THE CAUSES AND CONTROL OF ALGAL BLOOMS IN CLEAR LAKE

Clean Lakes Diagnostic/Feasibility Study
For Clear Lake, California

***prepared** for:*

Lake County Flood Control and Water Conservation District
California State Water Resources Control Board
United States Environmental Protection Agency

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1 Executive Summary

(1) Clear Lake is subject to frequent blooms of noxious scum forming blue-green algae which seriously reduce residents' enjoyment of the lake and limit its value to visitors. Economic losses due to the effects of algae on water quality are in excess of seven million dollars per year.

(2) A review of the historical literature suggests that the water quality of Clear Lake deteriorated significantly in the mid 20th Century. The first unambiguous reports of highly turbid condition occur in 1939-41 and 1946-47. Blue-green scums were apparently not a serious problem as late as 1925, although the lake has always been very productive.

(3) The Goldman-Wetzel and Home conclusions indicating that the lake is mainly nitrogen and iron limited are confirmed, although the timing and extent of blue-green blooms suggests more frequent phosphorus limitation in recent pre-drought years.

(4) The hypothesis that erosion is the main source of nutrients, especially iron and phosphorus, to the system is confirmed by a) the response of algae to variations in stream inflow over the last 23 years, b) by the lack of conspicuous response to improved sewage treatment, and c) by our monitoring measurements.

(5) Accelerated erosion (due to destabilization of creek channels by gravel mining, road construction, lakeside dredge and fill operations, the shoreline deposition of mine overburden and tailings, and similar disturbances) is the most important factor causing a doubling of sediment and phosphorus inflow into Clear Lake. Wetland and flood plain loss reduce the sedimentation capacity of the basin and increase delivery of nutrient rich sediments to the lake. About 160 metric tons (MT) of phosphorus enters the lake in an average year. Of this about 25 MT leaves via the Cache Creek outlet, and the rest is buried in the sediments. In the 1968-73 period around 250-500 MT of phosphorus cycled through the water of the lake, whereas for the 1974-86 period 100-200 MT cycled. During the recent drought years the phosphorus mass cycling rose to 550-700 MT.

(2) The hypothesis that phosphorus is in such great excess that no practical improvements can make it limiting is not convincing. The large drop in internal (and presumably external) phosphorus loading from the 1969-72 period to the 1974-86 period brought phosphorus close to limiting levels even during the summer maxima in some years, and probably played a role in the limitation of the spring bloom. There is statistical evidence from the historical data that phosphorus supply played some role in regulating blue-green algal abundance during the last 23 years. Cycles of drought and flood have strongly influenced the behavior of the phytoplankton community.

(7) It is apparent from the very high phosphorus (but clear water) conditions of the 1991-92 summers that iron can be sufficiently limiting to greatly reduce the abundance of scum forming species. Although the cycling of iron is poorly understood compared to phosphorus, it is known that the two elements tend to cycle together. A strategy for controlling phosphorus will probably control iron as well.

(8) The sediments store a fairly large pool of phosphorus during the winter that is released in summer to fuel algal growth. However this store is not so large as to preclude a reasonably prompt response to load reductions. The only measurements come from the recent anomalously high phosphorus drought period when several hundred tons of phosphorus left the iron/aluminum bound fraction of the sediments each summer and fall. Depending upon how the sediment storage term reacts to reduced loading, the time required to dilute away half of the stored phosphorus could range from few years to a few decades. Data from other lakes suggests an optimistic estimate for a lake like Clear Lake, with moderately elevated total phosphorus in sediments. It should respond relatively rapidly to reduced loading.

(9) Many alternative treatment strategies have been proposed for Clear Lake algal scums. Some, such as harvest of algae, and dredging of sediments, are impractical except on a small scale to solve local problems. Others, such as further improvements in sewage treatment, promise only marginal reductions in nutrient loading.

Control of erosion from creek channels, roads, and other sources of sediment and nutrients is a practical strategy. Wetland rehabilitation, especially the rehabilitation of Robinson Lake, would have an important impact on nutrient flows from Middle, Clover, and Scotts Creeks. Inactivation of sediment phosphorus by chemical means (alum) might be practical in Clear Lake, if the pessimistic scenario of high internal loading after erosion control and wetland rehabilitation proves correct.

(10) We recommend that Best Management Practices for erosion control be applied to degraded stream channel sections, road cuts and fills, and similar sources of erosion products that might enter Clear Lake. County, State, and Federal land management agencies in the basin should cooperate to seek budget authority, grants, and other sources of funding to implement Best Management Practices as widely and rapidly as is feasible. The objective should be to reduce stream sediment loads to natural levels, approximately half of today's delivery of nutrients to the lake.

(11) We recommend that a sediment retention wetland system be constructed in the former Robinson Lake to reduce the flow of sediment and nutrients into Clear Lake from Middle, Clover, and Scotts creeks. Smaller scale wetland and flood plain restoration projects may be useful under the Best Management Practices recommendation.

(12) We recommend a monitoring program to measure the effectiveness of management practices. Control of erosion in complex topography cannot be done by formula. The effectiveness of many practices must be evaluated in the field and adapted to local conditions. The lake itself is an ever-changing system; human causes of change have to be separated from natural ones in order to increase the effectiveness of management strategies. Some practices, such as algae skimming and spraying require timely data for - prompt responses.

(13) The County, State, and Federal agencies with responsibilities in the basin should actively support research and development projects with reasonable prospects of short or medium term payoffs. We recommend research on phosphorus inactivation (alum treatment), iron geochemistry, phosphorus recycling from sediments, wastewater and septic leachfield leakage, and algae skimming. Additional research on the economic benefits of improved water quality in Clear Lake would help define the size of investment that should be made by local, state, and federal interests in pursuit of better water quality.

Lake County in cooperation with individuals, voluntary organizations, local, state and federal agencies can implement practical actions and policies, primarily for erosion control, that will substantially improve lake water quality. Continued monitoring and research will ensure success and may develop other cost effective strategies.

2 Objectives

2.1 Objectives of the Clean Lakes Project

The water quality of Clear Lake is seriously impaired by frequent blooms of scum forming blue-green algae. The objectives of this project were: (1) to review past data and conduct new investigations to better understand the causes of this problem, (2) to review treatment options for control of blue-green algal blooms under the specific conditions found in Clear Lake, and (3) recommend the best options for current implementation and further investigation.

This project initiated a study of the problem under a US Environmental Protection Agency Clean Lakes Program contract with Lake County starting in April 1991. Additional support was provided by Lake County, and the University of California (via support from the Hopland Field Research and Extension Center, the Agricultural Experiment Station, the Ecotoxicology Training Grant, and an EPA funded grant to the Center for Ecological Health Research).

Six related investigations were conducted: (1) A study of the history of algae-related water quality problems and their investigation, (2) an analysis of the water quality data set collected by DWR, (3) a study of external nutrient loading, (4) a direct study of internal nutrient loading (recycling), (5) a series of nutrient bioassays, and (6) an analysis of wetlands and riparian vegetation. The objectives of these studies were to (1) retest the hypotheses formulated by Home and coworkers, and (2) to search further for practical control strategies.

The Findings **(Chapters 1-8)** of the study were circulated to approximately 120 reviewers in the form of a Draft Interim Report in October, 1993. We received detailed comments from 10 people. Additional comments in Algae Subcommittee meetings and from the Technical Advisory Committee meeting on November 18, 1993 were incorporated into the Draft Final Report, which included **Chapter 9**, Alternative Methods for the Control of Blue-Green Blooms, and **Chapter 10**, Recommended Strategies. The Draft Final Report was circulated to 75 individuals and made available to the public through copies deposited at county libraries and photocopy shops. The Draft Final Report was reviewed by the Technical Advisory Committee at a meeting on June 14, 1994, and at a public hearing on that same date. A number of comments received at these meetings and at May and June meetings of the Algae Subcommittee are incorporated into this Final Report.

2.2 Objectives of the Final Report

The objectives of the Final Report are to:

(1) Summarize the technical information available from past studies on Clear Lake,

(2) Analyze the large body of monitoring data collected by the Department of Water Resources accumulated since the last publication of these data in 1975 (including data from 1968-73),

(3) Analyze the data collected under this project on stream water quality, sediment phosphorus recycling, nutrient bioassay experiments, and stream channel conditions,

(4) Evaluate the many strategies known to be useful to control blue-green scums in light of the particular characteristics of Clear Lake,

(5) Recommend a practical course of action that the citizens of Lake County can undertake, with the assistance of local, state, and federal agencies, to control the occurrence of scums.

