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Waterways Experiment
Station

Water Quality Analysis of Alamo Lake: Feasibility of In-Lake Management for Water Quality Enhancement

by *Steven L. Ashby, John L. Myers, WES*
Robert J. Stuart, Los Angeles District

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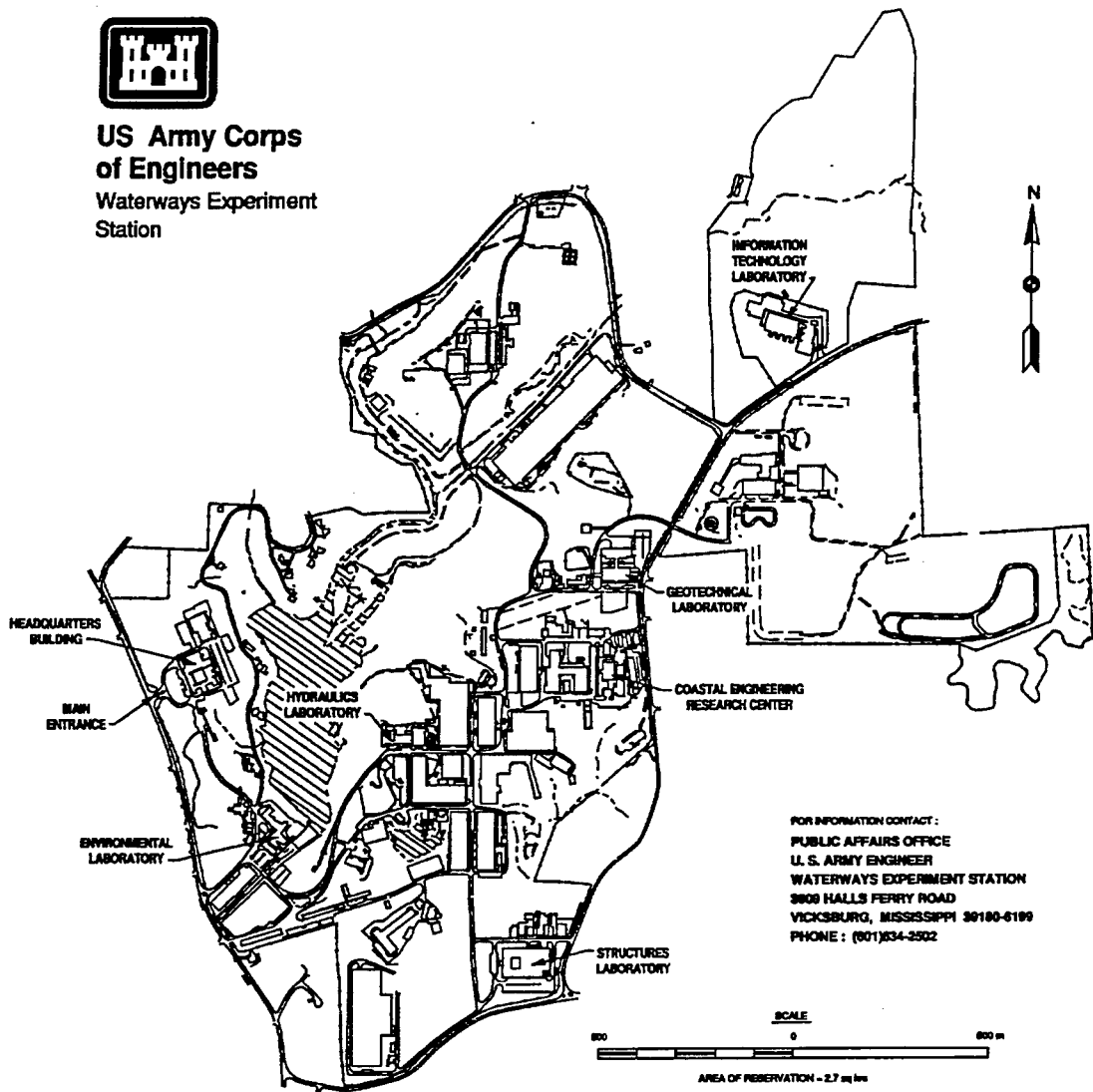
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Preface

A water quality assessment was conducted for Alamo Lake, Arizona, to delineate adverse water quality impacts associated with hydrogen sulfide production and to identify feasible in-lake management techniques. This study was a cooperative effort conducted by the U.S. Army Engineer District, Los Angeles, and the U.S. Army Engineer Waterways Experiment Station (WES). Additional support was provided by the Water Operations Technical Support Program. Dr. John W. Barko is the current Program Manager.

This report was prepared by Messrs. Steven L. Ashby and John L. Myers of the Environmental Laboratory (EL), WES, and Mr. Robert J. Stuart, Los Angeles District. The work was conducted under the direct supervision of Dr. Richard E. Price, Chief, Ecosystem Processes and Effects Branch, and under the general supervision of Mr. Donald L. Robey, Chief, Environmental Processes and Effects Division, and Dr. John W. Keeley, Director, EL.

Dr. Robert W. Whalin was Director of WES at the time of publication of this report. COL Bruce K. Howard, EN, was Commander.

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Conversion Factors, Non-SI to SI Units of Measurement

Non-SI units of measurement used in this report can be converted to SI units as follows:

Multiply	By	To Obtain
acres	4,047.9	square meters
acre-feet	1,233.5	cubic meters
cubic feet	0.02832	cubic meters
feet	0.3048	meters
miles (U.S. statute)	1.609347	kilometers
square miles	2.590	square kilometers

1 Introduction

Alamo Lake is a U.S. Army Engineer impoundment on the Bill Williams River in west-central Arizona (Figure 1). The lake provides flood protection for downstream communities while serving as a site for public access and water-based recreation activities. The lake can be utilized as a water conservation pool, although there has never been a local contractor for such a purpose (Corps of Engineers 1990). Adverse water quality problems, however, have resulted in recurring maintenance at the dam, increasing project operation costs.

Water quality problems at Alamo Lake are manifested in the degassing of hydrogen sulfide (H_2S) in reservoir releases. Alamo Lake water is relatively high in concentrations of sulfate and organic matter and low in concentrations of nitrate, iron, and manganese. This condition favors hydrogen sulfide production during anoxia (see Wetzel 1983, e.g., for a thorough discussion of the sulfur cycle and Gunnison and Brannon 1981; e.g., for a discussion of anaerobic processes in reservoirs). Hypolimnetic anoxia during the summer facilitates the start of the chemical and biological processes which form the hydrogen sulfide. The low-flow releases (approximately $10 \text{ ft}^3 \text{ sec}^{-1}$)¹ of anoxic hypolimnetic water during the summer season are conducive to degassing of hydrogen sulfide within the dam and throughout the outlet works. The hydrogen sulfide gas, which is frequently noticeable outside the dam, is potentially hazardous to project personnel when they are required to be in the lower portions of the concrete shaft beneath the control house. The gas causes corrosion to electrical connections in the concrete shaft structure, including the control house. Contact with concrete parts of the outlet works also causes etching to occur. While the production of hydrogen sulfide negatively impacts operations at Alamo Lake, it is the seasonal loss of dissolved oxygen in the waters which results in the chemical and biological processes causing the formation of hydrogen sulfide.

Study objectives were to evaluate the establishment of anoxic conditions in the hypolimnion and determine the feasibility of an in-lake enhancement technique such as aeration or oxygenation. An assessment of water quality of

¹ A table of factors for converting non-SI units of measurement to SI units is presented on page v.

Alamo Lake was conducted to describe the oxygen depletion and associated chemical changes during stratification. In addition to routine monitoring of physicochemical parameters, surveys of oxygen depletion rates and sediment quality were conducted. A hydrologic and physical assessment was included in the study since the relatively long residence time of hypolimnetic water during summer low-flow is likely a contributing factor. Physical assessments such as thermal patterns and lake stability were also conducted to provide additional information for evaluating in-lake enhancement techniques to improve water quality.

2 Site Description

Alamo Lake is located in the Bill Williams River Basin, a mountainous area in west-central Arizona and part of the Colorado River Basin (Figure 1). The lake is located at the confluence of the Big Sandy and Santa Maria Rivers, about 39 miles upstream from the confluence of the Bill Williams River with the Colorado River in Lake Havasu. The drainage area of the reservoir comprises approximately 4,770 mi², and normal water surface elevation is 1,100 ft National Geodetic Vertical Datum (NGVD) (Corps of Engineers 1990).

