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EFFECTS OF DIVERSION AND ALUM APPLICATION ON TWO EUTROPHIC LAKES

Environmental Research Laboratory
Office of Research and Development
U.S. Environmental Protection Agency
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EFFECTS OF DIVERSION AND ALUM APPLICATION ON TWO EUTROPHIC LAKES

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FOREWORD

Effective regulatory and enforcement actions by the Environmental Protection Agency would be virtually impossible without sound scientific data on pollutants and their impact on environmental stability and human health. Responsibility for building this data base has been assigned to EPA's Office of Research and Development and its 15 major field installations, one of which is the Corvallis Environmental Research Laboratory (CERL), in Oregon.

The primary mission of the Corvallis Laboratory is research on the effects of environmental pollutants on terrestrial, freshwater, and marine ecosystems; the behavior, effects and control of pollutants in lake systems; and the development of predictive models on the movement of pollutants in the biosphere.

This report provides a valuable contribution to knowledge of lake restoration and eutrophication control especially in areas of diversion and in-lake treatment.

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ABSTRACT

Two questions were asked: (1) Will diversion of septic tank drainage from East (ETL) and West (WTL) Twin Lakes, two shallow (<5M), morphometrically similar, dimictic lakes in Northeastern Ohio, bring about reduced symptoms of eutrophication and (2) Will a maximum hypolimnetic dose of aluminum sulfate to WTL, defined as the amount of alum which can be added before residual dissolved aluminum exceeds 0.05 mg Al/l, reduce internal phosphorus (P) loading and further improve the lake?

A return to normal fecal coliform levels in the study streams and lakes occurred after diversion. While P in streams was reduced, income did not fall to an "acceptable" level (Vollenweider, 1976). Groundwater was found to be an important P source. P in the lakes was lowered, and symptoms of eutrophication, measured as algal standing crop, were reduced just after diversion (1972), but P and biomass levels remained within ranges expected of moderately eutrophic lakes. Had diversion not occurred, the lakes would probably be closed to recreation due to excessive solidum levels and be classified as very eutrophic.

The P content of the lakes varied each year from a late winter high to a pre-thermal stratification low, followed by an increase over the summer, up to 100% of which was due to net internal P loading. Based on the hypothesis that this loading was hypolimnetic in origin, 139.55 M^3 (36,919 gals) of alum were added to the hypolimnion of WTL in July 1977, leaving ETL as a downstream control. Alum was earlier shown to be effective in retarding anaerobic sediment P release for up to two years in bottle lake, and up to 104 days in in situ columns. The dose as defined was shown to be non-toxic to fish and to certain littoral invertebrates.

Hypolimnetic P concentration in 1976 was greatly lowered, but net internal P loading, while reduced, continued at a significant rate, strongly suggesting the littoral as an internal P source.

Epilimnetic P was also lowered, and a Trophic State Index indicated some reduction in symptoms, but WTL remained eutrophic in 1976. No adverse effects or excessive concentrations of aluminum, sulfate or hydrogen ion were observed.

The treatment may continue to be effective, but could be overcome by continued excessive loading resulting from poor land management.

It is suggested that hypolimnetic alum applications may be more effective in large lakes where seiches and vertical entrainment of hypolimnetic P are more significant. The cost for these lakes could be excessive.

Details for determining and applying an alum dose are given.

This report was submitted in fulfillment of Project Number 16010 HCS, Contract Number R 801936, by Kent State University, under the (partial) sponsorship of the U.S. Environmental Protection Agency. This report covers a research period from November 1971 to January 1976, and work was completed in June 1977.

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SECTION 1

INTRODUCTION

Eutrophication is the process of enrichment of lakes with plant nutrients and particulate matter which leads to increased biological production and decreased lake volume. This is a natural process which occurs at varying rates, depending upon factors such as soil type and ratio of drainage area to lake surface area, and upon lake properties such as mean depth and flushing rate. Where activities on the land are intense, the rate may be greatly accelerated by erosion or sewage income, leading to what has been termed "cultural" eutrophication. The symptoms of eutrophication, such as excessive macrophytes, blooms of algae and fish kills may result in greatly decreased recreational, municipal, or industrial use.

Control of algal blooms must ultimately involve a reduction in external income to the lake. Experiments with algal assays (e.g., Miller et al. 1974), in situ enclosures (e.g., Powers et al. 1972), and whole lakes (e.g., Schindler, 1974) have shown that in most instances, a reduction in phosphorus concentration alone will bring about an algal bloom decrease. However, a means of macrophyte control, a more important nuisance than algae in many eutrophic lakes, is less well understood.

While there are few well-documented descriptions of the response of lakes to nutrient diversion, in those cases where diversion was sufficient to result in a large reduction in lake phosphorus concentration, the lakes have exhibited measurable improvements in quality (e.g., Edmondson, 1970) as measured by changes in plankton abundance and productivity.

Most, if not all, of the reported cases of nutrient diversion have involved the control of point sources, such as sewage outfalls. Perhaps a more common problem affecting hundreds of communities in the United States is the eutrophication of smaller, often kettletype lakes, surrounded by dense urbanization. Typically the homes are serviced by septic tanks and are located on small, completely cleared lots close to the lake shore. These lakes may have been somewhat eutrophic from their naturally rich soils, but became culturally eutrophic, with both algae and macrophyte problems, from a combination of septic tank drainage, erosion, storm run-off, and other non-point sources.

Will these lakes exhibit reduced symptoms of eutrophication if one of these several nutrient sources is diverted? One objective of this research was to address this question through long-term monitoring of the degree of

eutrophication of two small lakes after the diversion of septic tank drainage.

Several techniques have been employed to accelerate lake recovery after diversion (see review by Dunst et al. 1974). While some of these procedures are directed toward nutrient removal (e.g., flushing, harvesting) most are intended to control internal sources of nutrients. Mortimer (1971) and others have viewed the anaerobic, hypolimnetic sediments to be a potentially significant source of phosphorus. Stauffer and Lee (1973) have presented evidence to show that at least in large lakes such as Lake Mendota, thermocline erosion and co-occurring transport of P is significant in maintaining the algal blooms in the epilimnion. The second research objective was to determine whether or not the application of aluminum sulfate to the hypolimnion of one of the two lakes to control internal phosphorus loading from the sediments could accelerate the lake's recovery after septic tank diversion.

The organization of the presentation of methods and results in this report includes two principal sections. The Materials and Methods section describes the hydrological and limnological methods and a summary of our approach to aluminum sulfate dosage and application. The Results section contains the basic findings of both the immediate and longer term effects of the alum treatment and a description of the results of septic tank diversion. However, since a significant portion of this research dealt with a step-by-step development of aluminum sulfate dose, effectiveness, and tests for toxicity, a description of these findings is included as an Appendix to the hydrological-limnological findings. Thus the reader may turn promptly to the procedures, results and conclusions with a basic understanding of the aluminum sulfate methodology. Those who desire a more detailed statement of the basis for dose, or a step-by-step summary of how to ascertain dose, may refer to the Appendix and also to Kennedy (1978).

SECTION 2

CONCLUSIONS

A prompt return of groundwater, streams and lakes to near zero fecal coliform levels occurred after septic tank diversion. Leach fields, even when installed in ideal soils, did not operate properly and their use near lakes should be discouraged.

Diversion did not bring phosphorus income to an "acceptable" level (Vollenweider, 1976) over the five year post-diversion monitoring period because the lakes received untreated storm flow and run-off from diverse non-point sources typical of urban areas. Lake phosphorus concentrations remained at levels typical of eutrophic lakes. After an initial reduction in the year following diversion there was little change in the algal standing crop.

A maximum hypolimnetic dose of aluminum sulfate effectively retarded phosphorus release from lake sediments for at least one year. No adverse side effects were observed.

Net internal phsophorus loading was only partially reduced by the hypolimnetic treatment. The untreated upper zone of the lake, probably the littoral zone, was a significant internal source of phosphorus.

The alum treatment reduced epilimnetic phosphorus concentration and algal standing crop, but both remained within ranges expected in eutrophic lakes. The treatment appeared to alter the dominant bloom species from blue-green algae to co-dominance of blue-greens with dinoflagellates and diatoms.

Hypolimnetic aluminum sulfate treatment may be more effective in lakes where vertical entrainment of phosphorus-rich hypolimnetic waters is a dominant internal source of phosphorus. Such lakes are likely to be large and treatment costs may be prohibitive.

Septic tank diversion, unless accompanied by storm water control and land use regulations, will not bring about greatly lowered symptoms of eutrophication. However, diversion is essential to prevent the situation from becoming worse.

SECTION 3

RECOMMENDATIONS

Most lake restoration techniques are aimed at control of internal phosphorus loading. Contrary to expectations in this study, an important source of internal phosphorus loading appeared to be littoral in origin. These sources need to be identified and the use of aluminum sulfate to control internal phosphorus loading confined to those lakes where sediments are the primary source.

The role of groundwater in lakes should be further investigated, including studies of nutrient transport across the mud-water interface. Methods of measuring groundwater flow and knowledge of the distribution of groundwater flow over the lake bottom are especially undeveloped and require further study.

A land use plan to control non-point nutrient sources prior to lake shore development is essential. Septic tank systems, even where soils appear to be ideal, should not be used.

Where lake shore development has occurred, protection of the lakes will require, in addition to sewage diversion, a plan to prevent further development on the lake shore and to control non-point sources such as erosion and storm flows.

Macrophytes, which are at least as important as algae nuisances in the typical smaller eutrophic lake, should receive increased attention with regard to both means of control and their role in energy flow and nutrient cycling.

Continued monitoring to assess the long-term effectiveness of the hypolimnetic treatment of West Twin Lake in controlling internal phosphorus loading and reducing lake concentration is needed.

SECTION 3

WATERSHED CHARACTERISTICS

GEOLOGY

The Twin Lakes Watershed is glacial drift overlying [illegible] sandstone or shale bedrock. The flat-lying bedrock strata were deeply cut by a pre-glacial stream, and the resulting topography is a valley [illegible] meters deep and 2 kilometers wide, enclosed by walls of sandstone and shale. The valley has been completely filled with glacial drift. Figure 1 is a diagrammatic section of the buried valley in relation to the Twin Lakes.

The oldest bedrock is the Mississippian-Devonian shale that should be present only under the drift in the deepest part of the valley. Overlying the shale is Berea Sandstone. Interbedded shale [illegible] of the Mississippian Cuyahoga Group lies above the Berea Sandstone [illegible] Sandstone forms the upper part [illegible] Mississippian and Devonian shale [illegible] Illinoian deposits comprised of gravel and [illegible] gravel unit is the principal aquifer [illegible] Twin Lakes Watershed.

Overlying the Illinoian [illegible] Mogadore Till (Winslow and White, [illegible] is a sandy, silty, calcareous till, and [illegible] southern and eastern part of the watershed. [illegible] Till is overlain by Hiram Till, a gray silty [illegible]

The Twin Lakes Watershed lies within [illegible] Kent [illegible] Moraine of early Wisconsin age. [illegible] formation of kettle depressions separated by outwash deposits of sand and gravel. The kettles became [illegible] ice and thus developed [illegible] these Kent Till [illegible] in Figure [illegible]

The principal soil types are [illegible] upland gravel and sandy till (Chili), lowland [illegible] and clay [illegible] lowland peat and muck (Carlisle). These four major soil types [illegible] water holding and transmitting capacities, and potential for [illegible] Ravenna soils on the clay till [illegible] West Twin Lake are [illegible] erosion and have low infiltration capacity [illegible]

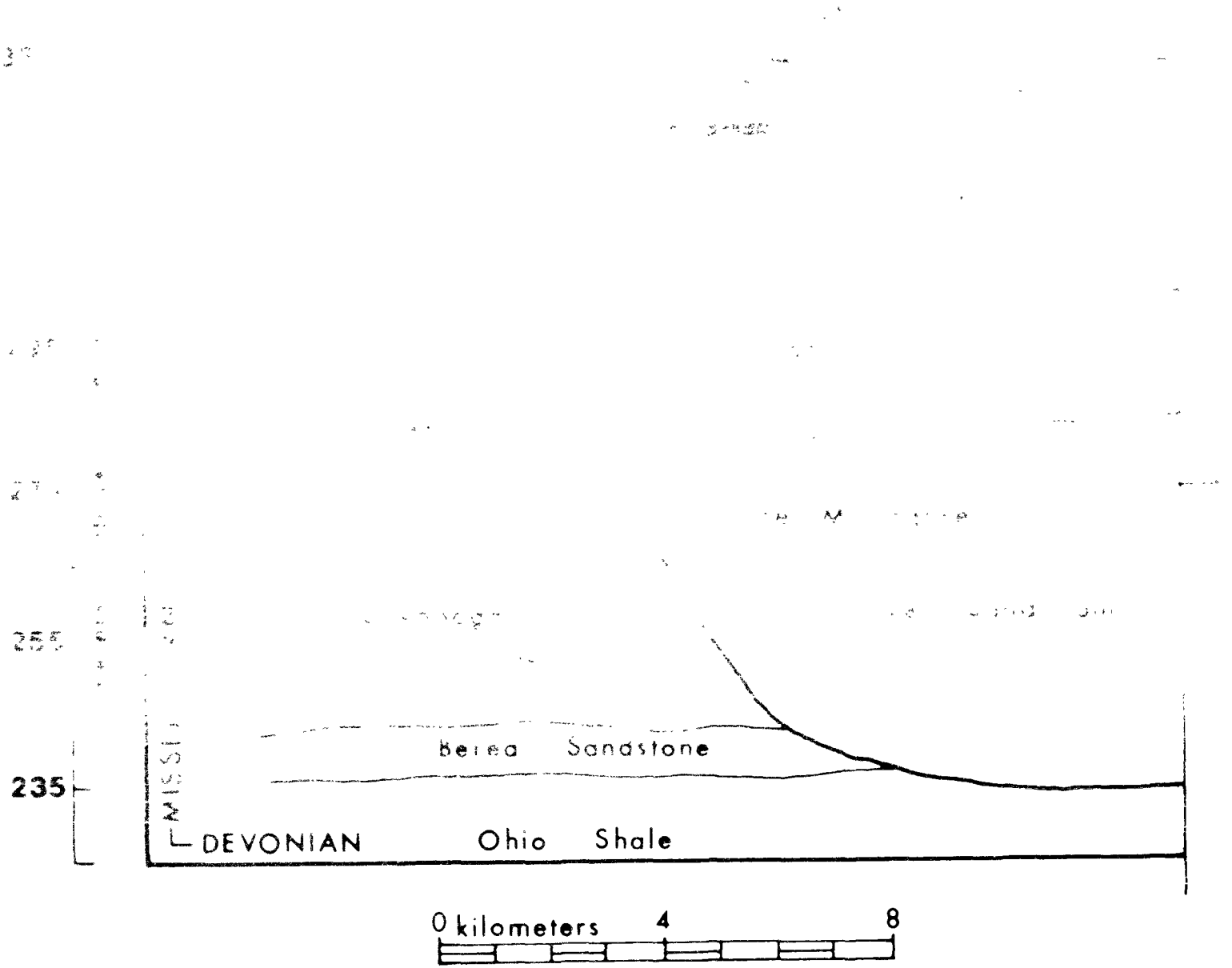
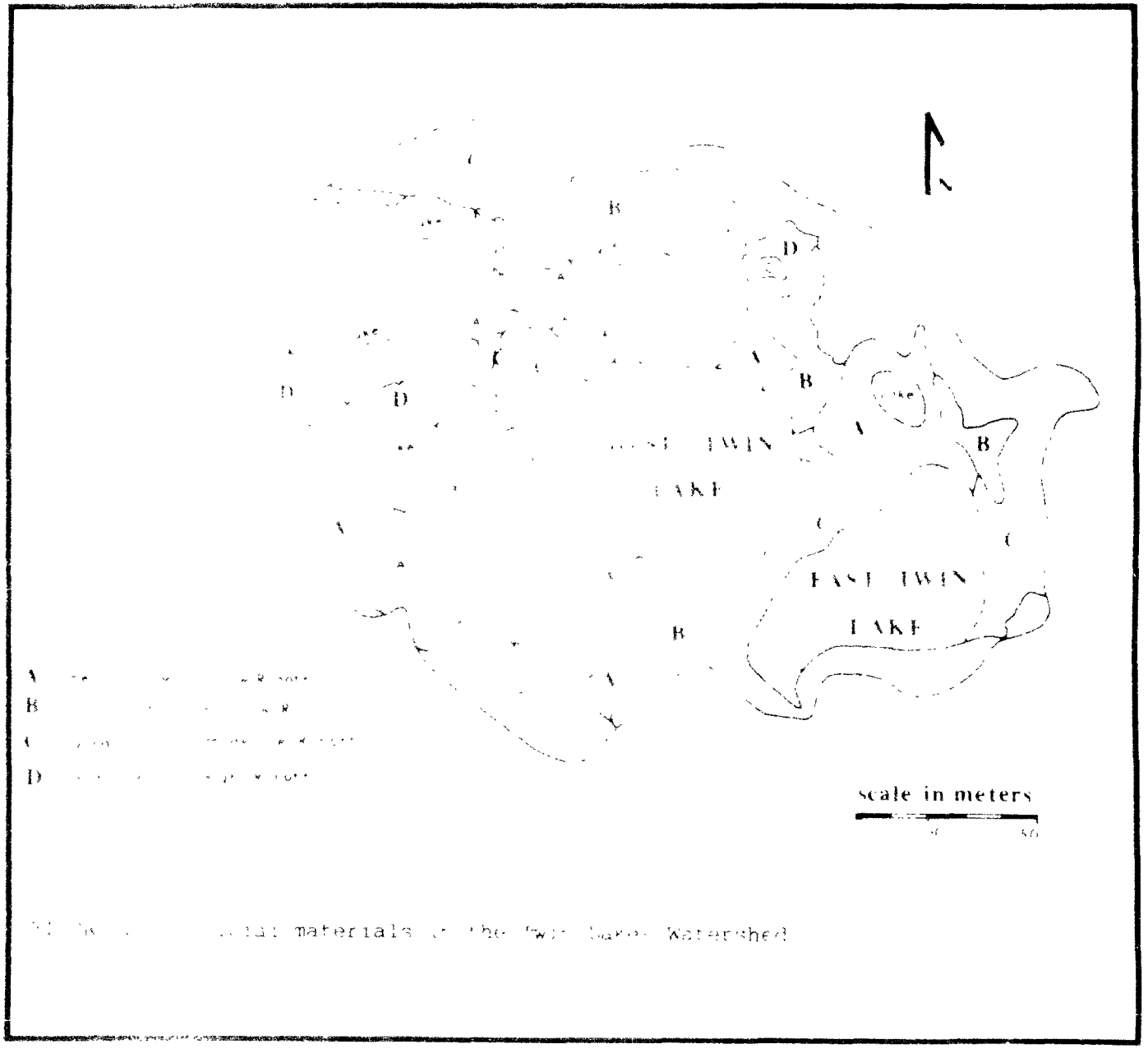


FIGURE 1. Glacial stratigraphy of the Twin Lakes Watershed



[illegible] materials [illegible] the [illegible] Watershed

Chili soils on the south of West Twin and around the edges of East Twin are less susceptible to erosion and have a high infiltration capacity.

The relationship of the soil [illegible] tank installation and subsequent problems was not considered in the [illegible]anization of the watershed. Ravenna type soils are not suitable for sep[illegible] tank installations because of the low permeability of the subsoil (Ritch[illegible] and Powell, 1973). Chili soils are suitable for most septic systems bec[illegible]se they may permit effluent to pass through subsoil. Carlisle muck and peat soils are not suitable for septic leach fields because of high water table conditions.

CULTURAL DEVELOPMENT

Earliest records indicate that by 1803 five families had established farms in the Twin Lakes area (Brown, 1885). Population changed little during the next 100 years. Tax maps filed at the Portage County Courthouse (Ravenna, Ohio) since 1850 show a change in land use from farming and woodlands to pasture (Table 1).

Table 1. LAND USE IN TWIN LAKES AREA, 1850 to 1880

	1850	1880
Plow	[illegible]	32.0%
Pasture	[illegible].6%	55.0%
Woodlot	[illegible].4%	12.5%
Land used	[illegible]	266.4 ha
Population density (persons/ha)	[illegible]	1.90

From Otterson, 1974

At the turn of the century, developers bought the farms surrounding the lakes and by 1918 the major subdivisions around the lakes had been laid out, and the shift from rural to residential use had begun. Today there are 374 houses on the Twin Lakes Watershed. Using an average of 4.2 persons/home (Statistical Abstracts of the United States, [illegible]), there are 1,571 persons and a population density of [illegible] persons per [illegible].

LIMNOLOGY

The principal morphological and hydrological features of East and West Twin Lakes are summarized in Table 2 and illustrated in Figure 3. Averages are for 1972-1976. A more complete description has been presented in Cooke et al. (1973) and Waller et al. (1976).

Table 2. LIMNOLOGICAL FEATURES OF THE TWIN LAKES WATERSHED

Latitude - Longitude	$41^{\circ}12'$ North; $81^{\circ}21'$ West	
Area of watershed (ha)	334.5 (including lakes)	
	West Twin Lake	East Twin Lake
Lake area (ha)	34.02 (30.22*)	26.88
Maximum length (km)	0.65	0.85
Maximum width (km)	0.60	0.50
Volume (M^3)	14.99×10^5	13.50×10^5
Maximum depth (M)	11.50	12.00
Mean depth (M)	4.34 (4.45*)	5.03
Elevation (M)	318.73	318.42
Subwatershed area (ha)	233.02	101.48
Area of other lakes in subwatershed (ha)	15.00	3.00
Mean water residence (yr)	1.28	0.53
Mean areal water load (M/yr)	3.50	9.12
Subwatershed population	1124	1571**

*Shallow canals are contiguous with West Lake, but are excluded as part of the lake in averaging some limnological features.

**Includes the population of the West Twin subwatershed since this lake drains into East Twin.

Both lakes are dimictic, second class lakes (defined by Hutchinson, 1957). In this report, seasons were based on water temperature as follows: winter (4° C), spring (unstratified), summer (metalimnion present), and fall (unstratified). Ice forms in December and remains until early March. The 1971-76 average onset of summer stratification was 15 April for both lakes, and the average summer lasted 210 days in East Twin 198 days in West Twin. By mid-November, both lakes had circulated.

Dissolved oxygen is rapidly depleted from lower waters following the onset of summer stratification and partially depleted after ice formation. Associated with thermal and dissolved oxygen stratification is a marked stratification of phosphorus and nitrogen forms. The annual ranges of nitrogen, alkalinity, pH, conductivity, and transparency are listed in Table 3. Changes in phosphorus concentration with depth and over the seasons are described in the Results section.

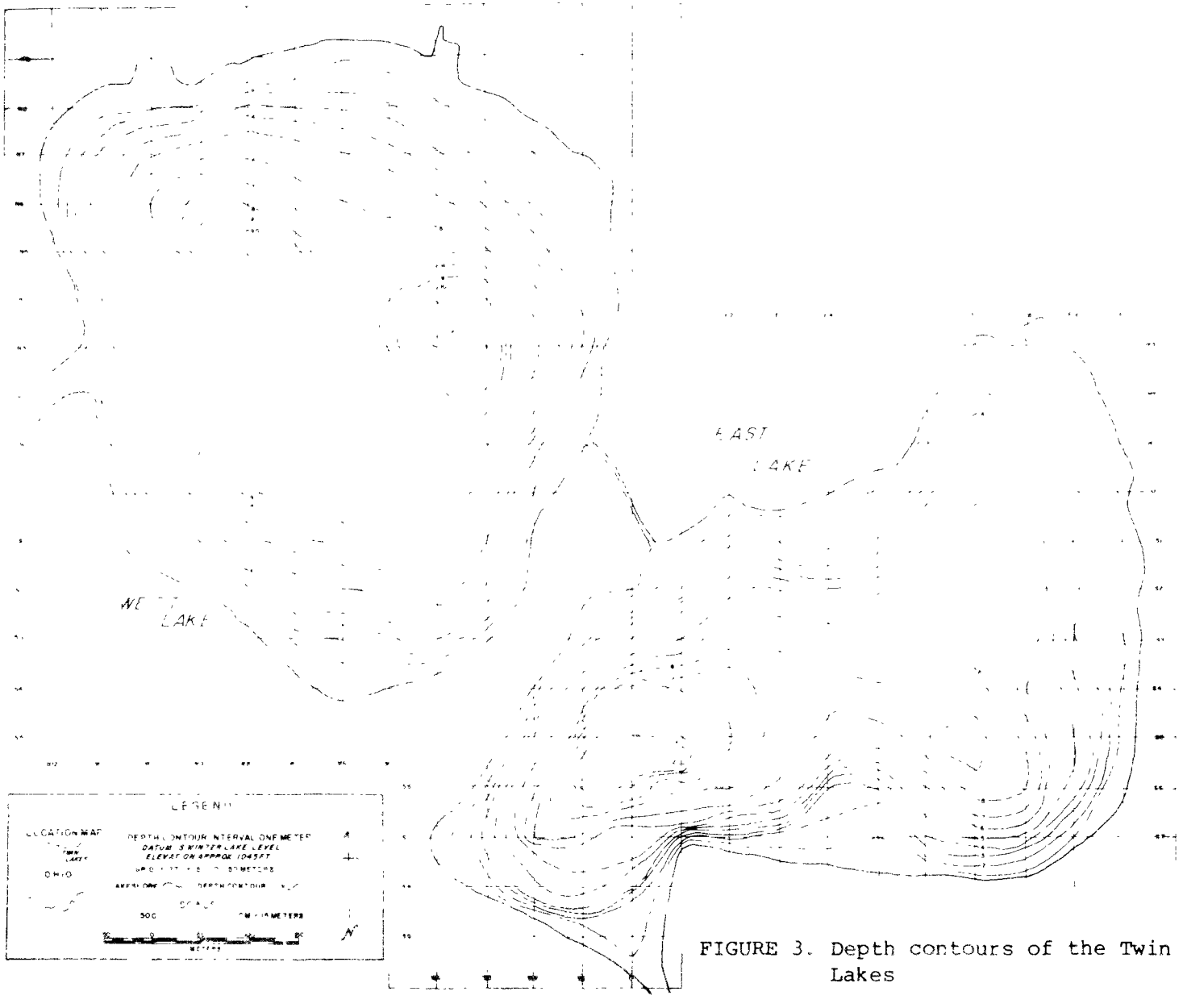


FIGURE 3. Depth contours of the Twin Lakes

Table 3. SEASONAL AND ANNUAL VOLUME-WEIGHTED AVERAGES (1971) OF SELECTED CHEMICAL AND PHYSICAL FEATURES OF THE TWIN LAKES (μg/l)

	Winter	Spring	Summer	Epilim	Hypolim	Fall	Year
1. Ammonia (NH_4-N)							
West Twin	293.4	223.2	745.6	387.3	3288.5	1336.0	673.0
East Twin	296.2	99.3	662.3	318.9	3190.5	924.9	582.2
2. Nitrate-nitrite (NO_3-N)							
West Twin	268.2	100.9	56.8	36.1	80.9	101.4	125.7
East Twin	129.1	69.9	65.7	43.2	102.0	95.6	89.1
3. Conductance (μmho)							
West Twin	411.8	411.8	401.2	393.0	435.0	385.3	402.3
East Twin	375.7	375.7	361.4	352.4	398.0	370.7	367.2
4. pH* (1973)							
West Twin	7.92	7.91	7.58	7.76	7.30	7.67	7.68
East Twin	7.54	7.54	7.53	7.94	7.23	6.81	7.38
*log of mean anti-log							
5. Alkalinity (mg $CaCO_3$/l)							
West Twin	109.4	109.4	107.9	101.9	144.5	115.7	109.9
East Twin	104.3	104.3	104.9	95.0	146.2	105.8	105.0

Characteristics of the littoral, sub-littoral, and profundal surficial (upper 2 cm) sediments were studied by Lardis (1973). Littoral sediments contain mainly decaying vegetation, shell fragments, and allochthonous debris. Sublittoral sediments are brown to black-gray with lesser and varying amounts of decaying vegetation. Profundal muds are black and very fluid. The organic content of dried sediment samples ranges from 14% in the littoral to about 40% in the profundal. The organic phosphorus content of the muds increases with depth of overlying water and is significantly ($p < .05$) less in ETL (mean=66 μg P/g) than in WTL (mean=85 μg P/g). The phosphorus content of profundal muds declines after the onset of thermal stratification.