3 Introduction To The Applied Limnology Of Clear Lake

3.0 Abstract

Clear Lake is the largest natural lake entirely within the borders of California. Its basin has been shaped by faulting, tilting, volcanism, and erosion over a period possibly as long as 2 million years. Clear Lake is a shallow, warm, nutrient rich system with three distinct regions, or arms, which behave like separate lakes in some respects. Minimum winter temperatures are well above freezing. In all seasons the lake is relatively well mixed by winds and cooling at night. Clear Lake is heavily used in the summer for recreational boating and supports considerable sport fishing year-round.

The nutrient richness of this "eutrophic" lake supports large fish and wildlife populations, but also frequent blooms of scum-forming floating blue-green algae (also called cyanobacteria). The latter can be a nuisance to commercial and recreational use of the lake and its environment. Algae problems are most serious at the eastern Arms of the lake, where prevailing winds can push the floating algae into mats that rot. In the worst cases, mats of algae can even restrict boat passage.

Available historical records suggest that algal blooms have become more of a problem in the last half of this century. In reports from the 19th and early 20th Centuries, scum forming algae are hardly mentioned, but growths of forms attached to the bottom were frequently observed. Bottom dwelling plants require relatively transparent water to thrive. Increased turbidity and blue-green scums were conspicuous by the late 1930s, and bottom dwelling algae and waterweeds have been absent in most years since that time. Causes for this shift are not well documented, but most likely an increase in the amount of fine sediment entering the lake during winter runoff was the most important influence. Fine sediments carry nutrients and trace elements which favor the blue-green algae over green algae. Disturbances to stream channels, filling and other earthmoving activities, and the removal of filtering capacity of marshes probably all contributed to the increased nutrient load and consequent onset of a situation in which scum-forming blue-green algae came to dominate the summer season.

3.1 Basic Facts About The Clear Lake Ecosystem

3.1.1 Physical/Chemical Characteristics

Clear Lake is the largest natural lake entirely within the borders of California. It occupies a shallow basin in the Central Coast Range about 80 miles north of San Francisco. The morphometry and watershed area of the basin is shown in **Figure 3.1**, and basic statistics are given in **Table 3.1**.

The Clear Lake Basin was shaped by a variety of processes over the last 1 to 2 million years. A nearly continuous sequence of lake sediments reaching back 480,000 years has been recovered by the US Geological Survey (Sims *et al.,* 1988) and other lake sediments in the region date back to the Early Pleistocene 1.8-1.6 million years ago (Hearn *et al.,* 1988). There is an excellent climate record from cores for the last 127,000 years of this sequence, which documents a shift from pine dominated to oak dominated forests at the end of the Pleistocene glacial Period 10,000 years ago, among other events (Adam, 1988). The diatom sequence in the cores indicate that Clear Lake has been a shallow, productive system, essentially similar to the modern lake, since the end of the Pleistocene (Bradbury, 1988). The basin was created primarily from the shear and torsional stresses of the San Andreas Fault System, the eruption and subsidence

	Area	Depth		Shoreline	Length	Width	Volume
		Mean	Max		Max	Max	Total
	(ha)	(m)	(m)	(km)	(km)	(km)	(m^3x10^6)
Upper Arm	12700	7.1	12.2	56	16.4	12.2	904
Lower Arm	3720	10.3	18.4	39	13.4	4.3	384
Oaks Arm	1250	11.1	18.4	19	8.5	2.6	138

Table 3.1. Basic morphometric information on Clear Lake. Based on Table II of Horne, 1975.

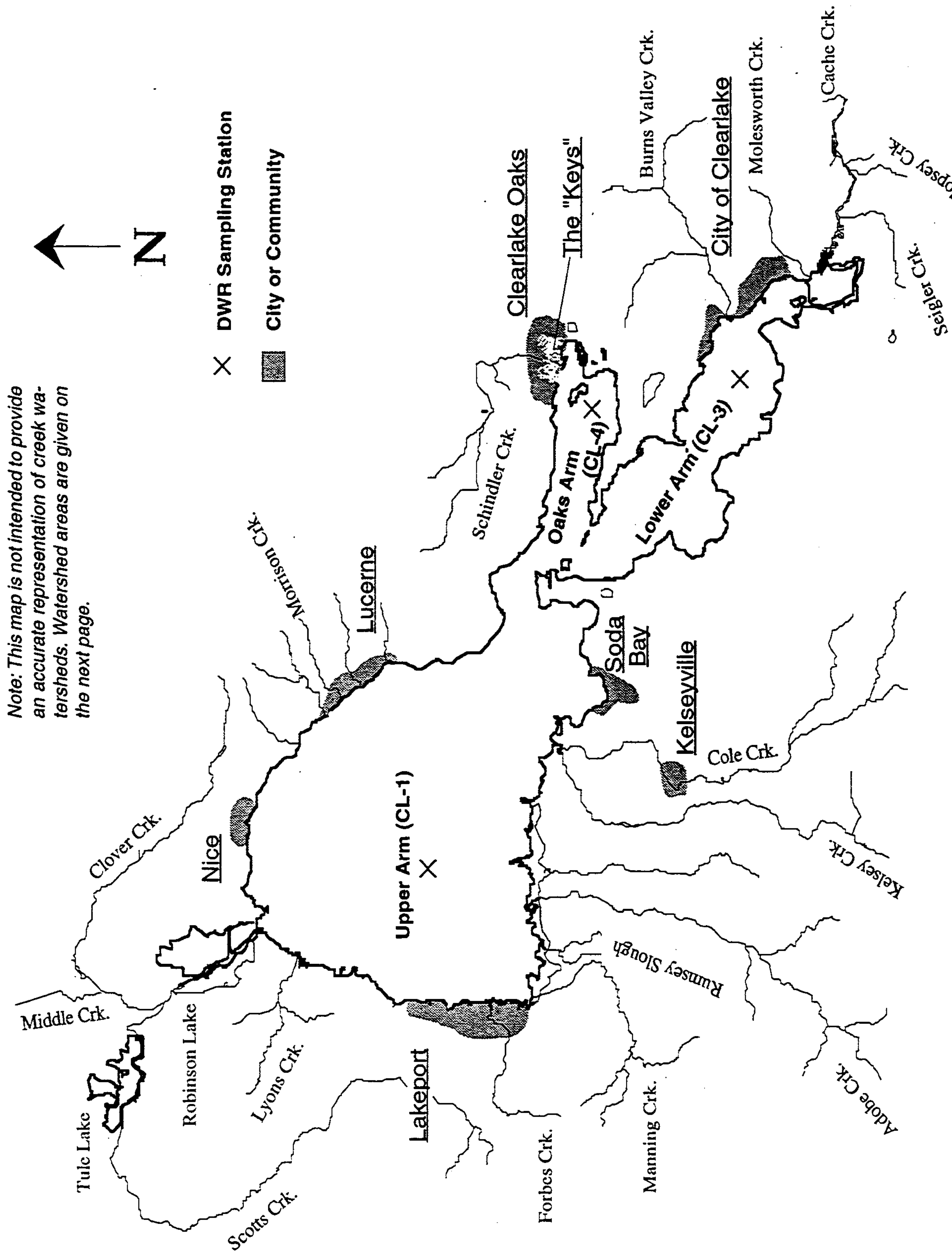
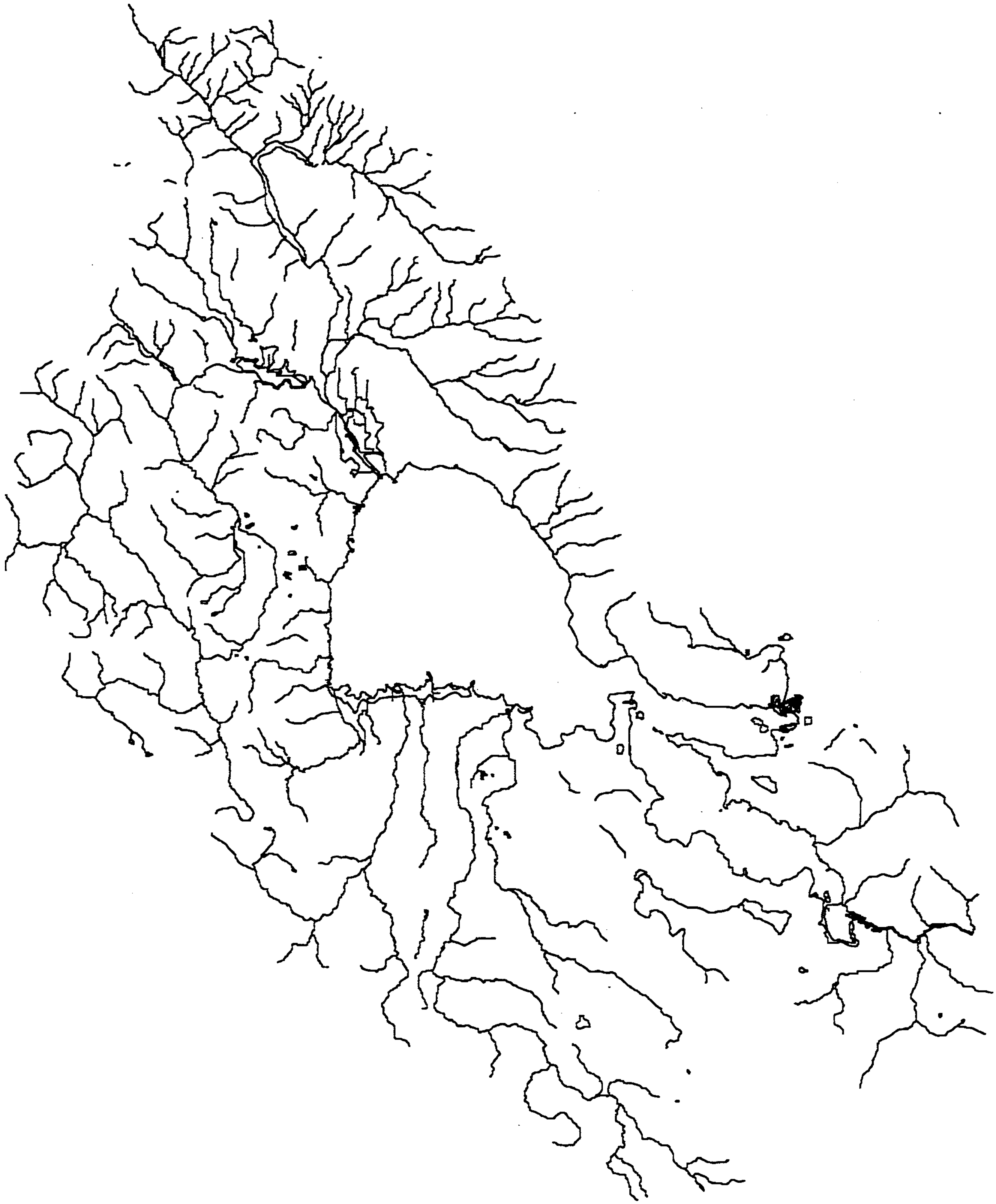