Since its construction in 1968, the reservoir has provided flood control for the lower Colorado River communities downstream from Lake Havasu by regulating flood flows on the Bill Williams River. Recently, the popularity of the lake has increased for water sports and recreation, especially fishing. Alamo Lake currently meets the needs for a number of water related sports activities for the western Arizona region. At the reservoir, the Arizona State Parks and Arizona Game and Fish Departments support an important warm-water fishery and other activities such as boating, hiking, swimming, and camping (Corps of Engineers 1990).

Alamo Dam is a rolled earth filled structure with a crest length of 975 ft, a crest width of 30 ft, and a maximum height of 283 ft above the original streambed. A detached, broad-crested spillway with crest elevation of 1,235 ft is located on the right abutment. The outlet works of Alamo Dam consist of a concrete-lined tunnel through the left abutment of the dam, an approach channel, a concrete intake structure at the upstream tunnel portal, a concrete shaft and gate chamber just upstream from the axis of the dam, and an outlet channel discharging directly into the streambed. The tunnel has an inside diameter of 12 ft and a length of 1,290 ft. The outlet channel is an unlined trapezoidal channel 18 ft wide and approximately 230 ft long. The gate section contains three passages, each with a pair of 5.5- by 8.5-foot slide gates installed in tandem, and an 18-inch conduit for low-flow discharges (Corps of Engineers 1964). Pertinent hydrologic features of the project are presented in Table 1.

3 Methods

Water quality sampling was conducted by the U.S. Fish and Wildlife Service at three stations in Alamo Lake and one station in the tailwater region: station ALP1 located near the dam, ALP2 in the mid-lake region, ALP3 in the upstream region of the lake, and ALP4 in the outflow (Figure 1). Physical parameters of temperature, dissolved oxygen, specific conductance, pH, Secchi depth, oxidation reduction potential, and turbidity were measured approximately once a month before, during, and after thermal stratification with a multi-parameter water quality meter (Hydrolab Surveyor, Hydrolab Corp.[®]). Data were collected at five meter (approximately 16.4 ft) intervals from the surface to near the bottom of the impoundment. Discrete samples were collected on a similar schedule for water quality analyses of nutrient species, iron, manganese, sulfate, calcium, solids, total organic carbon, sulfide, and chlorophyll *a*. Water chemistry analyses were conducted using standard methods (American Public Health Association 1992 and U.S. Environmental Protection Agency 1993). Sulfide analysis was conducted using a HACH[®] HS-C Test Kit. Although more intensive data collection was conducted in 1994, data collected from 1991 through 1994 were evaluated.

Sediment samples were collected at the three stations using a ponar dredge and a sieving bucket for the removal of excess water. Ten replicate samples were collected at each station for analysis of iron, calcium, manganese, sulfate, total organic carbon, alkalinity, total phosphorus, ortho-phosphorus, Kjeldahl nitrogen, ammonia nitrogen, sulfide, and nitrate. Bulk sediment samples were analyzed using standard procedures (US Environmental Protection Agency 1993, American Public Health Association 1992, and American Society of Agronomy 1982). Data reported were for concentrations of parameters measured after sediment digestion.

Oxygen depletion rates were evaluated with two methods of estimating oxygen demand. One method of estimating oxygen depletion rates utilized vertical dissolved oxygen measurements at station ALP1 taken periodically during 1992, 1993, 1994 coincident with the development of thermal stratification until the onset of anoxia. Data from these measurements were used as input to the computer program PROFILE (Walker, 1987) for the calculation of oxygen depletion rates. Calculations were limited to a period from mid-April to mid-June, coincident with available data prior to anoxia. This method calculates depletion rates prior to anoxia but does not account for any

additional oxygen deficit that may result with the formation of reduced metals. The second method employed incubation of water collected from mid- and near-bottom depths at stations ALP1 and ALP2. Samples were collected in 300 mL bottles fitted with ground glass stoppers and incubated at 20 °C for 10 days. Dissolved oxygen measurements were conducted on triplicate samples at day 0, 1, 3, 5, 7, and 9 for calculating oxygen depletion rates. Measurements were conducted in mid-July and mid-October in 1994 and samples were collected from anoxic water except for mid-depth at ALP2 in October. Information available from this effort was limited since samples were not adequately aerated at the time of collection resulting in low dissolved oxygen concentrations at the start of the measurements. However, this method provides an estimate of the oxygen deficit due to demand exerted by reduced metals upon reaeration when conducted appropriately.

4 Results and Discussion

Hydrology

Hydrologic records of mean daily inflow, outflow, and pool elevation were provided by the U.S. Army Engineer District, Los Angeles for describing the hydrology of Alamo Lake during the study period. Although several high flow events occur typically in winter and spring (flow greater than $100 \text{ ft}^3 \text{ sec}^{-1}$), flows during stratification are mostly less than $100 \text{ ft}^3 \text{ sec}^{-1}$ and often reduced to a nominal flow less than $10 \text{ ft}^3 \text{ sec}^{-1}$ (Figure 2). Inflows were greatest in 1993 (maximum flows above $9000 \text{ ft}^3 \text{ sec}^{-1}$) while flows above $100 \text{ ft}^3 \text{ sec}^{-1}$ occurred only once in 1994. In all years, a flow above $100 \text{ ft}^3 \text{ sec}^{-1}$ occurred at least once during late stratification. Discharge was mostly less than $40 \text{ ft}^3 \text{ sec}^{-1}$ except during high flows (Figure 2). Of interest is the effect of operations on the residence time of lake water, particularly during stratification. Daily retention times (calculated by dividing daily volume, based on elevation and the area-capacity curve, by mean daily outflow) varied from years during low flow periods which often occurred during the summer season to days during periods of high flow for each year of the study (Figure 3). Most importantly, increased retention during stratification results in progressively deteriorating water quality in the hypolimnion due to limited exchange with epilimnetic water. The relatively long hydraulic residence time during periods of low release should, therefore, be a major consideration in the evaluation of enhancement techniques for Alamo Lake.

Temperature and Dissolved Oxygen

Alamo Lake exhibits the temperature regime of a strongly stratified lake with considerable temperature differences between surface and bottom waters during stratification (Figure 4). Surface temperatures typically increase from April until late August, and bottom temperatures increase well into October. For example, in April 1992, surface temperatures were around $24 \text{ }^\circ\text{C}$, with the bottom temperatures slightly below $14 \text{ }^\circ\text{C}$. A distinct thermocline developed at a depth of 23 to 26 ft and surface temperatures increased to $26 \text{ }^\circ\text{C}$ with bottom temperatures increasing to $15\text{-}16 \text{ }^\circ\text{C}$ by mid-June. Thermal gradients became more established as surface temperatures increased to greater

than 30 °C until late August. Surface cooling to 25 °C with bottom temperatures increasing to 17 °C in October indicated the onset of autumnal destratification. A similar pattern was observed in 1993, even though flood control operations resulted in a relatively constant decrease in surface elevation from 1138 ft NGVD to 1105 ft NGVD. Similarly, in 1994, although surface temperatures were mostly less than 28 °C, bottom temperatures ranging from 12 to 14 °C resulted in a temperature differential between surface and bottom waters of 10 to 15 °C. This temperature differential was observed in all of the study years and may be attributed not only to seasonal surface warming but release of cooler, hypolimnetic water via bottom withdrawal at the project.

In all years, stable conditions of thermal stratification were apparent. Davis (1980) defines thermal stability as the minimum theoretical energy needed to mix a body of water from an initially stratified condition to an isothermal state and provides the following equations for estimating stability:

$$\text{Stability (J)} = \text{PEM} - \text{PES}$$

where

$$\text{Potential Energy of the Mixed System (PEM)} = g \sum_{i=1}^n p_{im} V_i h_i$$

$$\text{Potential Energy of the Stratified System (PES)} = g \sum_{i=1}^n p_{is} V_i h_i$$

where

p_{im} and p_{is} = the mean water densities of each layer for the mixed and stratified cases, respectively, (kg m⁻³)

V_i = volume of each layer, typically 1 meter deep (m³)

h_i = the height to the centroid of each layer above the bed of the reservoir (m)

g = the acceleration due to gravity (m s⁻²)

n = the number of layers

Using this definition, thermal stability was calculated for Alamo Lake to determine the lake's resistance to mixing events. A spreadsheet was employed to calculate the stability of a body of water on a particular day (Meyer 1991). The spreadsheet uses temperature profiles sorted by depth to calculate density.

Also, the volume of the lake as it varies with depth must be determined. Temperature profiles and lake volume determinations from area-capacity curves (Corps of Engineers 1964) were imported into the spreadsheet and stability was calculated for each sample date.

