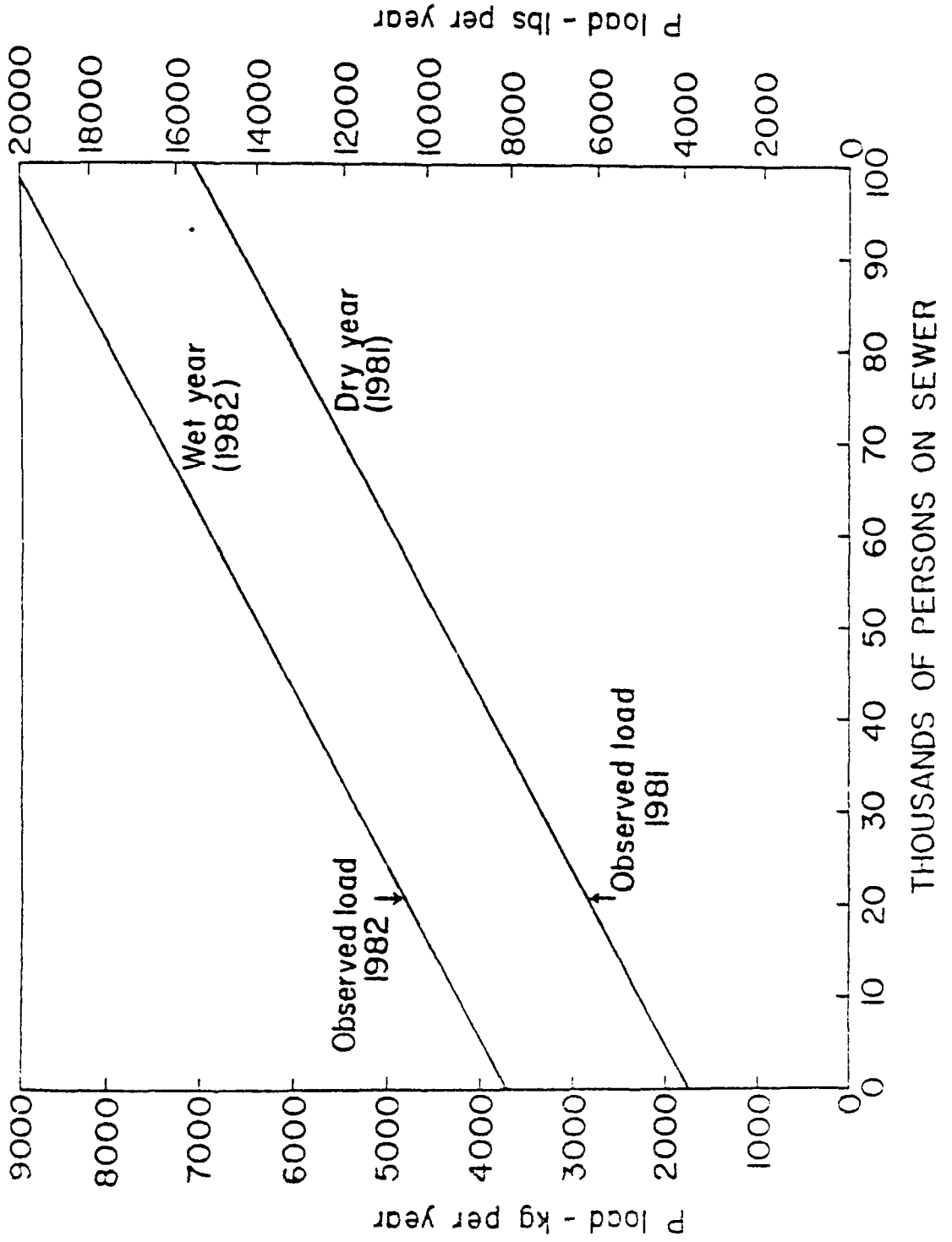


Figure 71. Relationship of number of persons on sewer to the total phosphorus loading of Lake Dillon if other loading sources are held constant. The projections include both the point and non-point contributions associated with sewered areas and assume that all point sources receive tertiary treatment of efficiency equal to that observed during 1981-1982.

The observed loadings for 1981 and 1982 are indicated on Figure 71 for reference purposes. The observed loadings correspond to populations on sewer of about 20,000. The actual number of persons on sewer in 1981 and 1982 was actually closer to 15,000. The observed values correspond to a slightly higher population because part of the population served by sewer (Copper Mountain) was not receiving tertiary treatment in 1981-1982. In the projections, it is assumed that all sewered areas are receiving tertiary treatment, which should be true for future years.

Figure 72 has a similar rationale to that of Figure 71 except that phosphorus loading is related to contributions from septic systems rather than sewers. Lines representing dry year and wet year conditions differ in slope because of the higher yield of phosphorus from septic sources when runoff is high. In addition, the inventory change function dictates that a larger percentage of the phosphorus originating in septic areas will reach the lake in a wet year than in a dry year. It should be noted that the loading scale (y-axis) covers three times the range of the scale in Figure 71 for sewer. Observed loadings in 1981 and 1982 are only a small fraction of the total potential loading if the population on septic were drastically increased.

Overview of Scenario Studies

The modelling and graphic studies indicate that Lake Dillon will move into the eutrophic category if diversion water rich in phosphorus is added in quantity to the lake or if high growth occurs without the adoption of nonpoint source controls or other measures not now in

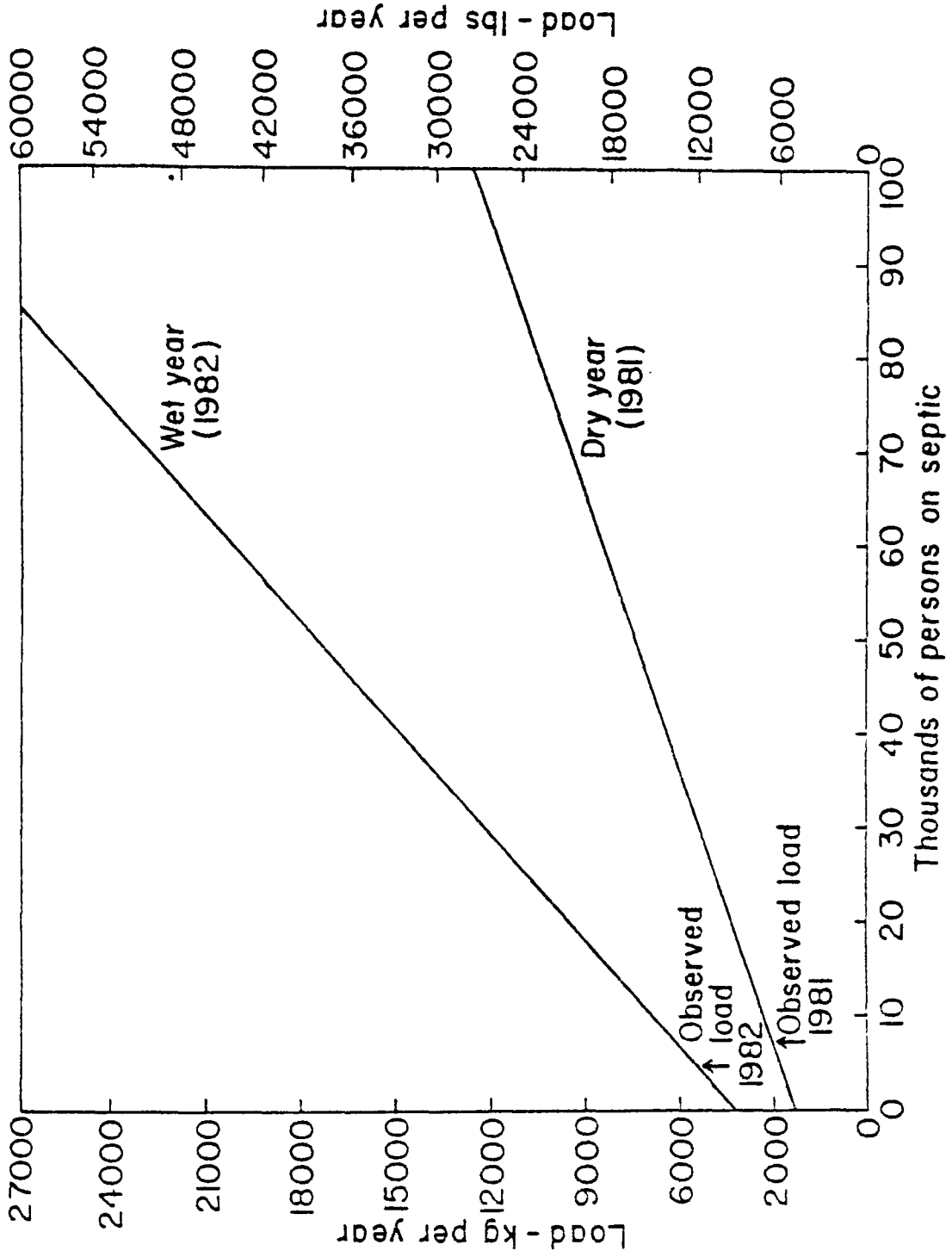


Figure 72. Relationship of populations served by septic systems to total phosphorus loading assuming other loadings are held constant at their 1981 or 1982 levels. The projections are based on average watershed runoff for 1981 and 1982.

practice to reduce the phosphorus loading of the lake. Under low growth or high growth with additional controls, the condition of the lake could be held within the mesotrophic range, but might suffer some trophic degradation or remain more or less the same, depending on the exact conditions.

Literature Cited

Alabaster, J.S., and R. Lloyd. 1980. Water Quality Criteria for Freshwater Fish. Butterworth, Boston. 297 pp.

American Public Health Assn. 1976. Standard Methods for the Examination of Water and Wastewater. American Public Health Assn. Washington, D.C. 1193 pp.