Figure 3.1A. Morphometry of the Clear Lake study area.

Clear Lake Watershed Map: (from U.S.G.S. GIS map)



of the Clear Lake Volcanics, and the erosion and deposition of the parent rock (Hearn *et al.,* 1988).

The east-west extension of the fault system and vertical movements of the faults have created and maintained the basin. Downward vertical movement within the basin created by these processes is at a rate approximately equal to the average sedimentation rate of 0.9 mm/yr in the lake basin. Since these rates are essentially equal, a shallow lake has existed in the upper basin for at least 475,000 years. Although the lake changed shape significantly over this time period, it was generally located in the same area as the Upper Arm (Sims *et al.,* 1988). If sedimentation rates were significantly different than the downshift, either a deep water lake or a dry valley would have resulted.

The Lower and Oaks Arms were created by a combination of faulting and subsidence resulting from volcanic vicinity. The Lower Arm was a shallow marsh from 40,000 to 9,500 years ago. The Oaks Arm was probably 2 m deep from 36,000 to 11,000 years ago (Sims *et al.,* 1988). The downshifting of the lake bottoms in the two arms is probably a combination of faulting and volcanic vicinity. It has been theorized by the locations of springs on the lake bottom that calderas have formed significant portions of the Lower Arm and the Narrows area (Norman Lehrman, personal communication). Lava flows have changed the shoreline in the Narrows, the Lower Arm, and the Oaks Arm.

The basin is on a topographic divide between the westerly Russian River system and the Sacramento Valley. The lake has drained alternately to the Sacramento and Russian River Systems. The lake currently drains to the Sacramento River. The outlet to the Russian River was blocked about 10,000 years ago by a landslide immediately west of Blue Lakes (LCFCWCD, 1970). The natural level of Clear Lake has been maintained by the Grigsby Riffle, a rock sill located at the confluence of Cache and Seigler Creeks. In 1872, Captain Rumsey established the low point of the sill as "Zero Rumsey" and all subsequent lake measurements are based on this elevation. Zero Rumsey is equivalent to 1318.256' 1929 NGVD. After a significant flood in 1938, the Riffle was excavated to an average depth of -2.3' Rumsey. Further excavation was stopped by the courts and is prohibited by the "Bemmerly Decree" in 1940.

Since 1914, the Clear Lake Dam, located approximately 3 miles downstream of the Riffle, has regulated the level of Clear Lake. Operation of the dam is controlled by two court decrees, the "Gopcevic Decree" 1920, and the "Solano Decree" 1978. The decrees attempted to balance the desire to store water in the lake for water supply and recreation and the desire to minimize flooding. The Gopcevic Decree regulates winter water levels by setting a lake stage, below which water may not be released and above which water must be released to reduce flooding. The critical lake stage is 5.50' Rumsey prior to January 8 and rises to 7.56' Rumsey on March 15. Because of the limited discharge capacity of Cache Creek, it is physically impossible to prevent the lake from flooding. Prior to construction of the dam, the highest recorded lake level was 13.66' Rumsey in 1890, with the lake level exceeding 10' Rumsey nine times between 1874 and 1914. Since 1914, the highest recorded lake level has been 11.34' Rumsey in 1986, with the lake exceeding 10' Rumsey seven times. The Solano Decree regulates summer water levels by establishing the allowable releases based on the spring water level. If the lake level equals or exceeds 7.56' Rumsey, up to 150,000 ac-ft (184,950,000 m^3) of water may be released through the dam, however, if the lake does not reach a level above 3.22' Rumsey on March 31, no water may be released. A release schedule, staged minimum lake levels, and an allowance for 3.1 feet of evaporation are also provided. Former Lake County District Attorney D. L. Luce (1972) gives a thorough account of the history and operation of the outflow.

Home (1975) and Rusk (1983) give good discussions of the physical limnology of Clear Lake. Due to its relatively modest depth, Clear Lake is usually well mixed from top to bottom, unlike deeper lakes which have a pool of colder water at depth during the summer which prevents vertical circulation. It does usually stratify during the day during summer due to solar heating of the top few meters of water, and usually circulates each night. **Figure 3.2** gives an example of this pattern. During prolonged periods of calm weather, stratification may persist for some days. Prevailing winds tend to blow down the main axis of the lake from the west and northwest. Wind conditions vary dramatically from day to day and place to place due to weather and topography. Mt. Konocti exerts an especially strong effect on winds in the Narrows and Lower Arm.

Lake temperatures fall to minima in the vicinity of 7° C in winter and rise to maxima of about 24° C in summer**. Figure 3.3** shows the average monthly pattern during the years of the DWR database record. There is substantial variation from year to year in both maxima and minima.