Maximum stability occurred during the summer months of the year with greatest values in 1993 due to the increased volume of warm, surface water during a high-flow year (Figure 5). Maximum stability at the height of thermal stratification suggests energy requirements to disrupt density gradients are quite high (e.g., greater than 2^{10} joules). It is not until late October or November of each year that stability decreases sufficiently, due to cooling of surface waters, to allow mixing to occur.

Patterns in dissolved oxygen in Alamo Lake are clearly influenced by the thermal regime and resultant isolation of bottom waters during periods of stability and by hydraulic retention time (Figure 6). At the onset of thermal stratification, dissolved oxygen concentrations in the bottom waters ranged from 2 to 4 mg L⁻¹, indicating considerable oxygen consumption during late winter and early spring. Anoxic conditions were well-established by April or May and lasted through October. During anoxia, surface concentrations remained at 7 to 8 mg L⁻¹, while oxygen concentrations below the thermocline depleted quickly. The duration of anoxia varied somewhat with earlier anoxic conditions coincident with warmer temperatures and increased retention times in 1992. In 1993, a high flow year (and decreased retention), anoxia was delayed until late May. Cooler bottom water temperatures and increased retention time in 1994 resulted in anoxic conditions in May, which was later than 1992 and earlier than 1993. The extent of anoxic conditions each year was similar with anoxia present throughout the hypolimnion (depths greater than 33 ft). Observed temporal trends were also apparent in spatial distribution of temperature gradients and dissolved oxygen concentrations, with similar conditions observed longitudinally within the lake in August of 1994 (Figure 7).

Oxygen depletion and oxygen consumption data are summarized in Table 2. In general, the two methods provided similar estimates of oxygen utilization rates with estimates ranging from 0.01 to 0.11 mg L⁻¹ day⁻¹, until anoxia, and an additional deficit rate (based on consumption of oxygen following reaeration of anoxic water) ranging from 0.06 to 0.51 mg L⁻¹ day⁻¹. These data provide an indication of oxygen necessary to maintain aerobic conditions in a localized area as well as additional oxygen necessary to meet deficits transported into the aerobic area from upstream.

Water chemistry data are summarized in Table 3. In general, the lake exhibits relatively high alkalinity (mean values near 180 mg L⁻¹ as CaCO₃), as would be expected with mean calcium concentrations near 40 mg L⁻¹, and a pH mostly greater than 7.0 standard units. Mean sulfate concentrations were near 66 mg L⁻¹ and mean total dissolved concentrations were near 350 mg L⁻¹, which represent about 95 per cent of the total solids. Mean values were calculated by sites for all years to describe spatial patterns. Concentrations of

total orthophosphate were relatively high with mean values ranging from 0.127 mg-PO₄ L⁻¹ in the outflow (ALP4) to 0.512 mg-PO₄ L⁻¹ in the upstream region of the lake (ALP3). A similar pattern was observed for chlorophyll *a* concentrations with maximum mean values in the upstream region. A dissimilar pattern was observed for nitrogen compounds, ortho-phosphate, manganese, and dissolved sulfide with greatest mean concentrations occurring in the downstream region of the lake (ALP1). Iron concentrations ranged from 0.4 mg L⁻¹ to 1.3 mg L⁻¹ with mean concentrations in the mid and downstream region of the lake near 0.6 mg L⁻¹. Although total organic carbon data were limited, concentrations ranged from detection limit (0.5 mg L⁻¹) to 30 mg L⁻¹ with a relatively high mean of 9.6 mg L⁻¹. Water clarity and turbidity were variable with Secchi disk depths from less than 1 meter (app. 3 ft) to greater than 2 meters (app 6.6 ft) and turbidity ranging from low values (less than 5 to greater than 100 nephelometric turbidity units (NTUs)) with minimum water clarity and maximum turbidity occurring during high flow periods.

Since the seasonal presence of hydrogen sulfide is a concern, dissolved sulfide data greater than the detection limit were plotted by month (Figure 8). Although detectable concentrations were observed coincident with seasonal anoxia, maximum concentrations were observed primarily in August, September, and October.

Sediment chemistry data are summarized in Figure 9. Except for total phosphorus and nitrate/nitrite nitrogen, variability among stations was minimal and longitudinal gradients were not apparent. Total phosphorus and nitrate/nitrite nitrogen concentrations decreased from the upstream station to the near-dam station.

5 Conclusions and Recommendations

Alamo Lake exhibits strong thermal stratification which begins in early April and persists until October. As a result of well-defined thermal structure and increased retention time due to operations, hypolimnetic oxygen depletion establishes throughout the hypolimnion of the lake. Chemical and biological processes during anoxia result in production of hydrogen sulfide which degasses during periods of low discharge causing noxious odors and creating hazardous and corrosive conditions within the outlet works of the project. The establishment of anoxic water available for release has been determined to be the basis for adverse water quality conditions.

Two possible methods are proposed for consideration, each dealing with improving release water quality to minimize adverse impacts in the outlet works and downstream. One method that should be considered is to develop a means to provide surface water to the outlet works for low flow releases or for supplementing bottom withdrawal. A second method would utilize aeration or oxygenation to maintain an aerobic hypolimnion in the vicinity of the withdrawal zone during stratification. Methods such as mechanical pumps and metalimnetic aeration were reviewed and were determined to be not applicable at Alamo Lake.

Altering the way by which water is released from the project can result in desired release water quality without addressing the in-lake processes which result in adverse water quality. Installing a siphon or by-pass tube is designed to withdraw surface water for discharge over the dam or rerouting to existing bottom withdrawal outlet works. The technique utilizes withdrawal of epilimnetic water to avoid the release of anoxic water and the subsequent degassing of hydrogen sulfide. The technique is limited to minimum releases and feasibility of structural modifications to the outlet works for the by-pass tube or dam and flood gates for use of the siphon. While the problems associated with the degassing of hydrogen sulfide are avoided, the adverse water quality in the lake remains on a seasonal basis.

Altering the in-lake water quality prior to discharge can also result in improved water quality in the releases. Techniques applicable to Alamo Lake

for in-lake water quality enhancement include methods of hypolimnetic aeration.

The main purpose of hypolimnetic aeration is to increase the oxygen content of the hypolimnion without warming the hypolimnion or destratifying the lake. Fast and Lorenzen (1976) reviewed 21 different hypolimnetic aerators and were able to group them into three categories: mechanical agitation, injection of oxygen, and injection of air.

Mechanical agitation consists of drawing water from the hypolimnion, aerating it on shore, and then returning the water to its original depth. This method has been used successfully, but is unpopular because of poor efficiency of gas exchange (Pastorak et al. 1982).

Pure oxygen can be directly introduced to the hypolimnion and forced downward with a pump, or it could be released at depth and allowed to rise through the water column. If the lake is sufficiently deep, the oxygen bubbles will completely dissolve before reaching the epilimnion. Another method of oxygenation involves withdrawing water from the hypolimnion, passing it through shore-based pumps where it is injected with oxygen, and then returning it to the deep waters.

Injection of air has probably been the most popular hypolimnetic aeration method. Options in this category involve the mass movement of water (i.e., full air-lift and partial air-lift, or gas transfer via diffusion). Full air-lift forces anoxic water to the surface by sending compressed air to the bottom of an inner cylinder. The rising bubbles lift the air-water mixture to the surface, the air bubbles are vented to the atmosphere, and then the water is returned to depth in an outer cylinder. Partial air-lift systems aerates the hypolimnetic water in place, separating air and water bubbles at depth. The air bubbles are then released to the surface, and the aerated water returned to the hypolimnion. Although the full air-lift system is least costly and more efficient (Fast et al. 1976), the partial design is probably most frequently used because of its commercial availability.

More recent aeration applications have employed gas transfer via diffusion with the use of various diffusers at or near the bottom of the lake. This technique utilizes a distribution system of diffusers for dispersing air bubbles similar to destratification systems but relies on gas transfer properties of the rising bubble column for supplying dissolved oxygen to the water column. While much less efficient than oxygen injection, this method is much less expensive if gas transfer is sufficient to meet oxygen demand.

Hypolimnetic aeration is highly effective for increasing dissolved oxygen. The hypolimnion will remain aerobic as long as the aerators are in operation. Negative features include unintentional destratification, supersaturation with N_2 gas, and the creation of a metalimnetic oxygen minimum.