The flora of the lakes are dominated by a large macrophyte community which covers about 25% of the lake area (Rogers, 1974) and by a succession of phytoplankton blooms. Potamogeton crispus is the dominant macrophyte in April, May, and June; late summer growth is dominated by dense populations of Elodea canadensis, Chara vulgaris, Najas flexilis, and Ceratophyllum demersum. After ice-out a bloom of diatoms, usually Fragilaria crotenensis and Asterionella formosa, occurs, followed by an abrupt change in April or May when high water clarity and low plankton density is typical. Oscillatoria rubescens, common in the spring, is the dominant species of the metalimnion. In the summer a series of blue-green algae blooms occurs, culminating in a heavy bloom of Aphanizominon flos-aquae or Anabaena limnetica in August and September.

The zooplankton have been studied by Heinz (1971). Large populations are found in spring and fall, dominated by Daphnia galeata mendotae and Cyclops bicuspidatus thomasae. The bottom fauna have not been examined quantitatively, but appear to be dominated by Chaoborus. The fish community is composed largely of members of the Centrarchidae.

SECTION 5

MATERIALS AND METHODS

HYDROLOGY

The equation for a hydrologic equilibrium was used to measure the water balance of the lakes from 1972 through 1976. It was assumed that the following equation must balance each year:

inflow + precipitation = outflow + change in storage.

Inflow was incoming streams + groundwater + overland flow, and outflow was outgoing streams + evaporation + groundwater.

Precipitation was [illegible] weighing bucket rain gauge located on the north side of the [illegible] station is shown on Figure 4. Data were [illegible] from nearby [illegible] km east) Lake Rockwell and from Kent (8 km [illegible] normal or 30 year average precipitation was taken from the Hiram [illegible] estimates 24 km north of Twin Lakes. For 1974-76 precipitation at [illegible] was [illegible] Hiram [illegible].

Evaporation was estimated for 197[illegible] temperature from the Hiram station [illegible] and [illegible] evaporation with a recording evaporimeter and [illegible] was placed near the lake for the spring, summer, and fall [illegible] gathered during periods of freezing temperatures. [illegible] installed on the roof of a low building in Kent. For the years 1975 and [illegible] evaporation data from Coshocton, Ohio, 160 km south of the watershed, were used.

The overland flow from individual precipitation events was computed using the U.S. Soil Conservation Service technique described by Mockus (1969). This technique is based on soil infiltration characteristics, 5-day antecedent precipitation, and land use. No actual measurements of overland flow were made.

Surface flow into West Twin, East Twin, and [illegible] Lakes (Figure 4) was measured at 14 stations. Flow at six stations was perennial and was continuously monitored at three locations. Table 4 gives the stream [illegible], type of flow, and measuring technique. The technique employed was determined by the flow regime of the stream and the stream gradient. Stream 8S, perennial stream discharging from West Twin Lake was [illegible] monitored

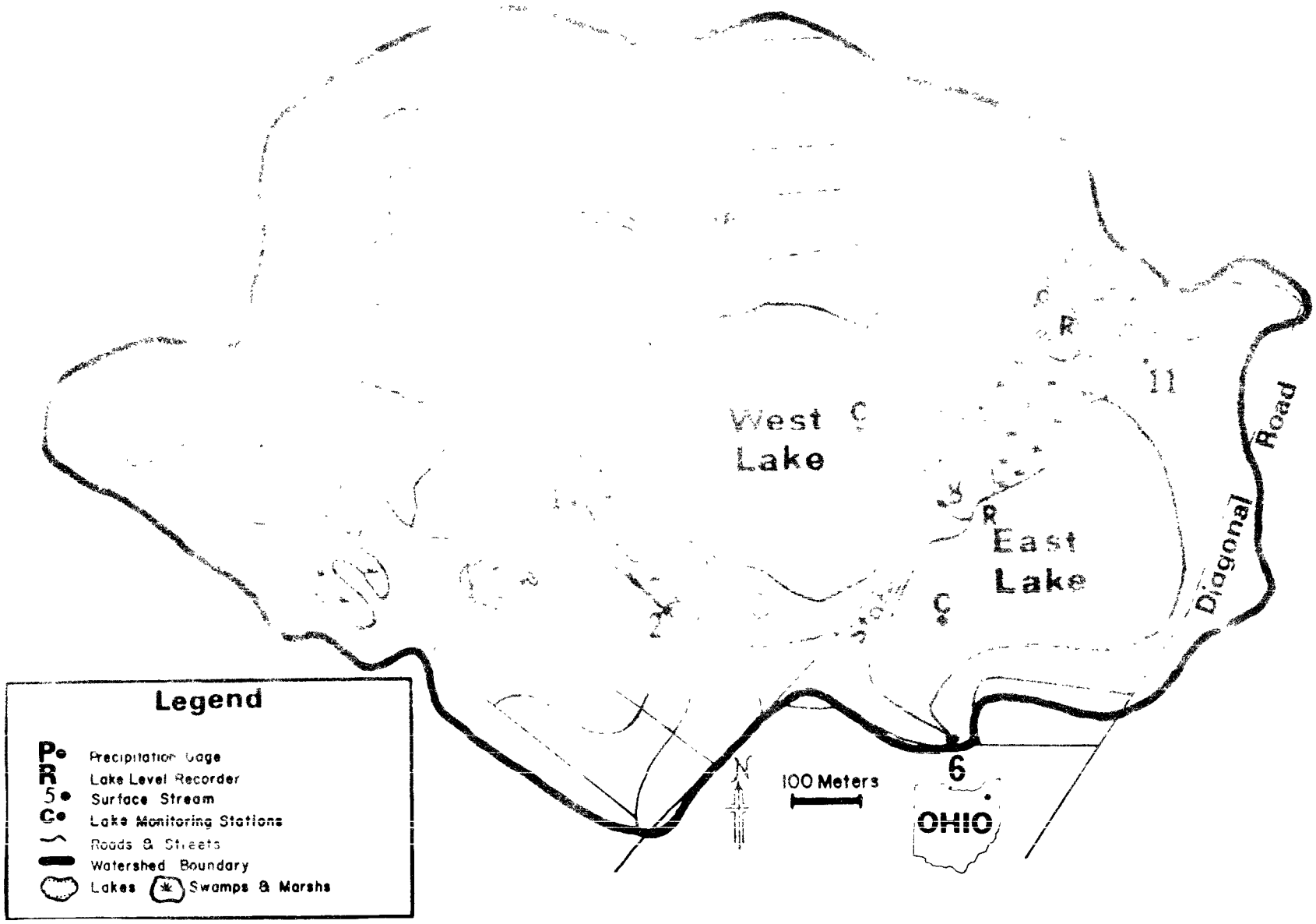


FIGURE 4. Location of streams entering Twin Lakes and Dollar Lake

Table 4. SOURCE, RECEIVING WATER AND MEASURING TECHNIQUES OF SURFACE STREAMS IN THE TWIN LAKES WATERSHED.

Stream No.	Source	Receiving Water	Flow	Measurement Method
1S	Pond overflow and storm drain	West Twin	Perennial	90° V-notch weir, stilling well and recorder
2S	Street drain and depression	West Twin	Perennial	90° V-notch weir, stilling well and recorder
3S	Island [illegible] overflow	[illegible] Twin	Dry in summer	[illegible] stilling well [illegible]
4S	Lake overflow and street drain	[illegible] Twin	In response to storms	[illegible] discharge
5S	Storm water in drain area	[illegible] Twin	In response to storms	[illegible] discharge
6S	East Twin Lake	[illegible] River watershed	Perennial	[illegible] recorder [illegible] stilling [illegible]
7S (4D)	Stormwater	[illegible]	In Response to storms	Culvert discharge
8S	West Twin Lake	[illegible] Twin [illegible]	Irregular in summer	Current meter and stage recorder 1972-1973
10S	House foundation drain	[illegible] Twin	Perennial	Culvert discharge
11S	Storm flow wooded area	East Twin	In response to storms	90° V-notch weir & bucket
12S (2D)	House foundation & storm drain	Dollar Lake	Perennial	Culvert discharge
1D	Foundation and lawn drainage	Dollar Lake	In response to storms	Culvert discharge
3D	Storm flow urban area	Dollar Lake	In response to storms	60° V-notch weir & stilling basin
5D	Storm flow-urban area	Dollar Lake	In response to storms	90° V-notch weir & measuring bucket.

continuously because of low gradient. This stream connects West Twin and East Twin Lake, and on occasion during the dry summer months the flow reversed so that East Twin Lake flowed into West Twin Lake. No structure was built on stream 5S because of flooding potential in and around this shallow channel low gradient stream. Flow into Dollar Lake was monitored because this small lake is separated only by a Sphagnum marsh from East Twin and thus is part of East's water budget.

Five streams, 1S, 2S, 3S, 4S, and 5S enter West Twin Lake. Three (1S, 2S, and 3S), which account for 50-75% of the annual stream income, were measured continuously or at a minimum, weekly. Two streams, 4S, an upland man-made lake outlet, and 5S, an urban stormflow channel, were measured infrequently in 1972 and not at all in 1973 and 1974. Their 1972-74 flows were estimated by regression equations based on data from 1975. The estimate of 4S outflow in 1972-74 was obtained by a seasonal linear regression of its daily 1975 outflow to that of daily outflow from stream 3S. Both streams are upland lake outflows which discharge to West Twin. The correlation coefficients were: summer [illegible] = 0.500 [illegible] = 0.[illegible]8, and spring [illegible] 1000 M^3 /day, r = 0.690; and spring [illegible] M^3 /day r = 0.805. Winter outflows of 4S were assumed to be a constant proportion of the outflow of 3S, based on the 1975 ratio of winter outflows. Stream [illegible] the urban drainage, was estimated by a log-log regression of its 1975 [illegible] flow against the 1975 monthly flow of all other streams entering West Twin [illegible] 0.481 for all seasons). In 1975 the rate of flow of all streams in the watershed was measured continuously or daily.

Change in lake water levels was [illegible] with a water level recorder and the data used to estimate change in storage.

To determine groundwater contribution to the Twin Lakes, a combination of lakeside wells and in-lake piezometers was installed. In 1971 16 wells were drilled in the watershed to determine the elevation of the water table. These wells established that groundwater flow was towards the lakes. Hydraulic conductivities were determined on each well using the slug injection method (Ferris et al., 1962). In 1972 and 1973 an additional 20 wells were drilled on the periphery of the lakes. Stratigraphy of the near shore wells showed an upper layer of saturated sand overlying a silt and clay layer. Shallow groundwater flow to the lakes was assumed to be the water passing through the upper sand layer into the lake. The hydraulic gradient was determined from the water table elevations at the phreatic divide and the elevation of the lake divided by the distance from the lake to the phreatic divide. Each lake was divided into 16 "pie" slices (Cooke et al. 1973), and total shallow flow was the sum of the flow across each of the slices. Measurements of the water table levels at the divide and at the lake showed only minor monthly fluctuations so the hydraulic gradients for each slice were assumed constant. Hydraulic conductivities for the tested wells varied by less than one third of an order of magnitude, so a constant value for hydraulic conductivity was used. The seepage value determined for West Twin Lake was 196 M^3 /day and 137 M^3 /day for East Twin Lake.

Deep groundwater flow was computed by installing two deep piezometers (13 and 6 meters deep) into the aquifer beneath the confining silt layer. Using the hydraulic gradient determined from these two wells as a vertical gradient, and the hydraulic conductivities of the surface, seepage flow was computed for the area of the lake basins. These values were 646 M^3 /day for West Twin and 535 M^3 /day for East Twin. These values were used in water budget computations in the first published reports from this project (Cooke et al. 1973).

Szczepanowski (1976) attempted to verify the flow rates into East Twin obtained from lakeside wells by installing piezometers in the lake. Thirty-two piezometers in 13 nests were installed to depths ranging from 0.63 to 17.5 meters below the water surface. The deepest penetration into the lake mud was 8.5 meters. His data gave an average inflow of 207.1 M^3 /day for East Twin Lake from both shallow and deep systems, a value 3.2 times lower than that computed using the flow net technique. His rate was believed to be too low because deep spring flow was not measured.

Analysis of the water budget and the lake outflow of East Twin suggested that the values of both 672 M^3 /day (137+535 M^3 /day) and 207.1 M^3 /day were probably too low; a high residual inflow value could not be accounted for. In 1975 daily stream monitoring and careful analysis of storm flows, precipitation and evaporation still did not eliminate a high residual.

Over the five-year monitoring period groundwater inflow was thus estimated by the use of lakewide wells (Buller, 1974), piezometers (Otterson, 1974; Szczepanowski, 1976), by the use of thermal gradient and seepage meters (Cooke et al. 1975), and by lake water balance residuals (Cooke et al. 1977). A comparison of these results is given in Table 5. No single technique proved to be entirely satisfactory in determining the groundwater inflow component, and the rates obtained varied by over an order of magnitude.

Table 5. RATE OF ANNUAL GROUNDWATER FLOW INTO THE TWIN LAKES, COMPUTED FROM FIVE TECHNIQUES

Method	Rate (M^3/yr)
Lakewide wells and flow net	2.5×10^5
Piezometers in the lake	7.6×10^4
Thermal gradient	3.6×10^6
Seepage meters	9.1×10^5
Water balance residual	1.1×10^5 to 7.0×10^5

In contrast to all earlier reports of the water-phosphorus budget of the Twin Lakes, the monthly groundwater inflow in this report was obtained by assigning all residual water income from the equation for a hydrologic equilibrium to groundwater. Measured groundwater flow rates were not used. If the value was negative, it was assumed that water was lost from the lakes via groundwater. This technique was chosen since much of the residual was probably interflow which was not measured, because there are numerous deep springs also not measured, and because accurate measures of surface flows, particularly in 1975, were obtained. When an alternative method was employed, as was done in Cooke et al. (1973), Cooke and McComas (1974), Cooke et al. (1975), and Waller et al. (1976), then the monthly residual was assigned to overland flow, giving this component of the equation an unrealistically high flow.

The methods employed [illegible] chemical analysis of incoming water (phosphorus, coliform bacteria [illegible] described in the Limnology section.

LIMNOLOGY

Each lake was sampled weekly over the deepest part (Figure 3), from the time of ice-out in the early spring until ice formation, and then weekly through the winter [illegible] permitted by ice conditions.

Temperature was measured to the nearest 0.5° C at one-meter intervals using a YSI resistance thermometer. Transparency was determined with a 20 cm diameter Secchi disc, with black and white alternating quadrats, and recorded to the nearest 0.1 M, averaged from three readings. Lake water was collected at one-meter intervals with a 1.9 l Kemmerer bottle and placed into 300 ml glass-stoppered BOD bottles for determination of dissolved oxygen concentration by the azide modification of the Winkler method (APHA 1965).

Lake water samples were also collected at 1 M intervals, placed in acid-rinsed polyethylene bottles, and returned to the laboratory for physical, chemical, or biological determinations. The following determinations were performed on whole lake water: conductivity, pH, alkalinity, total phosphorus and alkaline phosphatase. The following were performed on water filtered through a Whatman GF/A glass fiber filter: total phosphorus (to yield a "filterable total phosphorus" (FTP) estimation) sulfate, and aluminum.

Conductivity was determined with a YSI Model 31 conductivity bridge, pH with a Corning Model 7 pH meter, and alkalinity by titration of whole lake water with 0.02 N H_2SO_4 to an endpoint of pH 4.5 (APHA, 1965). Soluble reactive phosphorus (SRP) was determined by the Murphy and Riley (1962) procedure on 0.45 µm Millipore filtered water (USEPA, 1971). Total phosphorus (TP) was determined by the above colorimetric procedure, following persulfate-sulfuric acid digestion of unfiltered water (USEPA, 1971). Filterable unreactive phosphorus (FUP) was calculated as FTP-SRP. Alkaline phosphatase activity was determined by incubating 2.5 ml of whole lake water with 0.3 ml (0.1 M tris, .01 M $MgCl_2$ pH 9.0) and with 0.3 ml (0.3 mg p-nitrophenyl phosphate/ml). Activity was determined as the rate of increase of

absorbance at 395 nm over four hours, measured in 1 cm cuvettes (Heath and Cooke, 1975). Sulfate was determined turbidimetrically (Hach Chemical Co.), and aluminum colorimetrically (Hach Chemical Co.). On these tests the results were compared to our standard curves and not those provided by Hach Chemical Company.

Total seston was determined gravimetrically by filtering 500 ml of lake water onto tared Whatman GF/A glass filters, air dried at 98° C, and weighed to the nearest 0.1 mg without ashing (USEPA, 1971).

Chlorophyll a was determined by filtering 500 ml of lake water onto Whatman GF/A glass filters and extracting with 90 percent Mg CO_3 - buffered acetone in a tissue grinder (Long and Cooke, 1971). The trichromatic method of Strickland and Parsons (1968) was used without acid correction. Often samples were filtered and stored frozen before grinding and extraction.

Phytoplankton cell volume was measured by filtering 25 ml of lake water onto 0.45 µm Millipore (HAWP) filters. These were air dried, cleared with immersion oil, and cells counted at a magnification of 140X. Eleven Whipple fields were counted. Approximate geometric shapes were used to estimate the cell volume (McNabb, 1960).

Coliform bacteria were determined by filtering 100 ml of lake water onto gridded 2.5 cm Millipore 0.45 µm filters. These were incubated on sterile EMB lactose agar at 30° C for 24 hours. Coliforms were identified as the "metallic" colonies (APHA, 1965).

ALUMINUM SULFATE APPLICATION

Although aluminum sulfate (alum) applications to eutrophic lakes have been attempted (e.g., J.O. Peterson et al. 1973), additional studies of effectiveness and longevity were needed, and the dosage guidelines also needed evaluation. Our objectives were to establish dose determination methods, develop application methods, and then evaluate effectiveness, toxicity, and longevity through laboratory and field experiments.

Aluminum sulfate has been used in the clarification of water supplies and in the removal of phosphates from wastewater effluents. Dosage is commonly determined in "jar-tests" in which stepwise additions of alum are made to determine the minimum dose at which maximal removal of either suspended solids or phosphorus can be achieved. Effective dosages used for the removal of phosphorus from wastewater are expressed as aluminum to phosphorus molar ratios since theoretically one mole of aluminum is required to precipitate one mole of phosphorus as $AlPO_4$ (Stumm, 1964). In practice molar ratios of 1.5 to 2.0 are common (Nesbitt, 1969).

S.A. Peterson et al. (1974) suggested that the calculation of aluminum sulfate dosage for lake treatments could be based on molar ratios and reported maximal phosphorus removal (95%) from algal assay medium and pond water at molar ratios of 3.0, 4.0 respectively. Such a method places emphasis on the

removal of phosphorus from the water column. However, the success of any lake treatment will depend on the amount of alum added and the ability of the added alum to reduce the release of phosphorus from anoxic sediments, and only secondarily on the removal of phosphorus from the water column at the time of application. Accordingly, an alternative approach to dose calculation was sought.

The reaction of aluminum with carbonate alkalinity results in the formation of aluminum hydroxide. Since most eutrophic lakes are characteristically alkaline, dose can be based on this reaction:

$$Al_2(SO_4)_3 + 3Ca(HCO_3)_2 \rightleftharpoons 2Al(OH)_3 + 3CaSO_4 + 6CO_2$$

A dose based on alkalinity and not phosphorus concentration allows addition of aluminum sulfate up to a point at which most of the alkalinity is removed, pH values fall, and dissolved aluminum concentration begins to increase. Phosphorus in the water would be removed by entrapment of particulate matter in the polymerized amorphous aluminum hydroxide floc, and by adsorption of dissolved inorganic phosphorus to charged sites on its surface (Lea et al. 1954) during sedimentation. Depending upon the amount of phosphorus in the water column prior to treatment and upon the removal efficiency of the hydroxide, the resulting floc layer [illegible] the sediments could continue to be effective by retaining on [illegible] sediments and thus produce a "chemical barrier" over anoxic sediments [illegible] towards the reservoir of phosphorus in sediments rather than toward that in the water column at the time of application, this approach emphasizes the possibility of a long term means of improving water quality by control of internal loading or release of phosphorus from sediments.

A step-wise experimental program to substantiate this approach to dose determination involved laboratory studies to evaluate floc formation and phosphorus removal characteristics, controlled *in situ* enclosure experiments, laboratory and field toxicity studies, and the full scale pilot treatment of a small lake (Dollar Lake). These experiments are outlined in the Appendix and described in detail in Wilbur (1974) and Kennedy (1978).

It was found that maximum aluminum dose, defined as that dose above which residual dissolved aluminum concentration exceeds 0.05 mg/l, a value which Everhart and Freeman (1973) report can be toxic to fish, was linearly related to alkalinity and could be determined by simple laboratory tests. Dose determination for West Twin Lake is described in the Appendix (Section [illegible]).

The treatment of phosphorus-spiked [illegible] with a maximum aluminum dose and the elution of the [illegible] buffers (pH 6.0, 7.0, 8.0) indicated that phosphorus [illegible] was high (96%) even at high phosphorus concentrations (100 µg/l) and that the floc retained the capacity for continued phosphorus adsorption following sedimentation from the water column. Controlled field testing with columns of lake water (Kennedy, 1978) and field and laboratory toxicity tests (Wilbur, 1974) confirmed that long term (104 days) control of phosphorus release from

sediments could be achieved with an alkalinity-dictated maximum dose without adverse ecological impacts. The decision to treat hypolimnetically was based on results of preliminary enclosure experiments (Kennedy, 1978) in which surface applications of alum resulted in decreased removal efficiency in deeper, phosphorus-rich hypolimnetic [illegible], and to the desire to avoid the possibility of localized toxicity problems in the epilimnion due to decreased pH.

Based on an evaluation of the results of these studies, the decision was made to treat West Twin Lake, maintaining East Twin Lake (100 M downstream) as a diversion-only control. East Twin is not the perfect control since it receives over half its water from West Twin and, therefore, should ultimately benefit from a reduction of phosphorus concentration in West Twin. Dose and application methods developed during the pilot treatment of [illegible] Lake were used. The lateral extent of the upper boundary of the hypolimnion was marked at the surface with buoys and the entire enclosed area divided into treatment quadrats of 100 x 50 M. Measurement of the volume of the hypolimnetic portion of each quadrat and their alkalinities allowed the calculation of the maximum amount of alum to be applied to each. The application of the 139.55 M^3 (36,919 gals) of aluminum sulfate just above the hypolimnion (5 M) required two specially designed barges working 18 hours/day for a total of three days (July 29-31, 1975). A detailed description of the determination, calculation and logistics, including first-stage and [illegible] treatment, can be found in the Appendix. Also included [illegible]

[illegible]

[illegible] two parts. First, since it [illegible] and lower nutrient concentra-[illegible] phosphorus loading through the [illegible] Further, evidence is presented [illegible] phosphorus-limited and [illegible] in concentration. The most [illegible] the water-nutrient budget. This [illegible] of the lake may be altered [illegible], and so the effect of [illegible] immediate and longer-term [illegible] response of the biota to [illegible] final section quantify-[illegible] in a Trophic State [illegible]

[illegible]

[illegible] phytoplankton in freshwater eco-[illegible] lability of phosphorus. The [illegible] at the ecosystem, the [illegible] of organization. Here [illegible] with the view that [illegible] limited.

[illegible] phosphorus is among the [illegible] of the planktonic [illegible] driven to barely [illegible] to establish [illegible], particularly [illegible] (measured as [illegible] (less than 5 µg [illegible] persisted throughout [illegible]

[illegible] cell density attain-[illegible] altered natural lake-[illegible] water contained an [illegible] phosphorus (SRP)

greater than 10, maximum potential [illegible] was phosphorus limited, but with an initial weight ratio (N:P) less than [illegible] maximum potential cell density was nitrogen limited. Determination of inorganic N (NH_4 + NO_3 + NO_2) and SRP were used to examine the Twin Lakes N:P ratio according to this view. Table 6 provides values for 197[illegible] and 197[illegible] (the only years in which NO_3 was determined). Both lakes in each year appeared to be phosphorus-limited lakes from these considerations. This conclusion must be regarded cautiously because it is not certain that this laboratory method would yield similar results in the field and for species other than Aphanizomenon flos-aquae. Schindler (197[illegible]) recently suggest[illegible] that cyanobacteria dynamics are not as simple as the view that Chandler [illegible] Schindler points out that decreasing the N:P ratio in lakes may exacerbate the problem by stimulating bloom [illegible] N-[illegible]

Table [illegible] Ratios [illegible]

WEST TWIN LAKE

	Annual	Winter	[illegible]	[illegible]	[illegible]
1972	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]
197[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]
[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]
197[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]
[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]

[illegible] phosphorus [illegible] concentration [illegible] of [illegible] [illegible]

[illegible]

it may be interesting [illegible] availability [illegible] positions [illegible] 1973 fall with [illegible]

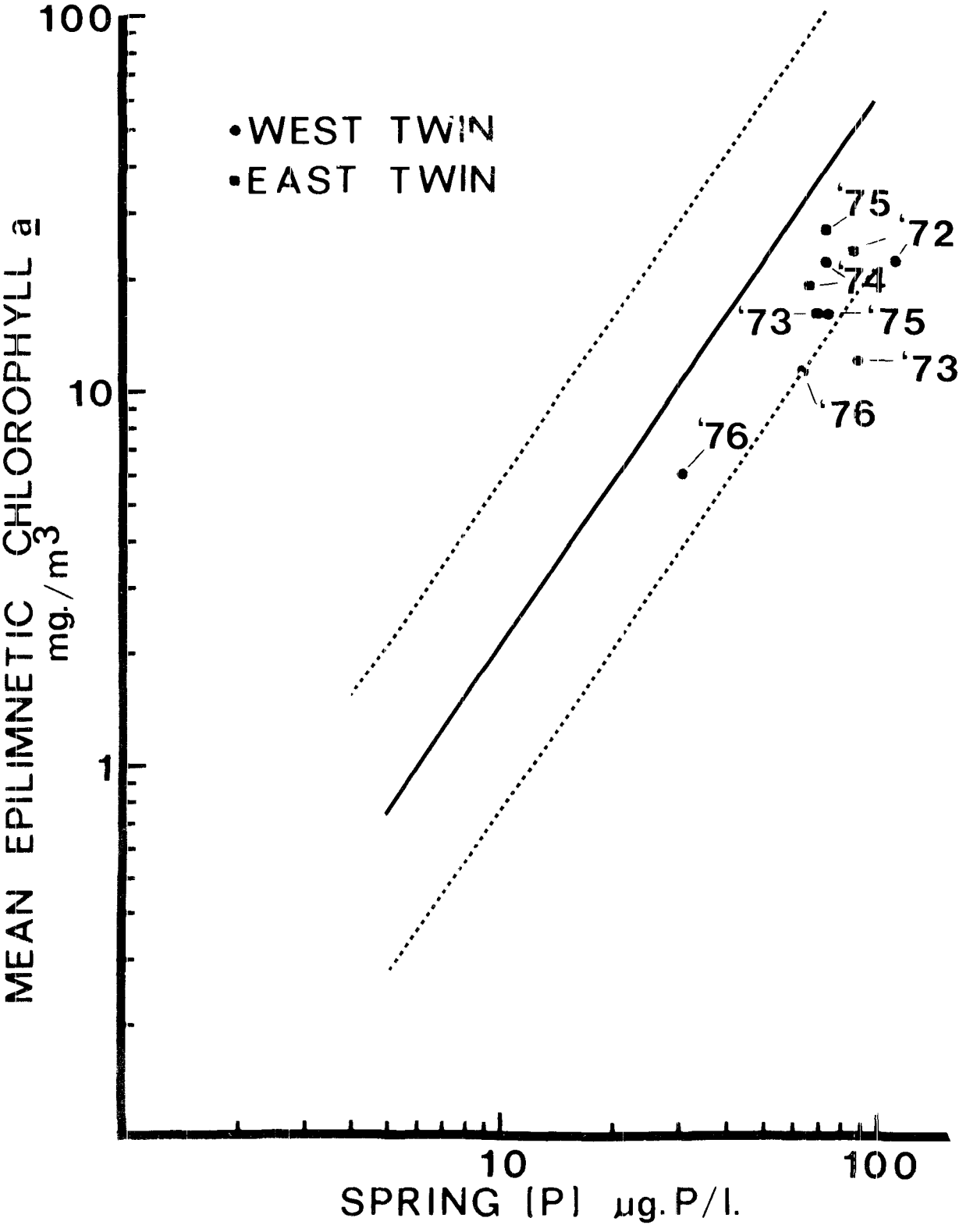


FIGURE 5. Relation between mean epilimnetic chlorophyll a and spring phosphorus concentration at Twin Lakes. Solid line is line of Dillon and Rigler (1974a). Dashed lines are 95% confidence limits.

noted in a later section on phytoplankton [illegible] and alum treatment, 1973 was a year of heavy alga[illegible] Twin Lake.