Numerous gas vents and water springs provide an additional source of vertical mixing (Suchanek *et al.,* 1993). The more vigorous and extensive springs en-

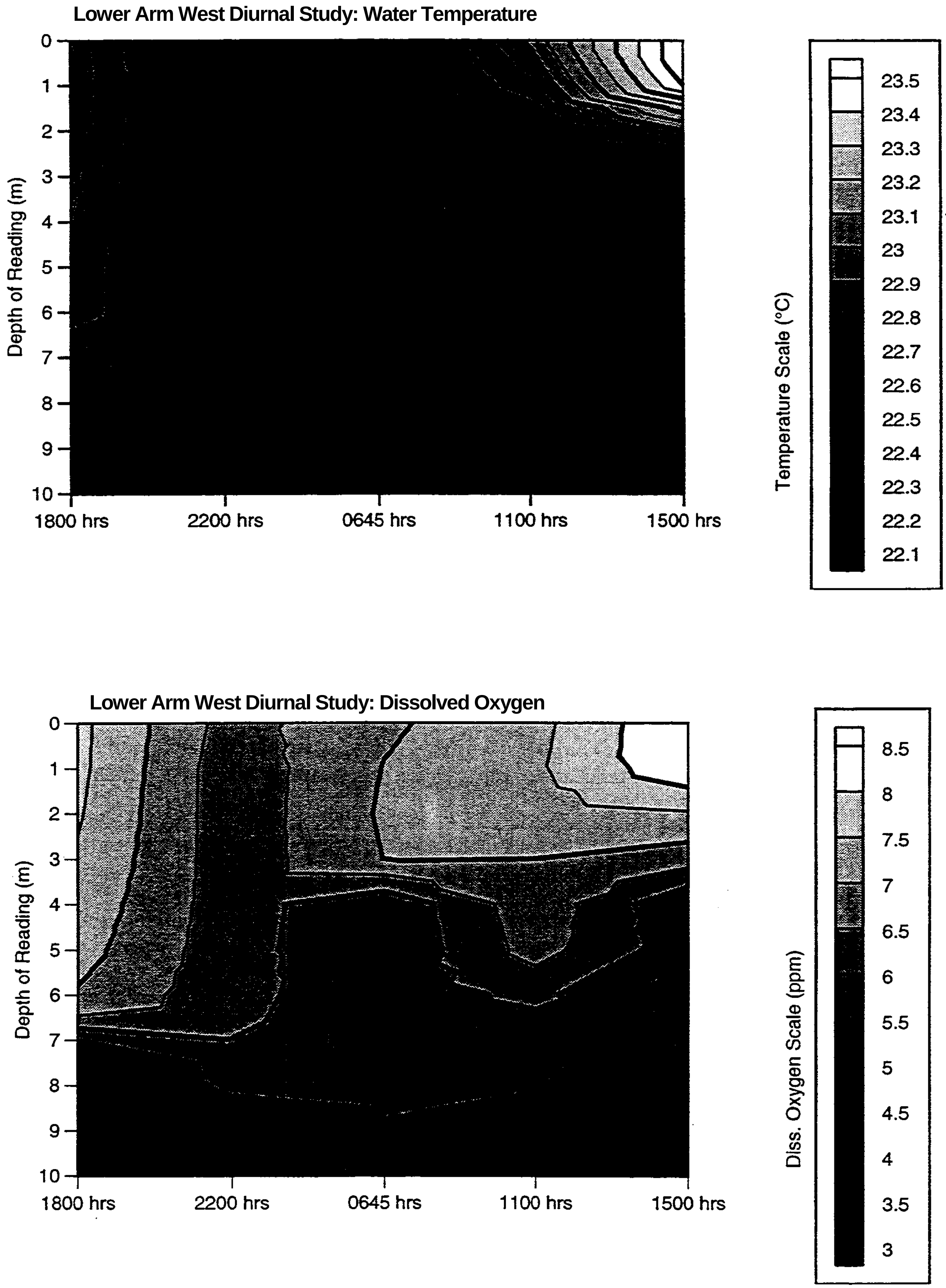


Figure 3.2 Example of the daily cycle of temperature and oxygen. Data from Lower Arm off Konocti Harbor (11-12 September, 1992).

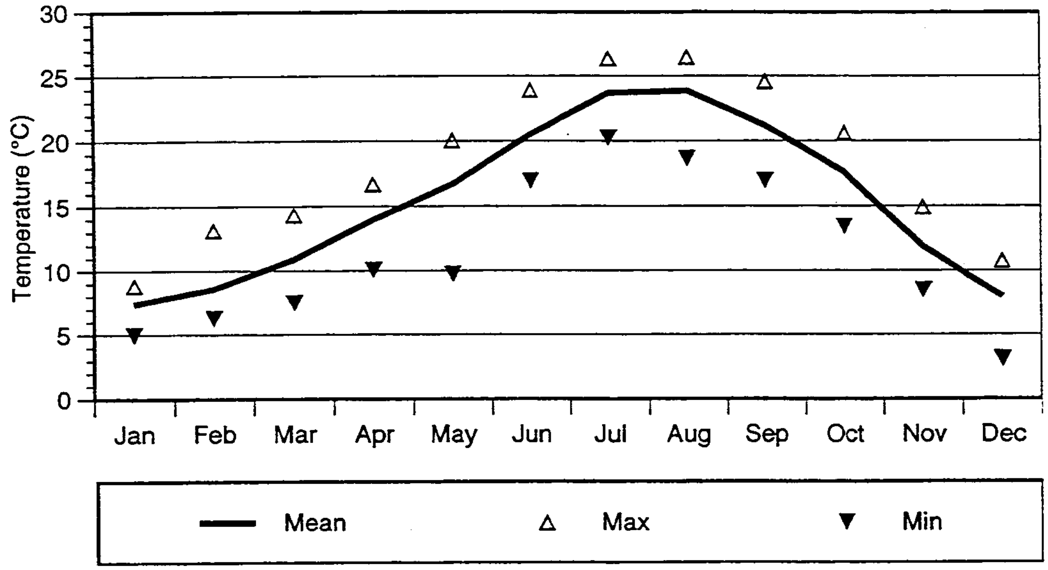


Figure 3.3. Surface temperature at the DWR Upper Arm station. The line is the 24 yr. average, and the maximum and minimum are from the pool of averages. Confidence intervals are ±95%.

train sediment and excavate pits and vents in the bottom. Some gas vents in the vicinity of the Narrows, toward Horseshoe Bend, were found to be 100' deep and 12' in diameter (Steve Why and Robert Reynolds, personal communication). The deepest parts of the lake, such as the area north of Monitor Island in the Lower Arm, have hard bottoms and appear to be maintained because a large area of gas seepage prevents sediment deposition. Limited measurements have been made on the water and gas composition of sublacustrine springs (gas is mostly CO_2, dissolved solids are dominated by calcium, magnesium, silicate and bicarbonate), but no volume measurements of water and gas flows exist (Thompson *et al.*, 1981).

Clear Lake is a moderately hard water system. Streams carry fairly heavy loads of dissolved solids, and salts are further concentrated by evaporation. Most chemical substances fluctuate markedly with cycles of drought and flood. Since the residence time of the lake (the time it would take to empty at the average annual outflow) is about 4.5 years (Chamberlin *et al.*, 1990), these variations include a modest annual cycle superimposed on longer term trends. **Figure 3.4** shows electrical conductivity (a measure of total dissolved salt concentration) from the 1969-92 DWR database record. The six year drought period of 1986-1992 is reflected in this data set which shows an increase in conductivity (higher concentration of salts) due to a high evaporation rate relative to outflow. Large flows, such as those of 1974, 1983, 1986 and 1993 measurably dilute the lake in a single winter. **Table 3.2** shows mean, maxima and minima recorded during the same period for a variety of chemical parameters for the same data set. For most of these constituents, as for conductivity, maxima coincide with sustained periods of low rainfall (mid-late 1970s, late 1980s-1992), and minima with rainier periods (late 1960s early 1970s, mid 1980s).

Clear Lake, belying its name, is currently a turbid system due to inorganic suspended particulate matter in the winter and algal blooms in the summer. Figure 3.5A shows the turbidity data from the DWR records over 24 years. **Figures 3.5B** and **3.5C** show how turbidity varies between winter and summer months, respectively. Turbidity is highest during winter runoff periods and can also be high during algal blooms at other times of the year. The best water clarity occurs during dry winters and during periods of moderate algal abundance in midsummer. There is a general trend toward lower turbidity over the last 23 years. The causes of this trend are discussed in **Chapters 4, 5** and **9**.