The following actions are recommended for consideration:

- a. SPL should evaluate the feasibility of surface withdrawal via a by-pass tube or siphon. This action will not eliminate the problem of hydrogen sulfide formation in the hypolimnion but it will provide better quality of release water. The presence of anoxic water in the hypolimnion may still be problematic periodically during reduced discharge.
- b. Hypolimnetic aeration should be considered using the following approach:

A pilot study using guidance provided by evaluations of enhancement techniques should be conducted to attempt aeration of the forebay region using diffusion of compressed air or oxygen. Rental of a compressor should be considered to reduce the cost of the pilot study if compressed air is used. The diffuser system should employ the use of porous garden hoses (commonly available soaker hoses) with a flexible design for the addition of diffusers as necessary to accomplish sufficient aeration. Comparisons of costs and efficiencies were conducted by the Hydraulics Laboratory of the USAE Waterways Experiment Station using compressed air and oxygen. Details and recommendations were provided to the Los Angeles District in a letter report.

Water quality monitoring should be conducted to describe conditions prior to installation of an aeration system, during operation of the system, and during periods of non-operation of the system to evaluate the impact of the system on water quality. This monitoring would include routine sample collection (as currently conducted) with increased sampling in the forebay and tailwater prior to and during operation of the system.

If an aeration system is not installed, continued water quality sampling (as currently conducted) is recommended should water quality enhancement be considered at a future time. Emphasis should be placed on the temporal and spatial development of thermal gradients, anoxia, and increased sulfide concentrations which would include monitoring nearly year round.

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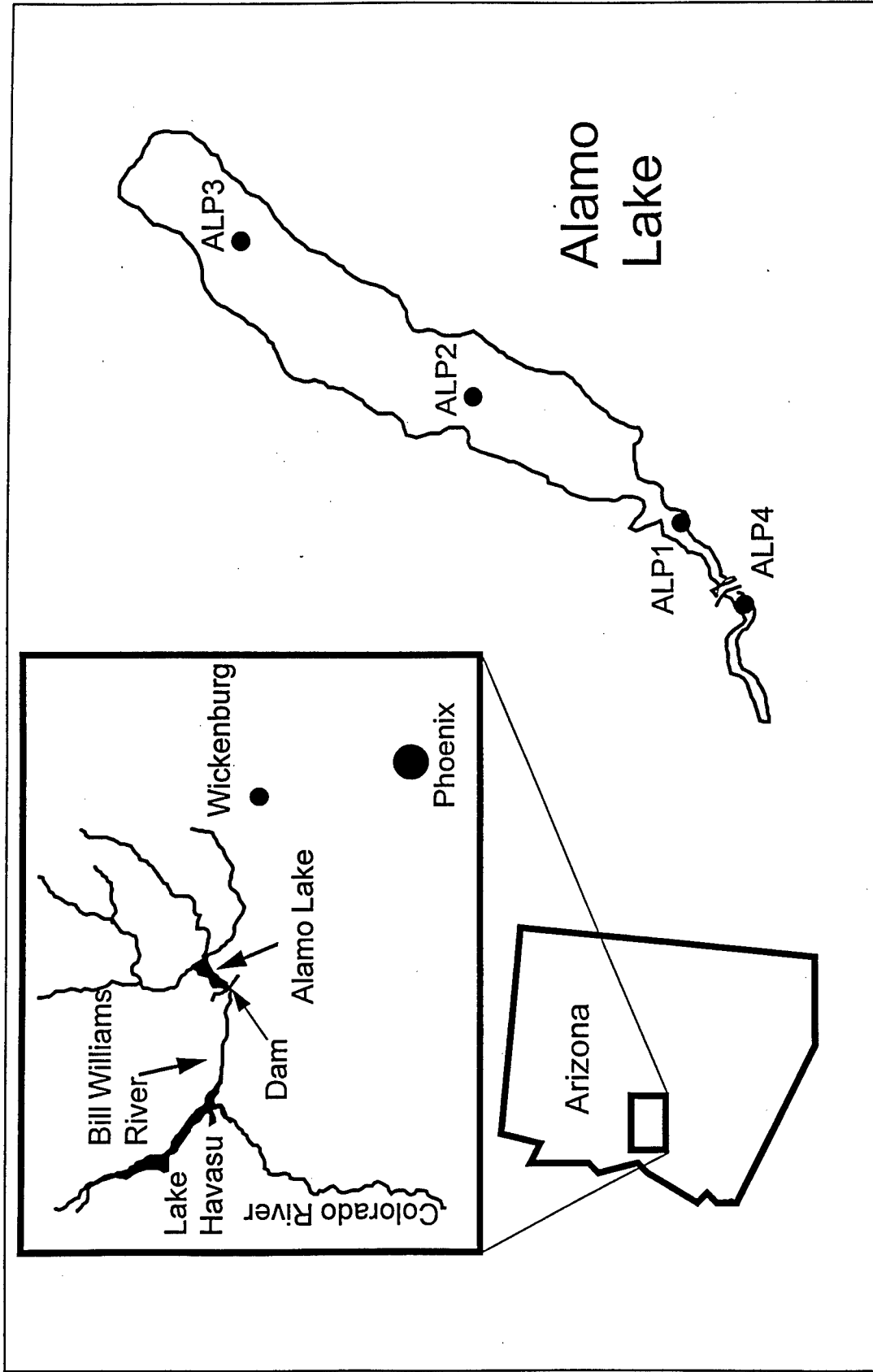