Bailey-Watts, A.E. 1978. A nine year study of the phytoplankton of the eutrophic and non-stratifying Loch Leven (Kinross, Scotland). J. Ecol. 66: 741-771.

Bendschneider, K., and R.J. Robinson. 1952. A new spectrophotometric method for the determination of nitrate in sea water. J. Mar. Res. 11: 87-96.

Bormann, F.H., G.E. Likens and J.S. Eaton. 1969. Biotic regulation of particulate and solution losses from a forest ecosystem. BioScience 19: 600-610.

Bormann, F.H. and G.E. Likens. 1979. Pattern and Process in a Forested Ecosystem. Springer-Verlag, New York. 253 pp.

Carlson, R.E. 1977. A trophic state index for lakes. Limnol. Oceanogr. 22: 361-368.

Cornett, J.R. and F.H. Rigler. 1980. Areal hypolimnetic oxygen deficit: An empirical test of the model. Limnol. Oceanogr. 25: 672-679.

Dillon, P.J. and F.H. Rigler. 1974. The phosphorus-chlorophyll relationship in lakes. Limnol. Oceanogr. 19: 767-73.

Edmondson, W.T. 1972. Nutrients and phytoplankton in Lake Washington. pp. 172-188 In G.E. Likens (ed.) Nutrients and Eutrophication. American Society of Limnology and Oceanography Special Symposia, Vol. 1.

Fee, E.J. 1976. The vertical and seasonal distribution of chlorophyll in lakes of the Experimental Lakes Area, northwestern Ontario: implications for primary production estimates. Limnol. Oceanogr. 21: 767-783.

Fogg, G.E., W.D.P. Stewart, P. Fay, and A.E. Walsby. 1973. The Bluegreen Algae. Academic Press, N.Y. 459 pp.

Fogg, G.E. 1975. Algal Cultures and Phytoplankton Ecology. Univ. of Wisconsin Press, Madison. 175 pp.

Ganf, G.G. 1975. Photosynthetic production and irradiance-photosynthesis relationships of the photoplankton from a shallow equatorial lake (Lake George, Uganda). Oecologia 18: 165-183.

Gibson, C.E. 1978. Field and laboratory observations on the temporal and spatial variation of carbohydrate content in planktonic blue-green algae in Lough Neagh, Northern Ireland. J. Ecol. 66: 97-115.

Glooschenko, W.A., J.E. Moore, and R.A. Vollenweider. 1973. Chlorophyll a distribution in Lake Huron and its relationship to primary productivity. Great Lakes Res. 16: 40-49.

Goldman, C.R. 1974. Eutrophication of Lake Tahoe emphasizing water quality. EPA Report 660/3-74-034. 408 pp.

Golterman, H.L. (ed.). 1976. Interactions between Sediments and Fresh Water. Junk, The Hague. 473 pp.

Grant, M.C. and W.M. Lewis, Jr. 1982. Precipitation chemistry in the Colorado Rockies. Tellus 34: 74-88.

Grasshoff, K. 1976. Methods of Seawater Analysis. Verlag Chemie, Weinheim. 317 pp.

Harris, G.P. 1978. Photosynthesis, productivity, and growth: the physiological ecology phytoplankton. Arch. Hydrobiol. Beih. Ergebn. Limnol. 10: 1-171.

Harris, G.P. 1980. Temporal and spatial scales in phytoplankton ecology. Can. J. Fish. Aquat. Sci. 37: 877-900.

Heaney, S.I., and J.F. Talling. 1980. Dynamic aspects of dinoflagellate distribution patterns in a small productive lake. J. Ecol. 68: 75-94.

Higgins, J.M. and B.R. Kim. 1981. Phosphorus retention models for Tennessee Valley Authority reservoirs. Water Resour. Res. 17: 571-576.

Hobbie, J.E. and J.L. Tiwari. 1977. Ecosystem models vs. biological reality: Experiences in the systems analysis of an arctic pond. Verh. Internat. Verein. Limnol. 20: 105-109.

Hutchinson, G.F. 1938. On the relation between oxygen deficit and the productivity and typology of lakes. Int. Rev. ges. Hydrobiol. Hydrogr. 36: 336-355.

Hutchinson, G.E. 1957. A Treatise on Limnology. Vol. I. Geography, Physics, and Chemistry. Wiley, New York. 1015 pp.

Hutchinson, G.E. 1967. A Treatise on Limnology, Vol. II. Introduction to Lake Biology and the Limnoplankton. Wiley, New York. 1115 pp.

Jones, A.R. and G.F. Lee. 1982. Recent advances in assessing impact of phosphorus loads on eutrophic-related water quality. Water Res. 16: 503-515.

Jones, J.R. and R.W. Bachmann. 1976. Prediction of phosphorus and chlorophyll levels in lakes. J. Water. Pollut. Cont. Fed. 48: 2176-2182.

Kirk, J.T.O. 1975. A theoretical analysis of the contribution of algal cells to the alteration of light within natural waters. New Phytol. 75: 11-20.

Komarek, J. 1974. The morphology and taxonomy of crucigenoid algae (Scenedesmaceae Chlorococcales). Arch. Protistenk. 116: 1-75.

Lambou, V.W., S.C. Hern, W.D. Taylor, and L.R. Williams. 1982. Chlorophyll, phosphorus, secchi disk and trophic state. Water Resources Bulletin. 18: 807-813.

Larson, D.P. and H.T. Mercier. 1976. Phosphorus retention capacity of lakes. J. Fish. Res. Bd. Canada 33: 1742-50.

Lasenby, D.C. 1975. Development of oxygen deficits in 14 southern Ontario lakes. Limnol. Oceanogr. 20: 993-999.

Lean, D.R.S., and F.R. Pick. 1981. Photosynthetic response of lake plankton to nutrient enrichment: a test of nutrient limitation. Limnol. Oceanogr 26: 1001-1019.

Leopold, L.B., M.G. Wolman, and J.P. Miller. 1964. Fluvial Processes in Geomorphology. W.H. Freeman, San Francisco. 522 pp.

Levine, S.N., and D.W. Schindler. 1980. Radiochemical analysis of orthophosphate concentrations and seasonal changes in the flux of orthophosphate to seston in Canadian Shield lakes. Can. J. Fish. Aquat. Sci. 37: 479-487.

Lewis, W.M., Jr. 1974. Primary Production in the plankton community of a tropical lake. Ecol. Monogr. 44: 377-409.

Lewis, W.M., Jr., 1980. Comparison of spatial and temporal variations in the zooplankton of a lake by means of variance components. Ecology 59: 666-671.

Lewis, W.M., Jr., and D. Canfield. 1977. Dissolved organic carbon in some dark Venezuelan waters and a revised equation for spectrophotometric determination of dissolved organic carbon. Arch. Hydrobiol. 79: 441-445.

Lewis, W.M., Jr. and F.H. Weibezahn. 1976. Chemistry, energy flow, and community structure in some Venezuelan fresh waters. Arch. Hydrobiol. Suppl. 50(2/3): 145-207.

Lewis, W.M., Jr. and M.C. Grant. 1978. Sampling and chemical interpretations of precipitation for mass balance studies. Water Resour. Res. 14: 1098-1104.

Lewis, W.M., Jr. and M.C. Grant. 1979. Relationships between stream discharge and yield of dissolved substances from a Colorado mountain watershed. Soil Science 128: 253-363.

Lewis, W.M., Jr. and M.C. Grant. 1980a. Change in the output of ions from a watershed as a result of acidification of precipitation. Ecology 60: 1093-1097.

Lewis, W.M., Jr. and M.C. Grant. 1980b. Relationship between snow cover and winter losses of dissolved substances from a mountain watershed. Arct. Alp. Res. 12(1): 11-17.

Lewis, W.M., Jr., and J.F. Saunders, III. 1979. Two new integrating samplers for zooplankton, phytoplankton, and water chemistry. Arch. Hydrobiol. 85: 245-249.

Likens, G.E. 1975. Primary production of inland aquatic ecosystems. In H. Lieth and R.H. Whittaker (eds.) The Primary Productivity of the Biosphere. Springer-Verlag, New York.

Likens, G.E. F.H. Bormann, R.S. Pierce, J.S. Eaton, and N.M. Johnson. 1977. Biogeochemistry of a Forested Ecosystem. Springer-Verlag, New York. 146 pp.

Lund, J.W.G. 1961. The periodicity of U algae in three English lakes. Verh. Internat. Verein. Limnol. 14: 147-154.

Lund, J.W.G. 1965. The ecology of freshwater phytoplankton. Biol. Rev. 40: 231-293.

Lund, J.W.G. 1970. Primary Production. Proc. Soc. Wat. Treat. Exam. 19: 332-358.

Lund, J.W.G. 1971. The seasonal periodicity of three planktonic desmids in Windermere. Mitt. Internat. Verein. Limnol. 19: 3-25.

Lund, J.W.G. 1979. Changes in the phytoplankton of an English lake, 1945-1977. Hydrobiol. Jour. 14: 6-21.

Margalef, R. 1978. Life-forms of phytoplankton as survival alternatives in an unstable environment. Oceanol. Acta 1: 493-509.

Mortimer, C.H. 1941. The exchange of dissolved substances between mud and water. J. Ecol. 29: 280-329.

Murphy, J. and J.P. Riley. 1962. A modified single solution method for the determination of phosphate in natural waters. Anal. Chim. Acta 27: 31-36.

Moss, B. 1972. Studies on Gull Lake, Michigan. Seasonal and depth distribution of phytoplankton. Freshwat. Biol. 2: 289-307.

Nalewajko, C., and D.R.S. Lean. 1980. Phosphorus. pp. 235-258 In I. Morris (ed.) The Physiological Ecology of Phytoplankton. Univ. Calif. Press, Berkeley.