3.1.2 Fauna

The lake's fauna reflects aspects of the geological history of the system, and its high productivity. Due to the historical importance of the noxious Clear Lake gnat, *Chaoborus astictopus*, the benthic and zooplanktonic invertebrates that form its prey and competitors, and potential fish predators have received considerable study, especially by the Lake County Mos-

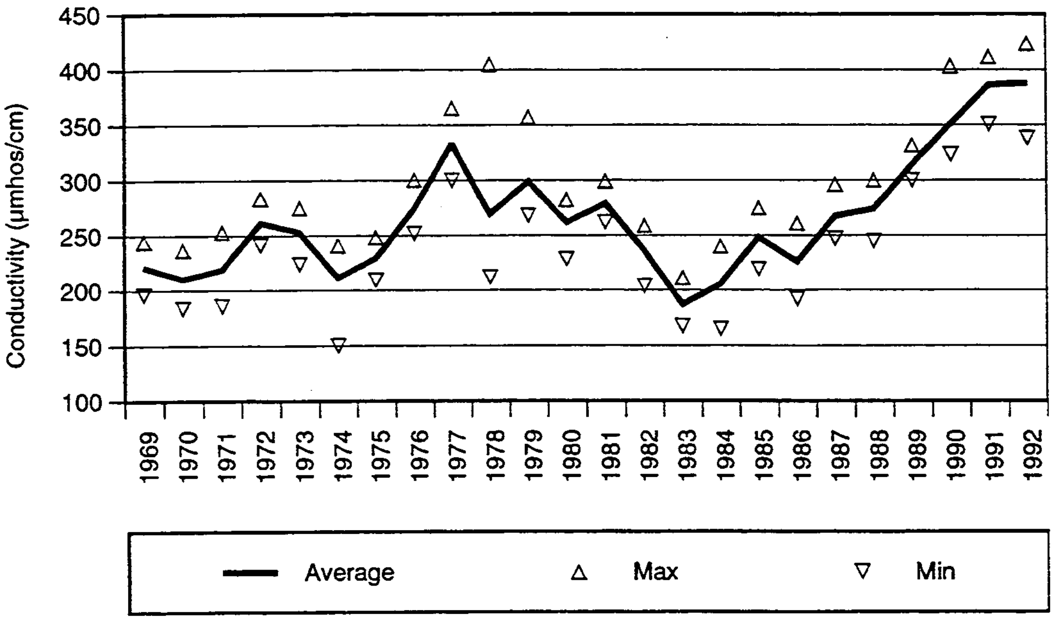


Figure 3.4. Annual values for electrical conductivity (an index of total dissolved salts) from the DWR data set for the Upper Arm. Values are averages of all collections in the year.

	pH	Alkalinity (m)	Hardness (m)	Sodium (m)	Sulphate (m)	Chloride (m)	Boron (m)
Maximum	**9.03**	**208.0**	**181.0**	**18.0**	**13.0**	**10.0**	**1.8**
Average	**7.96**	**126.3**	123.7	**11.7**	**8.5**	**6.8**	**1.0**
Minimum	**6.87**	74.3	**72.0**	**6.0**	**3.0**	**3.0**	0.3

Table 3.2. Mean, maxima and minima for selected chemical parameters in Clear Lake recorded by DWR 1969-1992, Upper Arm Station, top and bottom samples averaged.

quito Abatement District staff and their collaborators (Starrett, 1989, contains a good review).

The native fish fauna, especially rich in large minnows, was an interesting mixture of coastal forms derived from the Russian River and species from the Sacramento River System, reflecting the changing outflow (Hopkirk, 1973). The native fauna has been largely replaced by introduced warm-water fishes (Murphy 1951; Moyle 1976). Introductions of Florida strain black bass continue, and threadfin shad appeared in the lake in 1985 as a result of an accidental or unapproved introduction, and were abundant until their apparent eradication in the winter of 1989-90. The best record of fish abundances is maintained by LCMAD (A.E. Colwell, unpublished data). The trophy largemouth bass fishery has been the most important recreational fishery in recent years, but channel catfish are also important. Historically bluegill and crappie were more important. There is a commercial fishery based upon the native cyprinid black-fish, and upon carp. Week (1982) and Macedo (1991) have made important contributions to the under-standing of the large sport fishery on Clear Lake. The lake is heavily used by fish-eating birds and mammals, and over wintering waterfowl (Elbert, 1993). It has considerable value as wildlife habitat.

Clear Lake was treated with DDD in the 1940s and 1950s to control the Clear Lake gnat. DDD accumulated in the food chain and led to the reproductive failure of western grebes, becoming one of the first

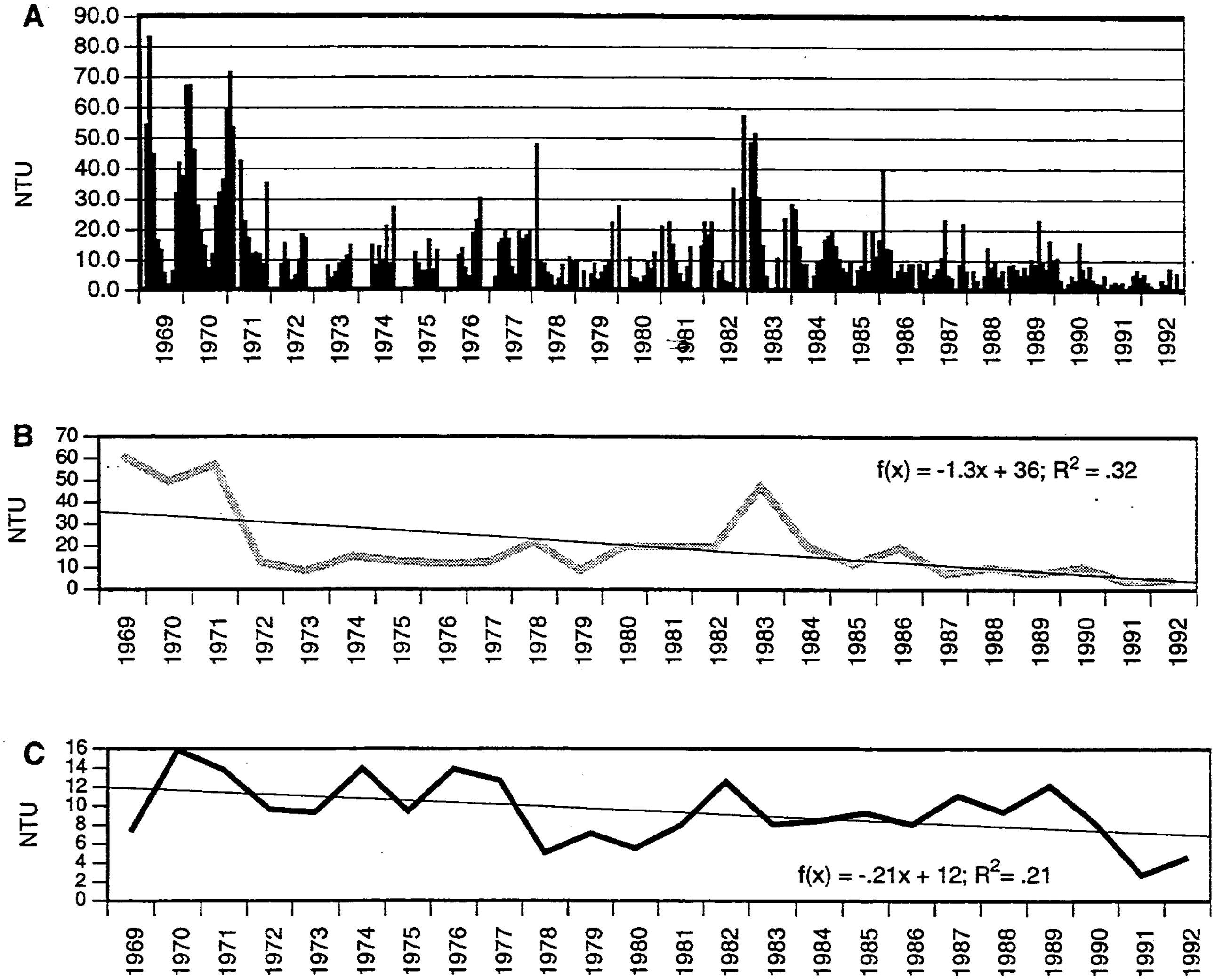


Figure 3.5. Turbidity for Upper Arm (CL1) as (A) time series of monthly averages, (B) average of winter observations and (C) average of summer observations. The other arms of the lake had similar plots. Summer months were defined as June through September, winter months as December through April. Averages were taken over the 24 years of observations provided in the DWR records.

documented examples of food chain accumulation of a chlorinated hydrocarbon pesticide (Hunt and Bischoff 1960; Carson 1962; Herman *et al.,* 1969). The lake is also contaminated with mercury from the abandoned Sulphur Bank Mercury Mine, now a USEPA SuperFund cleanup site (Chamberlin *et al.,* 1990; Suchanek *et al.,* 1993). Larger individuals of largemouth bass and other sport fish often have body burdens in excess of 0.5 ppm mercury, which has led to a health advisory on eating fish from Clear Lake. The problem stems from the large quantity of inorganic mercury stored in the sediments, mostly in the Oaks Arm. There may also be an interaction between excessive algal growth and the mercury problem. Some data indicates that heavy loads of organic matter to the sediments, as might result from the collapse of blue-green blooms, may fuel the microbial activity responsible for converting inorganic mercury to the toxic organic form (methyl mercury) that contaminates fish and other higher trophic level wildlife.