Figure 1. Location of Alamo Lake and water quality sampling stations

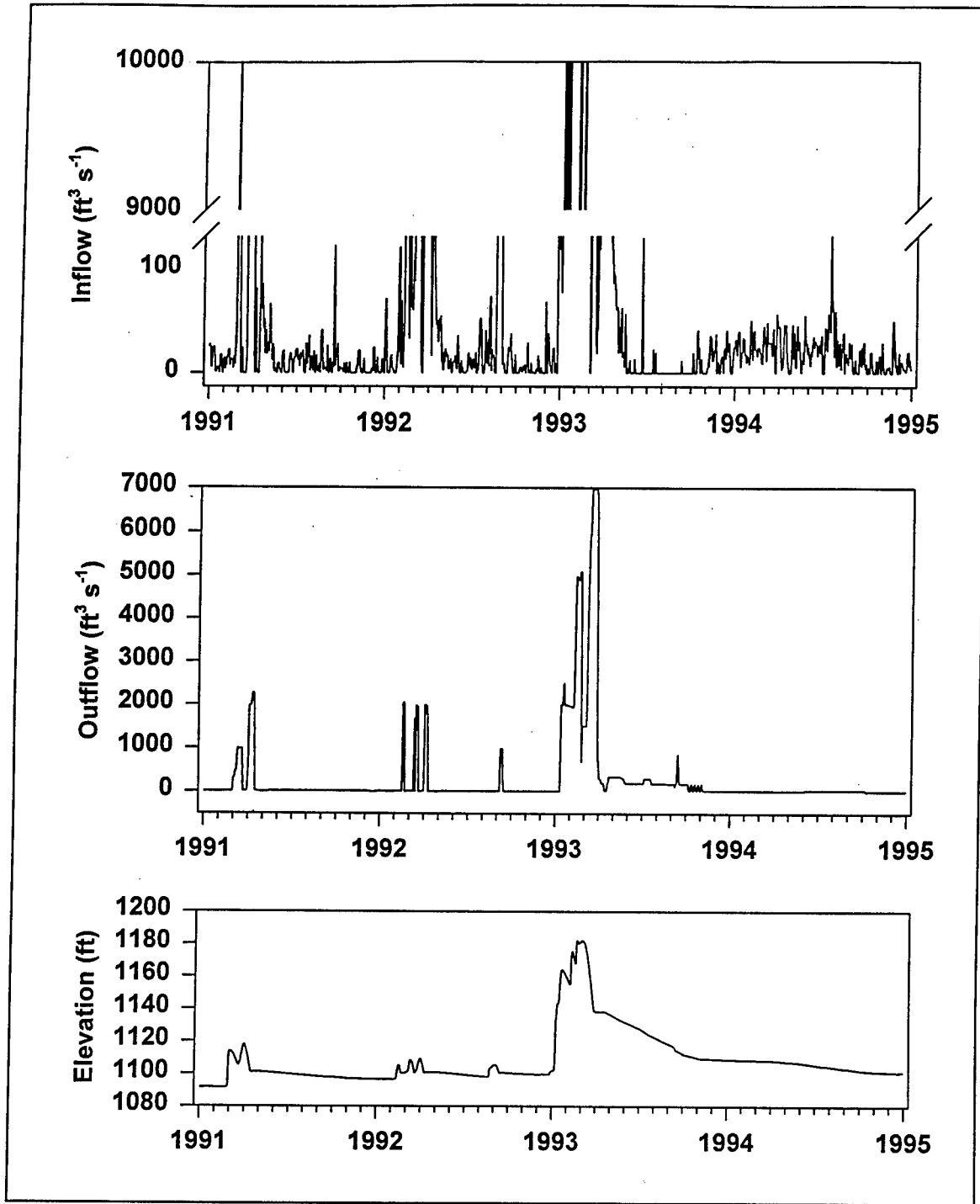


Figure 2. Inflow, outflow, and elevation of Alamo Lake

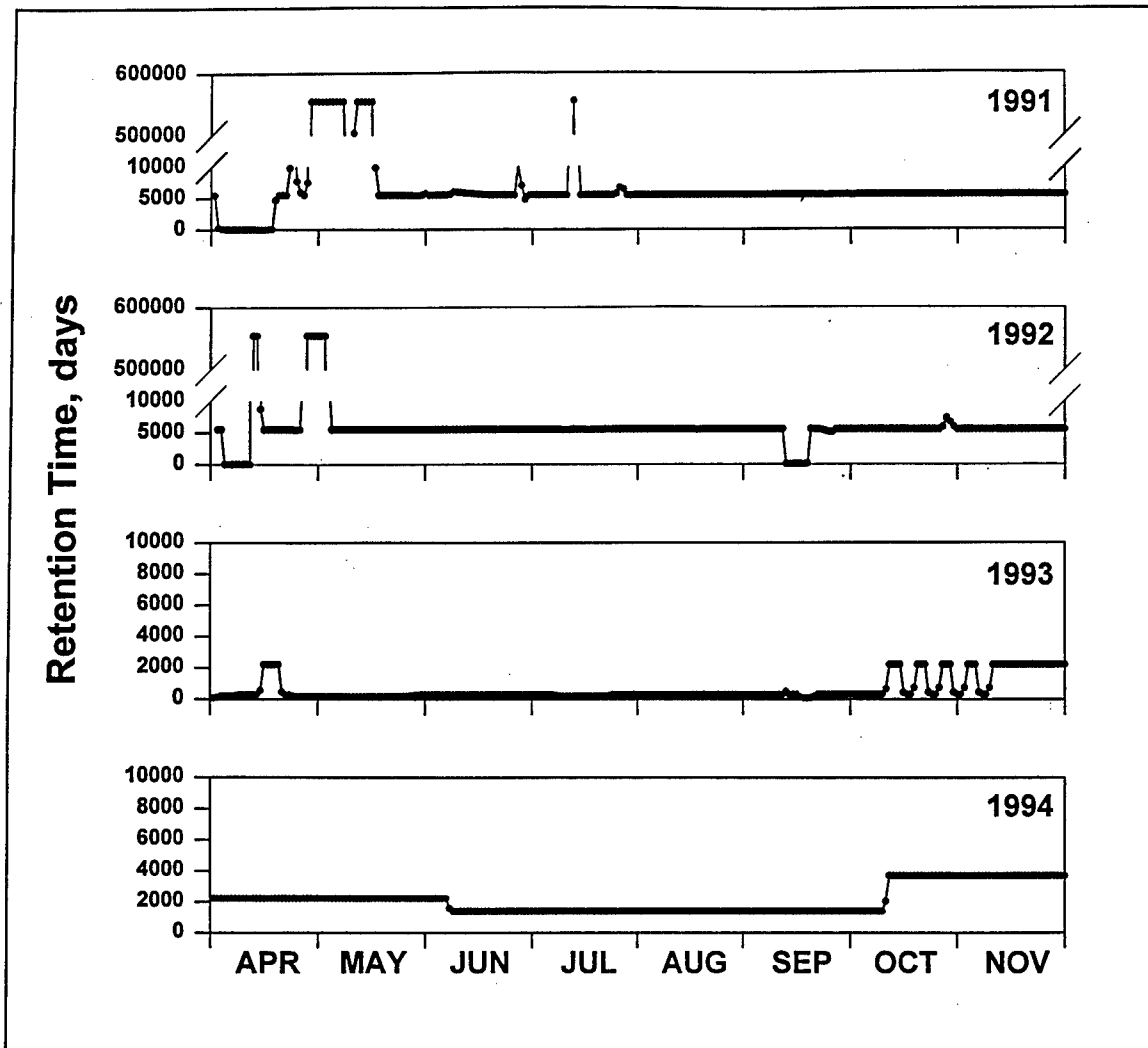


Figure 3. Calculated retention time in Alamo Lake

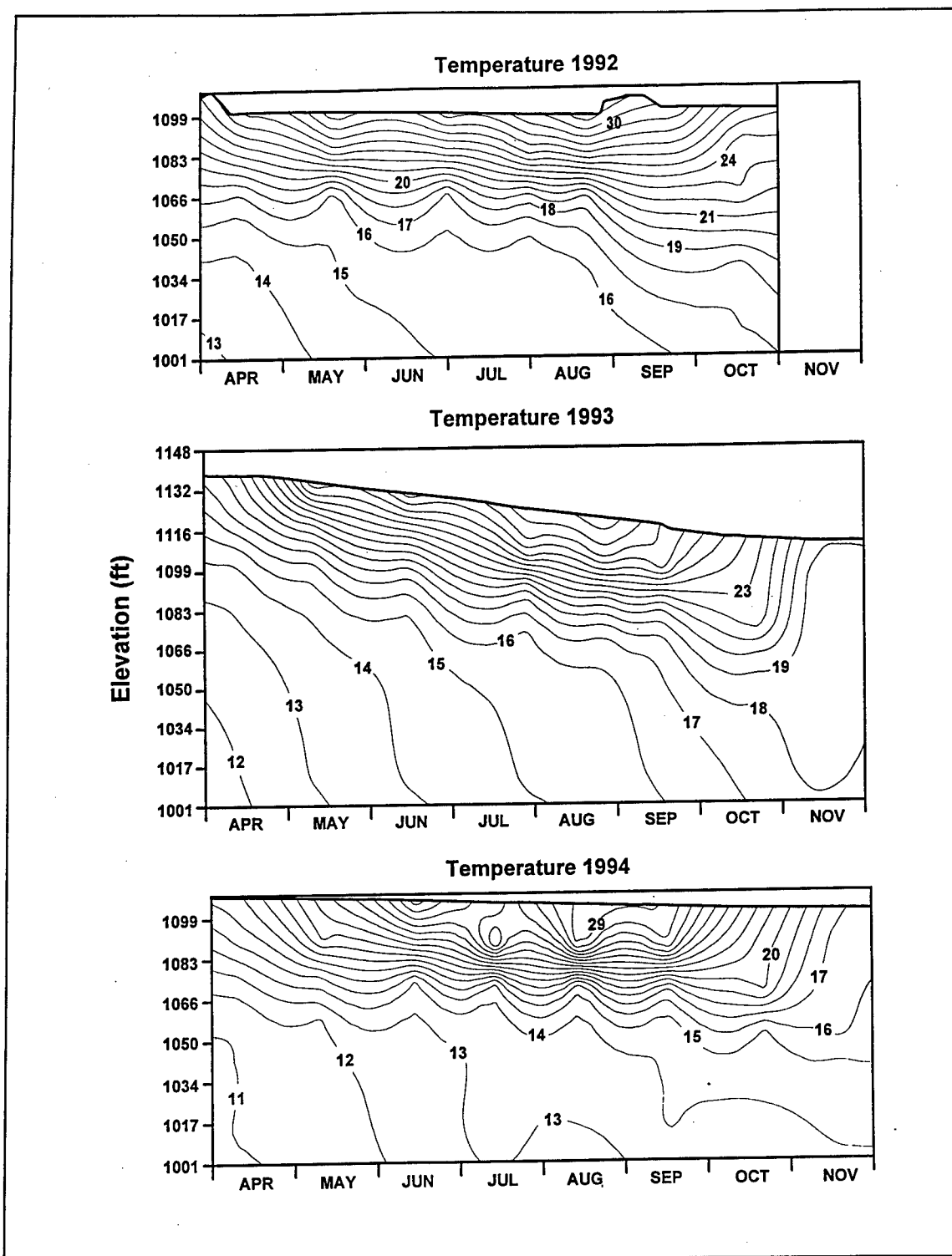


Figure 4. Temporal and vertical temperature structure at ALP1, April through November