The effect of additions of phospho[illegible] growth of individual populations growing in that wat[illegible] of phosphorus limitation. If the population [illegible] of phosphorus and only phosphorus would [illegible] corresponding response in the growth of the pop[illegible] determined by the use of certain "indicator spe[illegible] (USEPA, 1971). A modification of the B[illegible] Test [illegible] A[illegible] actual inoculation of 4X10[illegible] cells/ml [illegible] to replicate samples of lakew[illegible] through 0.45 μ membrane filter [illegible] Additions of phosphorus were [illegible] concentration was 100 [illegible] ed temperature and light conditions [illegible] the end of this time the cell den[illegible] Model [illegible] The significant [illegible] samples [illegible] judged [illegible] student[illegible] phosphorus limiting [illegible] least for [illegible]

[illegible] ASSAY PROBLEM

[illegible] ppm.

[illegible]	[illegible]
[illegible]	[illegible]
[illegible]	[illegible]
[illegible]	[illegible]
[illegible]	[illegible]
[illegible]	[illegible]
[illegible]	[illegible]
[illegible]	[illegible]
[illegible]	[illegible]
[illegible]	[illegible]

[illegible] Edwards T[illegible]

[illegible]

It w[illegible] supp[illegible] [illegible]

Algal cells resp[illegible] phosphor[illegible] of polyphosphates [illegible] called [illegible] producing acid or alkaline phosphatase [illegible] demonstrated a positive correlation between [illegible] alkaline phosphatase activity, indicating [illegible] plankton and bacterioplankton may be [illegible]

systems. Not all species are capable of this adaptive response (Kuenzler and Perras, 1968), but those which are may be identified as growing under phosphorus-limiting conditions by the appearance of enzyme activity. Fitzgerald and Nelson (1966) suggest this as a general procedure for identification of phosphorus limitation.

This view is complicated by the finding that some bacterioplankton and phytoplankton constitutively produce alkaline phosphatase, and a number of zooplankton also may continuously release phosphatases directly into the water. Therefore detection of phosphatase activity alone is not sufficient to document phosphorus limitation of the species producing it. It must be shown that the species producing it does so adaptively (i.e., that the cellular specific activity of the enzyme is variable and a function of available phosphorus).

Both East and West Twin were monitored for alkaline phosphatase activity at weekly intervals from 1972 through the summer of 1975; water was drawn from the 0.1, 2, 4, 7, and 10 M strata. Activity was consistently greatest in the epilimnion, least in the hypolimnion. Also, the majority of the activity was particulate (i.e., retained on filters) rather than soluble. Passage of whole lakewater through filters of differing porosity indicated that enzyme activity was filtered from the water coincident with the removal of phytoplankton, indicating that little if any of the activity was associated with unattached bacterioplankton or was dissolved in the water (Figure 6).

The results are graphed as activities (μmoles hydrolyzed/hr/l whole lakewater) and as specific activities (activity μl/l cell volume of species producing the enzyme) with respect to time (Figure 7). Only the results obtained from the 0.1 M stratum are presented. The greatest activities and the greatest specific activities occurred late in the growing season, and maximum activities were always coincident with phytoplankton blooms, especially blooms of Aphanizomenon flos-aquae. Previous studies have shown that this species is capable of adaptive production of alkaline phosphatase when grown under phosphorus limitation in laboratory cultures (Heath and Cooke, 1975).

Because the activity was predominantly associated with phytoplankton shown to be capable of adaptive production of this enzyme under phosphorus limiting conditions, and because the specific activity was variable and increasing with time, it can be concluded that at least those species responding in this manner were P-limited in their growth.

Eutrophication is a process occurring at the ecosystem level of organization, but its effects may be observed at lower levels. Not all studies reported here were carried out for the duration of the project and one (algal assay) was performed only on East Twin. Even so, the investigations at the ecosystem (phosphorus concentration, N:P ratio), at the subcommunity (chlorophyll a vs. spring overturn P), at the population (algal assay), and at the subcellular (alkaline phosphatase) levels of organization all are consistent with the notion that East and West Twin Lakes were phosphorus limited, at least late in the growing season and should therefore respond to significantly lowered lake concentrations by lowered cell volume and chlorophyll.

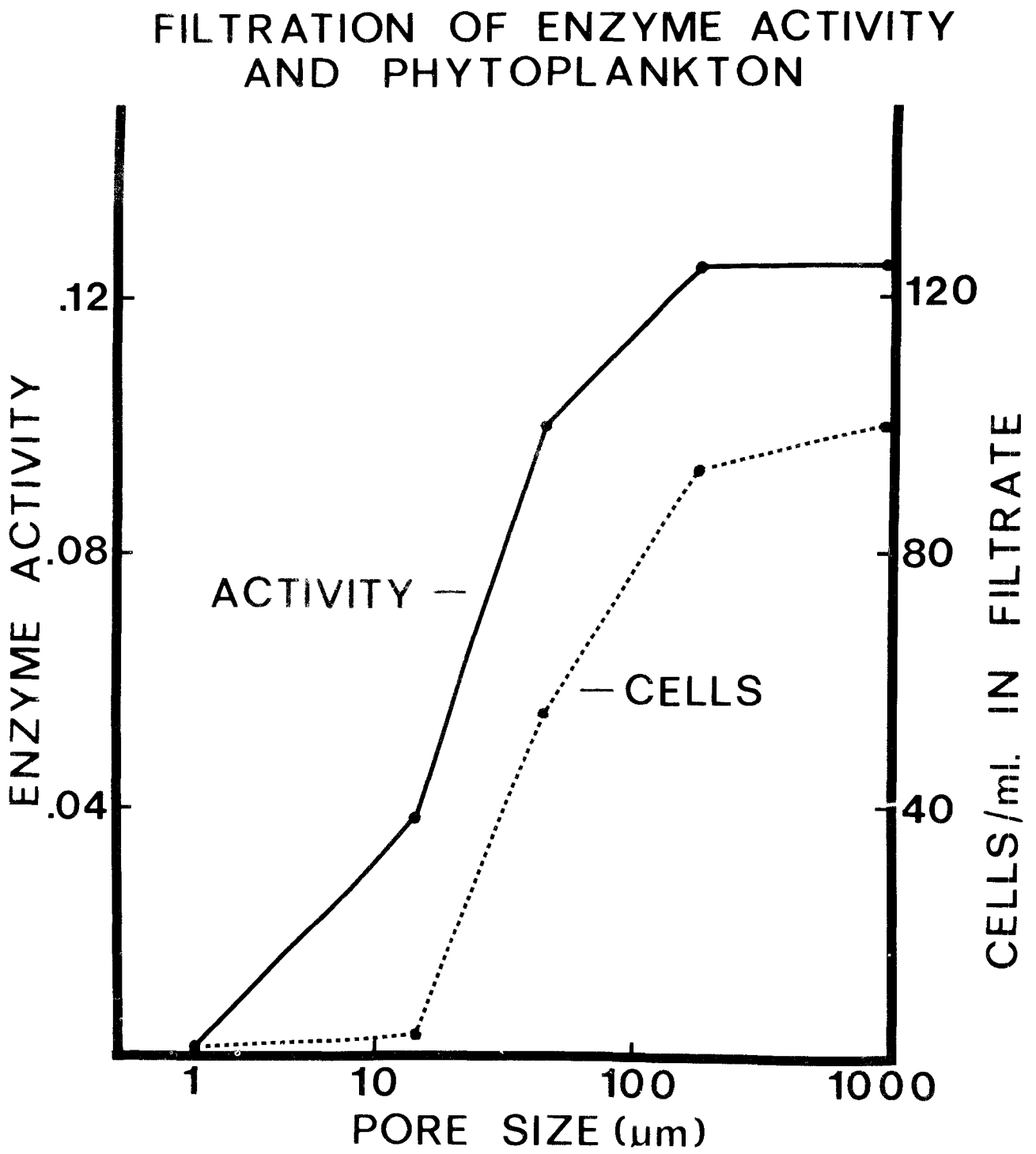


FIGURE 6. Relationship between algal cell size and alkaline phosphatase activity

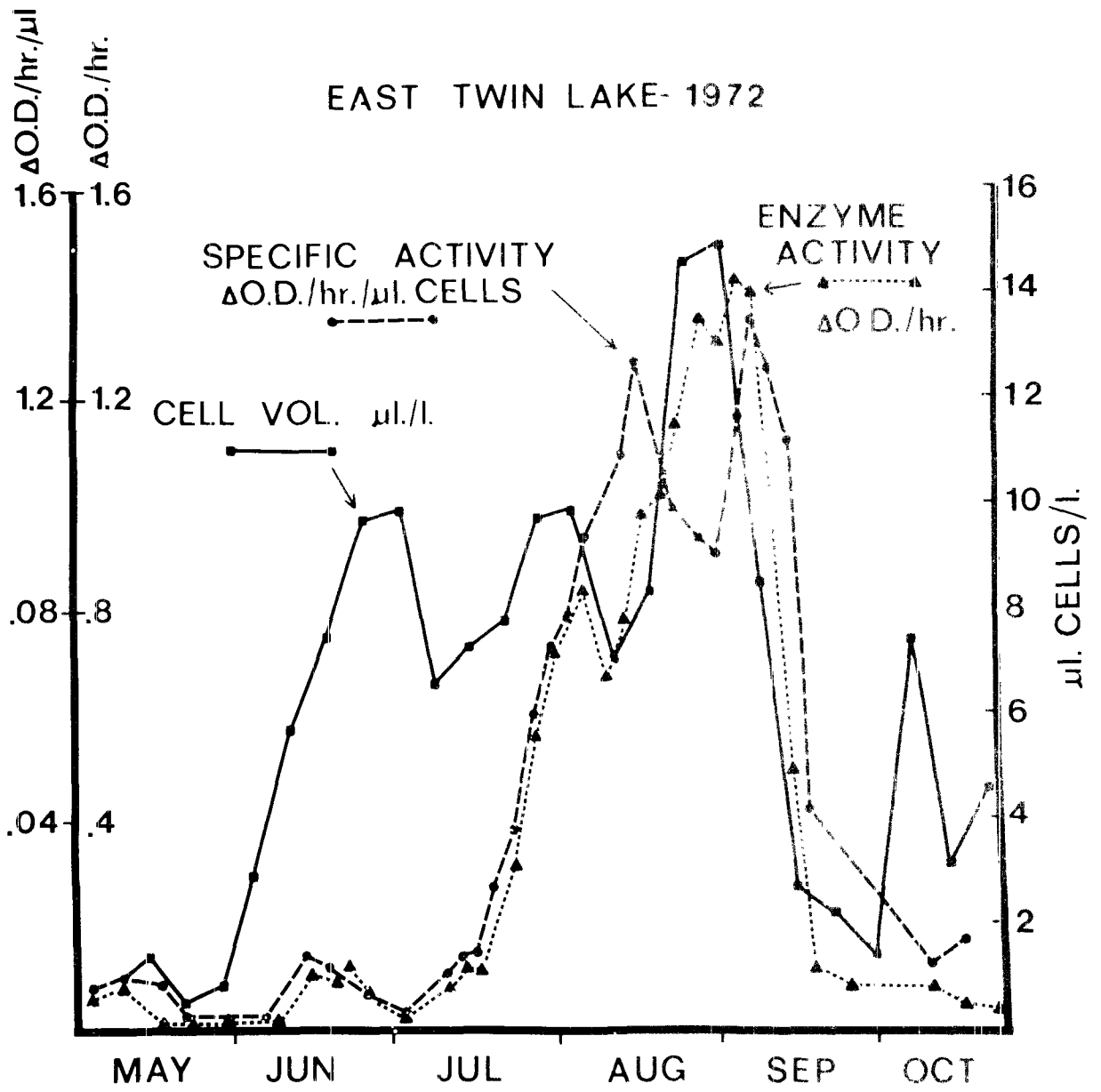


FIGURE 7. Alkaline phosphatase activity, specific activity and cell volume in East Twin Lake, 1972

WATER AND PHOSPHORUS BUDGETS

Septic Tank Diversion

A small sewage treatment plant was constructed outside the watershed to receive the diverted septic tank flows. A few homes were connected to a temporary "package" plant in 1969-71, but the [illegible] of the homes built before sewage service were not connected to the new treatment plant until [illegible] (Figure 8). About 85% of the homes were serviced by the end of 1972, and 9[illegible]% by the end of 1974. All new homes built since sewage service began have been connected to the sewage lines. There are presently a few (10-12) homes, some of them summer cottages, which have not been connected.

Effect of Diversion on Water and Phosphorus Budgets

Water Budget [illegible] East and West Twin Lakes are drainage lakes, but their principal sources [illegible] West Twin's water income was about evenly divided between precipitation, groundwater, and streams (Table 8), with groundwater usually the dominant source. East Twin received about half of its annual income from West Twin's outlet, except in 1976 when summer and fall seasons were unusually dry. Nearly all of the stream income to East Twin was from West's outlet. [illegible] overland flow [illegible] (Table 9) was much lower than West's because its drainage [illegible] was smaller (102 vs. 233 ha).

Groundwater, as estimated [illegible] residual in the hydrologic equilibrium, was high for both lakes, partly [illegible] Because of this large groundwater income plus the discharge [illegible] the mean water residence time of East was about half that of West (0.58 [illegible] years). The areal water load, or mean depth divided by water residence time (Table [illegible] and 12), was about twice as high for East than for West (8.[illegible] [illegible]).

The source of the large groundwater income to East Twin [illegible] by Szczepanowski (19[illegible]). Lake Rockwell, an impoundment [illegible] Twin, was filled in 1915 by damming the Cuyahoga River [illegible] determined that the level of the Twin Lakes rose significantly after [illegible] even though there is a high bluff between Rockwell and East Twin, and no interconnecting surface flows. Szczepanowski installed piezometers in Lake Rockwell, where the two lakes are closest, and estimated a groundwater discharge of 800 M^3/day towards East Twin.

The seepage from Lake Rockwell may have been partially responsible for maintaining the lake water levels in the Twin Lakes, even after sewage diversion. On other lakes water levels have often dropped following sewage diversion. Water income and water level of neither lake changed in a way which could be attributed to diversion, even though the diversion exported approximately 353 M^3/day out of the watershed which previously had entered the lakes as overland flow, stream flow, and groundwater.

While the several methods of computing groundwater income to the lakes were in general agreement, the sources of the water were not specifically located. During ice cover unfrozen "windows" were observed in both lakes

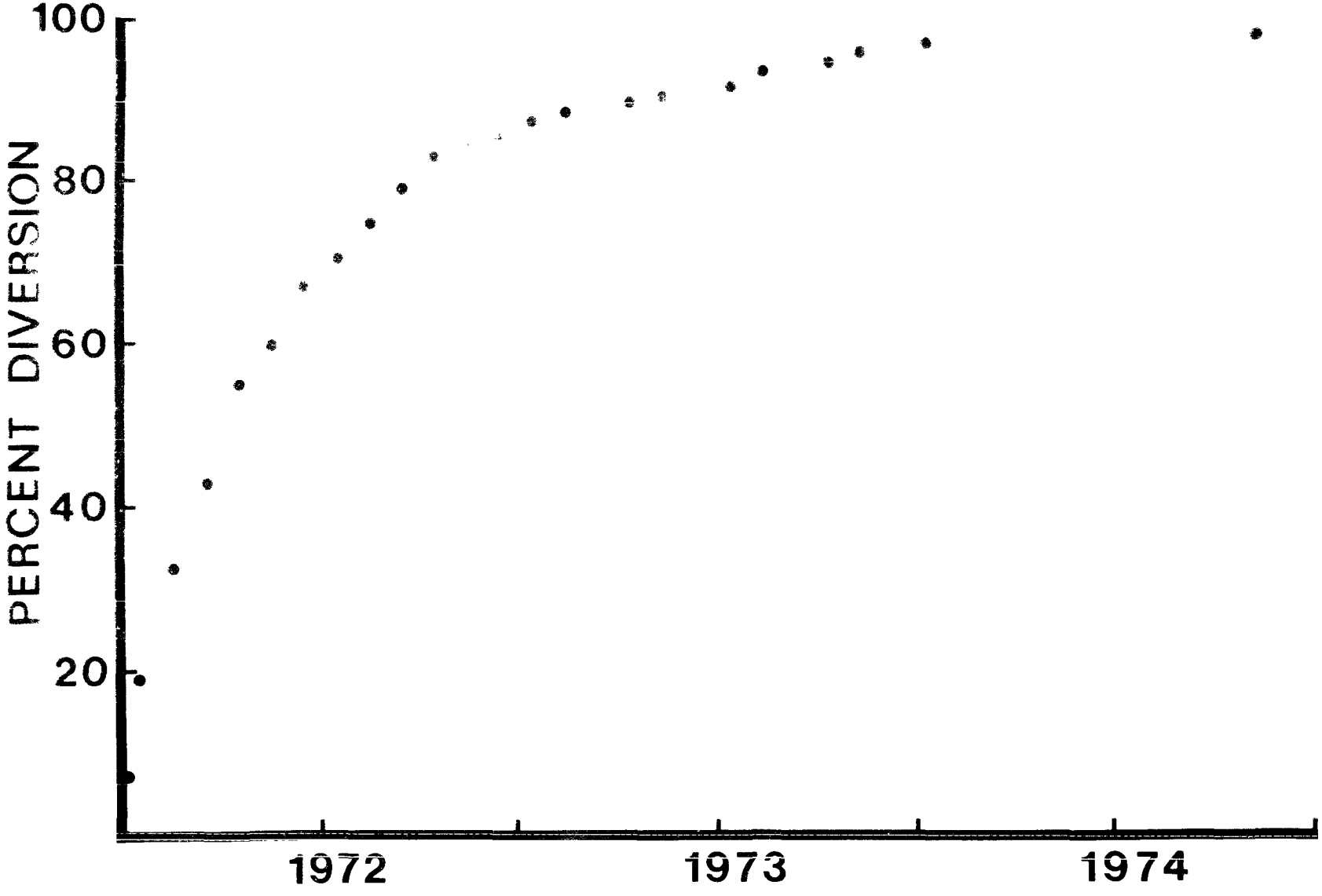


FIGURE 8. Percent septic tank diversion during 1972-1974

Table [illegible]. SOURCES OF ANNUAL WATER ($m^3 \times 10^3$), AND PHOSPHORUS (kg) INCOME
[illegible] LAKE [illegible] WEST TWIN LAKE

	[illegible]	1974	[illegible]	[illegible]
Annual Water [illegible]	108[illegible]	1679.8	[illegible]	[illegible]
[illegible]	[illegible]	367.[illegible]	[illegible]	[illegible]
[illegible]	[illegible]	[illegible]6.[illegible]	[illegible]	[illegible]
[illegible]	[illegible]	[illegible]95.[illegible]	[illegible]	[illegible]
[illegible]	[illegible]4	81.[illegible]	[illegible]	[illegible]
[illegible]		[illegible]44.		[illegible]
[illegible]		[illegible]07.[illegible]	[illegible]	[illegible]
[illegible]		[illegible]3.[illegible]	[illegible]	[illegible]
[illegible]	[illegible]	[illegible]9	[illegible]	[illegible]
[illegible]	[illegible]	[illegible]5.	[illegible]	[illegible]
[illegible]	[illegible]	[illegible]	[illegible]	[illegible]
[illegible]	[illegible]	[illegible].6	[illegible]	[illegible]
[illegible]		[illegible]	[illegible]	[illegible]

Table 9. SOURCES OF ANNUAL WATER ($M^3 \times 10^3$) AND PHOSPHORUS (kg) INCOME-OUTGO FOR EAST TWIN LAKE

	1972	1973	1974	1975	1976
1. Annual Water Income	2152.2	2056.9	2971.9	3192.9	2614.4
Groundwater (%)	659.4(30.6)	819.6(39.8)	1050.0(35.3)	1260.8(39.5)	1230.4(47.1)
Streams (%)	1202.7(55.9)	980.9(47.7)	1574.2(53.0)	1541.5(48.3)	1027.1(39.3)
Precipitation (%)	257.2(12.0)	251.1(12.2)	312.4(10.5)	315.5 (9.9)	292.9(11.2)
Overland (%)	33.0 (1.5)	5.4(<1.0)	35.3 (1.2)	75.1 (2.4)	64.2 (2.5)
2. Annual Water Outflow	1978.7	1921.9	2515.1	2996.6	2446.0
3. Annual Phosphorus Income	180.8	126.9	219.4	218.4	127.4
Groundwater (%)	33.0(18.3)	29.2(23.0)	73.2(33.3)	71.8(32.9)	79.9(62.7)
Streams (%)	137.6(76.1)	92.1(72.6)	137.0(62.4)	128.9(59.0)	33.3(26.1)
Precipitation (%)	5.4 (3.0)	5.4 (4.2)	7.0 (3.2)	11.4 (5.2)	7.0 (5.5)
Overland (%)	4.8 (2.6)	0.2 (0.1)	2.3 (1.0)	7.5 (3.4)	7.2 (5.7)
4. Annual Phosphorus Outflow	142.29	108.99	157.44	177.50	135.45

suggesting that fairly high volume springs entered the lakes. [illegible] would not be detected by piezometers, so that the rate of seepage from Lake Rockwell, and possibly from the four small lakes above West Twin [illegible] been much higher than Szczepanowski (1976) estimated. Th[illegible] indicates the decision to assign groundwater income [illegible] rather than depend upon in-lake and lakeshore piez[illegible] sample higher volume springs.

Phosphorus Budget - Phosphorus entered the [illegible] precipitation, dry fallout, and groundwater and stream flows [illegible] was not monitored.

The annual average concentration of phosphorus [illegible] (Table 10) did not change. The slightly higher [illegible] probably reflects the much larger number of determinations [illegible] annual [illegible] were underestimated [illegible] pr[illegible] temperature inversions [illegible] fallout was not measured.

A network of piezometers was establishe[illegible] in the phosphorus contributions from groundwater. Th[illegible] West Twin's groundwater increased [illegible] 19[illegible] 1976 were not monitored and were [illegible] Mean annual phosphorus [illegible] (Table 10 [illegible]

MEAN ANNUAL PHOSPHORUS CONC[illegible] WATE[illegible] SOURCES

	WEST TWIN			
	Groundwater	Stream	[illegible]	[illegible]
1972	[illegible]	[illegible]	[illegible]	
1973	[illegible]	[illegible]	[illegible]	
1974	[illegible]	[illegible]	[illegible]	[illegible]
1975	121	21	[illegible]	
1976	121	[illegible]	24	

	EAST TWIN			
	Groundwater	Stream	Precipitation	Overla[illegible]
1972	50	[illegible]		
1973	[illegible]	[illegible]		
1974	[illegible]	[illegible]		
1975	[illegible]	[illegible]		
1976	[illegible]			

To determine the impact of nutrient diversion on groundwater quality, especially in the vicinity of septic systems, a series of monitoring wells was drilled above, in, and below a leach field. The monitoring network was set up six weeks prior to connection of the sewers. The boundaries of the leach field were found using electrical earth resistivity techniques. All effluent was diverted from the septic system after sewer hook-up, but the leach field, by request, was not disturbed, and samples of the wells were taken weekly for the first two months and then monthly.

Several important factors about pre-diversion phosphorus income are indicated from this monitoring. First, the water table was encountered in the leach field 6.5 meters below the surface. Six meters above the leach field the water table was about 2 meters below the surface and 3 meters downslope from the leach field, the water table was about 2.5 meters below the surface (Figure 9). These observations indicate a perched water table in the leach field, perhaps caused by [illegible] pass which reduced the permeability of the leach bed. This [illegible] seen throughout the watershed, as evidenced by soggy lawns [illegible]. Nutrient-rich materials were probably washed from the lawns [illegible] and streams, and thence to the lakes, accounting for the high concentrations in streams in 1971 and early 1972.

No clay was [illegible] the area of the test leach field; the materials were coarse sand and gravel to the surface to 6 meters. Thus septic systems, placed in coarse sand and gravel (soils which are presumably ideal for septic tank-leach field systems) could discharge effluent to the surface as seeps which then contribute to surface water flows, especially during periods of high runoff and overbank [illegible].

The chemical monitoring of the leach field shows the effect of the perched water table and diversion on the groundwater phosphorus concentration. The [illegible] the control well 16G, and the well 9 meters below the leach field [illegible] very close (Figure 10), indicating little phosphorus contamination [illegible] groundwater away from the leach field. Well 17G, situated [illegible] field, had a logarithmic drop in concentration immediately after diversion. Well 18G, located at the toe of the field had a slightly higher concentration than the control and dropped little after diversion.

Surface water station 2S (Figure 4) is typical of the urban streams of the watershed where sources include baseflow, stormwater runoff and septic effluent. Stream 2S received the surface seepage from the experimental leach field and run-off from an urban area. Figure 11 illustrates the relationship between reduction in water discharge, phosphorus concentration, and the number of sewer connections from January 1972 through February 1973. In this interval the mean water discharge from 2S dropped from over 400 M^3 /day, to less than 40 M^3 /day. The temporary rise in concentration from June through August 1972 might be due to construction of sewer lines in the area during these months (Buller, 1974).