3.1.3 Food Web (Heath Carney Julia Camp)

The available published and unpublished data on the Clear Lake biota and food web have recently been compiled. This includes all known species, biomass and productivity, and feeding and interaction experiments. A total of 260 aquatic species has been found in Clear Lake. Most of these are algae (101 species) and invertebrates (94 species). The other major groups include macrophytes (23 species), microheterotrophs (8 species), and fishes (34 species). The major aquatic and nearby terrestrial species, and their feeding relations, are listed in **Table 3.3**. In this web there are 20 basal species (which have predators but no prey), 29 intermediate species (which have both predators and prey), and 10 top species (which have prey but no predators). This food web includes many benthic and wetland species as well as pelagic taxa. These species and the less edible blue-green algae are eaten by planktonic crustaceans and a broad diversity of fishes. The major amphibious/terrestrial links to these aquatic species are frogs, mink, otter, birds (*osprey*, heron, cormorants, grebes, mallards, coots), and humans.

3.2 Algal Blooms As A Management Issue

Clear Lake is subject to noxious blue-green (cyanobacteria) scums during the summer that seriously reduce residents' enjoyment of the lake and limit its value to visitors. Most residents and visitors understand that Clear Lake is a productive system and do not mind the fact that it is generally a "green" lake. Even the blue-greens, when dispersed in the water by winds, are not obnoxious. However, floating scums are quite unsightly, especially when they pack into solid mats in shallows and produce noxious odors. The water becomes completely opaque, and the scums resemble thick olive drab paint covering entire beaches and sometimes creating patches thousands of square meters in size in open water. If swimmers enter the water, the warm oily feeling material is repellent, and the colonies cling to skin and bathing suits. During bloom periods, scums of living algae appear repeatedly each morning as the buoyant forms have risen to the surface at night. As winds increase, the scums first drift about and, if winds are strong enough, are mixed back into the water to repeat the cycle the next day. If a colony remains too long at the surface, as often occurs during hot, calm weather, damage from the sun begins.

As colonies die in the sun, pigments are released, sometimes giving rise to improbable water colors, often iridescent blues. The dead (lysed) cells are white, and remain afloat for hours. Decaying scum material often acquires strong odors reminiscent of dead fish. In the most extreme situation we have observed, the notorious *Microcystis* bloom of September-October 1990, downwind areas such as Horseshoe Bend and the eastern end of Oaks Arm contained scums covering tens of acres up to 1 meter thick. Channels in the Clear Lake Keys subdivision, and a large open area of Oaks Arm outside the Keys, were too thickly clogged for small boats to navigate. Small stones would not penetrate the dried crust on top of the mat. Odors drove seasoned permanent residents from their homes for the duration (the cause of the decay odor was a very potent methyl mercaptan, R. Reynolds, personal communication).

Large blooms of blue-green algae are sometimes toxic to aquatic animals or animals that drink from impacted waters. A study by California Department of Health Services (1992) showed that some blooms in Clear Lake are mildly to somewhat toxic. A few cases of animal poisonings are known from other locations, but no confirmed cases of human deaths are reported in the toxicological literature. The DHS study concluded that the risks to public health posed by the blue-greens of Clear Lake are negligible; the problem of blooms is essentially aesthetic.

The scale of the problem at the economic level can be estimated from Hinton's (1972) surveys of tourists at Clear Lake and competing sites during 1969 and 1970. His data suggested that a total of 550,000 recreation days were lost at Clear Lake per year due to poor water quality conditions, of which the presence of algae was the most important cause. The total loss was nearly half of the actual recreation days. Put differently, by Hinton's estimate, the recreation industry at Clear Lake would have grown by nearly 50%, from 1.25 to 1.8 million visitor days, if water quality ceased to be a problem. In contemporary dollars, overnight recreators (the majority in Hinton's survey) will spend $50 or more per day. Even assuming no growth in lost days due to increased population in northern California since 1969, the problem is on the order of tens of millions of dollars of revenue lost to Lake County per year. Goldstein and Tolsdorf (1994) use Hinton's survey figures to estimate a $7 million per year loss just from lower visitation rates of Clear Lake recreators, not counting the more speculative estimates derived from responses of people who visited other lakes but might be attracted to Clear Lake if water quality were better. High algal mass also increases water treatment costs.

Long term, cost effective solutions to the scum problem require an understanding of the processes that favor the buoyant blue-greens in competition with more desirable suspended species like the greens and diatoms. The classic cause of blue-green dominance is excess phosphorus loading from sewage that makes nitrogen the limiting element, thus favoring nitrogen fixing scum formers (Cooke, et al., 1993). The classic control technique is to reduce phosphorus loading until it, rather than nitrogen, becomes limiting, depriving the blue-greens of their competitive advantage over other algae which cannot use atmospheric nitrogen. Clear Lake resembles this classic situation, with three important complications. First, iron plays an important role in limiting algal growth and atmospheric nitrogen fixation. Second,

Number	Species Name	Diet	Predator
A. ALGAE			
1	Aulacosira sp.	nutrients only	22, 36-38 (4)
2	Stephanodiscus sp.	nutrients only	22, 29-34, 36-38 (10)
3	Fragilaria sp.	nutrients only	22, 38 (2)
4	Navicula sp.	nutrients only	23, 25-34, 36-38 (14)
5	Ankistrodesmus sp.	nutrients only	22, 29-34, 38 (9)
6	Oocystis sp.	nutrients only	22, 24, 29-34, 38 (9)
7	Spirogyra sp.	nutrients only	22, 25-27, 36-38, 40, 41,44 (10)
8	Zygnema sp.	nutrients only	22, 25-27, 36-38, 40, 41,44 (10)
9	Microcystis sp.	nutrients only	22, 38, 40, 41,44 (5)
10	Anabaena circinalis	nutrients only	22, 25-28, 38, 40, 41,44 (9)
11	Aphanizomenon ovafisporum Forti	nutrients only	22, 25-28, 38, 40, 41,44 (9)
12	Aphanizomenon flos-aquae	nutrients only	22, 25-28, 38, 40, 41,44 (9)
13	Ceratium sp.	22, 23 (2)	22, 35, 38 (3)
B. MACROPHYTES			
14	Ceratophyllum demersum (Coon tail)	nutrients only	23 (1)
15	Ludwigia peploides (Water primrose)	nutrients only	23 (1)
16	Myriophyllum spicatum	nutrients only	23 (1)
17	Phragmites australis (= communis)	nutrients only	23 (1)
18	Potamogeton natens	nutrients only	23 (1)
19	Scirpus acutus (tule)	nutrients only	23 (1)
20	Scirpus californicus	nutrients only	23 (1)
21	Typha latifolia	nutrients only	23 (1)
C. MONERA, PROTISTA			
22	Planktonic bacteria/detritus	1-3, 5-13, 22 (13)	22, 24, 29-34, 38, 40, 41,44 (12)
23	Benthic bacteria/detritus	4, 14-21, 23 (10)	23-34, 37, 38, 40, 41 (16)
24	Zooflagellates	6, 22, 23 (3)	29-34, 37, 38 (8)
D. INVERTEBRATES			
25	Branchiura sowerbyi	4, 7, 8, 10-12, 23 (7)	38, 40, 43 (3)
26	Ilyodrilus frantzi	4, 7, 8, 10-12, 23 (7)	38, 40, 43 (3)
27	Potamothrix bavaricus	4, 7, 8, 10-12, 23 (7)	38, 40, 43 (3)
28	Asplanchna girodi	4, 10-12, 23 (5)	29-35 (7)
29	Bosmina longirostris	2, 4,-6, 22-24, 28 (8)	35, 43-45, 47, 48 (6)
30	Chydorus	2, 4-6, 22-24, 28 (8)	35, 43-45, 47, 48 (6)
31	Daphnia galeata mendotae	2, 4-6, 22-24, 28 (8)	35, 39, 4345, 46-48 (8)
32	Daphnia pulex	2, 4-6, 22-24, 28 (8)	35, 39, 4345, 46-48 (8)
33	Diacyclops bicuspidatus thomasi	2, 4-6, 22-24, 28 (8)	43-45, 48 (4)
34	Diaptomus franciscanus	2, 4-6, 22-24, 28 (8)	35, 43-45, 48 (5)
35	Chaoborus astictopus	13, 28-32, 34 (7)	38-40, 42, 43, 45 (6)
36	Chironomus plumosus	1, 2, 4, 7, 8 (5)	3841, 4345 (6)
37	Corisella decolor (fain. Corrixidae)	1, 2, 4, 7, 8, 23, 24 (7)	none
E. FISHES			
38	Cyprinus carp* (Carp)	1-13, 22-27, 35, 36 (21)	56 (1)
39	Gambusia affinis (Mosquitofish)	31, 32, 35, 36 (4)	56 (1)
40	Ictalurus catus (White catfish)	7-12, 22, 23, 25-27, 35, 36, 45 (14)	56 (1)
41	Ictalurus nebulosus (Brown bullhead)	7-12, 22, 23, 36, 44, 45 (11)	56 (1)
42	Ictalurus punctatus (Channel catfish)	35, 44, 45, (3)	56 (1)
43	Lavinia exilicauda (Hitch)	25-27, 29-36 (11)	56 (1)
44	Lepomis macrochirus (Bluegill)	7-12, 22, 29-34, 36, 45 (15)	41, 42, 46, 56(4)
45	Menidia beryllina (Inland silverside)	29-36 (8)	40, 41,44,46, 47 (5)
46	Micropterus salmoides (Largemouth bass)	31, 32, 44, 45 (4)	56(1)
47	Pomoxis annularis (White crappie)	29-32, 45 (5)	56(1)
48	Pomoxis nigromaculatus (Black crappie)	29-34 (6)	56(1)
49	Orthodon microlepidotus (Blackfish)		56 (1)
F.			
50	Frogs	Aquatic Invertebrates	54, 55 (2)
51	Cormorants	Fish	none
52	Herons	Fish	none
53	Osprey	Fish	none
54	Mink	50, Fish	none
55	Otter	50, Fish	none
56	Humans	Fish	none