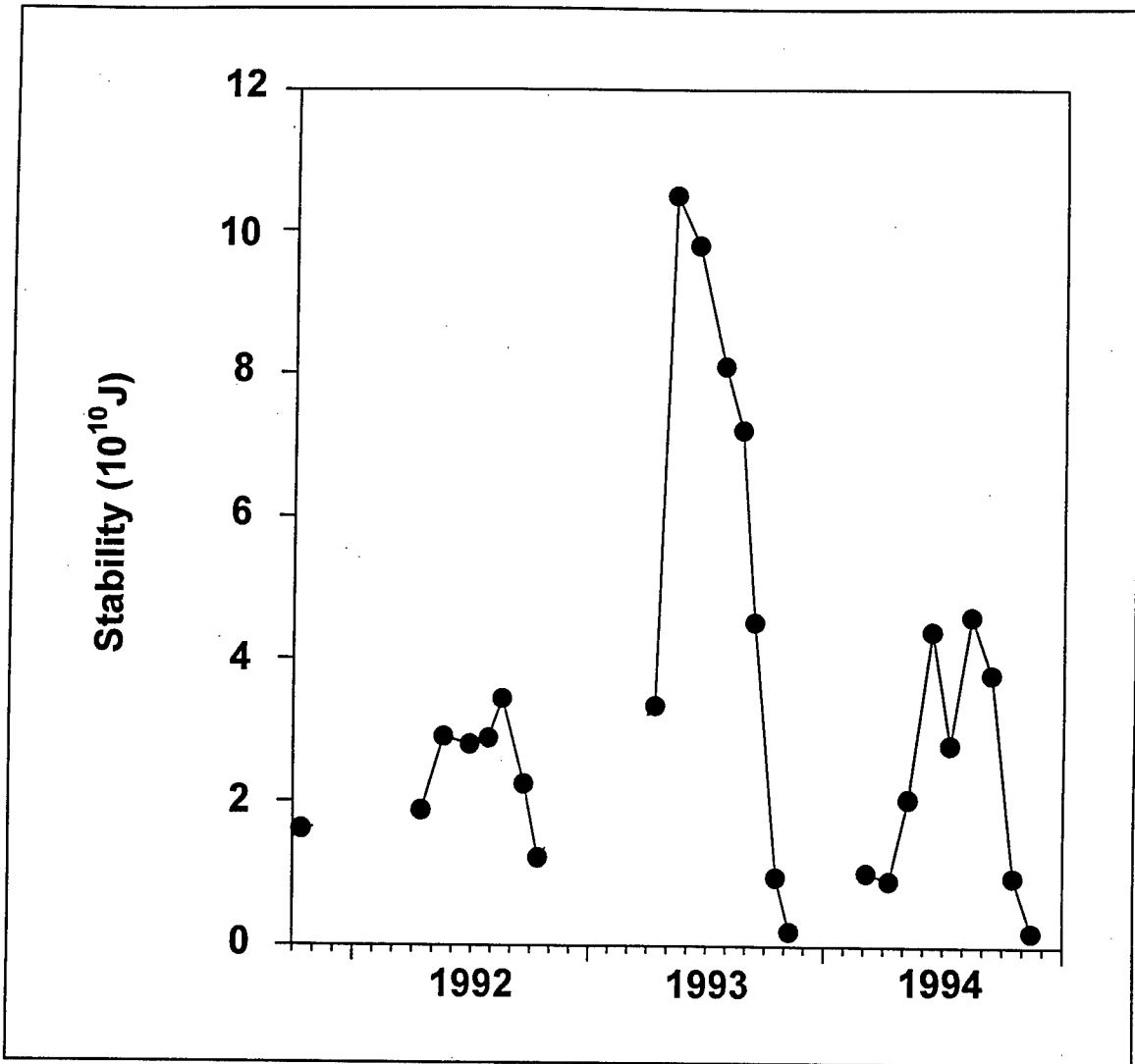


Figure 5. Calculated thermal stability of Alamo Lake

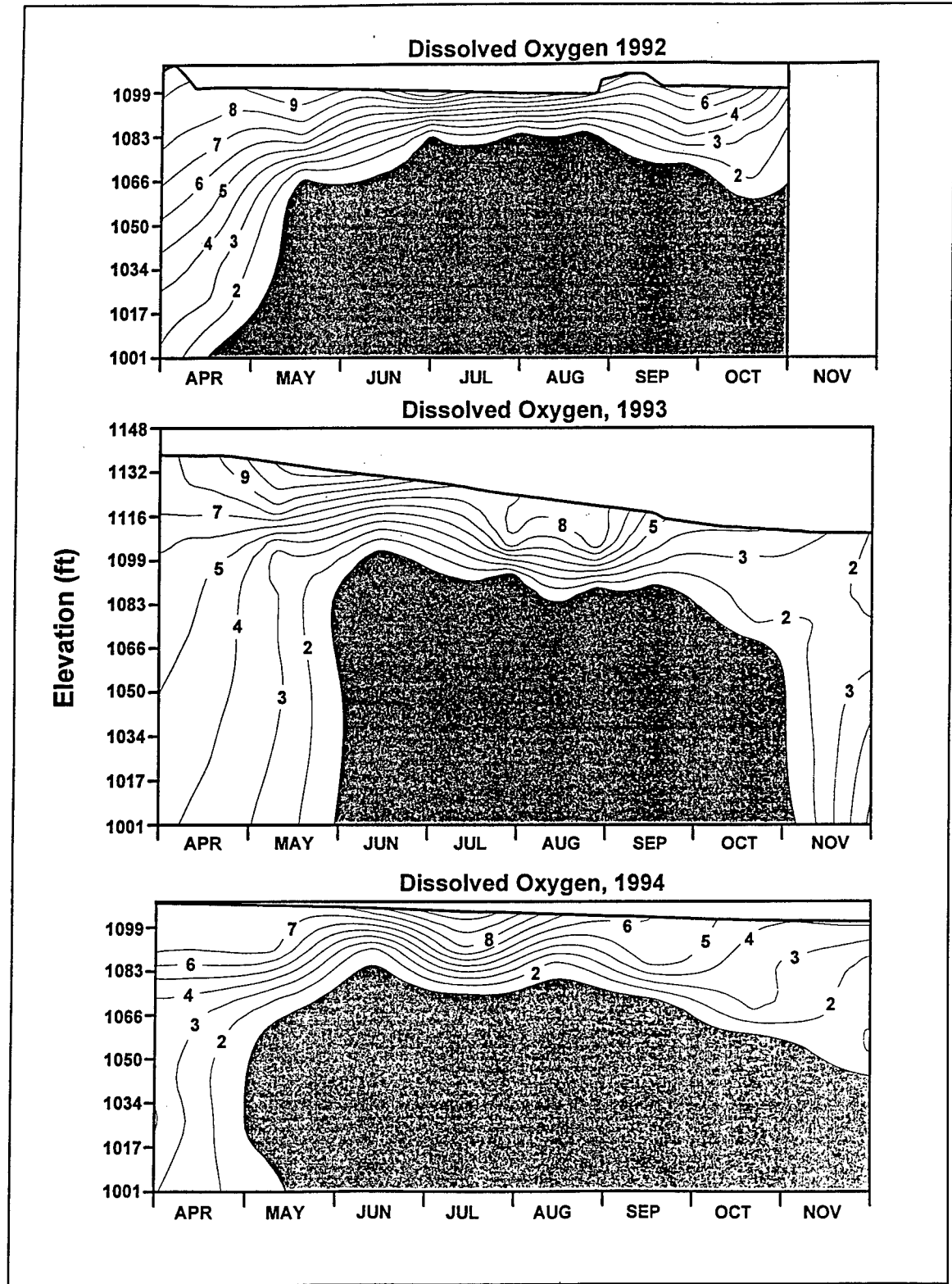


Figure 6. Temporal and vertical distribution of dissolved oxygen concentrations at ALP1 during thermal stratification