Figure 12 is a plot of phosphorus output past the weir at station 2S for the same period of time. The output dropped from almost 80 kg P in January

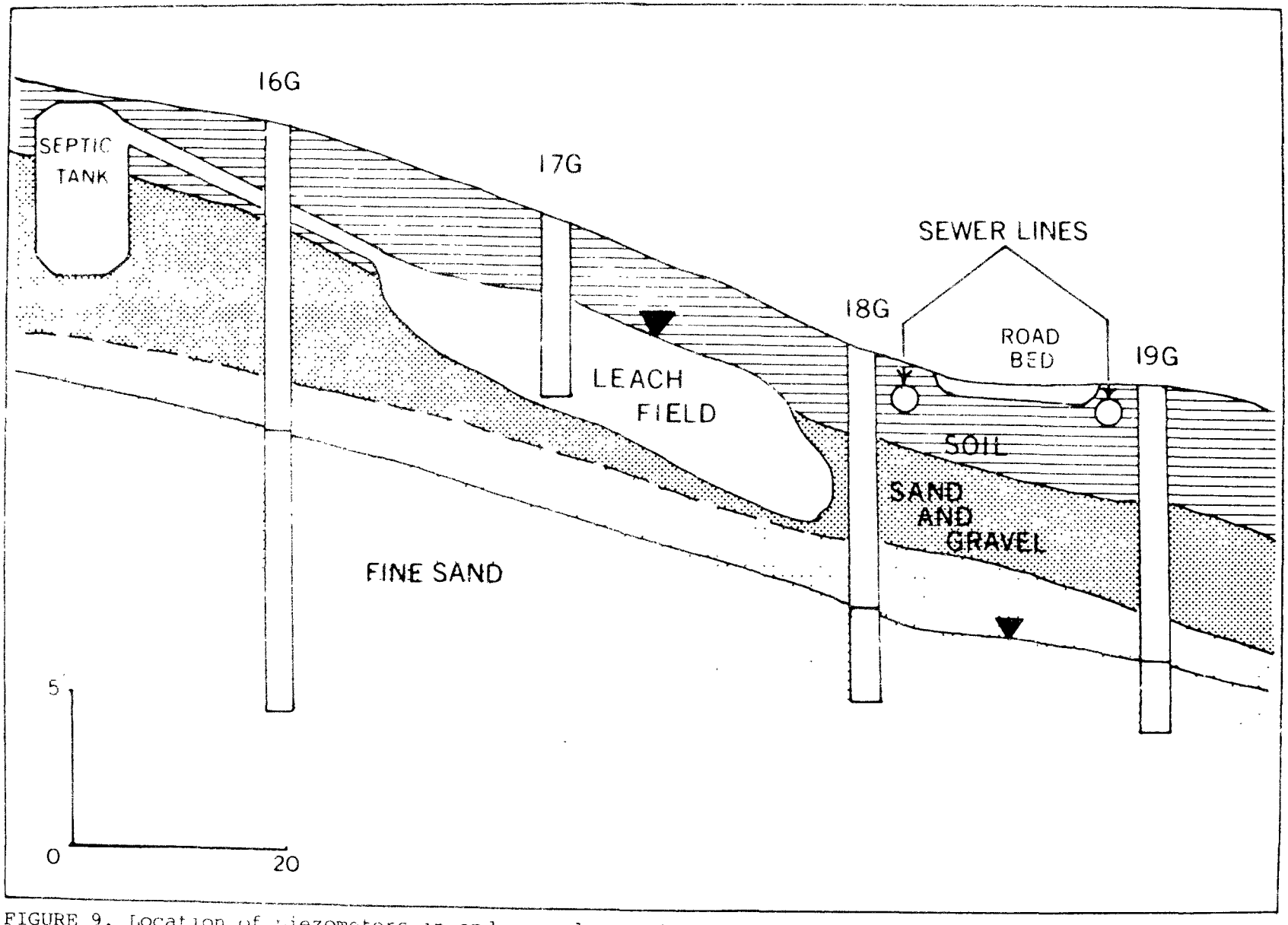


FIGURE 9. Location of piezometers in and around experimental leach field

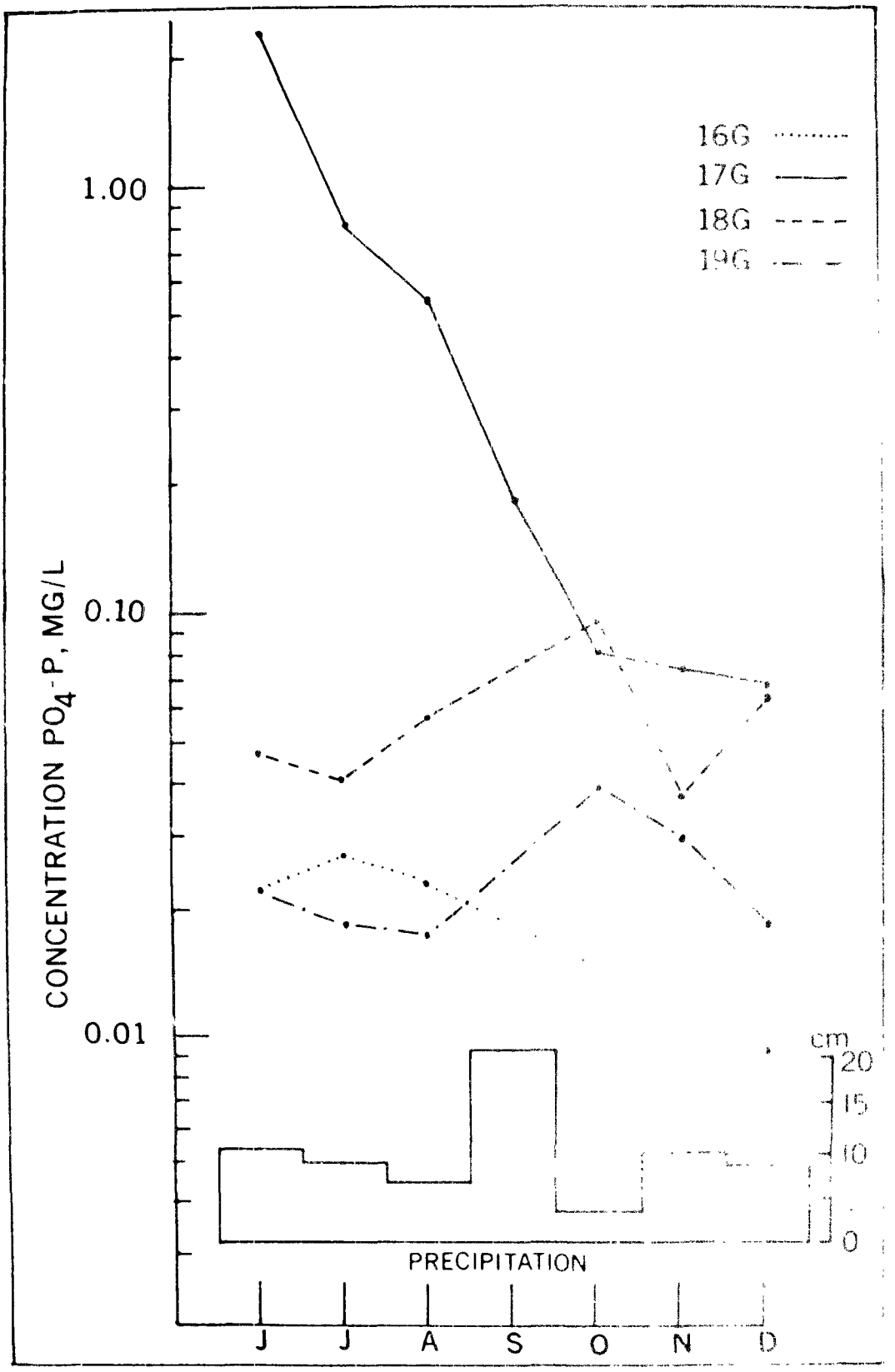
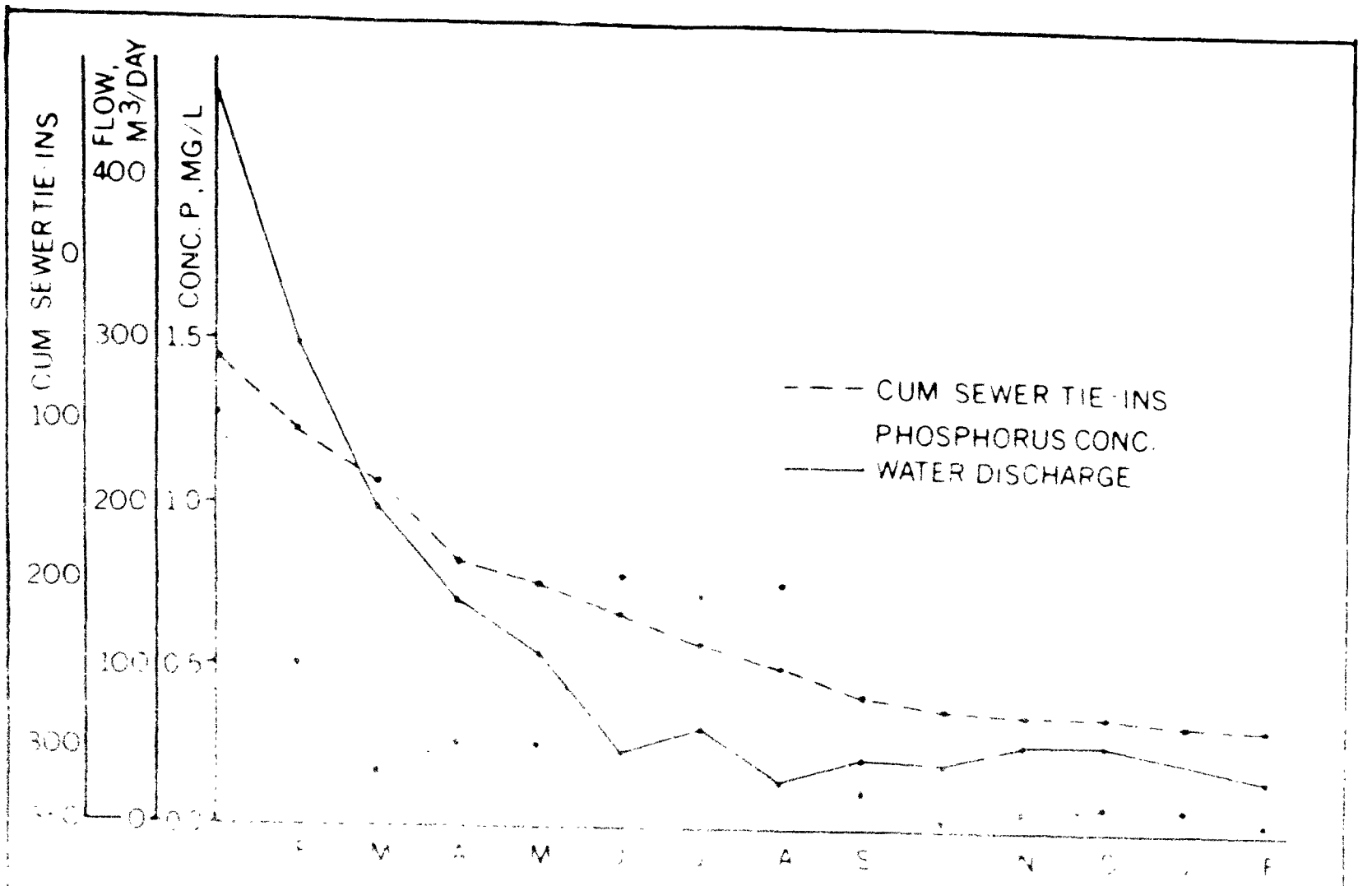


FIGURE 10. Total phosphorus concentration above, in, and below experimental leach field after diversion



PHOSPHORUS CONCENTRATION CUMULATIVE SEWER TIE INS AND WATER DISCHARGE, STATION 2S JAN. 12 - FEB. 17

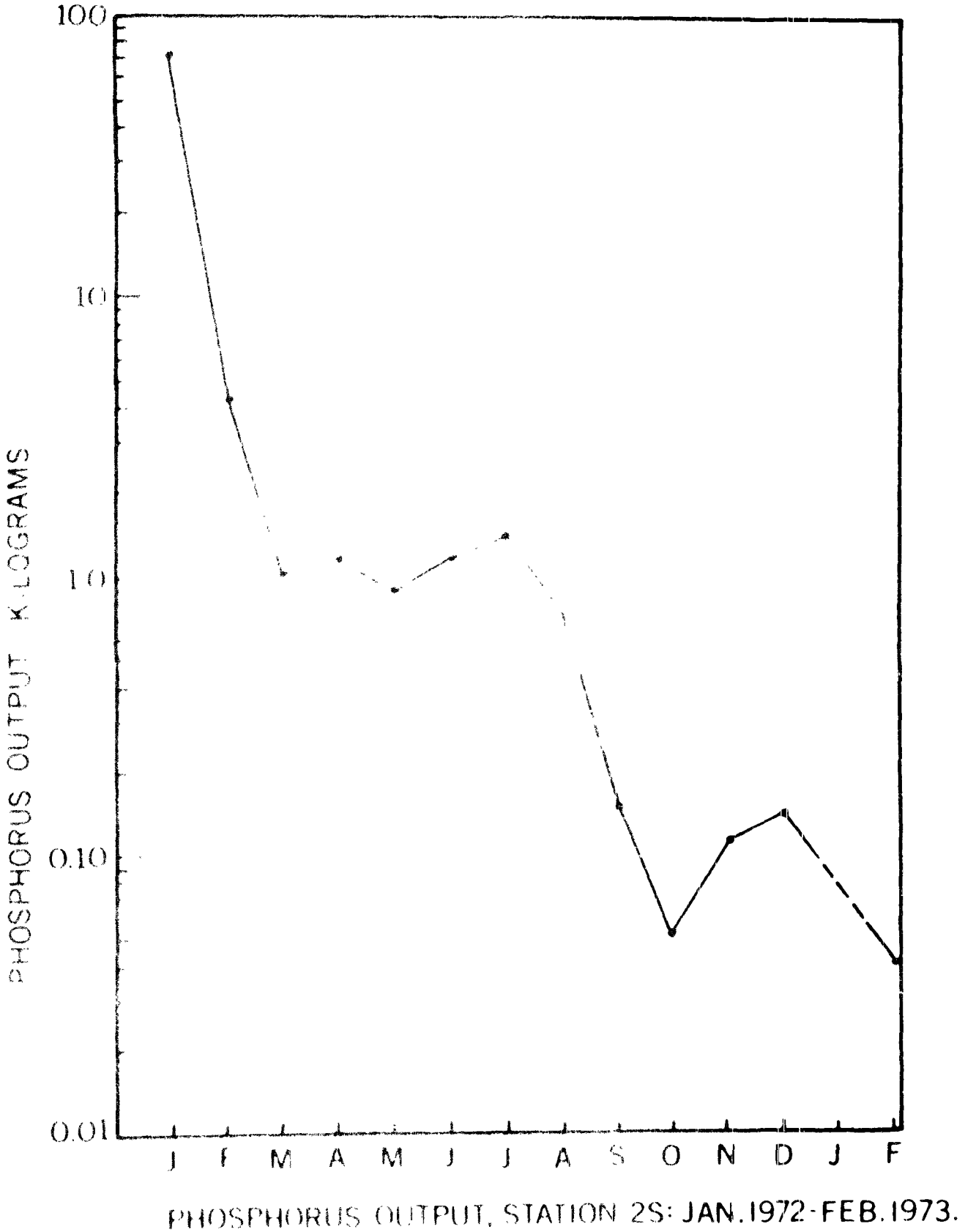


FIGURE 12. Phosphorus flood [illegible] stream 2S [illegible] Twin after diversion

Table 11. WATER [illegible] PHOSPHORUS BUDGETS OF WEST TWIN LAKE 1972-1976

	1972	1973	1974	1975	1976
1. Annual Water Inflow					
[illegible]	[illegible]	[illegible]	1679.8	1773.7	1137.1
[illegible] Water Inflow					
[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]
[illegible]					
[illegible]			[illegible]	[illegible]	1.40
[illegible]					
[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]
[illegible]					
[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	94.[illegible]
[illegible]					
[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]
[illegible]					
[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]
[illegible]					
[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]
4. [illegible]					
[illegible]			[illegible]	[illegible]	[illegible]
[illegible]					

Table 12. WATER - PHOSPHORUS BUDGETS OF EAST TWIN LAKE 1972-1976

	1972	1973	1974	1975	1976
1. Annual Water Inflow $M^3 x10^3$ (Q_i)	2152.2	2056.9	2971.9	3192.9	2614.4
2. Annual Water Outflow $M^3 x10^3$ (Q_o)*	1978.9	1921.0	2515.1	2996.6	2446.0
3. Water Residence (yrs) $Vol/Q_o = T_w$	0.68	0.70	0.54	0.45	0.55
4. Areal Water Load $\bar{Z}/T_w$ (M/yr)	7.40	7.19	9.31	11.18	9.15
5. Annual Phosphorus Income (kg) (J)	180.8	126.8	219.4	218.4	127.4
6. Annual Phosphorus Outflow (kg)	142.3	109.0	157.4	177.5	135.5
7. Phosphorus Retention $\frac{In-Out}{In}$	0.21	0.14	0.28	0.19	-0.06
8. Phosphorus Areal Loading (gm $P/M^2/yr$)	0.67	0.47	0.82	0.81	0.47
9. Mean Annual Inflow Concentration (μg P/l) (J/Q_i)	84.0	61.7	73.8	68.4	48.7

*Does not include water lost by evaporation

1972 to approximately 0.05 kg P in February 1973. Following the rapid change in total phosphorus in 1972, concentrations in stream 2S remained in the range of 50 to 70 µg P/l. These concentrations represent washoff from streets and lawns. At least one other stream entering West Twin (5S) was also influenced directly by sewage and runoff in a way similar to 2S.

The main source of phosphorus to West Twin was from streams in 1972 and 1973 and groundwater in 1974 through 1976 (Table 7). The mean annual phosphorus concentration in streams in 1972 was very high, particularly the two streams (2S, 5S) draining urban areas rather than those discharging from upland lakes (4S, 3S). Income from streams in 1973-76 was 50-75% lower than in 1972 but mean total annual inflow concentration was as high in 1975 and 1976 as in 1972, due in large part to the increased concentration in groundwater (Table 10). The reasons for this increase are not known.

The mean annual phosphorus concentration in the stream (8S) entering East Twin Lake fell steadily from 1972-1976 (Table 10), with the 1976 concentration 25% of the 1972 concentration, due to the alum treatment of West Twin in 1975. The concentration of phosphorus in other East Twin sources did not decline so that the mean annual inflow concentration, as in West Twin, remained fairly high through 1975.

In summary these drainage lakes received a significant volume of water from seepage and springs. The quantity of phosphorus in groundwater as it enters the lake was not measured, but was estimated from concentration in piezometers on the lake margin. Concentration generally increased over the study period. Surface water flows from urban areas were high in phosphorus before diversion, presumably from upward seepage of effluent from leach fields to the ground surface, where it was carried away as run-off to streams. Stream concentration fell over the study period. Total phosphorus income per year was variable from year to year and average inflow concentration to West Twin was as high in 1976 as in 1972. (Tables 11 and 12).

Vollenweider phosphorus loading model - Vollenweider (1976) has suggested that the critical factor leading to eutrophy in phosphorus limited lakes is the total phosphorus concentration, a value depending on average annual inflow phosphorus concentration and the water residence time. Eutrophic lakes tend to have high average incoming concentration, moderate residence times, and generally have a phosphorus concentration greater than 20 µg P/l. Vollenweider (1976) designated regions of a phase diagram (mean annual inflowing phosphorus concentration vs water residence time) to show the transition from oligotrophic to eutrophic, divided by boundaries designated as "excessive loading" (those which result in a potential concentration greater than 20 µg P/l) or "permissible loading" (those which result in a potential concentration less than 10 µg P/l).

Using data from Tables 11 and 12, trajectories were plotted with 1972 as the initial year for each trajectory (Figure 13). Over the duration of this study the phosphorus loading to each lake was excessive, and there was no trend from year to year, other than a small decrease from 1972, to suggest that septic tank diversion resulted in a significant decrease towards

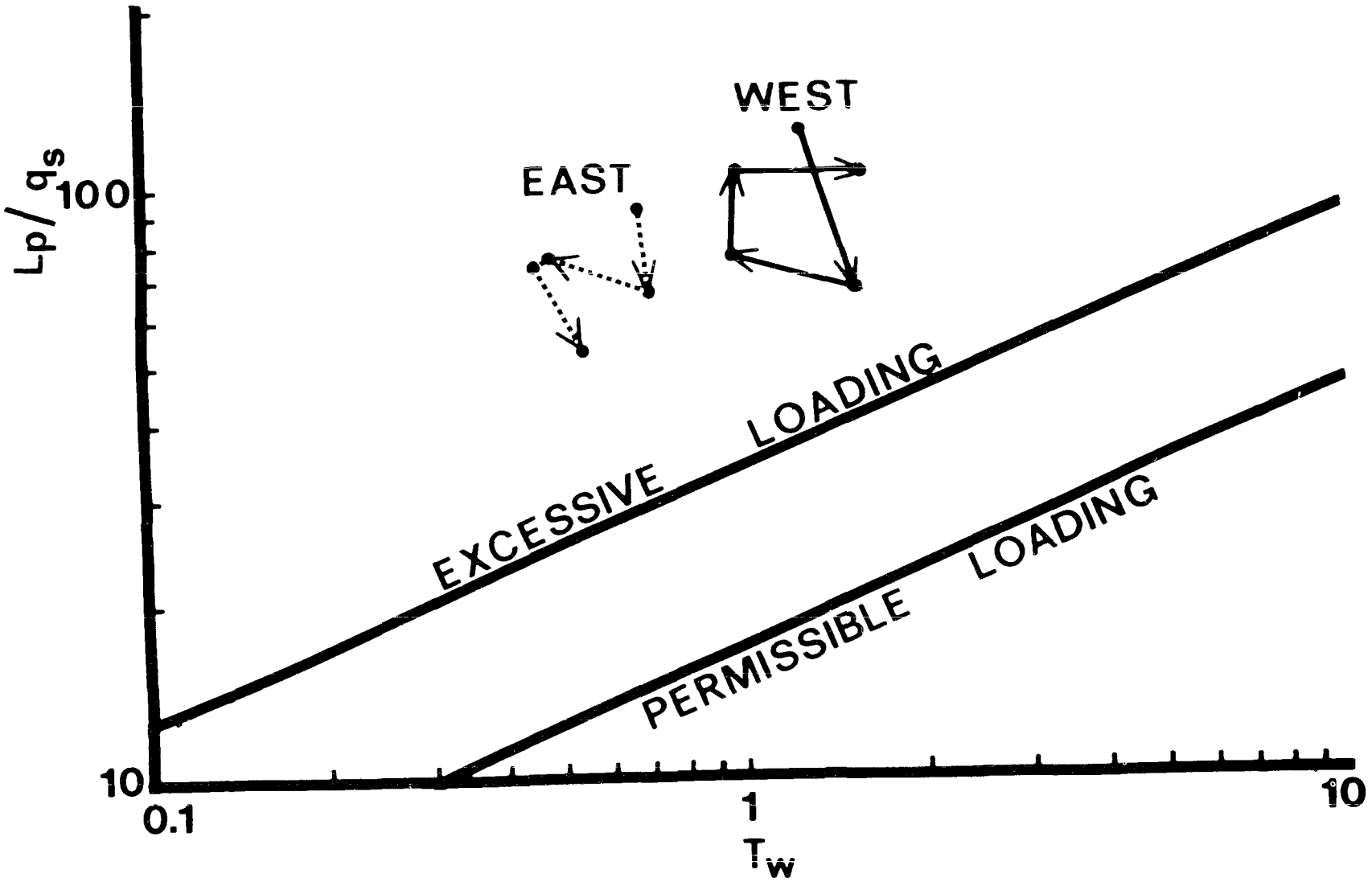


FIGURE 13. Positions of East and West Twin Lakes on Vollenweider (1976) loading diagram, 1972-1976

"permissible" loading. This observation is consistent with the small changes noted in mean annual incoming concentration to both lakes after diversion as well as with the small changes in lake concentration and in biotic indicators of the degree of eutrophication.

EFFECTS OF DIVERSION AND TREATMENT ON PHOSPHORUS

Immediate effects of aluminum sulfate treatment - West Twin Lake was monitored intensely just before, during, and after the aluminum sulfate treatment, and then at weekly intervals through 1976. The immediate responses of the lake were primarily chemical, and differed from any expected longer term response in that there was no immediate effect on biotic indicators of eutrophication in the epilimnion as measured by cell volume, seston, chlorophyll, or transparency.

Aluminum hydroxide particles appeared in deep water samples for several days after treatment. After two weeks the lake was examined through the use of SCUBA. The epilimnion at this time was murky due to an algal bloom, but the hypolimnion, in contrast to the absence of light in pre-treatment dives, was clear. The lake bottom could be seen from about the 5 M level. The mud had a uniform "carpet" of aluminum hydroxide floc about 1 cm thick with no bare spots, and it resembled a moonscape with pock marks where anchors had been dropped before treatment. There was no floc in water less than 5 M deep. Later, the floc consolidated with the sediment so that by 1976 floc particles could only be observed in dredge samples.

Dissolved aluminum remained below detection limits at all depths and locations, with the exception of one sample at 7 M just after application, where a concentration of 2.4 μg Al/l was observed (Table 13).

Table 13. PRE- AND POST TREATMENT ALUMINUM CONCENTRATIONS (μg Al/l) IN WEST TWIN LAKE AFTER 5 METER ALUMINUM SULFATE APPLICATION

	Date (1975)					
DEPTH (M)	7-10	7-29*	7-29**	7-30	7-31	8-1
Surf.	0	0	0	0	0	0
2	0	0	0	0	0	0
4	0	0	0	0	0	0
7	0	0	2.4	0	0	0
10	0	0	0	0	0	0

* 1 hour before treatment
** 1 hour after treatment

The alkalinity and pH of water below 5 M were altered by the alum treatment (Figure 14). Hypolimnetic alkalinity before treatment ranged from 110-140 mg $CaCO_3$ /l. Just after application alkalinity at 6 meters was 60 mg $CaCO_3$ /l, and the hypolimnion below this depth remained at about that alkalinity until fall circulation. Pre-treatment alkalinities returned after circulation and have remained at that level through 1976. The hydrogen ion concentration increased slightly in the hypolimnion, but returned to pre-treatment levels within a month (Figure 14).

Sulfate concentration in the hypolimnion was increased from a pre-treatment volume-weighted average of 35 mg SO_4 /l, to a post-treatment average of 50 mg SO_4 /l. The concentration below 7 M a month after application was over 100 mg SO_4/l, and averaged 50 mg/l during the fall, in contrast to pre-treatment fall averages of 40 mg SO_4/l (Table 14). Winter concentrations in earlier years, as well as winter 1975-76, averaged 50 mg SO_4/l. Summer 1976 concentrations were almost twice as high as previous years. The [illegible] change in the sulfate concentration of East Twin.

No toxic effects, such as fish kills, were observed after the application and fishermen report fishing to be as good just after treatment and in 1976 as before treatment.

The concentrations of phosphorus fractions in West Twin Lake were promptly affected by the alum application (Figures 15 and 16). Soluble reactive phosphorus (SRP) in the epilimnion before application ranged from 3-5 µg P/l, and in the hypolimnion, up to 500 µg P/l. Filterable total phosphorus was about 20 µg P/l in the epilimnion before treatment and above 400 µg P/l in the hypolimnion (no measurements of FTP fraction were taken at 8 and 9 meters so that SRP, which was measured at all depths appears to be higher). After treatment, the concentration of SRP and filterable total phosphorus in the epilimnion remained unchanged, but were greatly lowered in the hypolimnion. The lowest concentration was reached about one week after application. One year later hypolimnetic concentrations were slightly higher, but were still 10% or less of 1975 pre-treatment levels (Figure 15).

Total phosphorus concentration in the hypolimnion was also greatly lowered after the application (Figure 16) and remained low through 1976 even though oxygen was absent in the hypolimnion.

Effects on Phosphorus Content of the Lakes - The phosphorus content of the Twin Lakes varied in a regular pattern over each year (Figures 17 and 18). During spring circulation phosphorus content steadily fell to an annual low just prior to the onset of stratification, then increased, most of the increase being in the hypolimnion, to a late summer peak. In some years the increase continued through the fall to a winter peak. The spring decrease began shortly after ice-out.

The daily rate of increase in phosphorus content over the summer months became less in both lakes each year after septic tank diversion, except in West Twin in 1975 when the interval considered was 40 days shorter than other

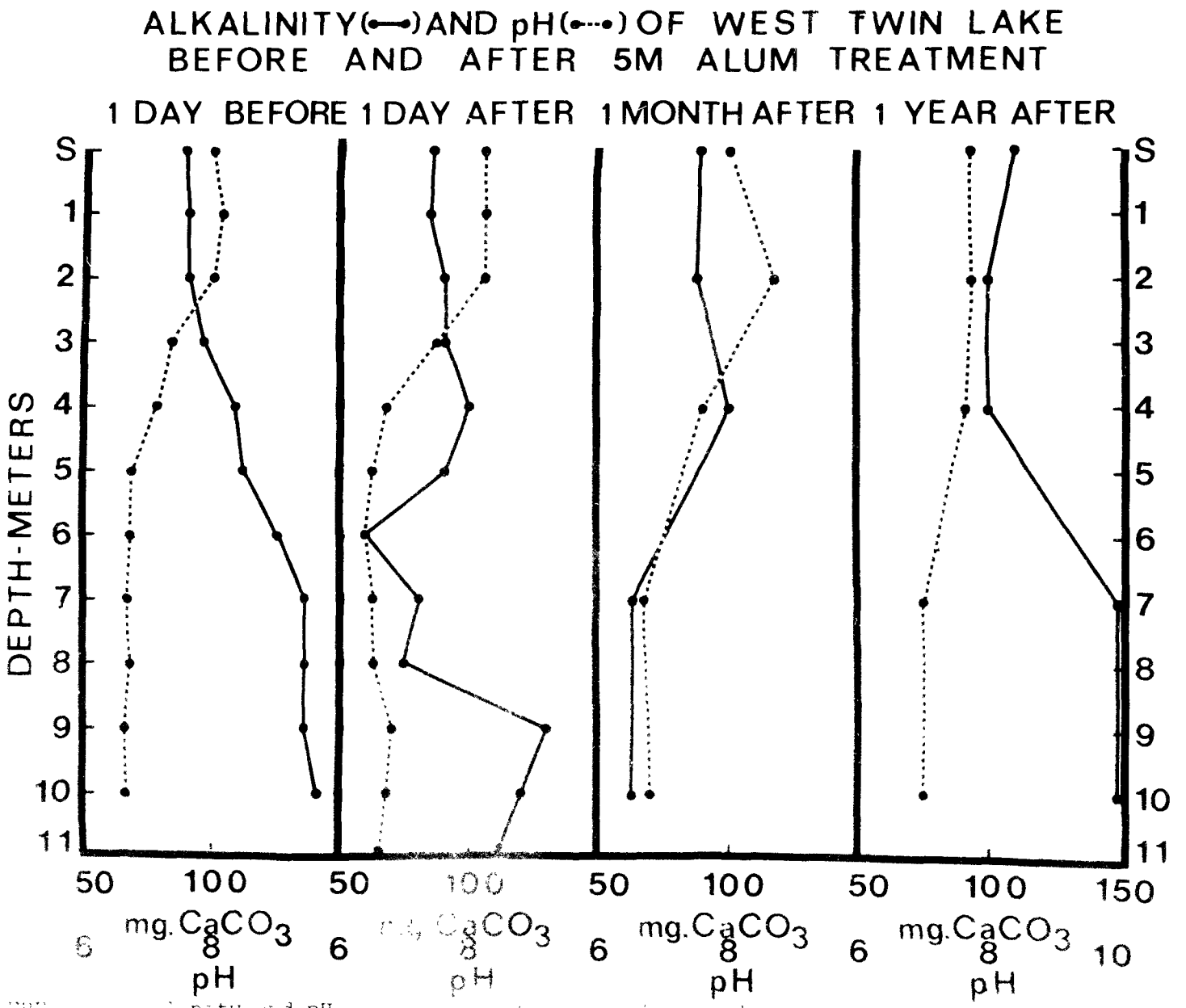

ALKALINITY(•—•)AND pH(•---•) OF WEST TWIN LAKE
BEFORE AND AFTER 5M ALUM TREATMENT
1 DAY BEFORE
1 DAY AFTER
1 MONTH AFTER
1 YEAR AFTER
DEPTH-METERS
S
1
2
3
4
5
6
7
8
9
10
11
50
100
150
mg.CaCO3
6
8
10
pH

Table 14. SULFATE (mg SO_4/l) IN EAST AND WEST TWIN BEFORE AND AFTER HYPOLIMNETIC ALUM APPLICATION

	West Twin Lake			
	[illegible] Jul [illegible]	[illegible] 1 [illegible]	28 Aug 75	[illegible] Jul 76
Depth				
0	[illegible]	[illegible]	49.66	59.35
1	[illegible]	37.6[illegible]	41.80	54.41
2	[illegible]	34.80	43.00	49.97
3	[illegible]	34.9[illegible]	43.00	48.80
4	[illegible]	[illegible]	43.00	48.14
5	[illegible]	[illegible]	65.07	47.71
6	24.[illegible]	[illegible]	87.13	47.27
7	[illegible]		[illegible]	46.84
8	[illegible]			[illegible].55
9	[illegible]	[illegible]	[illegible]	58.25
10	21.30	39.53	106.[illegible]	63.96
11	20.47	39.53	105.07	69.67

	East Twin Lake	
	8 Aug 75	13 Aug 76
Depth		
0	[illegible]	[illegible]
1	[illegible]	[illegible]
2	[illegible]	[illegible]
3	39.7[illegible]	[illegible]
4	39.8	[illegible]
5		[illegible]
6		
7		[illegible]
8	[illegible]	[illegible]
9	[illegible]	[illegible]
10	23.60	36.96
11	19.37	31.35

years (Table 15). Had the alum treatment not stopped the 1975 increase, the peak might have been little higher than the previous year and the rate thus equal to or less than 1974. The reduction in this rate of summer increase was particularly pronounced in East Twin in 1975 and 1976 (Figure 18, Table 15).