Nauwerck, A. 1963. Die Beziehungen zwischen Zooplankton und Phytoplankton im See Erken. Symb. Bot. Upsal. 17: 1-163.

Nelson, W.C. 1971. Comparative limnology of Colorado-Big Thompson project reservoirs and lakes. Colorado Division of Wildlife Report (mimeo).

OECD. 1982.

Park, A.R. 1979. Modifications to the model CLEANER requiring further research. pp. 87-108 D. Scavia and A. Robertson (eds.) Perspectives on Lake Ecosystem Modeling. Ann Arbor Science, Ann Arbor. 326 pp.

Parsons, T.R., M. Takahashi and B. Hargrave. 1977. Biological Oceanographic Processes. Pergamon, New York. 332 pp.

Rai, Hakumat (ed.) 1980. The measurement of photosynthetic pigments in freshwaters and standardization of methods. Arch. Hydrobiol. Beih. Ergeb. Limnol. 14: 1-106.

Ragotzkie, R.A. 1978. Heat budgets of lakes. pp. 1-19. In A. Lerman (ed.) Lakes: Chemistry, Geology, Physics. Springer-Verlag, New York. 363 pp.

Ramberg, L. 1976. Relations between phytoplankton and environment in two Swedish forest lakes. Scripta Limnol. Upsal. 426: 1-97.

Ramberg, L. 1979. Relations between phytoplankton and light climate in two Swedish forest lakes. Internat. Rev. ges. Hydrobiol. 64: 749-782.

Rieman, B. 1980. A note on the use of methanol as an extraction solvent for chlorophyll a determination. Arch. Hydrobiol. Beih. Ergebn. Limnol. 14: 70-78.

Rigler, F.H. 1968. Further observations inconsistent with the hypothesis that molybdenum blue method measures orthophosphate in lake water. Limnol. Oceanogr. 13: 7-13.

Rohlf, J.F. and R. Sokal. 1969. Statistical Tables. W.H. Freeman, San Francisco. 253 pp.

Saunders, G.W., F.B. Trama, and R.W. Bachmann. 1962. Evaluation of a modified C-14 technique for shipboard estimation of photosynthesis in large lakes. Great Lakes Res. Div. Publ. No. 8. 61 p.

Sawyer, C.N. 1947. Fertilization of lakes by agriculture and urban drainage. J. New England Water Works. Ass. 61: 109-127.

Schindler, D.W., R.W. Newbury, K.G. Beaty, and P. Campbell. 1976. Natural water and chemical budgets for a small Precambrian lake basin in Central Canada. J. Fish. Res. Bd. Canada. 33: 2526-2543.

Smith, R.C. 1968. The optical characterization of natural waters by means of an "extinction coefficient." Limnol. Oceanogr. 13: 423-429.

Smith, V.H. 1982. The nitrogen and phosphorus dependence of algal biomass in lakes: an empirical and theoretical analysis. Limnol. Oceanogr. 27: 1101-1112.

Solorzano, L. 1969. Determination of ammonia in natural waters by the phenol-hypochlorite method. Limnol. Oceanogr. 14: 799-801.

Solorzano, L., and J. Sharp. 1980a. Determination of total dissolved phosphorus and particulate phosphorus in natural waters. Limnol. Oceanogr. 25: 754-758.

Solorzano, L., and J. Sharp. 1980b. Determination of total dissolved nitrogen in natural waters. Limnol. Oceanogr. 25: 751-754.

Strom, K.M. 1931. Fetovatn: A physiographic and biological study of a mountain lake. Arch. Hydrobiol. 22: 491-536.

Stumm, W. and P. Baccini. 1978. Man-Made Chemical Perturbations of lakes. pp. 91-123. In A. Lerman (ed.) Lakes - Chemistry, Geology, Physics. Springer-Verlag, New York. 363 pp.

Stumm, W. and J.J. Morgan. 1981. Aquatic Chemistry. Wiley, New York. 780 pp.

Taguchi, S. 1976. Relationship between photosynthesis and cell size of marine diatoms. J. Phycol. 12: 185-189.

Talling, J.F. 1969. Measurement of photosynthesis using illuminated constant temperature baths. pp. 108-117 In R.A. Vollenweider (ed.) A Manual on Methods for Measuring Primary Production in Aquatic Environments. Blackwell, London.

Talling, J.F. 1971. The underwater light climate as a controlling factor in the production ecology of freshwater phytoplankton. Mitt. Internat. Verein. Limnol. 19: 214-243.

Talling, J.F. 1976. The depletion of carbon dioxide from lake water by phytoplankton. J. Ecol. 64: 79-121.

Talling, J.F. 1979. Factor interactions and implications for the prediction of lake metabolism. Arch. Hydrobiol. Beih. 13: 96-109.

Talling, J.F. 1982. Utilization of solar radiation by phytoplankton. pp. 619-631. In C. Helene, M. Charlier Th. Montenay-Garestier and G. Laustriat (eds.) Trends in Photobiology. Plenum, New York.

Vollenweider, R.A. 1968. Scientific fundamentals of the eutrophication of lakes and flowing waters, with particular reference to nitrogen and phosphorus as factors in eutrophication. Organization for Economic Cooperation and Development Report. DAS/CSI/68.27, 182 pp.

Vollenweider, R.A. 1969. Possibilities and limits of the elementary models concerning the budget of substance in lakes. Arch. Hydrobiol. 66: 1-36.

Vollenweider, R.A., W. Rast and J. Kerekes. 1980. The phosphorus loading concept and Great Lakes eutrophication. pp. 207-234 In Loehr, R.C., C.S. Martin, and W. Rast (eds.) Phosphorus Management Strategies for Lakes. Ann Arbor Science, Ann Arbor. 490 pp.

Wahlstrom, E.E. and D.Q. Hornback. 1962. Geology of the H.D. Roberts Tunnel, Colorado. Geological Society of America Bulletin 73:1477-1498.

Welch, E.B. 1980. Ecological Effects of Wastewater. Cambridge University Press, New York. 337 pp.

Wolk, C.P. 1973. Physiology and cytological chemistry of blue-green algae. Bacteriol. Rev. 37: 32-101.

Wright, R.F. 1974. Forest Fire: Impact on the hydrobiology, chemistry and sediments of small lakes in northeastern Minnesota. Interim Rept. No. 10, Limnology Research Center, University of Minnesota, Minneapolis, MN. 129 pp.

Summary of Public Participation

The Steering Committee of the Lake Dillon Clean Lakes Study was composed of the following members:

AMAX Inc.
Keystone Corporation
Copper Mountain Water and Sanitation District
Breckenridge Sanitation District
Frisco Sanitation District
Dillon/Silverthorne Joint Sewer Authority
Denver Water Department
Town of Breckenridge
Town of Frisco
Town of Dillon
Summit County
Northwest Colorado Council of Governments
Colorado Department of Health
Region VIII, U.S. Environmental Protection Agency

During the progress of the study one or more representatives of the contractor, Western Environmental Analysts, Inc., attended 12 meetings at which discussions or formal presentations were made concerning the Study. The date, place and nature of each meeting is listed below.

Jan. 6, 1981 - Frisco - Steering Committee meeting to discuss work plan.
Feb. 10, 1981 - Frisco - Steering Committee meeting to discuss addendum to work plan.
Apr. 8, 1981 - Frisco - Steering Committee meeting to discuss sampling plan and parameter measurements.
Nov. 23, 1981 - Frisco - Steering Committee meeting to discuss first year results and second year plans.
Apr. 13, 1982 - Breckenridge - presentation of first year results at public meeting.
June 18, 1982 - Frisco - description of Study and first year results to joint meeting of members of Colorado River and Middle Park Water Conservancy Districts.
July 7, 1982 - Frisco - discussion of comments on first annual report with Steering Committee and representatives from two consulting firms, Engineering Science and Culp Wessner Culp.

Feb. 16, 1983 - Frisco - status report on analysis of first and second year results to meeting of Dillon watershed discharge permit holders and others.

Mar. 1, 1983 - Denver - status report on analysis of first and second year results to meeting of American Waterworks Association members and guests.

May 19, 1983 - Frisco - Steering Committee meeting to discuss draft final report.

July 12, 1983 - Frisco - presentation of first and second year results to Dillon Reservoir Clean Lakes Policy Committee (policy-making representatives from Steering Committee membership as opposed to technical representatives described previously as the "Steering Committee").

July 26, 1983 - Breckenridge - presentation of final (first and second year) results at public meeting.