Table 3.3 Proposed food web for Clear Lake, showing major aquatic and terrestrial species, and their feeding relations. The numbers found under the headings "Diet" and "Predator" correspond with those under the heading "Number." The values in brackets are counts for each grouping of numbers.

not all scum forming blue-greens fix nitrogen; the non-fixing *Microcystis* is sometimes a major bloom species in Clear Lake. Third, the sewage waste stream is a small source of phosphorus compared to sediment eroded from the basin. Control of the sewage waste stream is expensive but technically straightforward. Control of erosion is more complex in that many differences of detail exist from one source to the next.

The scums are formed by various genera of cyanobacteria (blue-green "algae"), typically *Microcystis* and *Anabaena* in late summer to early fall and *Aphanizomenon* in the late spring and early summer and again in fall as shown in **Figure 3.6**. The magnitude, timing, and composition of blooms varies substantially from year to year over the 24 years for which counts are available from DWR. There are some statistical regularities for the 24 years of DWR monthly means, but the plots of maxima show considerable variability. *Aphanizomenon* can sometimes develop substantial populations even in winter, but it most typically blooms in June and November. *Anabaena* and *Microcystis* have outbreaks during anypart of the warmer season of the year, but typically in late summer. Thus, there is, during midsummer, a relatively low-abundance period in July and August that interrupts the blooms, as was originally described by Horne (1975).

These groups form scums because they are composed of relatively large colonies (visible to the naked eye), and because they form internal gas vesicles that al-low them to float. In the often turbid water of Clear Lake, the buoyant strains of blue-green algae obtain extra light for photosynthesis by remaining near the surface, although they also risk being stranded at the surface and being killed by excessive exposure to sunlight.

Aphanizomenon and *Anabaena* have a competitive advantage in the phosphorus rich waters of Clear Lake because, like peas and beans but unlike most other algae, they are among the atmospheric nitrogen fixing genera. Atmospheric nitrogen fixation costs blue-green algae considerable energy, so nitrogen fixers are at a competitive disadvantage unless nitrogen is low enough to limit other algae, and other nutrients are in high supply. With respect to phosphorus, nitrogen tends to become limiting below ratios of roughly 8 units of N to 1 unit of P by weight (16:1 by atoms) (Pick and Lean, 1987). On average the dissolved ion ratio is about 1:1 in Clear Lake, but fluctuates between large excesses of nitrogen in winter to large excesses of phosphorus in summer. Under bloom conditions in the warm seasons, dissolved forms of nitrogen are often undetectable in summer, while phosphorus levels exceed 100 µg/L. (See Figure 5.3). In a typical year, blooms may be expected on this chemical basis in all but the January-April period. The atmospheric nitrogen users take advantage of ample phosphorus supplies to support their fixation of nitrogen under warm season, high phosphorus conditions, but are typically outcompeted by diatoms and other non-scum forming species under high nitrogen, cool-season conditions. Home (1975) gives evidence that the substantial wintertime populations of *Aphanizomenon occur* in years with high inorganic turbidity. In such years the water column is too dark due to clay minerals suspended in the water to support diatom growth. A floating species tolerant of cool water can compete effectively quite independently of its ability to fix nitrogen **(Chapter 4)**.

Blue-greens, especially atmospheric nitrogen users, have a higher requirement for iron than more desirable forms of algae because iron is an important cofactor for the main enzyme system responsible for nitrogen fixation (Howarth *et al.,* 1988, Brand, 1991). Blue-greens are photosynthetic bacteria, not proper algae as far as a botanist is concerned, and bacteria tend to have a higher iron requirement than green algae, diatoms, and other more desirable forms.

The main sources of excess phosphorus and iron are typically sewage and erosion. Most forms of these elements are relatively insoluble in natural waters and hence are associated with particles. Excess nitrogen is a classical ground-water pollutant of intensive agriculture because the principle inorganic forms of nitrogen are very soluble. As we will see in **Chapter** 5, the groundwater inflow into Clear Lake is negligible, and the phosphorus loading from sewage is small. Home and coworkers noted the sensitivity of Clear Lake nitrogen fixation to iron in one of the early demonstrations of this effect

Clear Lake is naturally a productive lake. It has a fairly large drainage basin to contribute mineral nutrients to its waters. Its waters are shallow and well-mixed by the wind, and there is no summertime cold layer (hypolimnion) to trap nutrients. Since important fish and wildlife populations depend ultimately upon desirable algae for food, this is just as well. Our attempts at understanding and improving the system should be aimed at achieving a shift from buoyant scum forming species to smaller species that remain suspended in the water column rather than collecting at the surface. It is impractical, undesirable and likely impossible to make Clear Lake into a transparent, alpine style lake but not impossible to substantially improve the quality of its waters.