The effect of the West Twin hypolimnetic alum application on the annual pattern of phosphorus content and the hypolimnetic summer increase is illustrated in Figure 19. The phosphorus content was lowered by 67% of the peak July 1975 pre treatment content, and remained at that level through 1976, rising somewhat in the winter 1976. Autumnal and vernal circulation did not increase either the phosphorus or the aluminum concentration of the lake.

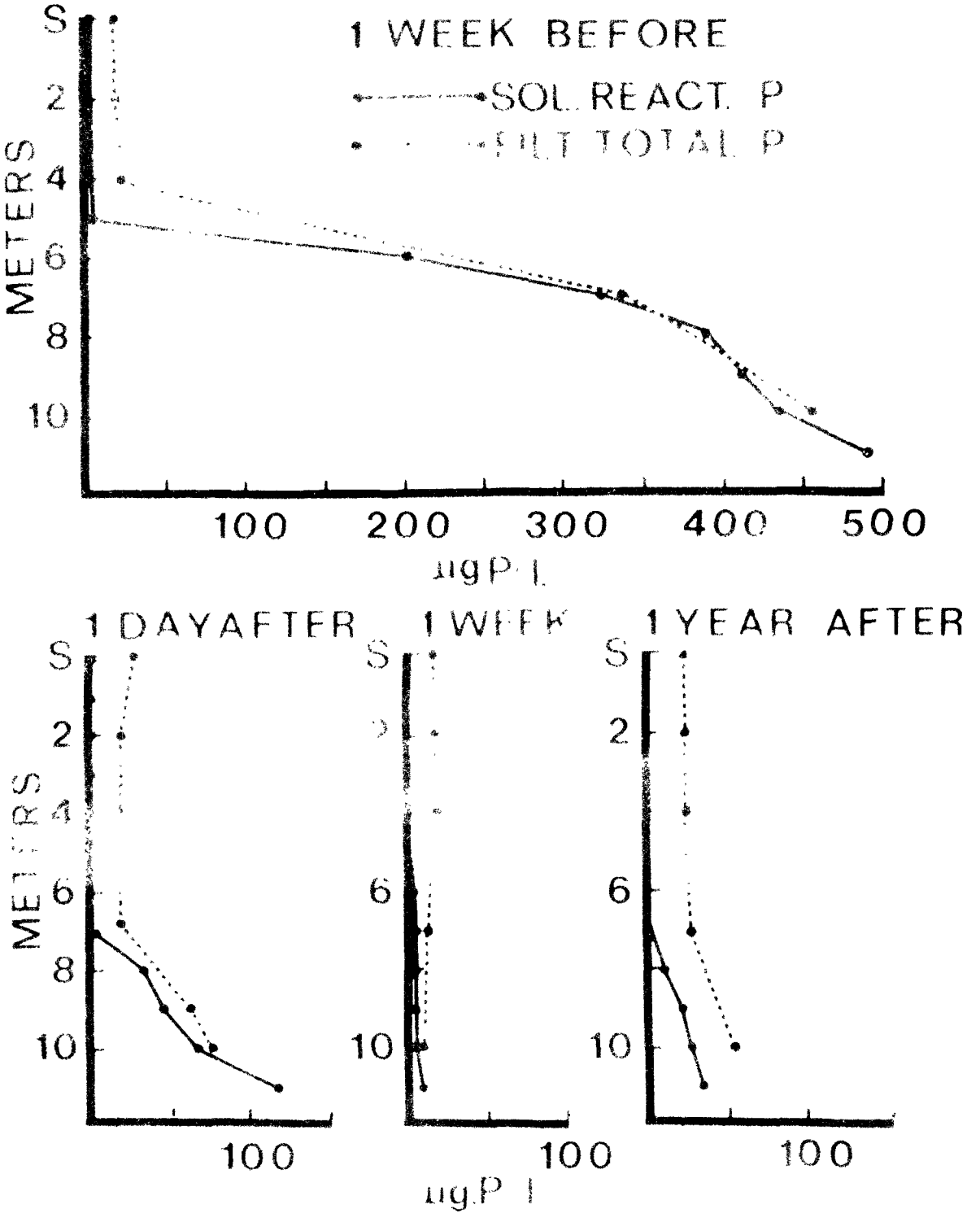


FIGURE [illegible]. Soluble reactive and filterable total phosphorus in West [illegible] before and after [illegible] application

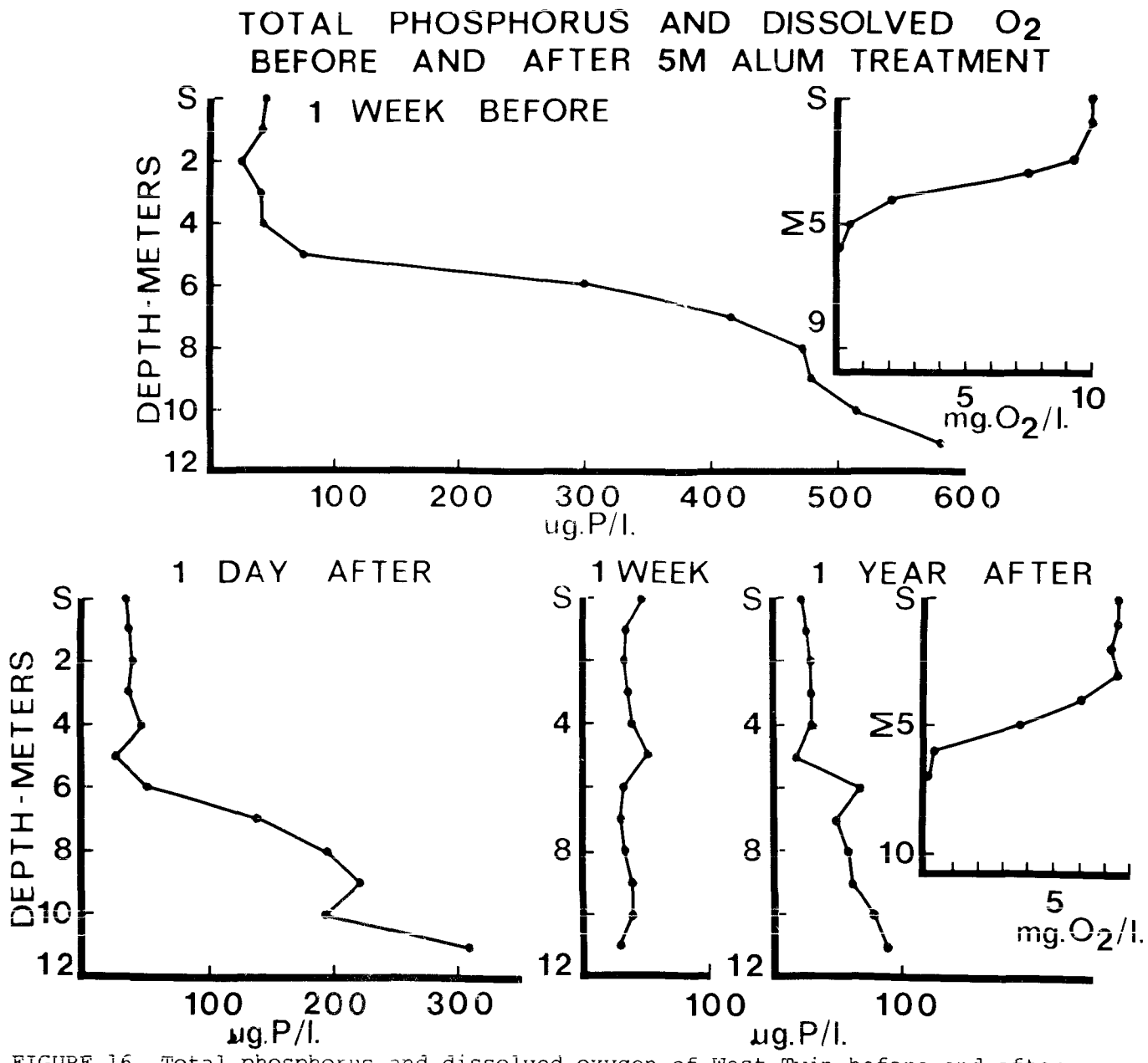


FIGURE 16. Total phosphorus and dissolved oxygen of West Twin before and after hypolimnetic aluminum sulfate application

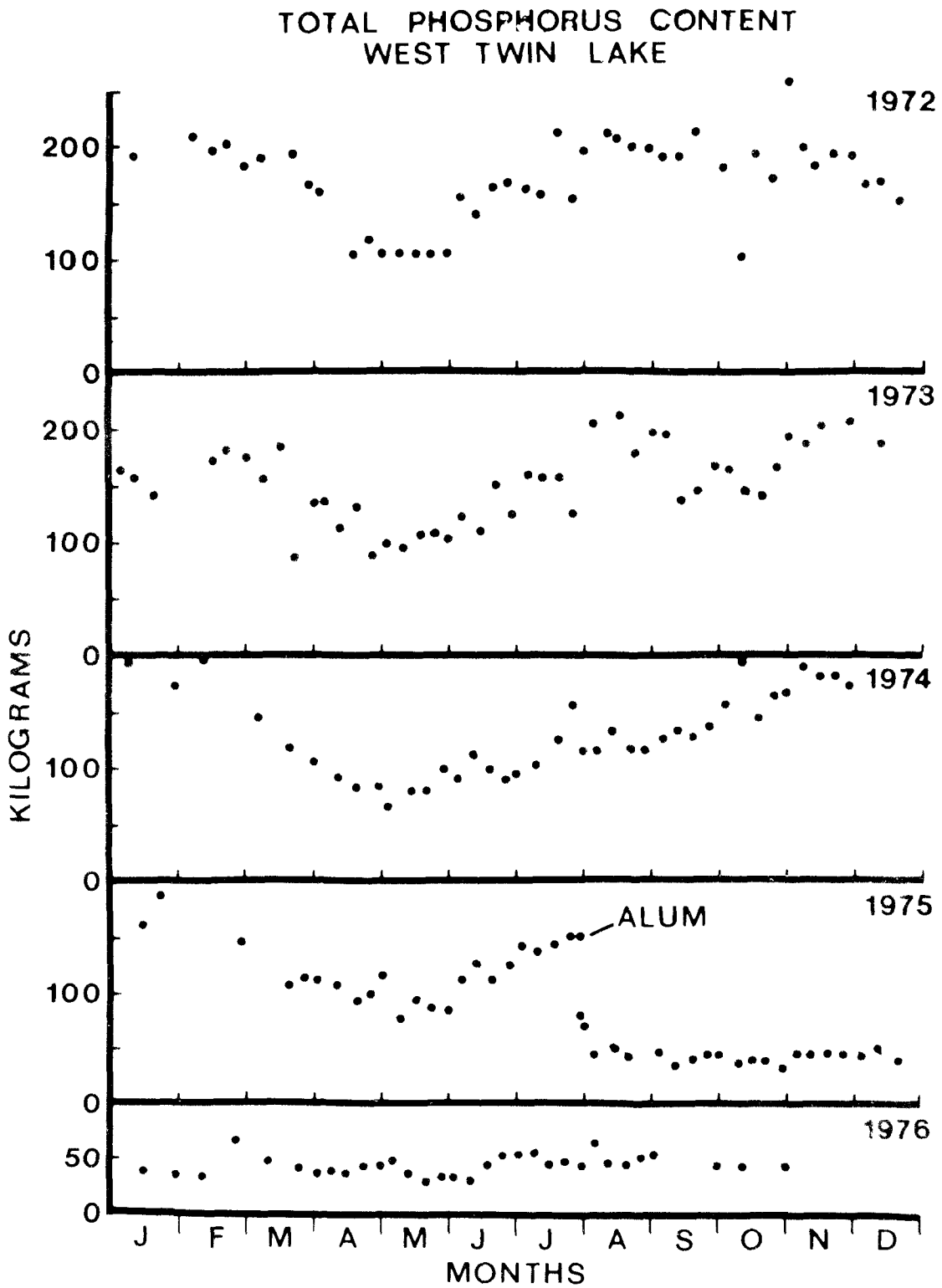


FIGURE 17. Phosphorus content of West Twin Lake, 1972-1976

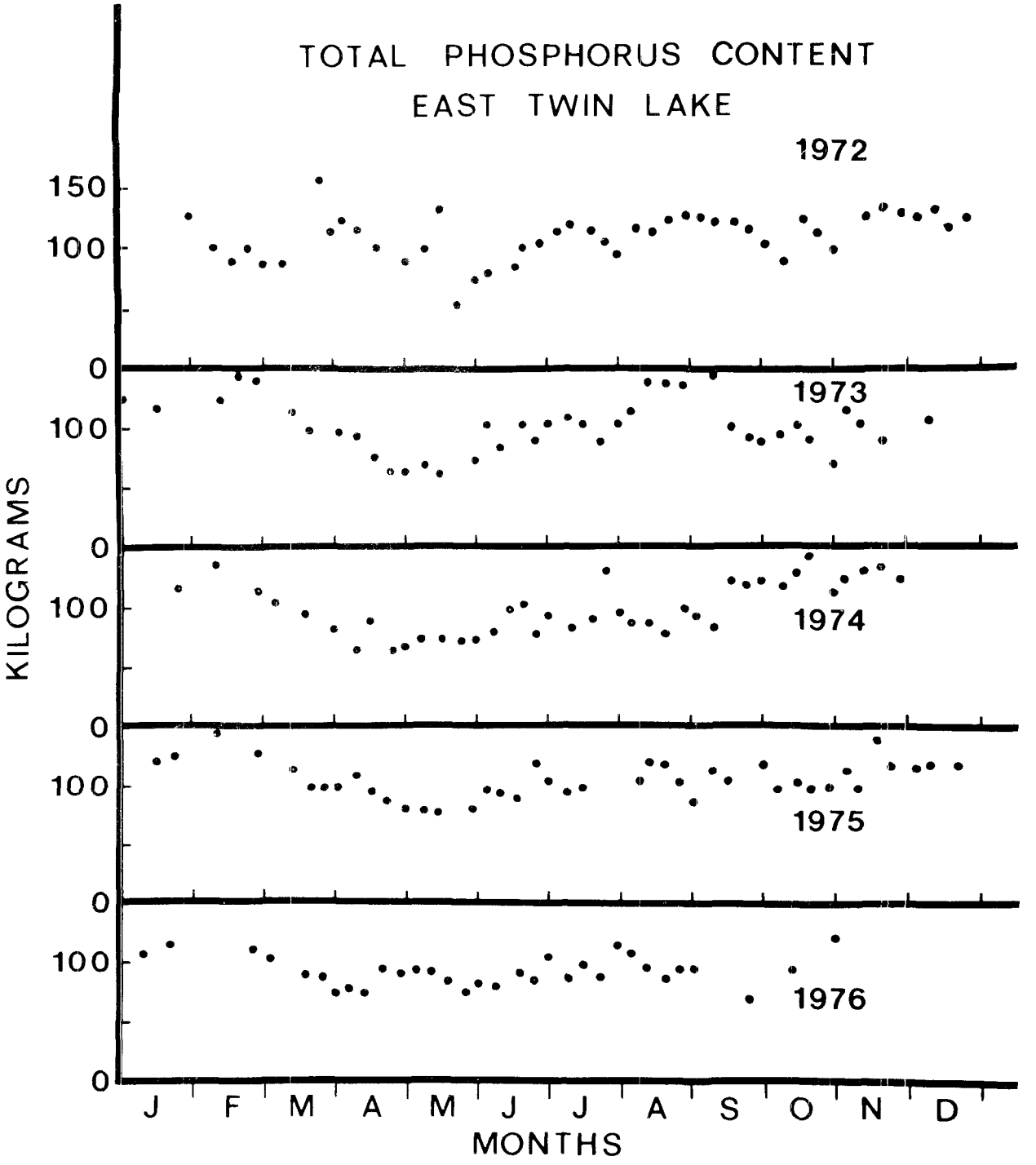


FIGURE 18. Phosphorus content of East Twin Lake, 1972-1976

Table 15. RATE OF INCREASE IN PHOSPHORUS CONTENT SPRING LOW TO SUMMER HIGH gms/day

	West Twin	East Twin
1972	751	769
1973	864	809
1974	4[illegible]1	292
1975	88[illegible]*	201
1976	228	147

*Up to day of alum application

Changes in the mean volume-weighted seasonal concentration of the three phosphorus fractions during 1971-1976 are illustrated in Figures 19, 20, and 21. Summer means include all lake strata. Changes during 1971-1974 are discussed first, since these changes may be related to diversion.

Mean total phosphorus (Figure 19) in both lakes was high in 1971 before diversion. West Twin, while initially higher in concentration, exhibited a steady decline from 1971 to alum treatment in 1975, while East Twin, after a small drop from 1971 to 1972, remained fairly stable at about 75 µg P/l. Spring mean concentration, which is strongly related to mean epilimnetic chlorophyll (Figure [illegible]) remained [illegible] 1970-76 [illegible] in East Twin, while West's declined [illegible] 1974.

Mean total phosphorus concentration in spring circulation in both lakes was determined by [illegible], lake morphometry, and water budgets. Predicted spring [illegible], using the model of Dillon and Rigler (1974b), was always higher than observed, except for 1972 and 1975 in West Twin, and always lower than [illegible] East Twin [illegible]. However, the correspondence between predicted and observed was close, supporting the conclusion that phosphorus [illegible], mean depth, flushing, and retention, determined spring [illegible]. Diversion thus appears to have been responsible for this reduction in the spring mean phosphorus concentration of West Twin.

Seasonal mean filterable total phosphorus (FTP) concentration (Figure 20) was much higher in West Twin in 1973 and 1974, and appeared to decline to East's level by [illegible]. [illegible] spring and summer 1973 for East Twin were questionable [illegible], tending to exceed SRP [illegible] and were therefore discarded. Mean seasonal [illegible] SRP concentration was almost always higher in West Twin [illegible] particularly during the fall (Figure 21). This difference in the fall may have been due to the export of autumn leaves, and leaf decomposition [illegible] West Twin. East Twin, it will be recalled, had no major inlet as entering it from its subwatershed. [illegible] relatively unaffected by diversion.

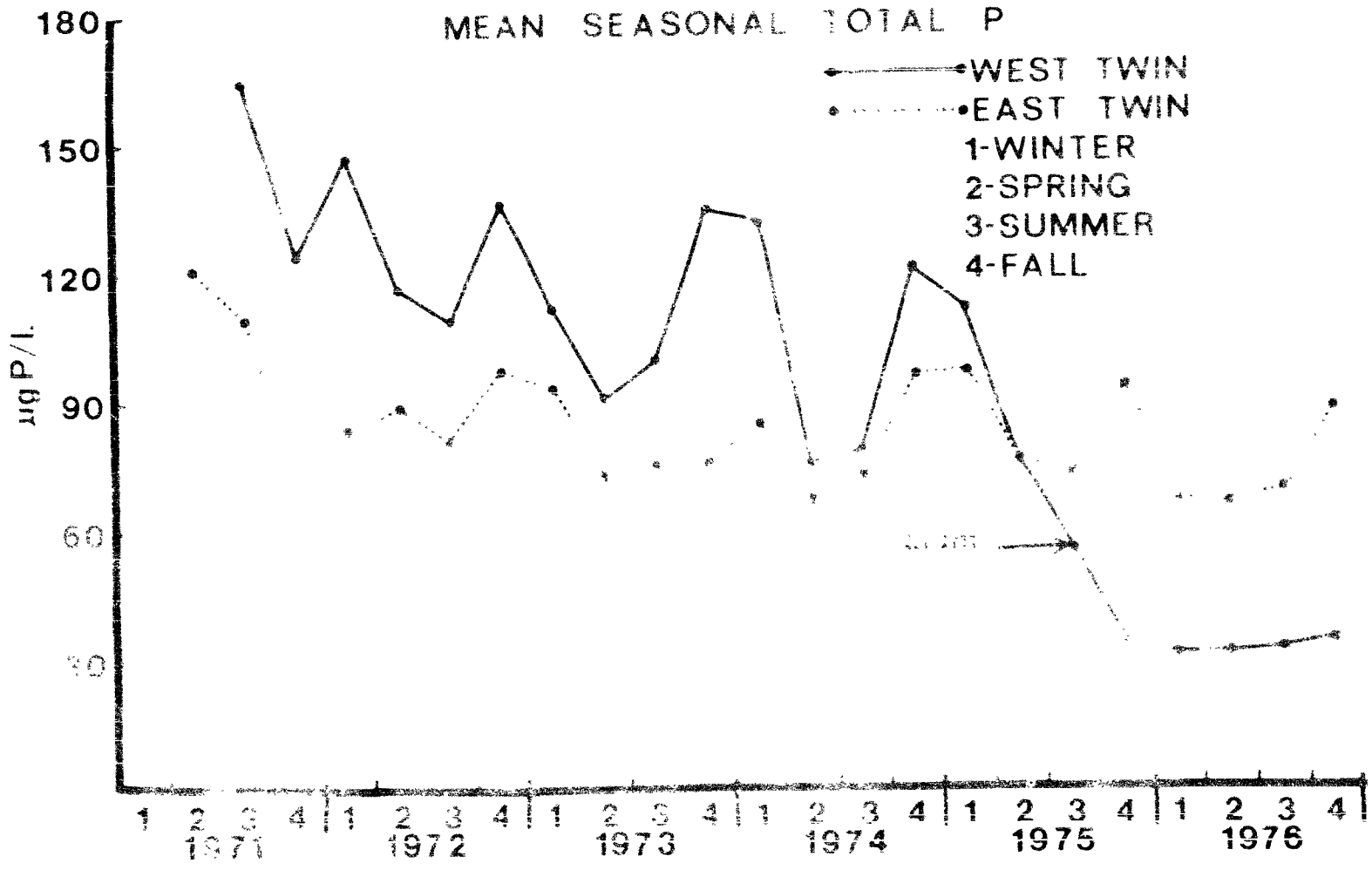
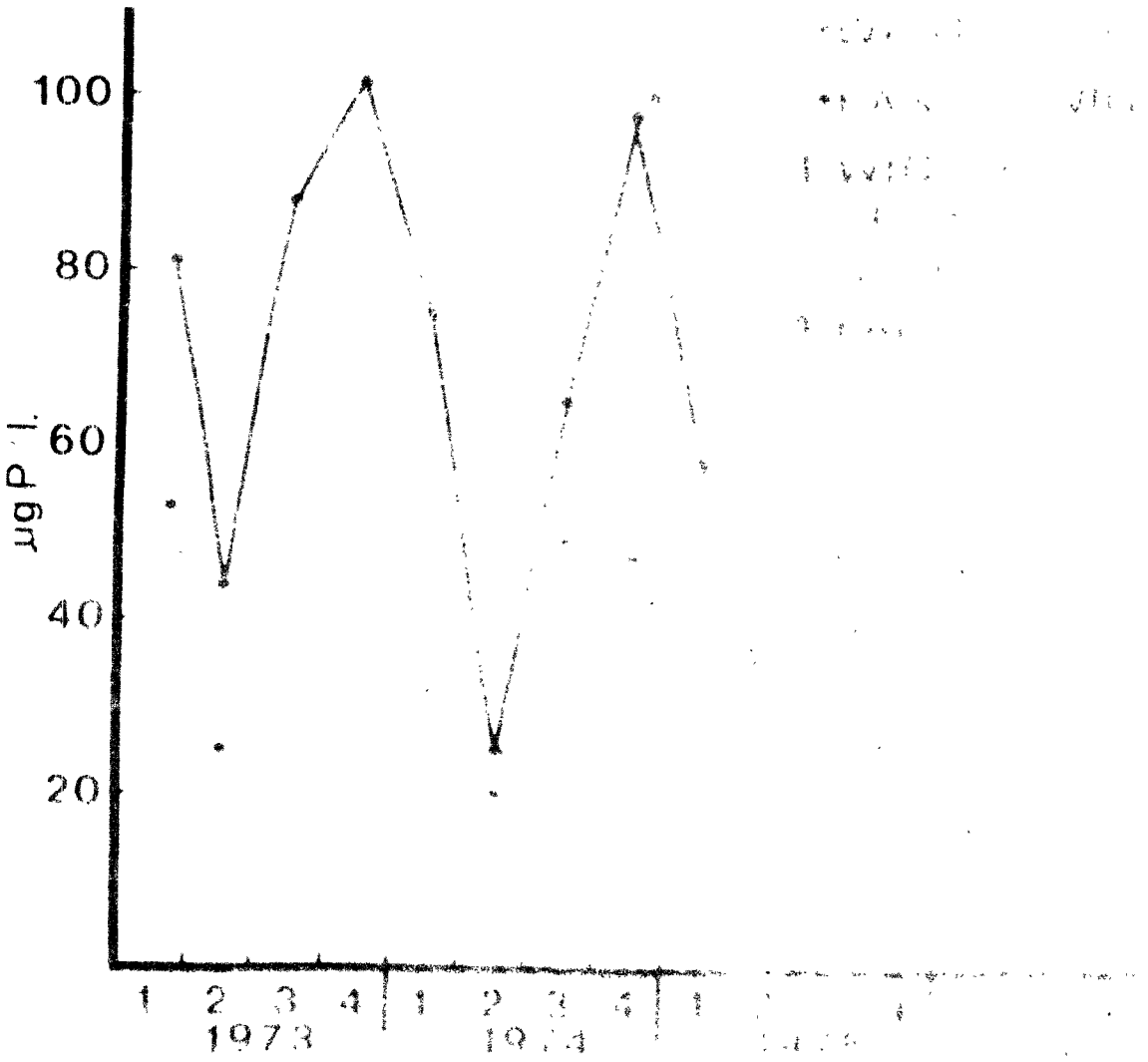


FIGURE 39. Mean seasonal total phosphorus concentration in East and West Twin Lakes, 1971-1976

MEAN SEASONAL



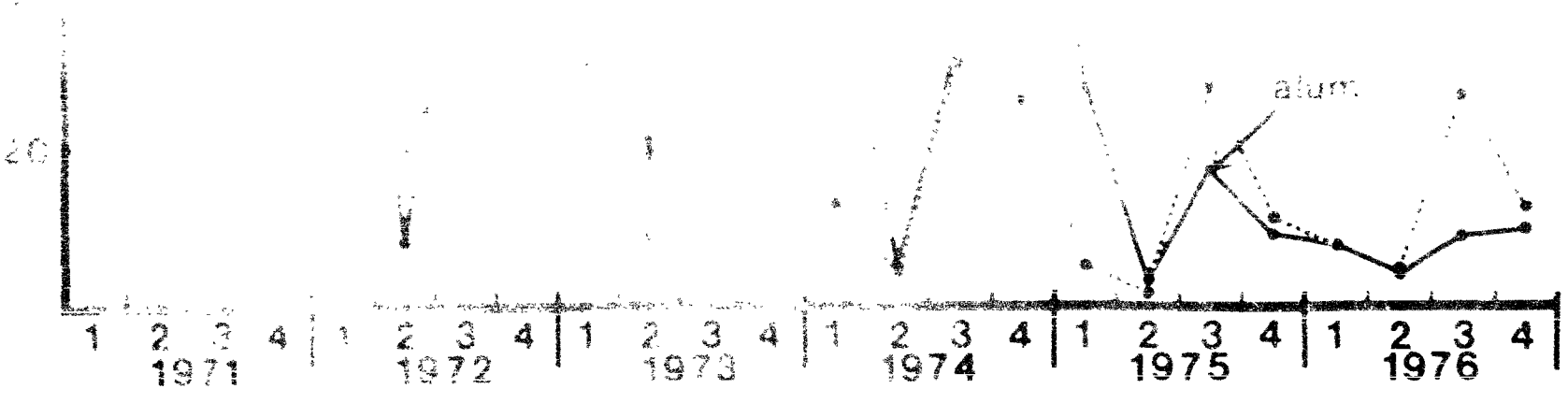


FIGURE 21. Mean [illegible] and West Twin Lakes, 1971-1976

West Twin, was greatly reduced from pre-treatment concentrations and in comparison to the hypolimnion of East Twin.