Appendix I
DILLON CLEAN LAKES MODEL
Input, Code, and Output

For a model run (Scenario 1A: Low growth, no diversions, wet year)

Input - Pages I-1 - I-2
Code - Pages I-3 - I-15
Output - Pages I-16 - I-21

```
 211
3572 424 231 253 235 319 310 500 364 433 406 513 325  77 325 325 325 325 325
2322  83 213 165 192 120 160 240 227 259 151 435 126  21 126 126 126 126 126
    0    0    0    0    0    0    0    0    0    0    0    0
    0    0    0    0    0    0    0    0    0    0    0    0
    0    0    0    0    0    0    0    0    0    0    0    0
BR13572.    25.                                  6700.
BR2 424. 11003.  63.       317.              28.
BR4 231   9719.1410.4263. 697        386.
BR6 253   6133.  11        647.
BR7 235.  5165. 178.       843.             178.
SR1 319.  1041  909.                         10.
SR2 310.  2678.1305.        64.       177.
SR3 500.  9789.            133.
SR4 364.  4122.                       78.  90.
TM1 433.  6329.       367. 135.  42.
TM4 406. 11422.       440.       67  302.      2500.
TM7 513.  6447                                      2650.
MC2 325.  1800.        86. 129.
SC2  77.  2212. 294.
AAA 325.  1143. 252.2388.   7.
BBB 325.  1226. 536.                            3230.
CCC 325.   619.             45.
DDD 325.    33.      1258.    .                 3300.
EEE 325.  1817.             43.  28.
```