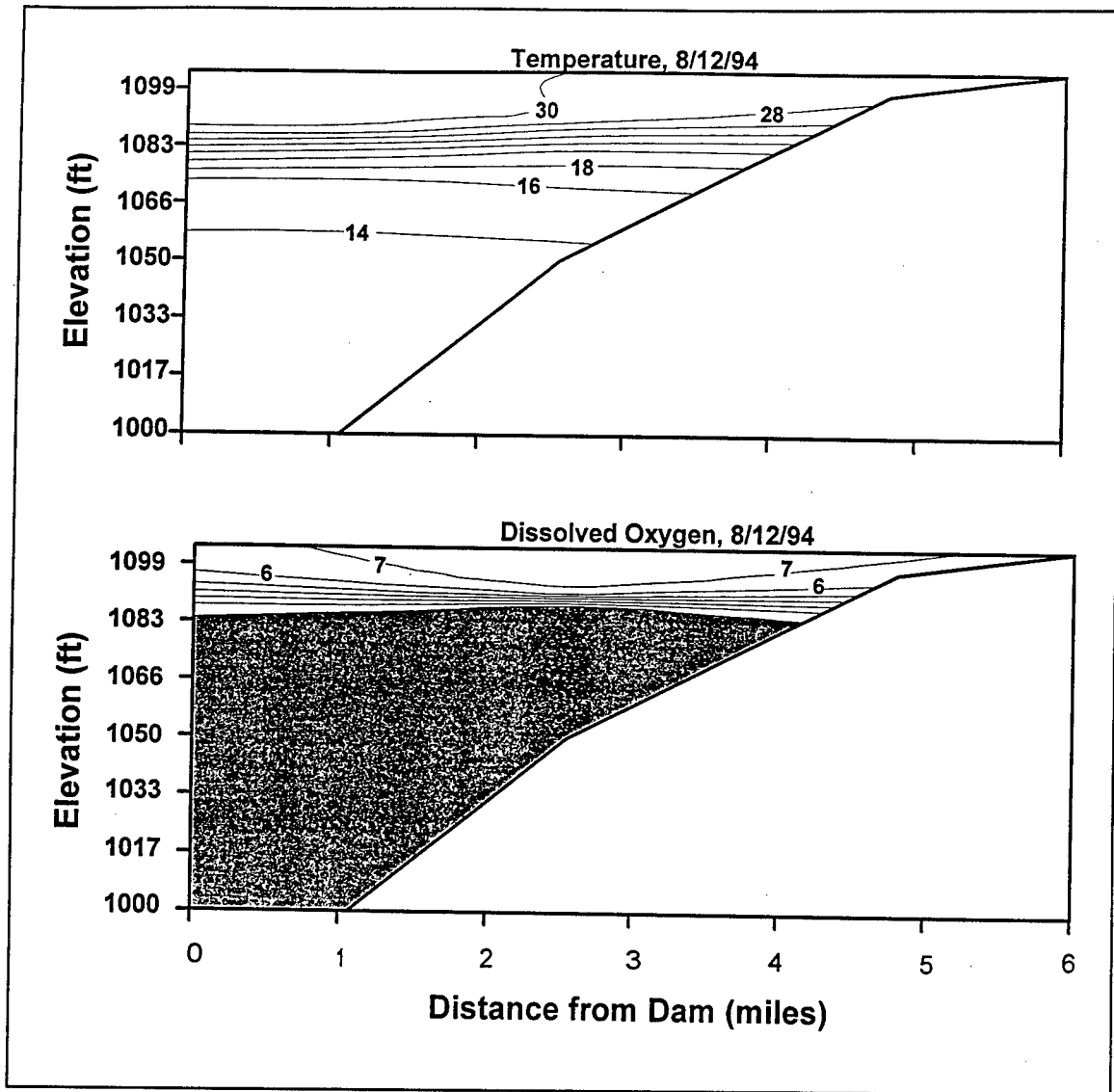


Figure 7. Longitudinal patterns in temperature and dissolved oxygen concentrations during stratification at Alamo Lake

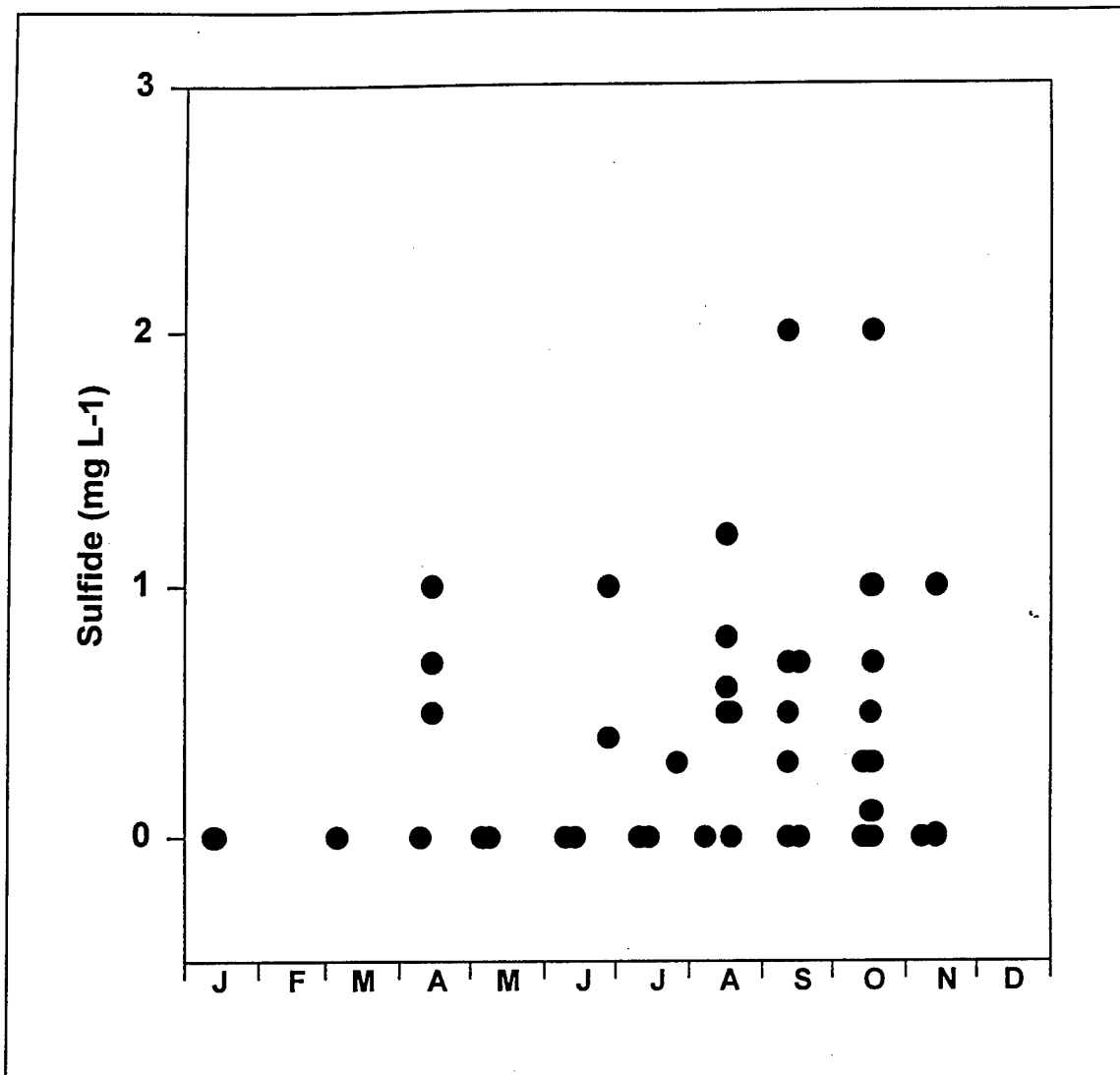


Figure 8. Dissolved sulfide concentrations measured during the study period at Alamo Lake