Table 17. AVERAGE EPILIMNETIC CONCENTRATIONS OF PHOSPHORUS FRACTIONS (µg P/l) (VOLUME-WEIGHTED)

		WEST TWIN LAKE		
		SRP	FTP	TP
1971		13.34	N.D.	40.59
1972		11.39	N.D.	58.09
1973		15.99	40.62	52.73
1974		10.25	29.31	47.46
1975	I*	3.50	26.42	43.80
	II*	5.40	15.55	28.93
	All	4.50	20.72	36.01
1976		3.05	19.16	28.26
		EAST TWIN LAKE		
		SRP	FTP	TP
1971		21.35	[illegible]	30.97
1972		8.84	[illegible]	44.50
1973		7.78	N.D.	37.61
1974		9.60	24.83	44.44
1975		5.02	23.15	39.38
1976		8.16	19.74	37.97

I = Days from stratification until [illegible] treatment
II = Days from treatment to autumn circulation
SRP = Soluble reactive phosphorus
FTP = Filterable total phosphorus
TP = Total phosphorus (See Lime [illegible] Methods, p. 26)

However, while mean total phosphorus in the epilimnion of West Twin was 25.5% less in 1976 than East Twin (28.2 vs 38.0 µg P/l), filterable total and soluble reactive phosphorus of the epilimnion were only 2.5% (19.2 vs 19.7 µg P/l) and 1.2% (8.1 vs 8.2 µg P/l) less, respectively (Table 18). Mean annual and epilimnetic concentrations of all fractions after treatment remained within the range expected of eutrophic lakes.

The small change in epilimnetic filterable total phosphorus may be of some significance since low molecular weight organic-phosphorus molecules

found as part of this fraction in the Twin Lakes (Francko and Heath, in press) may serve as sources of phosphorus to some phosphorus limited blue-green algae (Heath and Cooke, 1976).

In summary, sewage tank diversion apparently produced a steady decline in mean total phosphorus of West Twin, and the spring concentration in the lake declined to the level of East Twin's spring phosphorus. However, the 1975 spring concentration in both lakes, before the alum application, was about 75 µg P/L, a level which would produce a [illegible] chlorophyll concentration of a moderately eutrophic lake. This diversion was quite effective in lowering the concentration of phosphorus, and of biomass, over a four year post-diversion period to levels which were [illegible] improvement in symptoms of eutrophication.

Internal Phosphorus Loading - The release of phosphorus from anoxic lake sediments has been commonly observed in both laboratory and field experiments and is believed to be a significant internal source of phosphorus. The anoxic sediments of the hypolimnion of West Twin Lake were therefore treated with a maximum dose (as earlier defined) of aluminum sulfate in July 1975, with control of the [illegible] phosphorus loading to [illegible] the objective.

The [illegible] internal phosphorus [illegible] interface [illegible] (Cooke et al. [illegible]

[illegible] lake [illegible]

[illegible]

[illegible] in the phosphorus content [illegible] spring low [illegible] high.

[illegible] income

[illegible] phosphorus lost via outflow

[illegible] phosphorus gained (net release) [illegible] internal loading.

The [illegible] external and internal phosphorus loading, [illegible] lake outflow [illegible] the period of spring [illegible] lake [illegible] the summer [illegible] Table [illegible] loading [illegible] loading in Table [illegible] of the [illegible] were [illegible] West Twin was [illegible] This was [illegible] groundwater during the summer.

Net [illegible] internal loading rates declined [illegible] exception of a small increase in East Twin's 1975 [illegible] internal loading increased. The sum of rates indicating [illegible] 1974 [illegible] loading [illegible] twice that of 1974 [illegible]

Table 18. PRE- and POST- ALUM TREATMENT VOLUME WEIGHTED MEAN CONCENTRATIONS OF PHOSPHORUS FRACTIONS (μg P/l)

	WEST TWIN LAKE (treated)				EAST TWIN LAKE			
	Spring	Epi	Hypo	Year	Spring	Epi	Hypo	Year
	1. Total Phosphorus							
Pre-Treatment (1974)	74.9	47.5	420.5	96.6	67.3	44.4	405.6	76.9
Post-Treatment (1976)	30.8	28.3	75.2	32.1	65.4	38.0	268.8	70.2
Percent Change	-58.9	-40.4	-82.1	-66.8	-2.8	-14.4	-33.7	-8.7
	2. Filterable Total Phosphorus							
Pre-Treatment (1974)	25.9	29.3	358.0	69.5	20.0	24.9	N.D.	42.0
Post-Treatment (1976)	14.5	19.2	47.9	18.1	20.5	19.7	177.7	29.7
Percent Change	-43.3	-34.5	-86.6	-74.0	+2.5	-20.9	-45.8	-29.3
	3. Soluble Reactive Phosphorus							
Pre-Treatment (1974)	5.8	10.3	355.7	41.8	5.2	8.6	345.5	24.0
Post-Treatment (1976)	4.0	8.1	42.7	8.4	4.4	8.2	209.8	18.8
Percent Change	-31.0	-21.4	-88.0	-79.9	-15.4	-4.7	-39.3	-21.7

Table 19. NET SUMMER EXTERNAL, INTERNAL AND TOTAL PHOSPHORUS LOADING TO THE TWIN LAKES (mg P/M^2/day)

Year	External	Internal	Total	Days
		WEST TWIN		
1972	0.200	2.007	2.207	126
1973	[illegible]	.666	2.54[illegible]	126
1974	0.435	0.831	1.266	116
1975		NO CALCULATION	ALUM APPLICATION	
1976	0.607	.709	1.326	77
		EAST TWIN		
Year	External	Internal	Total	Days
1972	[illegible]	[illegible]	2.863	98
1973	[illegible]	.004	[illegible]	112
1974	[illegible]	.750	1.086	125
1975		NO CALCULATION -	ALUM APPLICATION	
1976	[illegible]	1.022	1.685	72

The 1975 alum application to the hypolimnion did not result in the expected sharp decline in 1976 net internal phosphorus loading of West Twin. Over 50% of the [illegible] in 1976 was due to internal loading and the rate was only [illegible] than the 1974 rate (.831 in 1974 vs .709 mg P/M^2 /day in 1976).

<u>In situ</u> experiments with polyethylene columns, where aluminum sulfate applied to the hypolimnion controlled phosphorus release for 104 days, and the inverted barrel experiments in Dollar Lake where alum treated (in 1974) lake muds exhibited very small phosphorus release (through 1976) when compared to untreated muds, strongly support the conclusion that the release of phosphorus from alum-treated hypolimnetic muds in West Twin is controlled (see Appendix and Kennedy, 1978, for further [illegible] of the [illegible] process). However, the similarity in mean epilimnetic total phosphorus concentration between the lakes in 1976 (WT = [illegible] mg P/l, [illegible]) and the very low hypolimnetic [illegible] (see Figure [illegible] and Table 18) support the conclusion that internal transport, described by Stauffer and Lee (1973) as a source of internal loading to the epilimnion, was not of great significance in the Twin Lakes. Significant portions of the net internal phosphorus release [illegible] the summer must have been from sources other than the anaerobic [illegible] deposits on

effective alum treatment and the concentration of phosphorus in the epilimnion of the two lakes was similar despite dramatic differences in hypolimnetic concentration.

Further studies are needed to confirm this conclusion, but it does suggest an explanation for other observations. For example, it is not intuitively clear why the late winter - early [illegible] phosphorus concentration should be strongly related to mean epilimnetic summer concentration (Dillon and Rigler, 1974a; Figure 5), when that phosphorus is lost from the water column during April and May each year (Figures 17 and 18). However, if the precipitated phosphorus is released as summer internal phosphorus loading, then the strong correlation may be explained. In small lakes, groundwater and the biota of the littoral may be especially important in returning phosphorus to the water. Macrophytes have been shown to transport phosphorus from roots through leaves to the water (e.g., McRae et al. 1972), and fish may release significant amounts (Lamarra, 1975). [illegible] and others strongly suggest that both abiotic and biotic [illegible] the littoral zone may be the major contributors of internal phosphorus loading in lakes like the Twin Lakes.

The annual phosphorus retention coefficient is related to internal loading. Kirchner and Dillon (1975) developed an equation from the data for 15 Canadian lakes which estimated R_p from the [illegible] water load ($r = 0.94$). In Table 20 the R_p's predicted from their equation and the observed R_p's for Twin Lakes are listed (see also Tables 11 and 12). Both of the Twin Lakes had R_p's far lower than predicted, which means that either incoming phosphorus was not sedimented in great quantities or was re-released after sedimentation as internal loading, and then lost through the outflow. The evidence of internal loading presented here for the summer period supports the belief that phosphorus was rapidly recycled in the Twin Lakes. Further, since it appears that the littoral zone may be a major source of the internal loading, a large amount of sedimentation must also occur here.

Table 20. PREDICTED AND OBSERVED PHOSPHORUS SEDIMENTATION COEFFICIENTS IN THE TWIN LAKES

	EAST TWIN		WEST TWIN	
Year	Observed	Predicted	Observed	Predicted
1972	0.213	0.592	0.093	0.724
1973	0.140	0.597	0.140	[illegible]
1974	0.283	0.542	0.161	0.688
1975	0.187	0.537	0.155	0.681
1976	-0.064	0.562	0.064	0.764

Lakes with a prolonged history of high [illegible] "saturated" with regard to their ability to efficiently retain phosphorus [illegible] the sediments and/or [illegible] Twin was [illegible] conclusion that [illegible] to the retention [illegible] retention of phosphorus [illegible]

In a hypolimnetic [illegible] the hypolimnion, [illegible] not the only major [illegible] results of Kerfoot [illegible] anaerobic [illegible] phosphorus [illegible] apparently [illegible] "release" [illegible] algae [illegible] to estab[illegible]

The [illegible] on the source of [illegible] where the [illegible] thermocline [illegible] be very [illegible] very expensive [illegible] littoral [illegible] more difficult [illegible] groundwater [illegible] alum [illegible] since Lee [illegible] declines [illegible] biotic [illegible] alum may not be sufficient to shift [illegible]

The littoral area alone of a small lake [illegible] where an accurate internal budget [illegible] allow some assessment of the [illegible] phosphorus to net internal loading [illegible]

Effect on Conductance, pH, Alkalinity, and [illegible]

The immediate [illegible] chemistry, [illegible] Levels [illegible] at [illegible] tions were [illegible] except for [illegible] twice the [illegible]

[illegible] INDICATORS

[illegible] with coliform [illegible]. Presumably [illegible] surface, in [illegible] 70 the lakes [illegible] (CFU) exceeded

[illegible] in 1972 was [illegible] East Twin's [illegible] the summer of [illegible] lowered [illegible] of [illegible], 2G (3 [illegible] West [illegible] of septic [illegible] coliforms [illegible] the water [illegible] ml). [illegible] was

[illegible] developed [illegible] representative of [illegible] very high [illegible] very [illegible] this [illegible]

[illegible] tank [illegible] water [illegible]

[illegible]

[illegible] must bring about a [illegible] the spring and [illegible] impact of [illegible] indicators [illegible] intervals [illegible] report data [illegible]

[illegible] OF [illegible] COLLECTION [illegible] CLONE FORMING UNITS

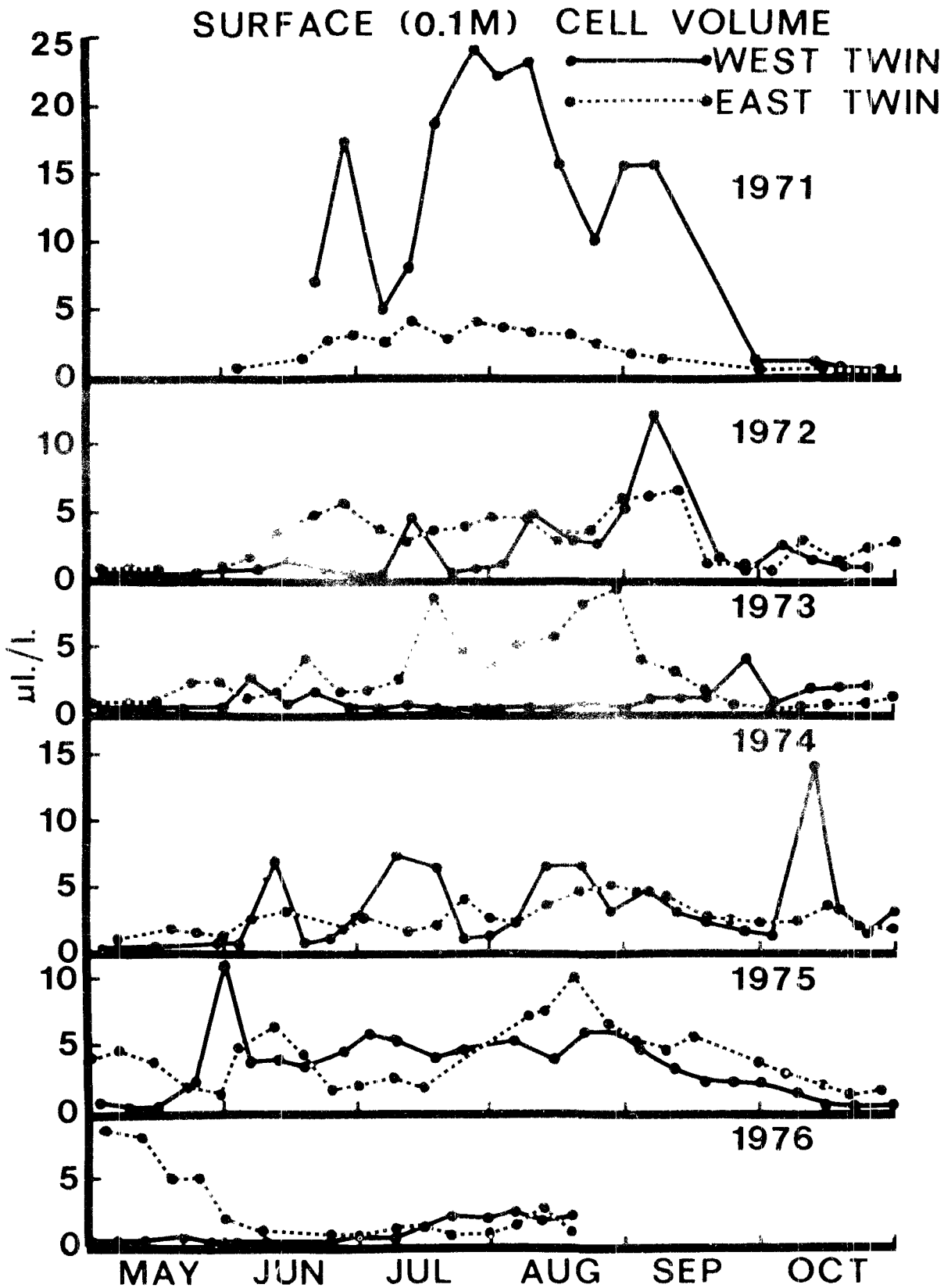


FIGURE 22. Changes in summer surface cell volume of the Twin Lakes 1971-1976

volumes of both lakes have remained at about 5 µl/l, a value expected of moderately eutrophic lakes (Vollenweider, 1968).

In 1976, the year after alum treatment, spring cell volumes were very different. West Twin was clear and had no blue-green algae, whereas East Twin was a greenish-brown color and supported a dense bloom of Oscillatoria rubescens. The same pattern occurred in spring 1977. By summer 1976 both lakes had a smaller cell volume than had occurred in any previous year, except 1973. Since algicide use was small in 1976, it appears that the alum application, which resulted in a lower epilimnetic total phosphorus but not SRP concentration, combined with the lower phosphorus income of 1976, was effective in lowering cell volume. It is suggested that the absence of a spring blue-green algae bloom in West Twin may also be due to the layer of aluminum hydroxide, since blue-green algae blooms apparently originate from populations in the sediments (Reynolds and Walsby, 1976).

The pattern of changes in surface chlorophyll [illegible] closely paralleled cell volume changes. Chlorophyll in West Twin dropped from a pre-diversion high in 1971 to the level of East Twin in 1972 (Figure 23). Both lakes were very low in 1976, as expected from the strong correlation between spring total phosphorus concentration and mean [illegible] (Figure 5). Sakamoto (1966) reported that eutrophic lakes [illegible] range of chlorophyll of 5-140 µg/m³. These lakes have had a [illegible] monthly average of about [illegible] µg/m³ since diversion and alum treatment, a level found in moderately eutrophic lakes.

Changes in the species composition of the summer surface phytoplankton followed a similar pattern each year. An early spring diatom bloom (Fragilaria, Asterionella) was followed by a sharp [illegible] cell volume in late April early May, and then a series of blue-green algae blooms occurred over the summer, with diatoms as sub-dominant [illegible] July. By August 90% or more of the cell volume was due to blue-green algae, either Anabaena limnetica or Aphanizomenon flos-aquae, and this pe[illegible] November. The summer dominant in both lakes in all years exce[illegible] 4 and 1975 was A. flos-aquae. A. limnetica was the 1974-75 dominant in both lakes.

In 1974 and 1975, Chlorophyta and Dinophyceae began to be a more abundant component of the late summer algae (Table 22). In 1976, the year after alum treatment, blue-green algae dominance was greatly reduced, particularly in East Twin, the downstream control lake.

The seston content of West Twin's surface waters was very [illegible] 1971 [illegible] as with [illegible] spring, declining [illegible] level through 1976 (Figure 24).

Changes in Secchi disk transparency were [illegible] cell volume, seston, and chlorophyll (Figure 2[illegible]). In 1975, 1976, and also in 1977 (not shown) the spring (April May) transparency of West Twin exceeded 4 meters while East Twin remained murky due to Oscillatoria. In 1976 the interval of high water clarity in West Twin persisted into July. The midsummer transparency has remained at about 2 meters following an initial increase between 1971 and 1972 for West Twin, a level found in moderately eutrophic lakes.

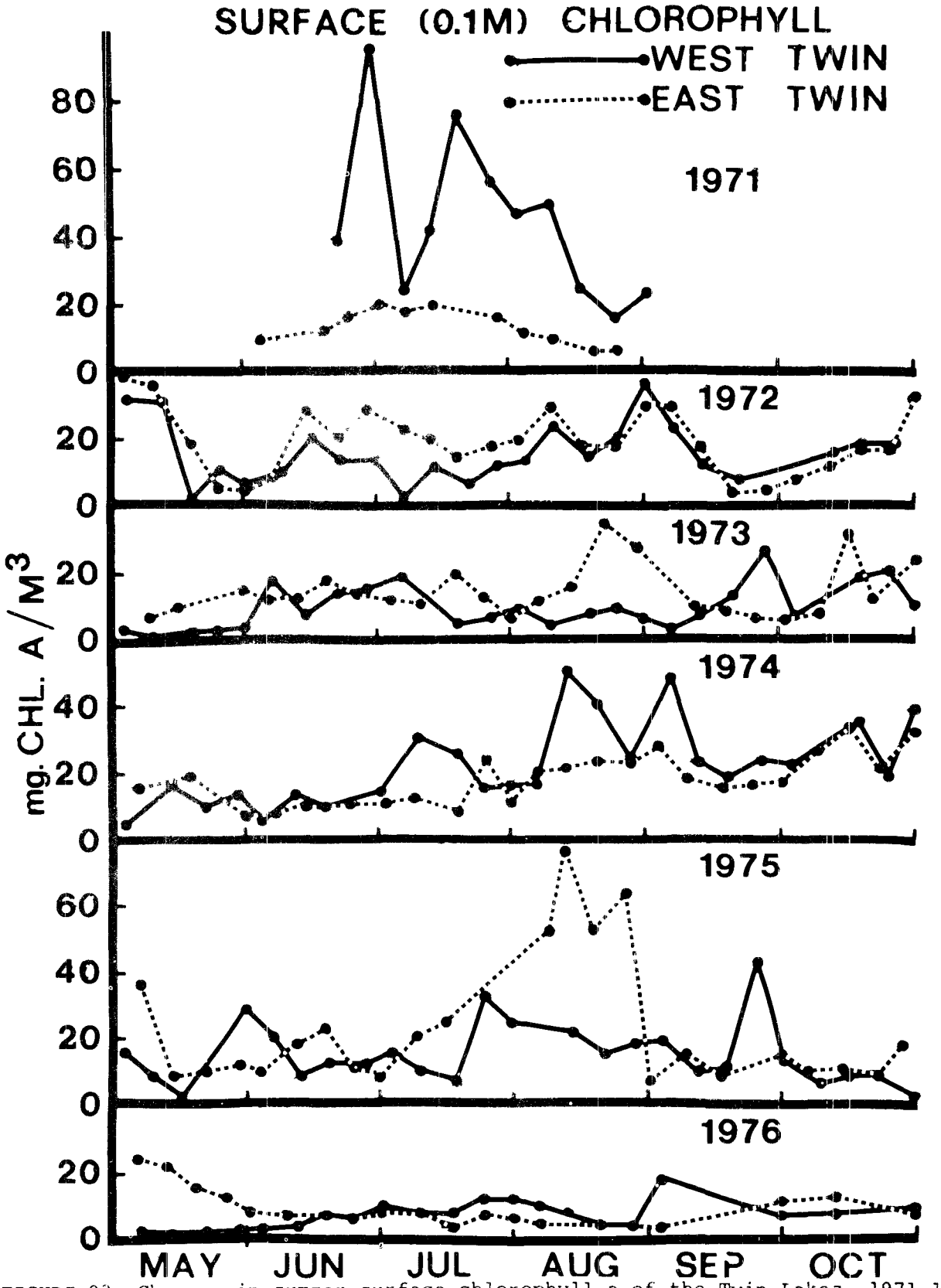


FIGURE 23. Changes in summer surface chlorophyll a of the Twin Lakes, 1971-1976

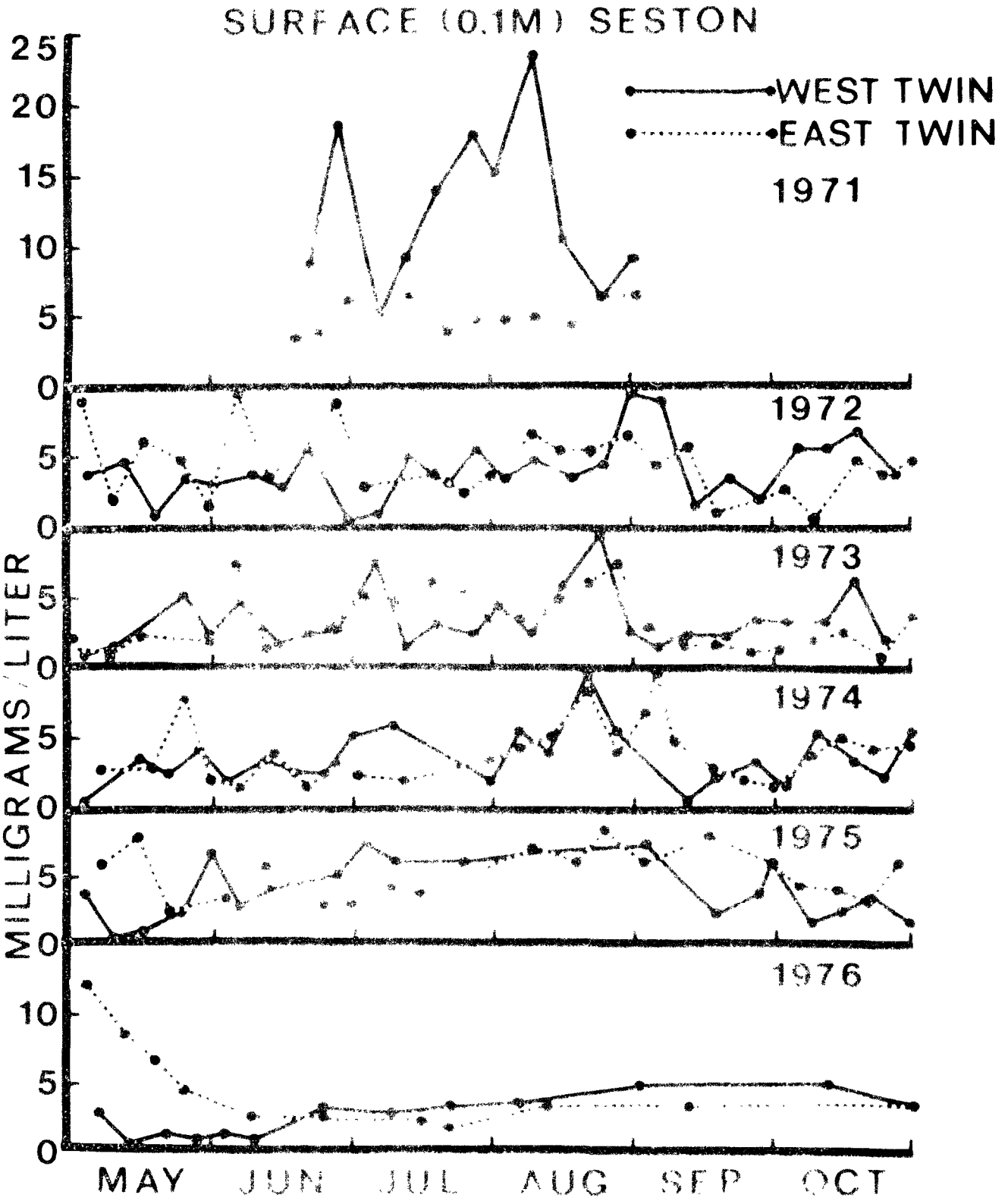


FIGURE 24. Changes in [illegible] Lakes [illegible] 1976

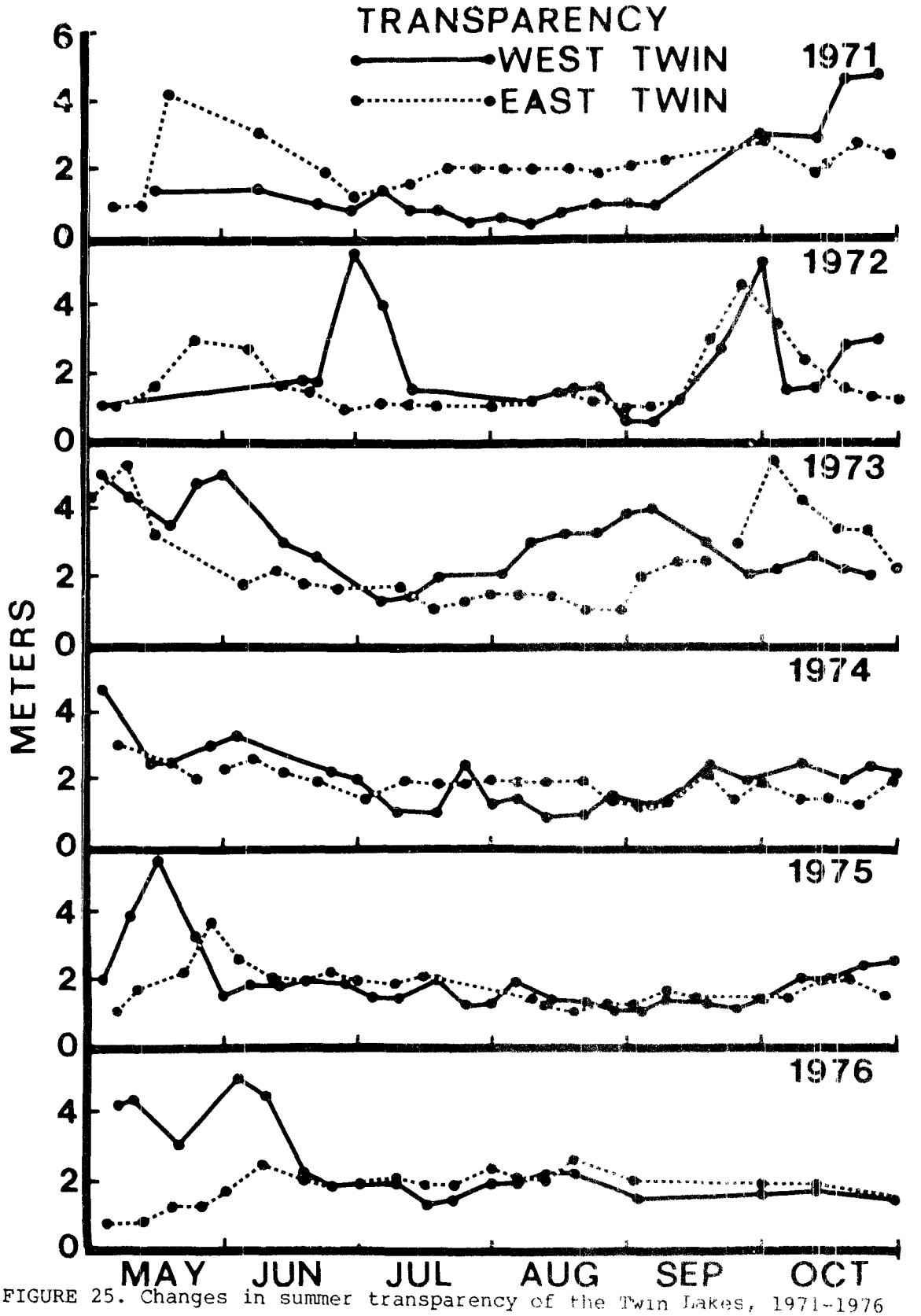


FIGURE 25. Changes in summer transparency of the Twin Lakes, 1971-1976

Table 22. MEAN PER [illegible] PHYLA AND [illegible]
SURFACE [illegible] MEAN [illegible]

[illegible]

1. [illegible]

2. [illegible]

3. [illegible]

4. [illegible]

Mean [illegible]

[illegible]

[illegible]

4 [illegible]

Mean [illegible]

[illegible]

* [illegible]

Oxygen deficits

The [illegible] pattern of change in the oxygen deficits (Table 23) which could be directly related to septic tank diversion or to the alum treatment. However, as shown by Stewart (1970) for Lake Mendota, the deficits were positively correlated with day of [illegible] (r = .797), suggesting that the oxygen deficits in lakes, even they are eutrophic, is not directly related to productivity

Table 23. OXYGEN DEFICITS OF EAST AND WEST TWIN LAKES mg/[illegible]/M²/Month

	[illegible]	EAST TWIN
[illegible]	[illegible]	[illegible]
1971	N.D.	2.25
[illegible]	[illegible]	1.76
[illegible]	[illegible]	1.56
[illegible]	0.76	1.00
[illegible]	[illegible]	[illegible]
[illegible]	[illegible]	[illegible]

[illegible]

Ma[illegible]

[illegible] phytes were surveyed by Rogers [illegible] qualitatively after that. These plants [illegible] the lakes, and their abundance [illegible] diversion or alum treatment.