1	15	8981	4.78	12.54	0.
2	45	8977	3.85	11 13	0.
3	74	8974	5.19	9.07	0.
4	105	8974	7 73	4.68	0.
5	135	8980	33.71	3.82	0.
6	166	9000	92.04	6.15	0.
7	196	9018	56.30	30.38	0.
8	227	9018	27.48	26.53	0.
9	258	9018	14.71	15.35	0.
10	288	9017	9.93	10.17	0.
11	319	9016	6.23	13.12	0.
12	349	9015	6.19	5.79	0.

```
                MNF,E=4,I=DILEXP,L=L.

1     000000B         PROGRAM DILEXP (INPUT,OUTPUT,TAPE1,TAPE2,TAPE3)
                C   THIS IS THE LAND USE ELEMENT OF THE DILLON CLEAN LAKES MODEL.
                C   IT PREDICTS NUTRIENT YIELD TO THE LAKE GIVEN LAND USE DATA
                C   AS INPUT AND AMOUT OF RUNOFF
2.    010335B         DIMENSION NAME(19),WATR(19),AREA(19),PTOT(19),NTOT(19),
                     1PBKG(19),NBKG(19),LUSE(19,8),PYLD(19,8),NYLD(19,8)
                     2,USTOP(8),USTON(8),USPCP(8),USPCN(8),HIWAT(8),
                     3LOWAT(8),DIV(12),DVCONP(12),DVCONN(12)
                C   TAPE1 ONE IS THE DATA SOURCE,STORED ON DISK AS DILEX
                C   THE FIRST RECORD ON TAPE1 IS THE RUNOFF(EXCLUDING ANY PUMPED
                C   WATER). THIS IS VARIABLE USGS AND IS GIVEN AS THOUSANDS
                C   OF ACRE FEET PER YEAR. IT IS THE SUM OF THE 4 USGS
                C   GAUGES (SNAKE,KEYSTONE GULCH,BLUE,TENMILE).  THE
                C   SECOND RECORD IS A LIST OF THE WATER YIELDS FOR THE
                C   19 SEGMENTS OF THE WATERSHED FOR 1982 IN MM PER YEAR.
                C   THE THIRD RECORD IS THE SAME FOR 1981.  THE FOURTH
                C   RECORD IS A LIST OF THE AMOUNT OF WATER DIVERTED
                C   OR PUMPED FROM OUTSIDE THE WATERSHED IN EACH OF THE
                C   12 MONTHS OF THE YEAR, AS THOUSANDS OF ACRE FEET
                C   PER MONTH.  THE NEXT RECORD GIVES THE P CONCENTRATION
                C   OF THE PUMPED WATER MONTH BY MONTH AND THE NEXT GIVE
                C   THE N CONCENTRATION. THE LAST 19 RECORDS GIVE THE
                C   AREA, WATER YIELD, AND LAND USE INFORMATION OF EACH OF
                C   THE 19 WATERSHED SEGMENTS.
3.    010335B         REAL LUSE,PYLD,NYLD,NTOT,PTOT,PGRAND,NGRAND,NBKG,PBKG
                     1,PRECP,PRECN,PRPCTP,PRPCTN,PCTSTOP,PCTSTON,NETYLDP,NETYLDN
                     2,HIWAT,LOWAT
                C   II SETS THE NUMBER OF RUNS
4.    010335B         DO 901 II=1,1
                C   THE AREA OF EACH SEGMENT IS ENTERED ON TAPE1 AS IS THE
                C   WATER YIELD.  IF THE WATER YIELDS ARE SPECIFIED ALONG WITH
                C   THE LAND USE DATA, THE SPECIFIED VALUES ARE USED.IF
                C   THEY ARE NOT SPECIFIED,THE DISTRIBUTION OF WATER YIELDS
                C   IS SET PROPORTIONAL TO THE 1981 YIELDS FOR A YEAR
                C   BELOW AVERAGE RUNOFF AND PROPORTIONAL TO THE 1982 YIELDS
                C   FOR AVERAGE OR ABOVE AVERAGE RUNOFF.
                C   THE RUNOFF IS VARIABLE USGS AND IS THE SUM OF THE 4 GAUGES
                C   IN THOUSANDS OF ACRE FEET PER YEAR.  THE MATRIX LUSE HAS 19 ROWS
                C   THAT ARE 19 SEGMENTS OF THE CATCHMENT AND 8 COLUMNS THAT
                C   ARE THE LAND USES AS FOLLOWS: 1)RESIDENTIAL SEWER NONPOINT,AS
                C   NUMBER OF PERSONS, 2)URBAN SEWER AS PERSONS,3)SEPTIC AS
                C   PERSONS,4)INTERSTATE HIGHWAY AS HECTARES ROAD PLUS RIGHT OF WAY,
                C   5)SKI AREA AS HECTARES OF CLEARED AREA, 6)PACKAGE PLANT OUTFALLS,
                C   AS NUMBER OF PERSONS SERVED,7)TERTIARY OUTFALLS SAME,8)AMAX
                C   MINING AS WORKFORCE SIZE
                C   THE GROSS YIELDS ARE UNCORRECTED FOR INVENTORY CHANGE IN THE
                C   VALLEY BOTTOMS, THE NET YIELDS ARE CORRECTED FOR THIS.
                C   READ IN THE AMOUNTS OF RUNOFF.
5.    011611B         READ (1,260) USGS,(HIWAT(K),K=1,19),(LOWAT(K),K=1,19)
6.    011626B     260 FORMAT (F5.0,/,19F4.2,/,19F4.2)
                C   READ IN THE INFORMATION ON PUMPING AND DIVERSIONS
7     011626B         READ(1,297) (DIV(K),K=1,12),(DVCONP(K),K=1,12),
                     1(DVCONN(K),K=1,12)
8     011645B     297 FORMAT (12F5.0,/,12F5.0,/,12F5.0)
9.    011645B         DO 201 I=1,19
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10.   011646B          READ (1,202) NAME(I),WATR(I),AREA(I),(LUSE(I,K),K=1,8)
11    011670B      202 FORMAT (A3,F5.0,F7.0,8F5.0)
12.   011670B      201 CONTINUE
                   C  COMPUTE THE LOADING BY DIVERSION AND PUMPING AND CONVERT
                   C  DIV TO MILLIONS OF M3 PER MONTH
13.   011672B          DVSUMP=0.0
14.   011672B          DVSUMN=0.0
15.   011673B          DO 298 I=1,12
16.   011675B          DIV(I)=DIV(I)*1000.*1233.5*.000001
17    011677B          DVSUMP=DVSUMP+DIV(I)*DVCONP(I)
18.   011702B          DVSUMN=DVSUMN+DIV(I)*DVCONN(I)
19.   011705B      298 CONTINUE
                   C  SET THE WATER YIELDS
20.   011706B          DO 280 I=1,19
21.   011710B          IF (WATR(2).GT.10.)GO TO 280
22.   011713B          IF (USGS.GT.180.)GO TO 281
23.   011715B          WATR(I)=LOWAT(I)*(USGS/116.5)
24.   011717B          GO TO 280
25.   011717B      281 CONTINUE
26.   011721B          WATR(I)=HIWAT(I)*(USGS/210.9)
27    011723B      280 CONTINUE
28.   011727B          WATR(1)=WATR(2)
29.   011730B          PGRAND=0.0
30.   011731B          NGRAND=0.0
                   C  APPLY LAND USE EQUATIONS
31.   011731B          DO 203 I=1,19
32.   011733B          PYLD(I,1)=.000001*LUSE(I,1)*2.95*WATR(I)**1.372
33.   011740B          NYLD(I,1)=.000001*LUSE(I,1)*536.*WATR(I)**1.154
34.   011746B          PYLD(I,2)=.000001*LUSE(I,2)*1.50*WATR(I)**1.372
35.   011754B          NYLD(I,2)=.000001*LUSE(I,2)*410.*WATR(I)**1.154
36.   011762B          PYLD(I,3)=.000001*LUSE(I,3)*3440.*WATR(I)**0.759
37    011770B          NYLD(I,3)=.000001*LUSE(I,3)*(44300.*WATR(I)-5205.)
38.   011774B          PYLD(I,4)=.000001*LUSE(I,4)*10000.*.00209*WATR(I)**1.799
39.   012003B          NYLD(I,4)=.000001*LUSE(I,4)*10000.*1.71*WATR(I)**1.13
40.   012012B          PYLD(I,5)=.000001*LUSE(I,5)*6.50*10000.
41.   012014B          NYLD(I,5)=.000001*LUSE(I,5)*42.*10000.
42.   012017B          PYLD(I,6)=.000001*LUSE(I,6)*270000.
43.   012021B          NYLD(I,6)=.000001*LUSE(I,6)*1197000.
44.   012024B          PYLD(I,7)=.000001*LUSE(I,7)*47000.
45.   012026B          NYLD(I,7)=.000001*LUSE(I,7)*1197000.
46.   012031B          PYLD(I,8)=.000001*LUSE(I,8)*98000.
47.   012033B          NYLD(I,8)=.000001*LUSE(I,8)*19930000.
48.   012036B          PBKG(I)=AREA(I)*10000.*.000001*.000782*WATR(I)**1.372
49.   012045B          NBKG(I)=AREA(I)*10000.*.000001*.0842*WATR(I)**1.154
50.   012054B          PTOT(I)=PYLD(I,1)+PYLD(I,2)+PYLD(I,3)+PYLD(I,4)+
                      1PYLD(I,5)+PYLD(I,6)+PYLD(I,7)+PYLD(I,8)+PBKG(I)
51    012065B          NTOT(I)=NYLD(I,1)+NYLD(I,2)+NYLD(I,3)+NYLD(I,4)+
                      1NYLD(I,5)+NYLD(I,6)+NYLD(I,7)+NYLD(I,8)+NBKG(I)
52.   012076B          PGRAND=PGRAND+PTOT(I)
53.   012100B          NGRAND=NGRAND+NTOT(I)
54.   012102B      203 CONTINUE
                   C  ADD 3.5 PERCENT FOR CONSTRUCTION ACTIVITIES P,2.2 PERCENT N.
55.   012103B          CONSTP=PGRAND*0.035
56.   012104B          CONSTN=NGRAND*0.022
57    012106B          PGRAND=PGRAND+CONSTP
58.   012110B          NGRAND=NGRAND+CONSTN
                   C  WRITE OUT PREDICTED YIELDS BY SEGMENTS
59.   012112B          WRITE (2,190)
60.   012116B      190 FORMAT(///////)
61.   012116B          WRITE (2,226)
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 62.    012121B     226 FORMAT (49H  GROSS P YIELD FOR WATERSHED SEGMENTS, KG PER YR)
 63.    012121B         WRITE (2,215)
 64.    012124B     215 FORMAT (15H STATION NAME  ,11H  MM RUNOFF,11H  AREA HA  ,
                       111H  RES SEW  ,11H  URB SEW  ,11H   SEPTIC  ,11H  HIWAY    ,
                       211H  SKISLOPE ,11H  PKG PLT  ,11H  TERT PLT ,11H   AMAX    ,
                       311H  BKGRND   )
 65.    012124B         DO 204 I=1,19
 66.    012125B         WRITE (2,205) NAME(I),WATR(I),AREA(I),(PYLD(I,K),K=1,8),
                       1PBKG(I)
 67.    012151B     205 FORMAT (/,4X,A3,5X,11F11.1)
 68.    012151B     204 CONTINUE
 69.    012153B         WRITE (2,190)
 70.    012157B         WRITE (2,227)
 71     012162B     227 FORMAT (49H  GROSS N YIELD FOR WATERSHED SEGMENTS, KG PER YR)
 72.    012162B         WRITE  (2,215)
 73.    012165B         DO 206 I=1,19
 74.    012166B         WRITE (2,205) NAME(I),WATR(I),AREA(I),(NYLD(I,K),K=1,8),
                       1NBKG(I)
 75.    012212B     206 CONTINUE
 76.    012214B         WRITE (2,190)
 77     012220B         WRITE (2,258)
 78.    012223B     258 FORMAT (42H GROSS P AND N YIELD BY SEGMENT, KG PER YR)
 79.    012223B         WRITE (2,257)
 80.    012226B     257 FORMAT (6H NAME ,8H PTOTAL ,8H NTOTAL )
 81.    012226B         DO 207 I=1,19
 82.    012227B         WRITE (2,208) NAME(I),PTOT(I),NTOT(I)
 83.    012241B     208 FORMAT (/,1X,A3,2F8.0)
 84.    012241B     207 CONTINUE
 85.    012243B         WRITE (2,209) PGRAND,NGRAND
 86.    012251B     209 FORMAT (/////,32H GROSS KG PER YR P IN RUNOFF IS,F6.0,/,
                       132H GROSS KG PER YR N IN RUNOFF IS,F8.0,/)
                  C  COMPUTE THE TOTAL P AND N YIELDS TOTAL FOR EACH LAND USE, GROSS
 87     012251B         PRECP=726.
 88.    012251B         PRECN=9872.
 89.    012253B         GRNDP=10.
 90.    012254B         GRNDN=1066.
 91     012256B         DO 210 J=1,8
 92.    012260B         USTOP(J)=0.0
 93.    012260B         USTON(J)=0.0
 94.    012260B     210 CONTINUE
 95     012262B         TBKGP=0.0
 96.    012262B         TBKGN=0.0
 97     012263B         DO 250 I=1,19
 98.    012265B         TBKGP=TBKGP+PBKG(I)
 99.    012266B         TBKGN=TBKGN+NBKG(I)
100.    012270B     250 CONTINUE
101     012271B         DO 211 J=1,8
102.    012273B         DO 212 I=1,19
103.    012275B         USTOP(J)=PYLD(I,J)+USTOP(J)
104.    012301B         USTON(J)=NYLD(I,J)+USTON(J)
105.    012303B     212 CONTINUE
106.    012301B     211 CONTINUE
107     012307B         WRITE (2,270)
108     012313B     270 FORMAT (50H GROSS YIELD BY SOURCE KG PER YR P(ABOVE),N(BELOW))
109.    012313B         WRITE (2,271)
110.    012316B     271 FORMAT (/,13H RESID. SEWER,10H  URB SEW ,10H  SEPTIC  ,
                       110H  HIWAY   ,10H  SKISLOPE,10H  PKG PLT ,10H  TERT PLT,
                       210H  AMAX     ,10H  BKGRND  ,10H PRECIP   ,10H  GRNDWAT ,
                       310H DIVERSION,10H CONSTRUCT)
111     012316B         WRITE (2,213) (USTOP(K),K=1,8),TBKGP,PRECP,GRNDP,DVSUMP,CONSTP,
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                1(USTON(K),K=1,8),TBKGN,PRECN,GRNDN,DVSUMN,CONSTN
112.  012343B         WRITE(2,190)
113.  012346B       213 FORMAT ( 13F10.1,/,13F10.1)
                  C  COMPUTE INVENTORY CHANGE AND ADJUST YIELD   ACCORDINGLY
114.  012346B         PCTSTOP=USGS*(-.40)+87.6
115.  012350B         PCTSTON=USGS*(-.57)+112.
116.  012353B         DO 290 I=1,6
117   012355B         USTOP(I)=USTOP(I)*(100.-PCTSTOP)*.01
118.  012360B         USTON(I)=USTON(I)*(100.-PCTSTON)*.01
119.  012362B       290 CONTINUE
120.  012364B         USTOP(8)=USTOP(8)*(100.-PCTSTOP)*.01
121.  012367B         USTON(8)=USTON(8)*(100.-PCTSTON)*.01
122.  012372B         CONSTP=CONSTP*(100.-PCTSTOP)*.01
123.  012376B         CONSTN=CONSTN*(100.-PCTSTON)*.01
124.  012402B         TBKGP=TBKGP*(100.-PCTSTOP)*.01
125.  012406B         TBKGN=TBKGN*(100.-PCTSTON)*.01
126.  012412B         WWTPP=PYLD(1,7)+PYLD(16,7)+PYLD(18,7)
                     1+PYLD(11,7)*(100.-PCTSTOP)*.01
127   012422B         WWTPN=NYLD(1,7)+NYLD(16,7)+NYLD(18,7)
                     1+NYLD(11,7)*(100.-PCTSTON)*.01
128.  012431B         NETYLDP=(PGRAND-USTOP(7))*(100.-PCTSTOP)*.01+WWTPP+PRECP+GRNDP
                     1+DVSUMP
129.  012443B         NETYLDN=(NGRAND-USTON(7))*(100.-PCTSTON)*.01+WWTPN+PRECN+GRNDN
                     1+DVSUMN
130.  012454B         USTOP(7)=WWTPP
131   012455B         USTON(7)=WWTPN
                  C  COMPUTE THE PERCENTAGES OF EACH CONTRIBUTION
132.  012457B         DO 295 J=1,8
133.  012461B         USPCP(J)=(USTOP(J)/(NETYLDP))*100.
134.  012463B         USPCN(J)=(USTON(J)/NETYLDN)*100.
135.  012465B       295 CONTINUE
136.  012467B         PRPCTP=(PRECP/(NETYLDP))*100.
137.  012471B         GRPCTP=(GRNDP/(NETYLDP))*100.
138.  012474B         TBKGPPC=(TBKGP/(NETYLDP))*100.
139.  012476B         DVPCP=(DVSUMP/NETYLDP)*100.
140.  012500B         CONSPCP=(CONSTP/NETYLDP)*100.
141.  012502B         TBKGNPC=(TBKGN/NETYLDN)*100.
142.  012504B         PRPCTN=(PRECN/NETYLDN)*100.