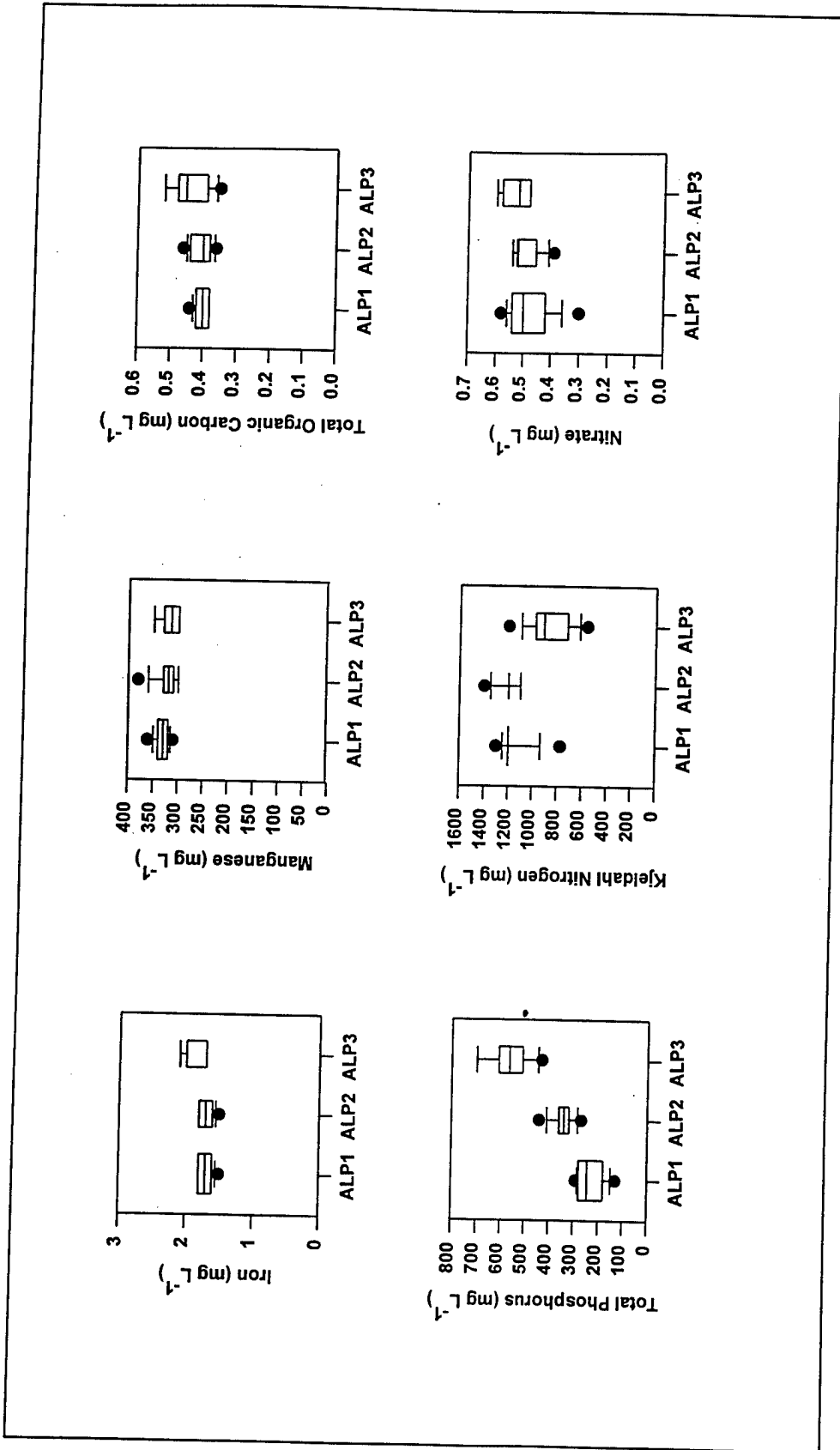


Figure 9. Sediment chemistry of Alamo Lake (upper and lower boundaries of the box plots indicate 75th and 25th percentiles, respectively; a line within the box marks the median; error bars above and below the box indicate 90th and 10th percentiles; and points indicate values outside indicated percentiles)

**Table 1
Hydrologic Features of Alamo Lake***

Drainage Area	4,770 mi ²
Recreation Pool:	
Elevation range	990-1,066 feet
Storage at top of pool	20,082 acre-feet
Surface at top of pool	946 acres
Water Conservation Pool:	
Elevation range	1,066-1,171.3 feet
Storage at top of pool	386,931 acre-feet
Surface area at top of pool	6,743 acres
Flood Control Pool:	
Elevation range	1,171.3-1,235 feet
Storage at top of pool	995,300 acre-feet
Surface area at top of pool	13,300 acres
Maximum water conservation discharge	2,000 ft ³ sec ⁻¹
Maximum flood control release through gates	7,000 ft ³ sec ⁻¹
* Area and storage values based on the 1993 Area-Capacity Curve.	

**Table 2
Oxygen Utilization Assessment**

Date	Oxygen Depletion Rate, Hypolimnion, mg L ⁻¹ day ⁻¹ 1	Oxygen Depletion Rate, Metalimnion, mg L ⁻¹ day ⁻¹ 1	Oxygen Depletion Rate, Both, mg L ⁻¹ day ⁻¹ 1	Oxygen Consumption Rate mg L ⁻¹ day ⁻¹ 2
4/14-5/18, 1992	0.11	0.05	0.075	
4/12-5/10, 1993	0.07	0.01	0.035	
5/10-6/14, 1993	0.085	0.07	0.077	
4/10-5/9, 1994	0.08	0.03	0.04	
5/9-6/13, 1994	0.02	0.08	0.06	
July 1994				ALP1, mid-depth 0.08
				ALP1, bottom 0.06
				ALP2, mid-depth 0.24
				ALP2, bottom 0.19
October 1994				ALP1, mid-depth 0.14
				ALP1, bottom NA ³
				ALP2, mid-depth 0.15
				ALP2, bottom 0.51

¹ Denotes data calculated using PROFILE.

² Denotes data calculated from incubated bottles.

³ Denotes data not available.

Table 3
Water Chemistry Data

SITE	N	MIN	MAX	MEAN
Total Alkalinity (mg L⁻¹ as CaCO₃)				
ALP1	121	110	330	182
ALP2	58	104	242	179
ALP3	33	104	257	175
ALP4	14	109	220	192
Sulfate (mg L⁻¹)				
ALP1	27	51	90	66
ALP2	0			
ALP3	0			
ALP4	0			
Total Dissolved Solids (mg L⁻¹)				
ALP1	122	177	530	357
ALP2	58	128	540	349
ALP3	33	190	560	353
ALP4	14	220	472	368
Total Phosphate (mg L⁻¹)				
ALP1	87	-0.1	2.2	0.171
ALP2	44	-0.1	2.2	0.198
ALP3	22	-0.1	9.2	0.512
ALP4	11	-0.1	0.278	0.127
Chlorophyll a (µg L⁻¹)				
ALP1	65	0.07	29.2	3.44
ALP2	29	0.29	40.4	5.42
ALP3	17	2.28	29.6	9.27
ALP4	8	0.39	9.42	2.94
Total Kjeldahl Nitrogen (mg L⁻¹)				
ALP1	120	0.16	1.69	0.71
ALP2	56	0.19	1.78	0.65
ALP3	31	0.27	1.2	0.69
ALP4	14	0.43	1.23	0.75

Table 3 (Continued)				
SITE	N	MIN	MAX	MEAN
Ammonia Nitrogen (mg L⁻¹)				
ALP1	89	-0.1	1.24	0.21
ALP2	41	-0.1	0.63	0.20
ALP3	16	-0.1	0.33	0.14
ALP4	10	-0.1	0.57	0.26
Iron (mg L⁻¹)				
ALP1	23	-0.1	1.3	0.58
ALP2	3	0.51	0.93	0.67
ALP3	1	1	1	1
ALP4	1	0.39	0.39	0.39
Total Organic Carbon (mg L⁻¹)				
ALP1	21	-0.5	30	9.6
ALP2	0			
ALP3	0			
ALP4	0			
Calcium (mg L⁻¹)				
ALP1	34	35	48	42
ALP2	3	43	43	43
ALP3	1	45	45	45
ALP4	1	44	44	44
Total Residue (mg L⁻¹)				
ALP1	115	192	550	384
ALP2	55	143	542	370
ALP3	32	200	710	374
ALP4	13	205	486	371
Total Suspended Solids (mg L⁻¹)				
ALP1	84	-10	170	17
ALP2	46	-10	75	15
ALP3	27	-10	190	24
ALP4	11	-5	443	55
<i>(Sheet 2 of 3)</i>				

Table 3 (Concluded)				
SITE	N	MIN	MAX	MEAN
ortho-Phosphate (mg L⁻¹)				
ALP1	65	-0.1	0.39	0.08
ALP2	35	-0.1	0.23	0.08
ALP3	12	-0.1	0.15	0.05
ALP4	10	-0.1	0.14	0.08
Pheophytin (µg L⁻¹)				
ALP1	62	-2	26.8	3.0
ALP2	28	-2.7	82.1	8.5
ALP3	16	-2	11.2	5.3
ALP4	8	0.5	6.5	3.0
Total N (NO₂ + NO₃, mg L⁻¹)				
ALP1	63	-0.1	0.64	0.16
ALP2	32	-0.1	0.47	0.17
ALP3	12	-0.1	0.19	0.09
ALP4	12	-0.1	0.58	0.15
Manganese (mg L⁻¹)				
ALP1	31	-0.05	2.09	0.44
ALP2	3	0.13	0.16	0.14
ALP3	1	0.13	0.13	0.13
ALP4	1	0.11	0.11	0.11
Dissolved Sulfide (mg L⁻¹)				
ALP1	100	-0.5	2	0.2
ALP2	48	-0.5	2	0.1
ALP3	25	-0.5	1	0.1
ALP4	11	0	15	1.4
Turbidity (ntus)				
ALP1	34	3.32	108	12.90
ALP2	12	4.14	20.4	8.01
ALP3	5	4.72	38.2	15.58
ALP4	4	3.53	78.1	26.40

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