Prob[illegible]

[illegible] the eutrophic state [illegible] for certain [illegible] most recent of [illegible] 1977)

[illegible] low transparency, high [illegible] phosphorus. The Carlson [illegible] growing season (May 1 [illegible] of secchi depth (SD), [illegible] total phosphorus (TP), [illegible] a pre-diversion TSI [illegible] for 1969. Averages [illegible] are reported in Table 24.

The TSI for East Twin did not vary much from one year to the next, and for the years 1969 to 1975 the same can be said of West Twin, with the exception of 1973 determinations. In 1973 the TSI determination from Secchi depth and from chlorophyll a observations are substantially lower than other years. That the 1973 TSI determined from total phosphorus is not different from 1972 or 1974 suggests that the phytoplankton community was limited by some factor other than phosphorus, probably a very heavy addition of herbicides. The average TSI determined from each set of observations in 1976 was lower than in other years. Although this decrease is not statistically significant (one-way analysis of variance) it does suggest that the application of a [illegible] has had some impact on the degree of eutrophy. This view is supported by the reduction in cell volume and the change in algal species composition.

SECTION 7

DISCUSSION

The results of this study have implications for lake management, for lake classification and for general understanding of the eutrophication problem in small lakes.

Septic tank systems have been commonly installed in residential developments around lakes, and probably will be the usual method for domestic waste disposal in most future lake developments. This study has shown that these systems, even when properly installed in adequate soils, may not operate as designed and will discharge materials to the ground surface.

This finding is significant because it has been commonly believed that phosphorus will not enter lakes via groundwater in large amounts because of the sorptive capacity of soils. As earlier pointed out by Dudley and Stephenson (1973), old systems or those with high or perched water tables in [illegible] may be expected to [illegible] groundwater with phosphorus. The present study further supports the belief that septic tank systems should not be used near lakes because the leach fields appear to plug, inhibiting downward movement of water. The continued water income to the field from the home eventually forces water and nutrients to the ground surface. This is the cause of the commonly observed soggy lawn and resulted in very high concentrations in stream draining urban areas prior to diversion at two lakes.

The diversion of the septic tank system alone did not result in a reduction of phosphorus income to the Vollenweider (1976) "acceptable" level. This was confirmed by levels of algal standing crop five years after diversion and by a Trophic State Index (Carlson, 1977).

This response to diversion was to be expected. The watershed was developed as a residential-recreation area away from the usual urban setting [illegible], traffic, and congestion. However, development occurred without [illegible] that erosion and street wastes as well as seeps from septic systems [illegible] the lake and not [illegible] the natural [illegible] of the water and [illegible] the lake. The result was lakes that do not provide the complete [illegible] envisioned by either the developers or the residents.

Unlike lakes which receive point sources of nutrients, the lakes have become culturally eutrophic from diffuse sources. Development continues on the watershed, in many cases on lots which are not suitable for homes unless the land is completely cleared, a marsh filled, or where some other operation is required which has a large impact on the lake. The usual development

practice is to leave the cleared land without cover for a year or more during home construction, thus adding tons of soil to the lake. As a result of years of these practices, the littoral zone has expanded from erosion, creating a favorable environment for the major nuisance, macrophytes.

It is not likely that this very common type of eutrophication problem will be solved by reducing the income of nutrients alone through septic tank diversion. A rigorous land use policy and storm water diversion might slow the further deterioration of the lakes, but these steps are expensive and politically unpopular, and in some cases impossible if the land owner has the deeded right to develop the land, regardless of its suitability for such development. Even if phosphorus income can be greatly reduced, this may not alter the macrophyte problem and these plants and their associated biota may continue to add phosphorus to the epilimnion at rates sufficient to support algae growth, even after further nutrient diversion.

The prognosis for the Twin Lakes, and the many hundreds of lakes like them, is not favorable unless steps are taken to manage the land in ways which have less impact on the lakes. Restoration of such lakes then might be possible, although the cost could be high.

The diversion of septic tank drainage at Twin Lakes, however, was essential to prevent further deterioration of water quality and to return the lakes to a condition more acceptable to lake shore residents. Based upon measurements in 1971 and early 1972, before diversion, the pre-diversion phosphorus income may have been more than twice as high as the immediate post-diversion income and would have continued to climb had there been no diversion, due to leach field failures and continued planless construction.

This study has provided evidence, from the hypolimnetic aluminum sulfate application to West Twin, that the littoral zone may be a very significant internal source of phosphorus. The source(s) within the littoral was not identified. This suggests the hypothesis that there may be two classes of eutrophic lakes. The first class, like the Twin Lakes, has small pelagic-littoral area ratios and internal loading is dominated by sources such as macrophytes-epiphytes, fish and groundwater. Lakes of this class are likely to be small in total area and perhaps protected from wind by features of the landscape, thus reducing the possibility of large internal seiches. The second class has a large pelagic-littoral area ratio, is usually much more exposed to winds and thus has large seiches, and vertical entrainment of phosphorus from the hypolimnion is the main internal source of phosphorus. Lake Mendota as described by Stauffer and Lee (1973) may be of this type, although the littoral of that lake is extensive and must be important.

Based upon this hypothesis and the observations of internal loading after alum treatment of the Twin Lakes, hypolimnetic application might be most effective on large lakes. However, cost and ease of application might be prohibitive.

The hypolimnetic aluminum sulfate application may continue to reduce the phosphorus concentration in West Twin since sedimented material, as well as

that which might have been released from hypolimnetic muds, appears to be satisfactorily controlled. This may lead to smaller summer algal blooms if spring concentrations remain low.

One of the [illegible] further [illegible] based upon the results of this investigation, will be to identify and quantify the sources of internal phosphorus [illegible] from the littoral, and to assess the significance of groundwater [illegible] phosphorus into the water of the littoral zone as it passes through [illegible] sediment. The next alum application should be in a spring to the littoral area of a small lake for which there is a good phosphorus [illegible].

The preponderance [illegible] eutrophication has been directed toward algal problems, and [illegible] correctly identified this nuisance as the most significant [illegible] particularly those which deserve protection due to aesthetic, [illegible] commercial reasons. However, of equal significance is the [illegible] real estate lake development. In many of these [illegible] eastern states [illegible] equally difficult and important problem. This [illegible] not only do these [illegible] productive littoral zones [illegible] water use, they may also release sufficient [illegible] algal blooms after a significant [illegible].

As [illegible] (1[illegible]), Patten (1971), and Cooke (1973), understanding [illegible] requires an ecosystem perspective [illegible] nutrient concentration - algal biomass approach. [illegible] investigations have observed that the lakes [illegible] accumulated. This probably [illegible] zone with most active development. [illegible] strongly related to [illegible] predicted [illegible] phosphorus [illegible] the water column in summer by both [illegible] the littoral zone. The control of algal blooms at [illegible] lakes will involve not only a reduction in nutrient income through [illegible] wastes, but land management and control of the internal [illegible].

The [illegible] lakes appears to be more complex than for large [illegible] of the lake-watershed system are significantly [illegible]. A [illegible] problem for smaller lakes, however, may be more [illegible] water [illegible] regional.

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APPENDIX

INTRODUCTION

A general outline of our approach to aluminum sulfate dose determination, experiments to test its effectiveness, and application procedures was given in the Materials and Methods section. Here the step-by-step approach taken during this investigation is described in detail. Experiments were confined at first to the laboratory, then to in situ enclosures to test toxicity and effectiveness, and finally to a pilot lake treatment (Dollar Lake). It was planned that if, at any step, there was evidence of the possibility of damaging side effects then no full-scale lake treatment would occur. The successful treatment of the eutrophic Dollar Lake (1974) was the final step before treatment of West Twin in 1975. A further description of these experiments, including a complete description of the Dollar Lake treatment, is given in Wilbur (1974) and Kennedy (1978).

The Appendix is organized to give the reader a summary of the experiments. The procedure for the determination of West Twin's dose is given in detail. Finally, a section is presented on the equipment and application design, the application procedure for the West Twin treatment, and a summary of costs. Non-metric units are used in portions of the Appendix since aluminum sulfate is supplied as gallons or pounds and temperature and other corrections are based on these units.

DOSE DETERMINATION EXPERIMENT

Lake waters collected at varying depths (and thus with varying total alkalinity) in West Twin Lake and Dollar Lake were treated with aluminum sulfate at dosages ranging from 0-50 mg Al/l to confirm the relationship between the amount of aluminum added and initial alkalinity. One liter jars containing 500 ml of lake water were stirred at 40 RPM for two minutes following the addition of a liquid concentrate of aluminum sulfate. Changes in pH were measured during the first two minutes. Supernatants were subsampled following a 12 to 24 hour settling period and measurements of alkalinity and dissolved aluminum taken.

Results obtained from the treatment of West Twin Lake 3 meter water (initial total alkalinity = 137.3 mg $CaCO_3$/l) are typical (Figure A-1). Doses of 1 to 26 mg Al/l produced a large settleable floc, while doses above 26 mg Al/l resulted in a white turbid reaction mixture which did not settle after 18 hours. No measurable floc was produced at dosages below 1 mg Al/l. Alkalinity dropped rapidly between 0.5 and 26 mg Al/l (137.3 to 6.0 mg $CaCO_3$/l) and was completely removed above a dose of 30 mg Al/l. pH was reduced from 7.2 to 5.13 at 26 mg Al/l and to 4.9 at 30 mg Al/l.

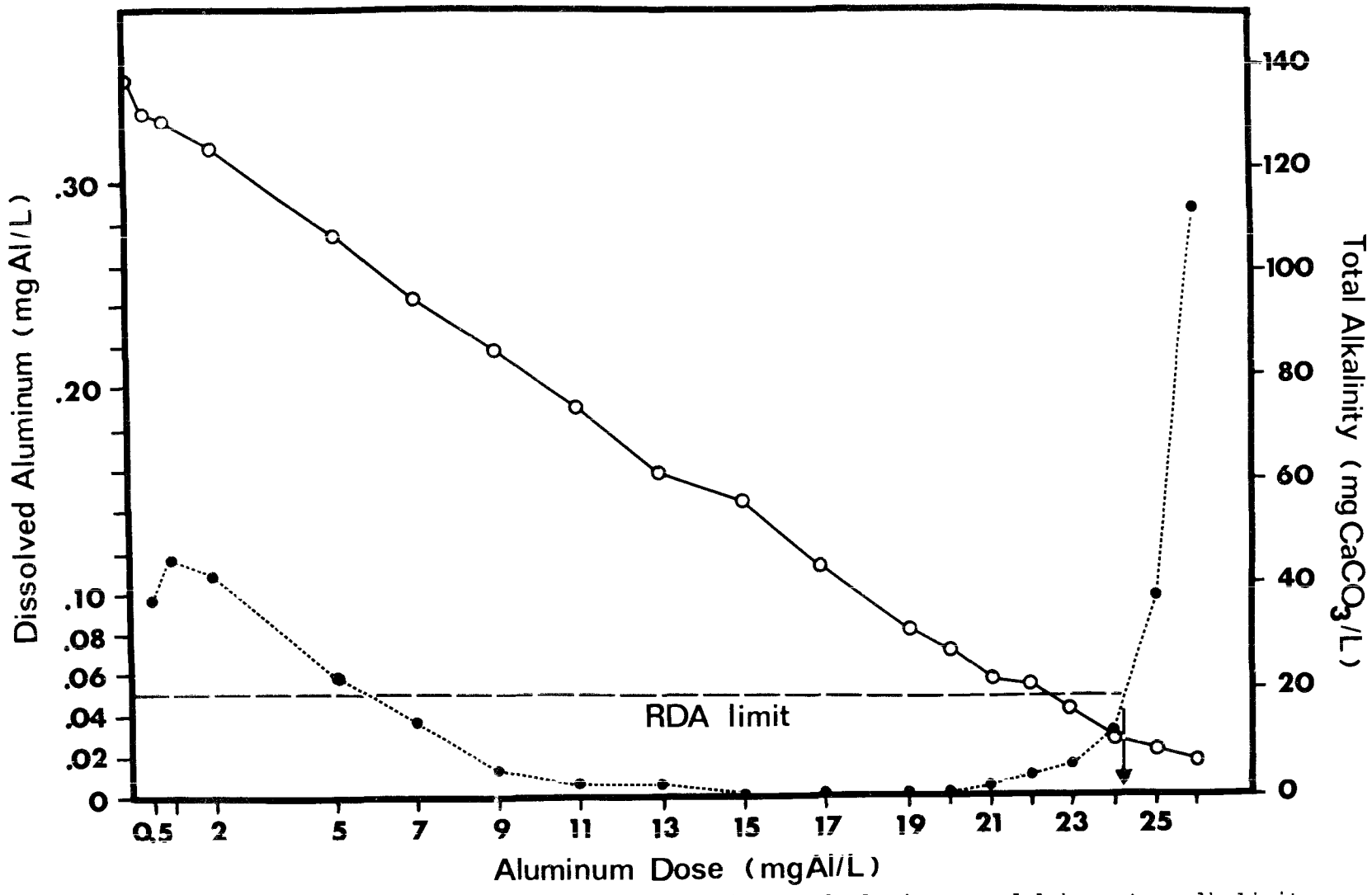


FIGURE A-1. Relationships of aluminum dose, residual dissolved aluminum, and lake water alkalinity

Residual dissolved aluminum, high in the dose range 1 to 7 mg Al/l, remained low (.02 mg Al/l) in the dose range 9 to 23 mg Al/l. Doses above 24 mg Al/l resulted in elevated residual dissolved aluminum concentrations. The maximum dose for this alkalinity was calculated to be 24.2 mg Al/l since this dosage represents the maximum amount of aluminum which can be added (under the conditions of this test, i.e., 2 minute mixing) before residual dissolved aluminum concentration exceeds 0.05 mg Al/l, a level reported to be tolerated by trout (Everhart and [illegible] 197[illegible]).

Maximum dosages [illegible] in like manner for the full range of alkalinities found [illegible] these lakes [illegible] used to determine the relationship between alkalinity and aluminum dose using linear regression techniques. The relationship is expressed as

$$y = [illegible].4460 + 0.1950X$$
$$r = .967$$

Where:

y = maximum dose (mg Al/l)
x = total alkalinity (mg $CaCO_3$/l)

The slope of this line was compared to that based on stoichiometry and was found not to be significantly different (P [illegible] .05). The low intercept value, as yet unexplained, suggests the necessity for performing such tests prior to any lake treatment [illegible] low residual dissolved aluminum values and to avoid unnecessary [illegible] that procedure to be followed in the calculation of [illegible] the maximum doses to [illegible] depth in the water column [illegible] waters found in the lake, [illegible] alkalinity [illegible] the resulting linear relationship being used to determine [illegible]

PHOSPHORUS REMOVAL [illegible]

Phosphorus [illegible] was validated by treating lake water spiked with [illegible] concentrations ranging from [illegible] mg P/l to 1158 [illegible] .14 mg Al/l, the maximum dose for water with an alkalinity [illegible] following settling, final total phosphorus concentrations ranged from 15 to 52 mg P/l indicating that efficient removal could be achieved over the range of phosphorus concentrations commonly experienced in the lake during summer stratification. Phosphorus concentration and [illegible] with [illegible] formation.

A second [illegible] to determine the maximum phosphorus retention capacity [illegible] flow and [illegible] essential for retarding the [illegible] phosphorus [illegible] water interface. Four [illegible] The barrels [illegible] packed with glass wool to prevent [illegible] received a known quantity of [illegible] whose aluminum content ranged from 0.957 mg to 8.959 mg. The [illegible] contained only [illegible] wool as a control. [illegible]

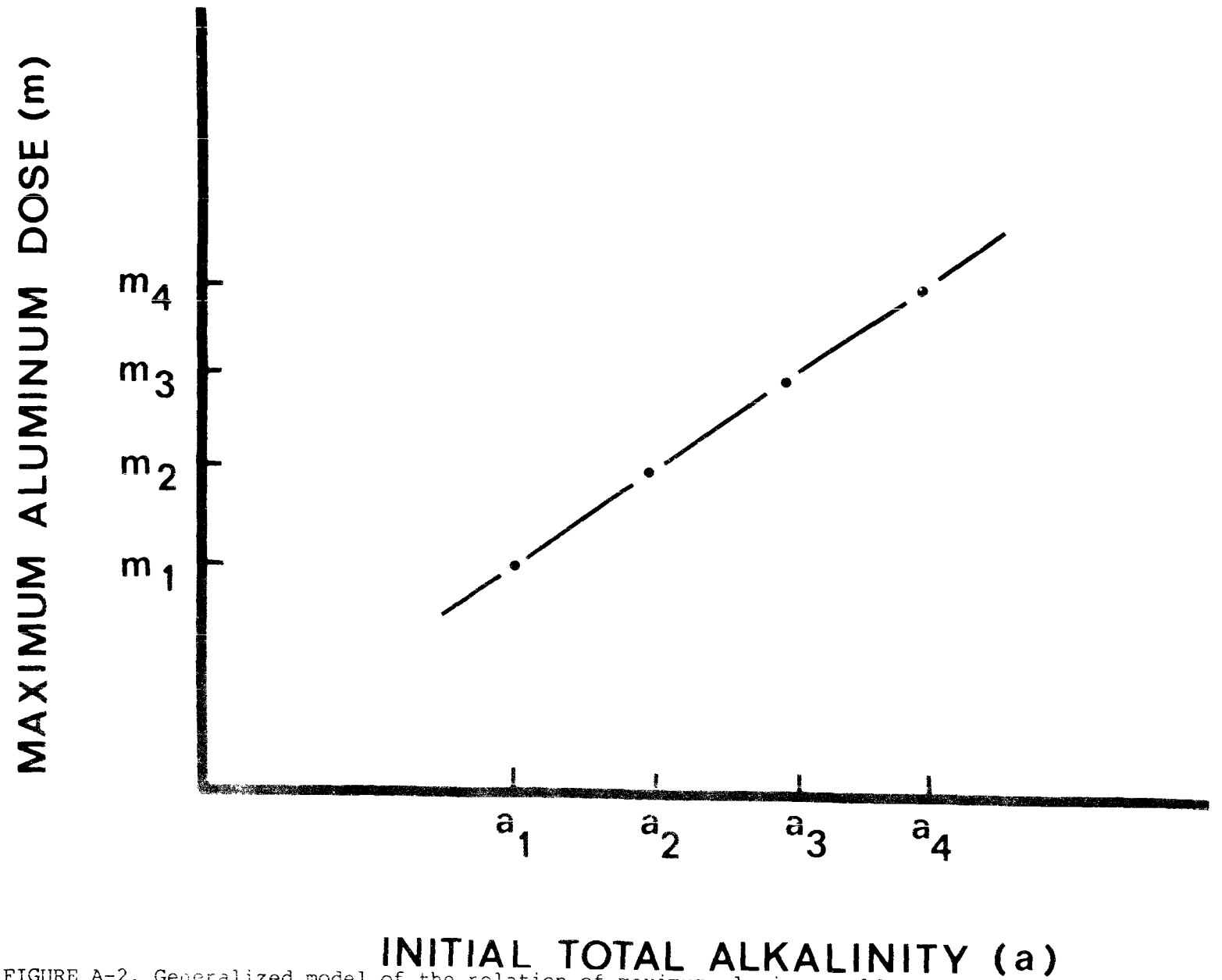


FIGURE A-2. Generalized model of the relation of maximum aluminum sulfate dose and lake water alkalinity

column was capped and fitted with inflow and outflow ports and eluted at a rate of 8.31 ml/h with a phosphorus solution containing 10 mg/l. The pH of the three experimental columns was maintained at pH 6, 7 and 8 to bracket the pH values which might be found at the water interface. Effluent was collected at 100 ml intervals for phosphorus determination. The amount of phosphorus lost to the columns was calculated by difference.

The maximum phosphorus removal capacity at pH 6, under the conditions of the test, was reached at Al/P molar ratios of 2.863, [illegible] and 3.311, respectively. There was no significant [illegible] from the columns.

This experiment provided further evidence to support the notion that dosage should be based upon water alkalinity and not upon the pH content of the lake. The objective of the treatment is to prevent release from lake sediments. To achieve this effectively, a lake with high pH requires the largest amount of aluminum in the sediments, consistent with upper limit based on RDA. The effectiveness of the aluminum is related to pH of the sediments. Eutrophic lakes with low alkalinity and with high initial water pH cannot be [illegible] than in the opposite [illegible]

IN SITU COLUMN EXPERIMENTS

In [illegible] ascertain [illegible] tions. [illegible] sediment [illegible] diameter [illegible] steel rod, [illegible] of 485.6 [illegible] interval [illegible] columns as [illegible] (in both experiments [illegible] long handled [illegible] volume of water [illegible] treatment employed [illegible]

Water [illegible]

[illegible]

[illegible]

[

[illegible] curred during the first [illegible] ranging from 68 µg P/l to 7 µg [illegible] Maximum reduction in total [illegible] hours after treatment, at [illegible] 33 µg P/l and 43 [illegible] concentrations at 5 and 7 [illegible] was a small reduction in [illegible] of continued sedimenta-

[illegible] enclosures by SCUBA [illegible] 8 cm floc layer [illegible] had no [illegible] between 7 M

[illegible] that the applica- [illegible] sediment phosphorus [illegible] concentration at [illegible] day 32 to 390 µg [illegible] phosphorus concentration [illegible] to 30 µg P/l [illegible] source of [illegible] can be [illegible] in [illegible] the ex- [illegible] of treatment, [illegible] effective in [illegible]

[illegible] following [illegible] recover- [illegible] aluminum levels, [illegible]

[illegible] of [illegible] and

[illegible] method [illegible] received [illegible] pH's. [illegible] µg Al/l, [illegible] West Twin [illegible] at each [illegible] from [illegible] in [illegible] treatment with

Complete [illegible] mortality [illegible] in aquaria maintained at pH 4.9 with doses of [illegible], [illegible] and [illegible] mg Al l (Table A-1), and appeared to be related to elevated dissolved aluminum concentrations (1.55, 1.80, and [illegible] mg Al l, respectively). High losses to waters maintained at pH's above [illegible] or waters maintained at pH [illegible] down to pH 4.9 and receiving no aluminum [illegible], [illegible] mortality. Based on the response of this species, [illegible] sulfate, which is, or definition results in [illegible] aluminum concentration of equal to or less than 0.05 mg Al l, [illegible] sensitive [illegible] to it.

[illegible] A [illegible] MEPER [illegible] MEPER

[illegible]

this case, two sampling dates. If the two have no species in common, both CC and PS will equal zero. The maximum value of CC (i.e., 100) occurs when both samples have identical species lists, and the maximum value of PS (i.e., 100) occurs if both samples have the same species in the same proportions (Pielou, 1975). Results of these analyses suggest that the application of aluminum sulfate, or its accompanying side-effects (high sulfate concentrations and low pH), does not seriously effect macrobenthic organisms or macrobenthic community structure. With the exception of the buffered-alum treatment enclosures, all enclosures experienced a similar decrease in numbers of organisms during the interval day one to day fourteen (Table A-2). All of the enclosures showed little change in total number of species (Table A-2) Comparisons of diversity (similarity) indices indicated a high affinity between sampling dates for all treatment groups (Table A-3).

Table A-2. BENTHIC MACROINVERTEBRATES OF GROUPS AFTER TREATMENT

Group	# of Organisms After One Day	# of Organisms After 2 Weeks	# of Species After One Day	# of Species After 2 Weeks
A (alum)	868	415	30	26
B (alum-buffer)	602	657	26	2[illegible]
C (sulfate)	1007	463	26	28
D (control)	727	424	25	24

Table A-3. DIVERSITY INDEX VALUES COMPARING SAMPLING DATES OF EACH GROUP

Group	CC	PS
A	55	69
B	63	56
C	68	74
D	58	73

From Wilbur 1974

DOLLAR LAKE TREATMENT

The results of these experiments with effectiveness and toxicity led to a pilot lake treatment. Dollar lake, a small (2.2 ha), dimictic, shallow (Z_{max} = 7.5 M) eutrophic lake north of East Twin was given a hypolimnetic treatment in July 1974. The complete results are described in Kennedy (1978). Two principal observations were made which formed the basis for the West Twin treatment. First, no toxicity to fish was seen. Secondly, the alum was very effective in removing phosphorus from the water column and in retarding release from the sediments.

An experiment was conducted to measure phosphorus release from treated and untreated sediments. Plastic wastebaskets, fitted with sampling ports to which a plastic bag could be attached, were inverted and placed in hypolimnetic muds before treatment. After treatment additional baskets were inverted over floc-covered sediments next to the control baskets. Samples were taken over a period of one week in 1974 and 1976 and the water analyzed for total phosphorus. Treated muds (Table A-4) released very small amounts of phosphorus, particularly in 1974, further substantiating the conclusions from the in situ columns that alum-treated anoxic sediments have low phosphorus release rates, and the conclusions that continued internal phosphorus loading in West Twin after treatment must come in part from a source other than anoxic hypolimnetic sediments.