143.  012507B         GRPCTN=(GRNDN/NETYLDN)*100.
144.  012511B         DVPCN=(DVSUMN/NETYLDN)*100.
145.  012513B         CONSPCN=(CONSTN/NETYLDN)*100.
146.  012515B         WRITE (2,190)
147.  012521B         WRITE (2,296)
148.  012524B       296 FORMAT(42H  VALUES UP TO THIS POINT UNCORRECTED FOR ,/,
                     155H   INVENTORY CHANGE IN RIVER BOTTOMS, NOW CORRECT THIS.,//)
149.  012524B         WRITE (2,370)
150.  012527B       370 FORMAT (50H NET LOADING BY SOURCE KG PER YR P(ABOVE),N(BELOW))
151.  012527B         WRITE (2,271)
152.  012532B         WRITE (2,213) (USTOP(K),K=1,8),TBKGP,PRECP,GRNDP,DVSUMP,CONSTP,
                     1(USTON(K),K=1,8),TBKGN,PRECN,GRNDN,DVSUMN,CONSTN
153.  012557B         WRITE (2,275)
154.  012562B       275 FORMAT (//,48H THE NET YIELDS BY PERCENT ARE P(ABOVE),N(BELOW))
155.  012562B         WRITE (2,271)
156.  012565B         WRITE (2,213) (USPCP(K),K=1,8),TBKGPPC,PRPCTP,GRPCTP,DVPCP,
                     1CONSPCP,(USPCN(K),K=1,8),TBKGNPC,PRPCTN,GRPCTN,DVPCN,CONSPCN
157.  012612B         WRITE (2,190)
158.  012615B         WRITE(2,265) NETYLDP,NETYLDN
159.  012622B       265 FORMAT(48H  THE NET P YIELD TO THE LAKE IN KG PER YEAR IS,F6.0,
                     1,/,48H  THE NET N YIELD TO THE LAKE IN KG PER YEAR IS,F8.0)
160.  012622B         WRITE (3,902) NETYLDP,PRECP,USGS,(DIV(K),K=1,12),
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                     1(DVCONP(K),K=1,12)
161     012640B     902 FORMAT (3F7.0,/,12F6.2,/,12F6.2)
162.    012640B     901 CONTINUE
163.    012642B         STOP
164.    012643B         END
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UNIVERSITY OF MINNESOTA FORTRAN COMPILER (VERSION 5.4 - 79/03/01) ON THE 6400 UNDER KRONOS 2.1.0 ON 83/04/01 AT 16.03
UNIVERSITY COMPUTING CENTER - UNIVERSITY OF COLORADO

                  MNF,E=4,I=DILTP,L=LL.

    1.   000000B        PROGRAM DILTP (INPUT,OUTPUT,TAPE11, TAPE2,TAPE3,TAPE4)
                     C  THIS IS THE TROPHIC STATUS ELEMENT OF THE DILLON
                     C  CLEAN LAKES MODEL.  THIS VERSION IS SET UP TO
                     C  INTERFACE WITH LAND USE ON THE INPUT SIDE AND
                     C  EFFECTS ON THE OUTPUT SIDE.  THE PROGRAM TAKES
                     C  USER SUPPLIED INPUT ON TAPE11 AND INPUT SUPPLIED
                     C  BY DILEXP ON TAPE3. KRONOS COMMANDS FOR INTERFACE
                     C  ARE ON DILPROC.  VARIABLES COMING IN BY MONTH
                     C  ON TAPE11 INCLUDE MONTH NUMBER (MON), DAY NUMBER
                     C  (DAY), LAKE LEVEL IN FEET (LEV), TOTAL SURFACE
                     C  WATER IN MILLIONS OF M3 PER MONTH (VIN),TOTAL
                     C  OUTFLOW IN MILLIONS OF M3 PER MONTH (VUT). VARIABLES
                     C  COMING IN ON TAPE3 FROM DILEXP ARE TOTAL P LOAD IN
                     C  KG PER YEAR (NETYLDP), P LOAD DUE TO PRECIPITATION
                     C  IN KG PER YR (PRECLO), AMOUNT OF DIVERTED OR
                     C  PUMPED WATER IN MILLIONS OF M3 PER MONTH FOR ALL
                     C  12 MONTHS, AND P CONCENTRATION MONTHLY IN PUMPED
                     C  OR DIVERTED WATER, AS UG PER L.
                     C  THE PROGRAM WRITES TO TAPE2 AND OUTPUT IS
                     C  PACKED BY DILPROC.
    2.   012411B        DIMENSION MON(12), DAY(12),LEV(12),VIN(12),
                       1VUT (12), PLD (12), ARE (12), VOL (12),  RLD(12), DIV(12)
                       2,DVCONP(12)
    3.   012411B        REAL MON, LEV, PLD, PLDTO, PLAKE, LANDLO, PRECLO,NETYLDP
                     C  SET NUMBER OF DATA SETS TO BE TREATED
    4.   012411B        DO 107 II=1,1
                     C  READ INPUT DATA FOR EACH OF 12 MONTHS
                     C
    5.   012623B        READ (11,3) (MON(K), DAY (K), LEV (K), VIN (K), VUT(K)
                       1 , K=1,12)
    6.   012644B      3 FORMAT (3F5.0,2F7.0)
    7.   012644B        READ (3,101) NETYLDP,PRECLO,USGS,(DIV(K),K=1,12),
                       1(DVCONP(K),K=1,12)
    8.   012662B    101 FORMAT (2F7.0,F7.0,/,12F6.2,/,12F6.2)
    9.   012662B        DVSUMP=0.0
                     C  COMPUTE LOADING FRACTIONS
   10.   012662B        DO 150 I=1,12
   11.   012664B        DVSUMP=DVSUMP+DIV(I)*DVCONP(I)
   12.   012664B    150 CONTINUE
   13.   012671B        LANDLO=NETYLDP-PRECLO-DVSUMP
                     C  TOTAL UP THE WATER EXCLUDING DIVERSION INPUT AND ADD GROUNDWATER
                     C  VTOT WILL BE TOTAL NONDIVERSION WATER INPUT PER YR IN M3
   14.   012673B        VTOT=0.0
   15.   012674B        DO 103 I=1,12
   16.   012676B        VIN(I)=VIN(I)+.575
   17.   012677B        VTOT=VTOT+VIN(I)*1000000.
   18.   012702B    103 CONTINUE
                     C  COMPUTE MONTHLY P INPUTS IN KG PER MONTH
   19.   012703B        DO 102 I=1,12
   20.   012704B        RLD(I)=0.063*PRECLO
   21.   012705B        PLD(I)=LANDLO*(VIN(I)*1000000./VTOT)+DVCONP(I)*DIV(I)
   22.   012712B    102 CONTINUE
   23.   012714B        RLD(6)=0.3*PRECLO
                     C  WRITE PRELIMINARY DATA DATA ONTO TAPE 2
   24.   012715B        WRITE (2,2)
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25.   012722B      2 FORMAT (//////,'   *MONTH*     ,'     *DAY*     ',
                    1' *LEVEL FT* ',  *MIL M3MO ROFF* ',' *MIL M3MO DIVER*   ,
                    2' *MIL M3MO OUT* ',' *KG P NONPREC* ',' *KG P PREC* ')
26.   012722B       WRITE (2,6) (MON(K),DAY(K),LEV(K),VIN(K),DIV(K),
                    1VUT(K),PLD(K),RLD(K),K=1,12)
27    012750B      6 FORMAT (F7.0, F14.0, F12.0,F12.2, 2F18.2, 2F17.1)
                C  COMPUTE LAKE VOLUME AND AREA FROM LEVEL
28.   012750B       DO 7 I = 1,12
29.   012751B       IF (LEV (I).GT.8000..AND.LEV(I).LE.8950.)ARE(I)=(580./50.)*(LEV
                    1(I)-8900.)+1000.
30.   012761B       IF (LEV(I).GT.8900..AND.LEV(I).LE.8925.)VOL(I)=(28.15/25.)*(LEV
                    1(I)-8900.)+40.
31    012767B       IF (LEV(I).GT.8925..AND.LEV(I).LE.8950.)VOL(I)=(37./25)*(LEV
                    1(I)-8925.)+67.
32.   012775B       IF (LEV (I).GT.8950..AND.LEV(I).LE.8975.)ARE(I)=(400./25.)*(LEV
                    1(I) - 8950.)+1600.
33.   013003B       IF (LEV (I).GT.8950..AND. LEV (I).LE.8975.)VOL (I)=(47./25.)*(LEV
                    1(I) - 8950.)+103.
34.   013011B       IF (LEV (I).GT.8975..AND.LEV (I).LE.9000.)ARE(I)=(700./25.)*(LEV
                    1(I) - 8975.)+2000.
35.   013017B       IF (LEV(I).GT.8975..AND.LEV (I).LE.9000.)VOL(I)=(60./25.)*(LEV
                    1(I) - 8975.)+150.
36.   013025B       IF (LEV (I).GT.9000..AND.LEV (I).LE.9025.)ARE(I)=(1050./25.)*(LEV
                    1(I) - 9000.)+2700.
37.   013033B       IF (LEV (I).GT.9000..AND.LEV(I).LE.9025.)VOL(I)=(75./25.)*(LEV
                    1(I) - 9000.)+210.
38.   013041B       VOL(I)=VOL(I)*1000.*1233.
39.   013043B       ARE(I) = ARE(I)*.4047
40.   013044B      7 CONTINUE
                C  SUM VARIABLES OVER YEAR VOLS IN M3
41    013047B       VUTTO = 0.0
42.   013047B       PLDTO = 0.0
43.   013050B       ARETO = 0.0
44.   013051B       VOLTO = 0.0
45.   013051B       VINTO=VTOT
46.   013052B       DO 8 I = 1,12
47    013054B       VUTTO = VUTTO + VUT (I)*1000000.
48.   013056B       PLDTO = PLDTO + PLD (I) + RLD(I)
49.   013061B       ARETO = ARETO + ARE(I)
50.   013063B       VOLTO = VOLTO + VOL(I)
51    013065B       VINTO=VINTO+DIV(I)*1000000.
52.   013070B      8 CONTINUE
                C  COMPUTE P LOAD PER UNIT AREA IN MG PER M2 PER YR, HYDRAULIC
                C   RESIDENCE TIME IN YR,
                C  VOLUME LOADING IN MG PER M3 PER YR, AND MEAN DEPTH IN M
53.   013071B       PLDTO = PLDTO/365.
54    013072B       AREMN = ARETO/12.0
55.   013074B       VOLMN = VOLTO/12.0
56.   013076B       ARELD = (PLDTO*1000.*1000.*365.)/(AREMN*10000.)
57    013103B       VOLLD = (PLDTO*1000.*1000.*365.)/VOLMN
58    013106B       DEPMN = VOLMN/(AREMN*10000.)
59.   013111B       RESID=VOLMN/VINTO
                C  COMPUTE THE EFFECTIVE SEDIMENTATION COEFFICIENT,SIGMA
60    013113B       SIGMA=USGS*0.0036+.63
                C   COMPUTE THE PREDICTED IN-LAKE P CONCENTRATION
                C  USING VOLLENWEIDER MODIFIED INDEX
61    013116B       PLAKE=(ARELD/(DEPMN*(1/RESID +SIGMA)))
                C  COMPUTE THE CHLOROPHYLL OVER THE 0-5M LAYER,
                C  STRATIFICATION SEASON MEAN
62    013124B       CHLA=1.449*ALOG10(PLAKE)-.398
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      CHLA=10**CHLA
C  WRITE OUT RESULTS
      WRITE (2,10) AREMN, VOLMN, VINTO
   10 FORMAT (/////,2X,'MEAN LAKE AREA OVER THE YEAR IN HECTARES IS',
     1F6.0,/,2X,
     2'AVERAGE LAKE VOLUME OVER THE YEAR IN M3 IS ',
     3F11.0,/,2X,
     4'TOTAL ANNUAL INFLOW IN M3 INCLUDING DIVERSION IS', F11.0)
      WRITE (2,11) VUTTO, PLDTO
   11 FORMAT (/,2X,'TOTAL ANNUAL OUTFLOW IN M3 IS',F11.0,/,2X,
     1'TOTAL LOADING IN KG PER DAY IS', F7.3)
      WRITE (2,12) ARELD, VOLLD, DEPMN
   12 FORMAT (/,2X,'AREAL LOAD IN MG PER M2 PER YR IS ',
     1F6.1,/,2X,'VOLUME LOAD IN MG PER M3 PER YR IS',
     2F6.1,/,2X,'MEAN DEPTH IN M IS',
     3F5.1)
      WRITE (2,13)RESID,PLAKE
   13 FORMAT (/,2X,'TAU IN YR IS', F5.2,/,2X,
     1'PREDICTED  P CONCENTRATION IN TOP 15 M IS',F5.1,//)
      WRITE (2,14)CHLA
   14 FORMAT (/,2X,'MEAN CHLOROPHYLL A 0-5M DURING STRATIFICATON IS',
     1F7.2,//)
C  WRITE TO TAPE4
      WRITE (4,60) USGS,PLAKE,CHLA,DEPMN
   60 FORMAT (F8.0,2F8.2,F8.1)
  107 CONTINUE
      STOP
      END
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UNIVERSITY OF MINNESOTA FORTRAN COMPILER (VERSION 5.4 - 79/03/01) ON THE 6400 UNDER KRONOS 2.1.0 ON 83/04/01 AT 16.03
UNIVERSITY COMPUTING CENTER - UNIVERSITY OF COLORADO

                MNF,E=4,I=DILEFP,L=LLL.

 1    000000B          PROGRAM DILEFP (INPUT,OUTPUT,TAPE4,TAPE2,TAPE5)
 2.   010335B          REAL PCON
 3.   010335B          DO 61 II=1,1
                C   THIS IS THE EFFECTS ELEMENT OF THE DILLON CLEAN
                C   LAKES MODEL.  IT TAKES INPUT FROM THE TROPHIC
                C   STATUS ELEMENT ON TAPE4 AND WRITES TO TAPE2.
                C   THE PURPOSE OF THE PROGRAM IS TO CONVERT
                C   THE PREDICTED CHANGES IN P CONCENTRATON TO
                C   PREDICTED CHANGES IN OTHER VARIABLES OF INTEREST.
 4.   010337B          READ (4,1) USGS,PCON,CHLA,ZBAR
 5.   010347B        1 FORMAT (F8.0,2F8.2,F8.1)
                C   PREDICT THE MEAN AND MIN SECCHI DEPTH, JULY-OCTOBER
 6.   010347B          SD=2.4-0.68*ALOG(CHLA)
 7    010353B          SD=EXP(SD)
 8.   010355B          SDMIN=SD*.70
 9.   010356B          WRITE (2,2) SD,SDMIN
10.   010364B        2 FORMAT (//////,36H MEAN SECCHI DEPTH IN M JULY-OCT IS,F5.2,//,
                       135H MIN SECCHI DEPTH IN M JULY-OCT IS,F5.2)
                C   PREDICT THE CARLSON TSI
11    010364B          CTSI=10.*(6.-(SD/ALOG(2.)))
12.   010370B          WRITE (2,3) CTSI
13.   010376B        3 FORMAT (//,32H CARLSON TROPHIC STATE INDEX IS,F5.0)
                C   PREDICT THE AOHD, SUMMER DEPLETION, AND MINIMUM O2
14.   010376B
15.   010376B          AOHD=-0.67*ALOG10(SD)+0.47*ALOG10(ZBAR)+2.45
16.   010406B          AOHD=10.**AOHD
17.   010411B          CORR=-1.43*USGS+296.1
18.   010414B          AOHD=AOHD+CORR
19.   010415B          DEPL=0.00698*AOHD
20.   010416B          IF (DEPL.GT.9.0)DEPL=9.0
21    010421B          BOTO2=9.0-DEPL
22.   010422B          WRITE(2,4)AOHD,DEPL,BOTO2
23.   010431B        4 FORMAT (//,36H THE AREAL HYPOLIM O2 DEFICIENCY IS,F5.0,//,
                       132H THE SUMMER DEPLETION IN PPM IS,F5.1,//,
                       245H THE MINIMUM O2 5M OVER THE BOTTOM IN PPM IS,F5.1)
24.   010431B          PCON=((PCON/7.3)*100.)-100.
25.   010433B          CHLA=((CHLA/7.1)*100.)-100.
26.   010435B          IF (PCON.GT.100.)PCON=100.
27    010440B          IF (CHLA.GT.100.)CHLA=100.
28.   010442B          SD=100.-((SD/2.9)*100.)
29.   010444B          BOTO2=100.-((BOTO2/4.8)*100.)
30.   010447B          WRITE (5,66) PCON,CHLA,SD,BOTO2
31.   010457B       66 FORMAT (3H  1,3H  1,F5.0,/,3H  1,3H  1,5H    0,/,
                       13H  2,3H  2,F5.0,/,3H  2,3H  2,5H    0,/,
                       23H  3,3H  3,F5.0,/,3H  3,3H  3,5H    0,/,
                       33H  4,3H  4,F5.0,/,3H  4,3H  4,5H    0,/,
                       43H  5,3H  0,5H    0,/,3H  5,3H  5,5H    0,/,
                       53H  6,3H1.8,5H  13.,/,3H  6,3H2.2,3H13.,/,
                       63H  7,3H1.8,5H-59.0,/,3H  7,3H2.2,5H-59.0)
32.   010457B       61 CONTINUE
33.   010461B          STOP
34.   010462B          END
```