Table A-4. TOTAL PHOSPHORUS CONCENTRATION IN WATERS FROM TREATED AND UNTREATED ANAEROBIC DOLLAR LAKE SEDIMENTS

Depth of Half-Barrels	1974 µg P/l	1976 µg P/l
6M untreated	701	1021
6M treated	130	224
4M untreated	315	200
4M treated	198	705
4M untreated	392	1519
4M treated	33	600

DOSAGE DETERMINATION FOR WEST TWIN LAKE TREATMENT

Preparatory to the treatment of West Twin Lake, a series of jar tests was conducted using water collected at 5, 7, 9 and 11 M to confirm the relationship between alkalinity and dosage. Treatment was to be confined to the hypolimnion (5 M and below).

As before, 500 ml subsamples of lake water from a given depth received increasing amounts of aluminum sulfate, were stirred for 2 minutes, and then allowed to settle for 24 hours. Each jar in each alkalinity series was then sampled and alkalinity and dissolved aluminum measurements made.

Total alkalinities in mg $CaCO_3$/l determined immediately prior to alum addition were:

Depth	Total Alkalinity
5 M	115.33 mg $CaCO_3$/l
7 M	137.33
9 M	143.93
11 M	149.00

With the exception of 7 M water, which received an aluminum dose ranging from 0.5 mg Al/l to 26 mg Al/l all depths received a range of doses intended to bracket the critical portion of the titration curve. Results of these jar-tests are given in Table A-5. In each case, dissolved aluminum concentrations increased exponentially as the critical concentration of 0.05 mg Al/l was approached and exceeded. Therefore, dissolved aluminum concentrations determined at each dose level in each alkalinity series were plotted versus dosage and fitted with an exponential curve in order to determine the maximum aluminum dose at each alkalinity:

$$y = ae^{bx}$$

where:

y = dissolved aluminum concentration (mg Al/l)

x = aluminum dose (mg Al/l)

Values for b and a determined for each data set were:

Depth	a	b	r
5 M	8.518×10^{-9}	0.7875	0.98
7 M	9.40×10^{-11}	0.8294	0.99
9 M	7.121×10^{-15}	1.1938	0.96
11 M	3.30×10^{-8}	0.8401	0.98

Table A-5. JAR TEST RESULTS-DISSOLVED ALUMINUM CONCENTRATION (mg Al/l)

Dose (mg Al/l)	5 meters	7 meters	9 meters	11 meters
17	0.007	0.0005	-	-
18	[illegible]	[illegible]	-	-
19	[illegible]	[illegible].0008	-	-
20	[illegible]	[illegible]	-	-
21	[illegible]	[illegible]	-	-
22	[illegible]	[illegible]	[illegible]	-
23	[illegible]	[illegible]	0.0040	[illegible].0059
24	[illegible]	[illegible]	0.0317	0.0286
25	[illegible]	[illegible]	0.0850	0.0505
26	[illegible]	[illegible]	0.1540	0.1055
27	[illegible]	[illegible]	-	0.2050

A dissolved aluminum concentration of 0.05 mg Al/l was substituted for y allowing the calculation of x, the maximum dose of aluminum (in mg Al/l) required at each alkalinity.

Depth	Total Alkalinity	Maximum Dose
5 M	[illegible] (mg $CaCO_3$/l)	19.791 (mg Al/l)
7 M	[illegible]	24.221
9 M	143.93	24.778
11 M	149.00	25.167

The apparent relationship between alkalinity and maximum dosage was determined by subjecting these data to linear regression analysis.

$$y = 0.849066 + 0.165972x \quad (r=0.98[illegible])$$

where:

y = aluminum dose (mg Al/l)

x = initial total alkalinity (mg $CaCO_3$/l)

This relationship was then used to determine the maximum dose for all alkalinities within the range 115.33 to 149.0 mg $CaCO_3$/l.

The next step was to determine the dosage to be applied to each one meter thick strata. The alkalinity of each one meter depth from 5M to 11M was determined in triplicate, with the mean alkalinity of each one meter interval (e.g., 5 M to 6 M, 6 M to 7 M, etc.) obtained by averaging. Based on the above relationship and these average alkalinities, the (volumetric) maximum dose for each depth interval was calculated. Based on the formula weight of 594.19 (assuming $Al_2(SO_4)_3 \cdot 14H_2O$) these maximum doses were converted to pounds of dry alum per cubic meter of lake water by multiplying by 0.02428, the conversion factor to change mg Al/l to pounds alum/M^3, as follows:

Depth Interval	Average Alkalinity	Max. Dose	lb Alum (dry)/M^3
5-6 M	121.35 mg $CaCo_3$/l	20.907 mg Al/l	0.5076
6-7 M	132.35	22.934	0.5568
7-bottom	142.91	24.880	0.6041

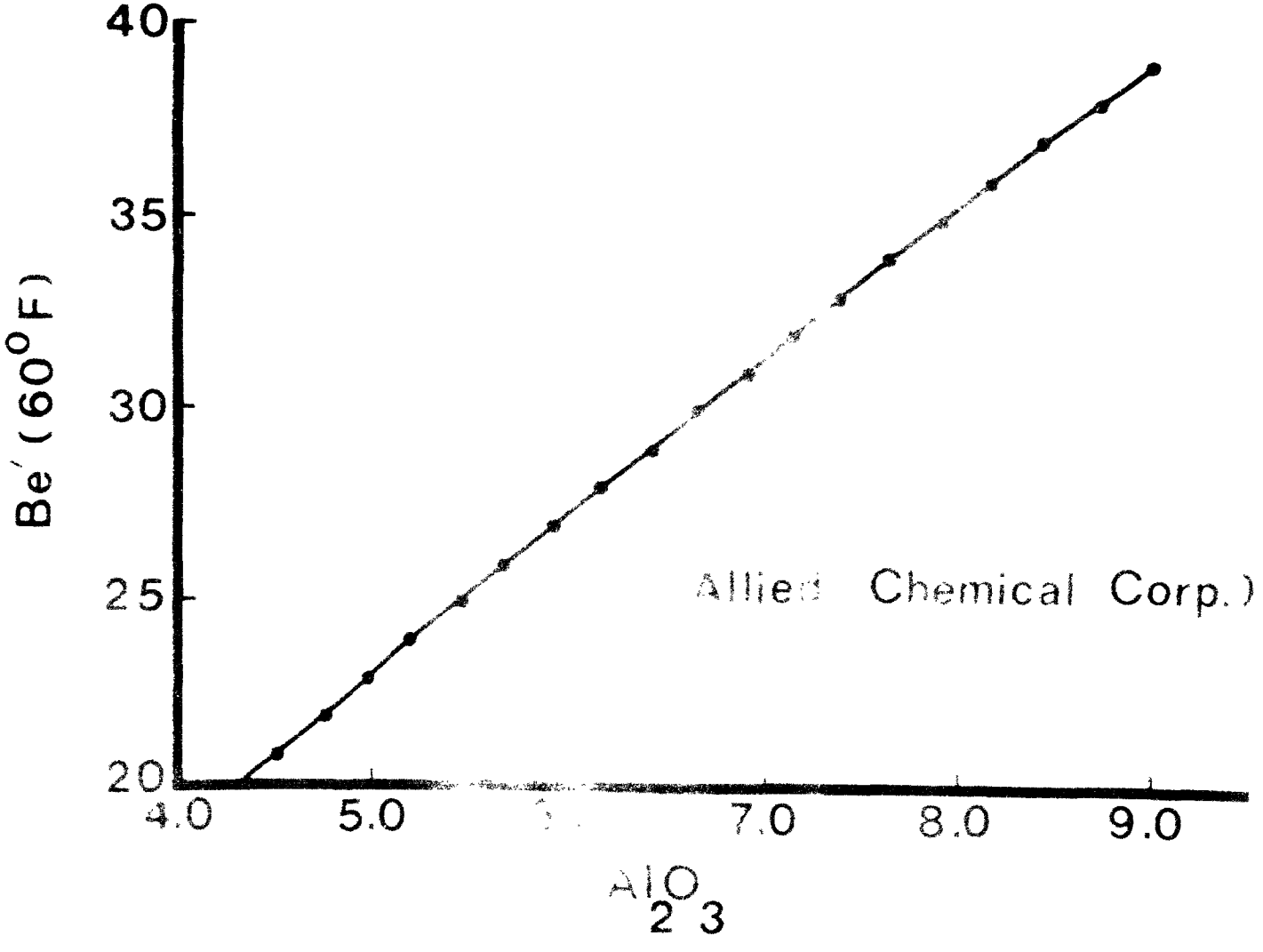
A dose expressed in terms of pounds of dry alum per cubic meter was sufficient to determine the total amount to be added to the lake if dry alum was to be used. However if liquid alum was to be used, as was the case for the treatment of West Twin Lake, further calculations were necessary to express dose in terms of gallons per cubic meter.

The standard strength of commercial alum liquors ranges from 8.0 to 8.5% Al_2O_3, which is equivalent to 5.16 to 5.57 pounds dry alum per gallon (at 60°F). Since liquid alum is shipped at temperatures considerably higher than 60°F, correction must be made to allow for the resulting lower density. For example, one gallon of 8.5% Al_2O_3 liquor will contain 5.57 pounds at 103 F. Alum suppliers provide results of Al_2O_3 assays with each shipment. It is important to note that such assays are reported as percent Al_2O_3 at a standard temperature of 60° F. The supplier should also be able to estimate the temperature of the liquor at the proposed time of application by taking transport and storage time into account. It is advisable that the liquor temperature be re-checked prior to treatment in case such an estimate is grossly in error. Knowledge of both temperature and Al_2O_3 is required in order to determine dosage in terms of gallons per cubic meter.

The following procedure was used to determine dosage in terms of gallons per cubic meter.

1. Percent Al_2O_3 (at 60° F) was converted to density, expressed as degree Baume (at 60° F) using Figure A-3. The liquor purchased for the West Twin Lake treatment assayed at 8.3% Al_2O_3, or 36.5° Be (60° F).

2. A temperature correction was applied against this Baume to account for the decrease in density using Figure A-4. The estimated temperature of the liquor at time of application was 100-105° F (a temperature of 103° F


40
35
30
25
20
Be′ (60°F)
4.0
5.0
7.0
8.0
9.0
$AlO_{2\ 3}$
Allied Chemical Corp.)

TEMPERATURE CORRECTIONS

[illegible] quors)

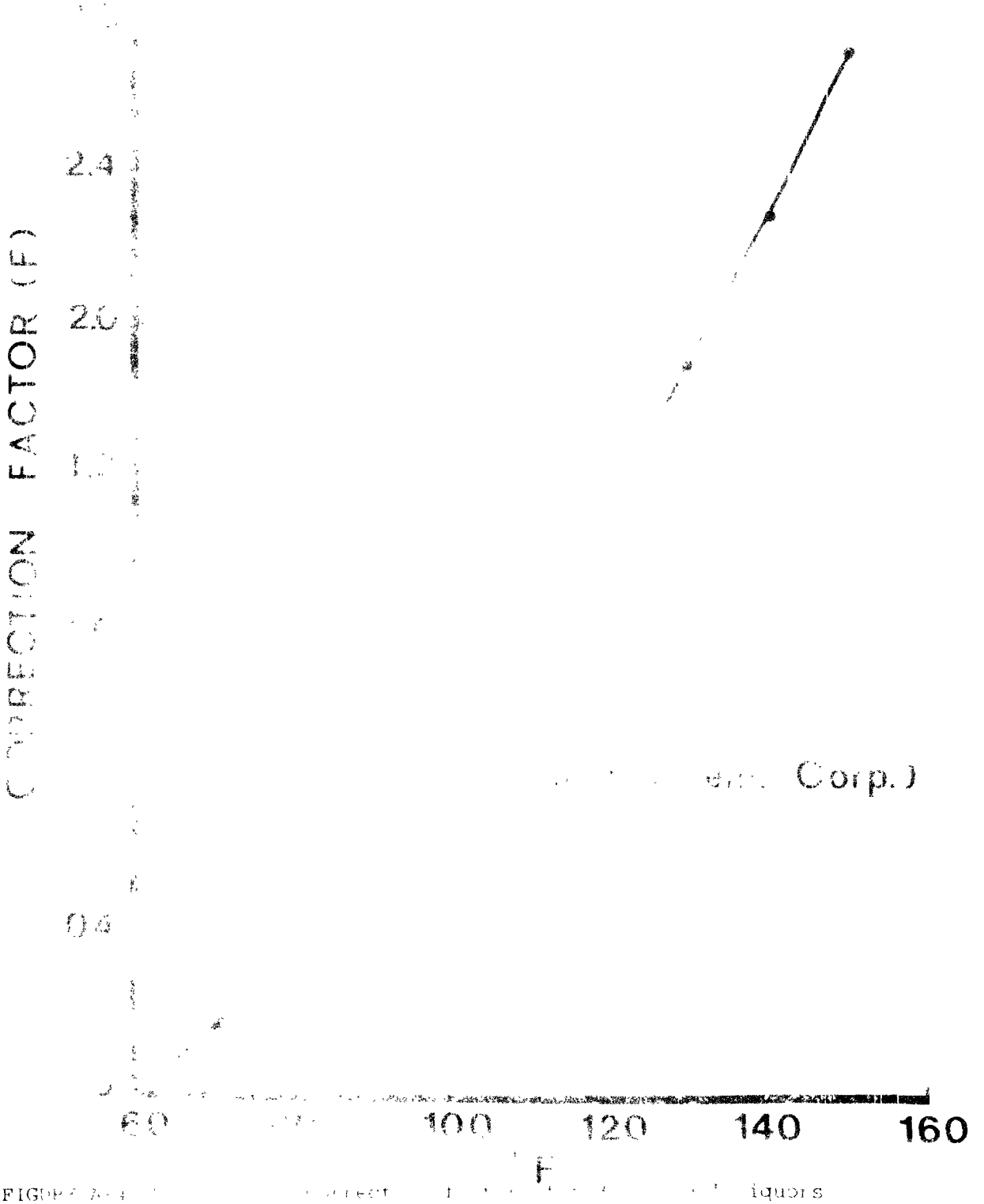


FIGURE [illegible] Correction [illegible] liquors

was used). The corresponding correction factor (for 32° - 36° Be liquors) (Anonymous, 1970) was 1.003. The [illegible] Be was determined by subtract- [illegible]

[illegible]

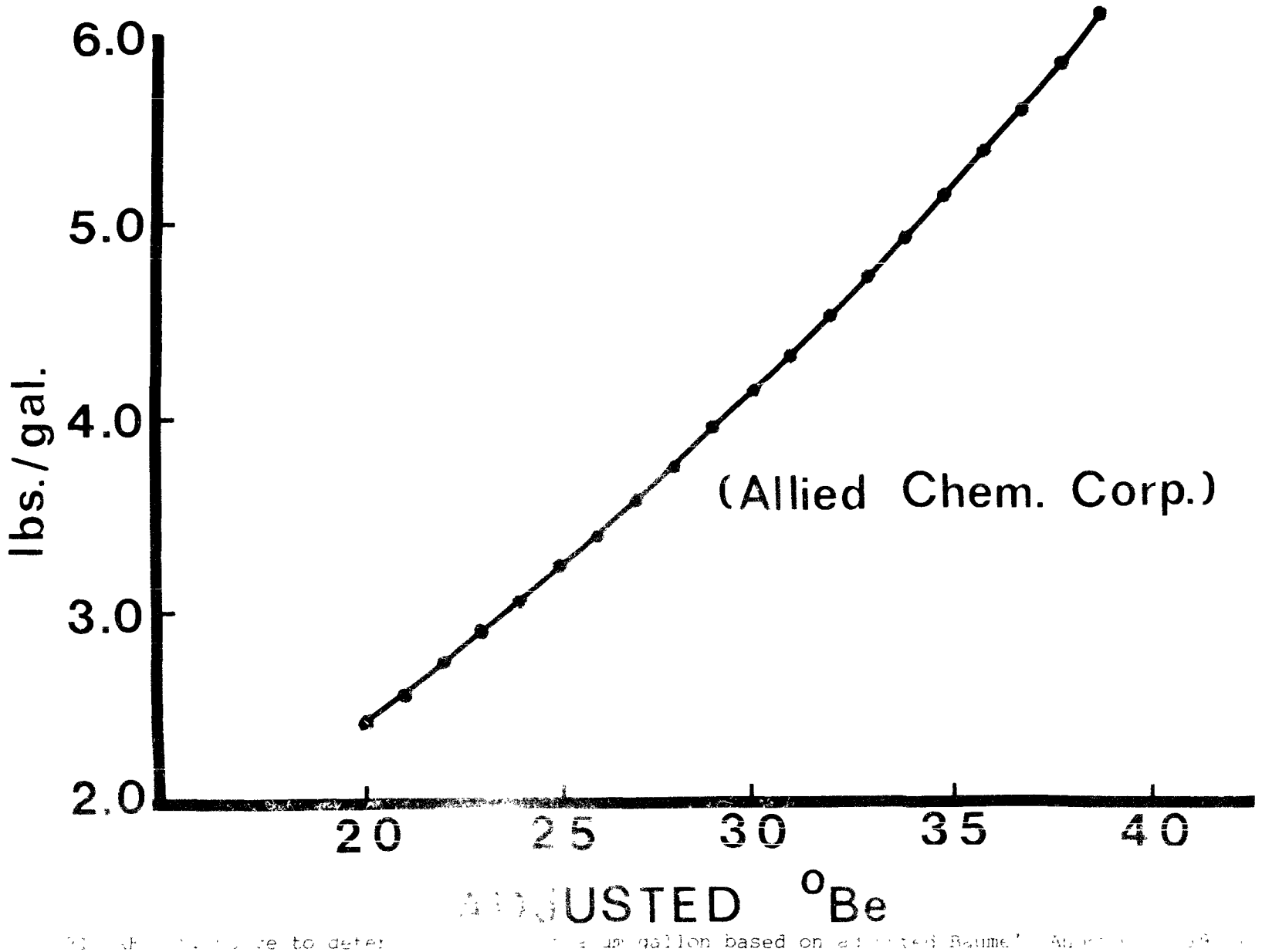
[illegible] on the basis of the adjusted [illegible] temperature thus contained [illegible]

[illegible] follows:

[illegible]

[illegible]

[illegible]



[illegible] to deter[illegible] gallon based on [illegible] Baume' [illegible]

Treatment quadrats were established by locating east-west transects at 50 M intervals along the original north-south transect. Coded buoys marked all intersections of east-west transects with either a north-south transect or the perimeter of the treatment zone. Specific quadrats, identifiable by a four number code, could be easily located by locating the appropriate four corner buoys. The correct location of each of the corner buoys was confirmed by triangulation from shore points.

Depth soundings were taken at each buoy. These readings were used to determine the volume of water below a depth of 5 M in each quadrat. Volumes of depth intervals uninterrupted by lake bottom were calculated by multiplying the area of the quadrat by one meter. Volumes of depth intervals in which the bottom was reached before the next even depth or passed through any part of the depth interval were calculated individually, calculating the volume of the [illegible] Volumes below a depth of 7 meters were summed [illegible]

[illegible] treatment [illegible] was calculated [illegible] volume [illegible] depth interval by the appropriate unit dose and then summing the doses [illegible] depth intervals. Quadrat doses are listed in Table A [illegible]. The total dose [illegible] the sum of all [illegible] doses [illegible] was [illegible] independent [illegible] density, the alkalinity [illegible] each depth interval [illegible] morphology are used instead, [illegible] the calculations by quadrat [illegible] 47 [illegible] %. This difference [illegible] shape of the lake partial meter [illegible] quadrats. This problem leads some [illegible]

[illegible] CALCULATION

[illegible] treatment of West [illegible] Lake [illegible] treatment [illegible] very system, and application vehicles.

1. Storage facility. Although the decision to use liquid alum simplified treatment procedures by not requiring equipment for mixing, it did require [illegible] liquid storage be provided. An [illegible] foot diameter, 4 foot deep [illegible] with a maximum capacity of 7,600 gallons was used. The capacity of the pool was more than sufficient to hold one truck load [illegible] approximately 4,000 gallons [illegible] at the [illegible]

[illegible] paved [illegible] approximately [illegible] of the lake. A flat area was covered with [illegible] layer of sand to level the site. The site was approximately 10-[illegible] the lake [illegible] sloping to the lake.

Table A-6. GALLONS OF ALUMINUM SULFATE APPLIED TO WEST TWIN LAKE

Quadrat	Gallonage	Quadrat	Gallonage
1	38	21	2986
2	234	22	2751
3	1436	23	1105
4	3077	24	791
5	1866	25	927
6	500	26	848
7	91	27	996
8	11	28	554
9	176	29	60
10	547	30	61
11	627	31	380
12	910	32	1220
13	2347	33	1070
14	2541	34	450
15	2282	35	274
16	855	36	430
17	595	37	843
18	208	38	786
19	181	39	211
20	1654		

(2) Delivery system - The delivery system consisted of three components: (1) a 12 hp, 2 1/2 inch pump, (2) a system of pipe leading from the pool-side pump out into the lake, and (3) a floating platform, which served as a loading point for application vehicles.

The pump was located at pool side with the intake port attached to a siphon laid over the rim of the pool and the output port attached to the delivery pipe. Although alum solutions are caustic there appeared to be no damage to the impeller, casing, or fittings after use. All of these parts were brass.

The piping system consisted of coupled 20 foot sections of 2 3/8" I.D. (1/4" wall) PVC rigid wall pipe floated under anchored 55 gallon drums spaced 10 M apart. Two and one half [illegible] was used for the segment from the pump to the lake shore to avoid problems presented by uneven terrain. The pipe terminated at a butterfly valve lowered on a 8' by 8' platform situated in the lake and anchored on all four [illegible]. A 20 [illegible] section of flexible tubing (of the same diameter), connected to the [illegible] side of the valve, served for loading tanks on application [illegible]

(3) Application vehicles - Two barges were used for application of the alum. The barge design developed for the Pond [illegible] Lake treatment (Rennedy, 1978) was adopted with modifications. A [illegible] constructed by welding five capped 55 gal drums end-to-end, [illegible] and lashed with 3 8" steel cable to a steel frame constructed [illegible] 4 inch channel iron. Five rows (25 drums) of pontoons were used for [illegible] and six rows (30 drums) for the second. The long axes of the pontoons were [illegible] bow to stern. Two 250 gallon fuel storage tanks, the bottom ends of which were joined by 2 1/4" galvanized pipe, were strapped [illegible] steel frame, allowing a total payload of 500 gallons.

The buoyancy of one drum [illegible] its own weight [illegible] is approximately 400 lb. The total weight of the [illegible] 500 gallons of alum at approximately 11.5 lb/gallon, [illegible], and miscellaneous on-board equipment was approximately 8 [illegible] buoyancy provided by five and six rows of pontoons was 10,000 [illegible] respectively.

[illegible] constructed from PVC pipe (2 3/8" [illegible] 1/4" wall), with the [illegible] of 7 M below each barge by attaching [illegible] steel frame [illegible] of two A-frames, the legs of each being at [illegible] the bow and the stern. The apexes [illegible] below the [illegible] equal in length to the width [illegible]

The manifold was attached [illegible], capped at the ends, and drilled with 1/4" holes, [illegible]. A 1 1/2 hp gasoline pump, located [illegible] of each barge, was [illegible] with a tee coupling at the intake, allowing [illegible] water from just below the surface to be drawn simultaneously [illegible] flow from each source, adjusted using two ball valves [illegible] maintained at 50/50 during application [illegible] the manifold. The only other [illegible] treated by the forward movement [illegible] manifold. A [illegible], gauged in gallons, allowed [illegible] in the tanks.

The barges were propelled by [illegible] hp fitted with work prop[illegible] which provided adequate [illegible] and throttle controls were located [illegible] of the tubs [illegible]

ALUM APPLICATION PROCEDURES

Treatment began on July 29 and required three full days to complete. A total of 12,693 gallons was added on day one, 12,555 gallons on day two, and 11,671 gallons on the third day. Operations started at approximately 0700 hr and continued till dark (2200 hr), giving an application rate (per barge) of between 0.26 hr/100 gallons and 0.24 hr/100 gallons. This rate includes time to load barge tanks and re-position equipment.

The platform which received alum from the shore and served as a "filling station" for the barges was positioned each day near the quadrats to be treated. Each platform location allowed the barges to approach from opposite sides of the platform. To be loaded, barges were moored, bow in. When the filling hose was in place and the line valve opened, instructions to begin pumping were radioed to the pump operator at pool side. Pumping was terminated when the tanks were nearly full and were allowed to completely fill by gravity flow. When it had been confirmed by radio that the pump was off, the valve was slowly closed.

Alternately colored floats spaced seven meters apart (the length of the manifolds) on a 50 meter line served as navigational aids. A line strung between the north and south buoy at the eastern end of the quadrat and another line, similarly marked with floats, was strung along the western end. The first float on each line was 3.5 meters from the end, with the remaining six buoys being spaced 7 meters apart. The barge moved back and forth across the quadrat by lining-up on like-colored floats, thus insuring even coverage of the area. Barge speeds were kept below 4 mph. When a quadrat was completed, the navigational lines and markers were moved by persons in smaller boats, while the barges returned for refilling of the tanks.

Surface water and alum, mixed 50/50, were pumped down to the manifold by an on-board 1 1/2 hp gasoline pump. At the end of each run, the pumps were allowed to pump water only while returning to the platform to flush out the steel pump casing.

Equipment dismantling and removal required approximately one week.

COST OF WEST TWIN ALUM APPLICATION

The cost of the West Twin treatment is itemized in Table A-7. Labor costs are not given, but the number of hours spent on construction and application are listed. The costs for equipment and alum are likely to increase over the years, but the labor hours could be greatly reduced by the employment of experienced personnel or by the development of a faster means of application.

Table A-7. COST FOR THE ALUMINUM SULFATE TREATMENT OF WEST TWIN LAKE

A.	Itemized Costs	
	1. Equipment (re-usable)	$ 4,058
	2. Supplies	646
	3. Rentals	262
	4. Aluminum sulfate	6,803
	TOTAL	$11,779
B.	Labor Hours	
	1. Total hours, construction and application	2,000
	2. Application	590
	(divided among 11 people over [illegible] days)	

TECHNICAL REPORT DATA

Please read Instructions on the reverse before completing

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4. TITLE AND SUBTITLE Effect of Diversion and Alum Application on Two	5. REPORT DATE March 1978
	6. PERFORMING ORGANIZATION CODE
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7. AUTHOR(S) [illegible] Robert T. Heath, Robert H. Kennedy[1], [illegible] McCombas	
9. PERFORMING ORGANIZATION NAME AND ADDRESS [illegible] of Biological Sciences [illegible] Geology [illegible] University [illegible]	10. PROGRAM ELEMENT NO. 1BA031
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15. SUPPLEMENTARY NOTES

16. ABSTRACT

[illegible] questions were asked: (1) Will septic tank diversion reduce symptoms of eutrophication in the Twin Lakes, two similar dimictic lakes, and (2) Will a maximum hypolimnetic dose (residual dissolved aluminum ≤ 0.05 mg/l) of aluminum sulfate (alum) reduce internal phosphorus (P) release and further improve them?

[illegible] diversion, P in lakes and streams declined and plankton biomass declined, but [illegible] of monitoring after diversion, P and biomass remained those of moderately [illegible], and P income remained above "permissable". Non-point sources were [illegible]

[illegible] laboratory and field tests, maximum dose, which is related to alkalinity and [illegible] tests", was effective in retarding P release from sediments. A [illegible] of Dollar Lake in 1974 was done as a test of the technique. 139.55 M^3 [illegible] to the hypolimnion of West Twin (WTL) Lake in July, 1975. Compared [illegible] Lake, hypolimnetic P was lowered in 1976 but net internal P loading, while [illegible] continued, suggesting littoral sources. Epilimnetic P also declined in WTL, [illegible] the eutrophic range [illegible] indicated by a trophic state index, but remains moderately eutrophic. [illegible] fish were observed.

[illegible] for dose determination and application are given.

17. KEY WORDS AND DOCUMENT ANALYSIS

a. DESCRIPTORS	b. IDENTIFIERS/OPEN ENDED TERMS	c. COSATI Field/Group
[illegible] [illegible]rophytes [illegible]	eutrophication lake restoration	08H

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