UNIVERSITY COMPUTING CENTER
UNIVERSITY OF COLORADO

S P S S - - STATISTICAL PACKAGE FOR THE SOCIAL SCIENCES

VERSION 8.3 (NOS) -- MAY 04, 1982

070000 CM MAXIMUM FIELD LENGTH REQUEST

```
RUN NAME          PLOT
VARIABLE LIST      VAR001 TO VAR003
N OF CASES         14
INPUT MEDIUM       TAPE5
INPUT FORMAT       FIXED(F3.0,F3.0,F5.0)
```

ACCORDING TO YOUR INPUT FORMAT, VARIABLES ARE TO BE READ AS FOLLOWS

VARIABLE	FORMAT	RECORD	COLUMNS
VAR001	F 3. 0	1	1- 3
VAR002	F 3. 0	1	4- 6
VAR003	F 5. 0	1	7- 11

THE INPUT FORMAT PROVIDES FOR 3 VARIABLES. 3 WILL BE READ.
IT PROVIDES FOR 1 RECORDS (*CARDS*) PER CASE.
A MAXIMUM OF 11 *COLUMNS* ARE USED ON A RECORD.

CPU TIME REQUIRED.. .115 SECONDS

```
PLOT               PLOTS=VAR003(-100,100) WITH VAR002(0,5) BY VAR001(1,7)/
                    TITLE=AS APPROPRIATE/
                   TITLEX=  TOTAL P      CHLA      SECCHI    BOTTOM 02/
                   TITLEY=BETTER,PERCENT                    WORSE,PERCENT/
                   SIZE=25,25/
                     SYMBOLS=0,0,0,0,0,-3,-3/
                    PLOTS=VAR003(-100,100)WITH VAR002(0,5) BY VAR001(1,7)/
                       TITLE=AS APPROPRIATE/
                   TITLEX=  TOTAL P    CHLA    SECCHI   BOTTOM 02/
                   TITLEY=BETTER,PERCENT                WORSE,PERCENT/
                   SIZE=9,9/
                    XDIV=10/
                    YDIV=10/
                   SYMBOLS=0,0,0,0,0,-3,-3/
READ INPUT DATA
```

GIVEN 3 VARIABLES, INITIAL CM ALLOWS FOR 271 CASES
MAXIMUM CM ALLOWS FOR 949 CASES

- - - WARNING - - -

NO PLOTS WILL BE PRODUCED UNLESS THE PLOT FILE (TAPE99) IS DISPOSED.
THE FOLLOWING JOB SETUP WILL YIELD MEDIUM LINE ON PLAIN PAPER-

```
SPSS.
DISPOSE,TAPE99,GP=CP3.
```

EE THE CALCOMP MANUAL FOR OTHER PAPER/PEN COMBINATIONS.

OPTION - 1
IGNORE MISSING VALUE INDICATORS
(NO MISSING VALUES DEFINED...OPTION 1 MAY HAVE BEEN FORCED)

PLOTS WILL NEED APPROXIMATELY 53 INCHES OF PAPER

FILE NONAME (CREATION DATE = 83/04/01)

PLOT 1
ON THE X-AXIS..
VAR002
ON THE Y-AXIS..
VAR003

1	VALUES PLOTTED -	2
2	VALUES PLOTTED -	2
3	VALUES PLOTTED -	2
4	VALUES PLOTTED -	2
5	VALUES PLOTTED -	2
6	VALUES PLOTTED -	2
7	VALUES PLOTTED -	2

PLOT 1.
ON THE X-AXIS...
VAR002
ON THE Y-AXIS..
VAR003

1	VALUES PLOTTED -	2
2	VALUES PLOTTED -	2
3	VALUES PLOTTED -	2
4	VALUES PLOTTED -	2
5	VALUES PLOTTED -	2
6	VALUES PLOTTED -	2
7	VALUES PLOTTED -	2

CPU TIME REQUIRED.. .835 SECONDS

FINISH

TOTAL CPU TIME USED.. .965 SECONDS

RUN COMPLETED

NUMBER OF CONTROL CARDS READ 21
NUMBER OF ERRORS DETECTED 0

GROSS P YIELD FOR WATERSHED SEGMENTS, KG PER YR

STATION NAME	MM RUNOFF	AREA HA	RES SEW	URB SEW	SEPTIC	HIWAY	SKISLOPE	PKG PLT	TERT PLT	AMAX	BKGRND
BR1	424.0	25.0	0	0	0	0	0	0	314.9	0	.8
BR2	424.0	11003.0	.7	0	107.6	0	0	7.6	0	0	346.3
BR4	231.0	9719.0	7.3	11.2	149.2	0	25.1	0	0	0	133.0
BR6	253.0	6133.0	.1	0	148.4	0	0	0	0	0	95.1
BR7	235.0	5165.0	.9	0	182.8	0	0	48.1	0	0	72.3
SR1	319.0	1041.0	7.3	0	0	0	0	2.7	0	0	22.2
SR2	310.0	2678.0	10.1	0	17.1	0	11.5	0	0	0	54.8
SR3	500.0	9789.0	0	0	51.2	0	0	0	0	0	386.3
SR4	364.0	4122.0	0	0	0	0	5.1	24.3	0	0	105.2
TM1	433.0	6329.0	0	2.3	46.6	48.6	0	0	0	0	205.0
TM4	406.0	11422.0	0	2.5	0	69.0	19.6	0	117.5	0	338.7
TM7	513.0	6447.0	0	0	0	0	0	0	0	259.7	263.5
MC2	325.0	1800.0	0	.4	35.8	0	0	0	0	0	39.3
SC2	77.0	2212.0	.3	0	0	0	0	0	0	0	6.7
AAA	325.0	1143.0	2.1	10.0	1.9	0	0	0	0	0	25.0
BBB	325.0	1226.0	4.4	0	0	0	0	0	151.8	0	26.8
CCC	325.0	619.0	0	0	12.5	0	0	0	0	0	13.5
DDD	325.0	33.0	0	5.3	0	0	0	0	155.1	0	.7

EEE	325.0	1817.0	0	0	11.9	19.3	0	0	0	0	39.7

GROSS N YIELD FOR WATERSHED SEGMENTS, KG PER YR

STATION NAME	MM RUNOFF	AREA HA	RES SEW	URB SEW	SEPTIC	HIWAY	SKISLOPE	PKG PLT	TERT PLT	AMAX	BKGRND
BR1	424.0	25.0	0	0	0	0	0	0	8019.9	0	22.7
BR2	424.0	11003.0	36.3	0	5952.6	0	0	33.5	0	0	9972.5
BR4	231.0	9719.0	403.6	933.5	7129.0	0	162.1	0	0	0	4370.6
BR6	253.0	6133.0	3.5	0	7248.1	0	0	0	0	0	3063.3
BR7	235.0	5165.0	52.0	0	8771.7	0	0	213.1	0	0	2369.2
SR1	319.0	1041.0	377.7	0	0	0	0	12.0	0	0	679.4
SR2	310.0	2678.0	524.6	0	878.6	0	74.3	0	0	0	1691.0
SR3	500.0	9789.0	0	0	2945.3	0	0	0	0	0	10731.5
SR4	364.0	4122.0	0	0	0	0	32.8	107.7	0	0	3132.8
TM1	433.0	6329.0	0	165.9	2588.9	684.7	0	0	0	0	5877.0
TM4	406.0	11422.0	0	184.7	0	1015.6	126.8	0	2992.5	0	9846.8
TM7	513.0	6447.0	0	0	0	0	0	0	0	52814.5	7280.2
MC2	325.0	1800.0	0	27.9	1856.6	0	0	0	0	0	1200.3
SC2	77.0	2212.0	23.7	0	0	0	0	0	0	0	280.0
AAA	325.0	1143.0	107.0	775.4	100.7	0	0	0	0	0	762.2
BBB	325.0	1226.0	227.5	0	0	0	0	0	3866.3	0	817.6
CCC	325.0	619.0	0	0	647.7	0	0	0	0	0	412.8
DDD	325.0	33.0	0	408.5	0	0	0	0	3950.1	0	22.0
EEE	325.0	1817.0	0	0	618.9	330.1	0	0	0	0	1211.7

GROSS P AND N YIELD BY SEGMENT, KG PER YR

NAME	PTOTAL	NTOTAL
BR1	316	8011

BR2	462.	15995.
BR4	326.	12999.
BR6	244.	10315.
BR7	304.	11406.
SR1	32.	1069.
SR2	94.	3169.
SR3	437	13677.
SR4	135.	3273.
TM1	302.	9316.
TM4	547	14166.
TM7	523.	60095.
MC2	75.	3085.
SC2	7.	304.
AAA	39.	1745.
BBB	183.	4911
CCC	26.	1060.
DDD	161.	4381.
EEE	71.	2161

GROSS KG PER YR P IN RUNOFF IS, 4435.
GROSS KG PER YR N IN RUNOFF IS, 185155.

GROSS YIELD BY SOURCE KG PER YR P(ABOVE),N(BELOW)

RESID. SEWER	URB SEW	SEPTIC	HIWAY	SKISLOPE	PKG PLT	TERT PLT	AMAX	BKGRND	PRECIP	GRNDWAT	DIVERSION	CONSTRUCT
33.2	31.6	765.0	136.9	61.3	82.6	739.3	259.7	2175.1	726.0	10.0	0	150.0
1755.9	2496.0	38738.0	2030.3	396.1	366.3	18828.8	52814.5	63743.4	9872.0	1068.0	0	3985.7

VALUES UP TO THIS POINT UNCORRECTED FOR
INVENTORY CHANGE IN RIVER BOTTOMS, NOW CORRECT THIS.

ET LOADING BY SOURCE KG PER YR P(ABOVE),N(BELOW)

ESID.	SEWER	URB SEW	SEPTIC	HIWAY	SKISLOPE	PKG PLT	TERT PLT	AMAX	BKGRND	PRECIP	GRNDWAT	DIVERSION	CONSTRUCT
	32.2	30.6	740.5	132.5	59.3	80.0	735.5	251.4	2105.5	726.0	10.0	0	145.2
	1901.1	2702.4	41941.6	2198.2	428.8	396.6	19076.3	57182.3	69015.0	9872.0	1068.0	0	4315.3

IE NET YIELDS BY PERCENT ARE P(ABOVE),N(BELOW)

ESID.	SEWER	URB SEW	SEPTIC	HIWAY	SKISLOPE	PKG PLT	TERT PLT	AMAX	BKGRND	PRECIP	GRNDWAT	DIVERSION	CONSTRUCT
	.6	.6	14.7	2.6	1.2	1.6	14.6	5.0	41.7	14.4	.2	0	2.9
	.9	1.3	20.0	1.0	.2	.2	9.1	27.2	32.8	4.7	.5	0	2.1

THE NET P YIELD TO THE LAKE IN KG PER YEAR IS, 5049.
THE NET N YIELD TO THE LAKE IN KG PER YEAR IS, 210098.

MONTH	*DAY*	*LEVEL FT*	*MIL M3MO ROFF*	*MIL M3MO DIVER*	*MIL M3MO OUT*	*KG P NONPREC*	*KG P PREC*
1.	15.	8981.	5.35	0	12.54	84.2	45.7
2.	45.	8977.	4.42	0	11 13	69.6	45.7
3.	74.	8974.	5.76	0	9.07	90.6	45.7
4.	105.	8974.	8.30	0	4.68	130.5	45.7
5.	135.	8980.	34.29	0	3.82	538.9	45.7
6.	166.	9000.	92.61	0	6.15	1455.7	217.8
7	196.	9018.	56.88	0	30.38	893.9	45.7
8.	227	9018.	28.06	0	26.53	441.0	45.7
9.	258.	9018.	15.28	0	15.35	240.2	45.7
10.	288.	9017	10.50	0	10.17	165.1	45.7
11	319.	9016.	6.80	0	13.12	107.0	45.7
12.	349.	9015.	6.76	0	5.79	106.3	45.7

MEAN LAKE AREA OVER THE YEAR IN HECTARES IS 1130.
AVERAGE LAKE VOLUME OVER THE YEAR IN M3 IS 262365960.
TOTAL ANNUAL INFLOW IN M3 INCLUDING DIVERSION IS 2750-10000.

TOTAL ANNUAL OUTFLOW IN M3 IS 148730000.
TOTAL LOADING IN KG PER DAY IS 13.819

AREAL LOAD IN KG PER M2 PER YR IS .446.2
VOLUME LOAD IN KG PER M3 PER YR IS .19.2

MEAN DEPTH IN M IS 23.2

AU IN YR IS .95
PREDICTED P CONCENTRATION IN TOP 15 M IS 7.9

MEAN CHLOROPHYLL A 0-5M DURING STRATIFICATON IS 7.97

MEAN SECCHI DEPTH IN M JULY-OCT IS, 2.69

MIN SECCHI DEPTH IN M JULY-OCT IS, 1.88

CARLSON TROPHIC STATE INDEX IS, 21.

THE AREAL HYPOLIM O2 DEFICIENCY IS, 631

THE SUMMER DEPLETION IN PPM IS, 4.4

THE MINIMUM O2 5M OVER THE BOTTOM IN PPM IS, 4.6

DATE. 1983/04/01

JOB. 18669 F253CCV REEL. 140652 START..1641:30 STOP..1641:50
ACCT PAGES. .000022 STATUS. .80008000

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