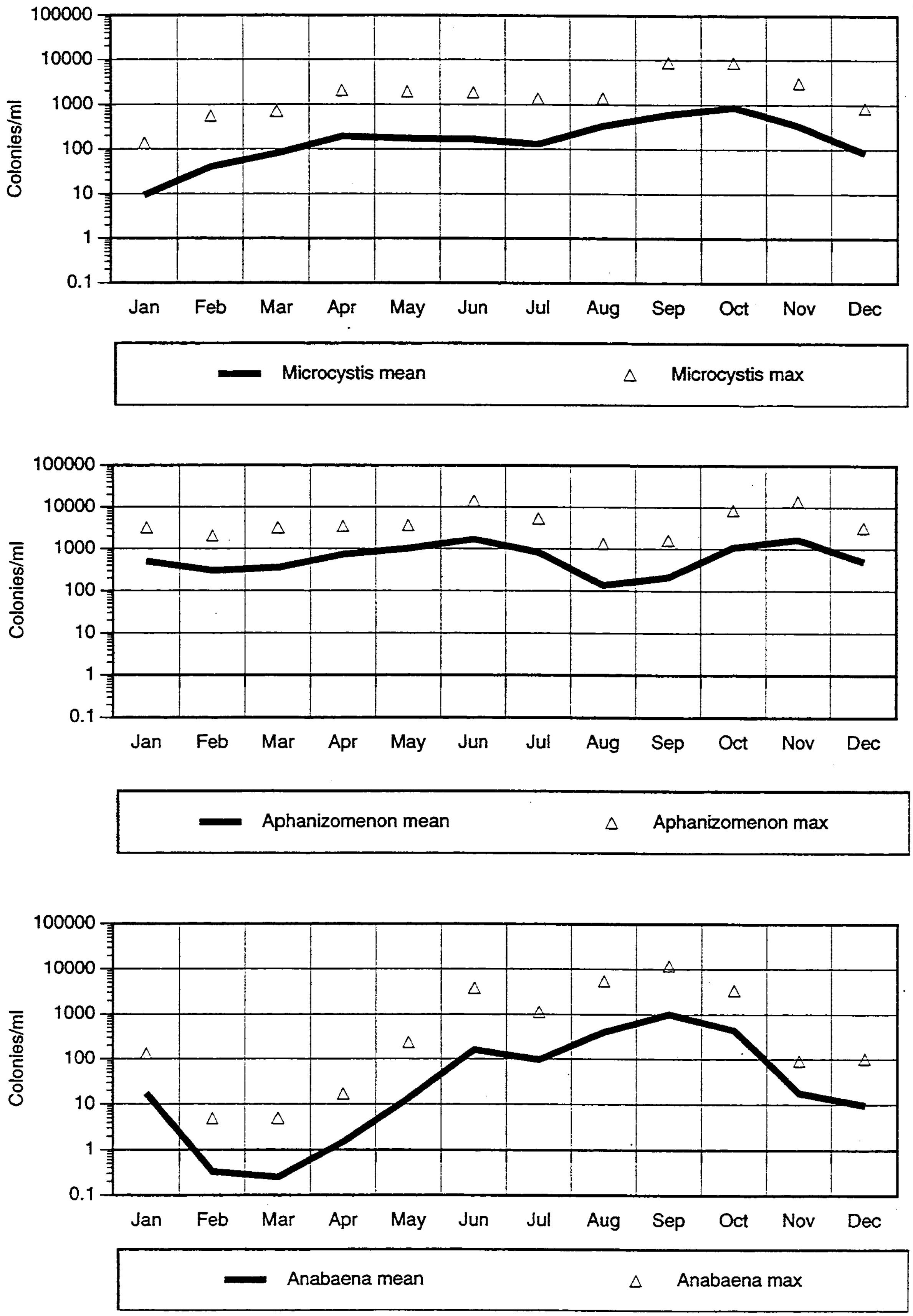


Figure 3.6. Major species of blue-green algae (cyanobacteria) and their occurrences during the year in surface samples. Based on 24 yr. DWR data set, the heavy line is the average of all data for that species, and the maximum value is the maximum of monthly averages.

Any reduction in total algal production caused by this shift will not likely have dramatic effects on fish and wildlife consumers due to the fact that large blue-green colonies are inedible. Scum formers enter the food chain only when bacterial decay reduces them to particles that zooplankton can eat, an inefficient process compared to direct consumption of smaller algae. Thus, improvements in water quality for swimming, boating, esthetics, and potable water can be achieved with minimal risk to the fish and wildlife productivity.

Nonetheless, it is impossible to reduce the quantity of scum forming algae without causing other changes. Nutrient reduction may lead to clearer water, and hence the growth of rooted waterweeds in shallow zones. Waterweeds are excellent fish habitat, although weeds seriously interfere with navigation and enjoyment of beaches, and can make fish harder to catch (Knight, 1993). Weeds, on the other hand, are technically more easily and cost effectively managed by harvesting. Some changes in the relative abundance of fish species could occur. Under blue-green algal reduction the supply of mercury to fish might also be reduced as the mass of algae decaying near the bottom is reduced. The abundance of noxious aquatic insects could be affected. Some of these effects will be the result of complex chains of cause and effect, and are hence difficult to predict.

3.3 Review Of Previous Studies

3.3.1 Historical Observations Of Water Quality

Was Clear Lake ever clear? This is an important question for two reasons. First, the natural background conditions are likely to define the opportunities and limits of any practical effort to reduce scum-forming blooms. Second, many people feel that humans are responsible for cleaning up their messes, but not so responsible for correcting the "faults" of nature. For some, but not all, citizens and decision-makers blue-green scums are more tolerable if they are a natural part of the Clear Lake ecosystem, and more objectionable if they are caused by human disturbances.

Paleoecological evidence from the USGS coring project indicate that Clear Lake has been productive throughout its history (Bradbury, 1988). Since warming began after the last glacial cold period (beginning about 15,000 years ago), the lake has had a diatom flora dominated by the same species of *Melosira* and *Stephanodiscus* that dominate the current diatom community. Unfortunately, blue-green algae do not leave as obvious remains in the sediment, and no attempt was made by the USGS group to infer their abundance directly. Bradbury does suggest, by analogy to the diatoms, that blue-green blooms have also been common for the past 15,000 years but it is impossible to infer directly the levels of productivity in prehistoric times.

Direct observational data on algae are quite limited but do not support the hypothesis that scum forming blue-greens have always been a major component of the algae. Rather they strongly indicate that Clear Lake was considerably clearer up to the 1920s than in recent decades.

The first observations of Clear Lake water quality that we have discovered were made by fish biologist Livingston Stone (1874) based on field work in 1872-73. Stone made the following comments on water quality as part of his efforts to establish whitefish in the lake:

> "It is a singular fact, illustrating the inaptness with which names are often given to natural objects, that the water of Clear Lake is never dear. It is so cloudy, to use a mild word, that you cannot see three feet below the surface. The color of the water is a yellowish brown, varying indefinitely with the varying light. The water has an earthy taste, like swamp-water, and is suggestive of moss and water-plants. In fact, the bottom of the lake, except in deep places, is covered with a deep, dense moss, which sometimes rises to the surface, and often to such an extent in summer as to seriously obstruct the passage of boats through the water."

Stone also mentions a reddish-brown froth produced by the evaporation of soda-water, plants and moss from the bottom floating on the lake, and a green scum in September and October.

Stone's description is frustrating because of his use of imprecise terms. The key issue is interpreting his term "moss". Aquatic mosses in a strict botanical sense exist, but are unlikely to have been important in Clear Lake. If "moss" refers to blue-green algae, Stone's description would tend to fit today's lake. However, his description of the bottom being covered by "moss" to considerable depths strongly suggests waterweeds such as *Potamogeton* or possibly the large weed-like bottom algae, *Nitella* or *Chara*. Blue-greens never carpet soft bottoms, although they may be a component of the growth on rocky bottoms. The algal community growing attached to rocks might also be referred to as "moss". All forms of attached bottom growth have been scarce in Clear Lake in the past few decades because of low water transparency, and "moss" growth "except in deep places" suggests reasonably clear water by modem standards. In recent decades, only the unusually clear years of 1991 and 1992 have had conditions that resemble this description. If the remark "cannot see three feet below the surface" is interpreted as a bloom condition (Secchi disk minimum of roughly 1 meter), average water clarity would seem to have been sufficient to support considerable bottom growth. Water routinely less transparent than a meter, as in recent decades, is not consistent with substantial bottom growth.

His green scum in September and October is reasonably interpreted as blue-greens, but he does not attribute the main problems with water quality to this phenomenon. The description of the water color as "yellowish brown" perhaps reflects blooms of diatoms or cryptomonads, rather than blue-greens, during most of the year (or possibly suspended inorganic sediment in winter). Diatoms and cryptomonads are typical of clear winter water due to low runoff in re-cent decades. They do give the water a brownish cast, as opposed to blue-greens which are usually quite green. An exception is the brownish blue-green *Gleotrichia* which was abundant during the dear 1992 year and again in 1993 and 1994.

Stone was also impressed with the high fish productivity of the system, and gives a list including trout and squawfish. Trout require relatively high-quality water and both these fish are sight-feeding predators. Larry Week (personal communication) believes that squawfish require more transparent water than is common in recent years to feed successfully.

Despite his stout terms of disparagement, Stone's description would seem to best fit 1991-92 in terms of water quality in recent years. However, a water-weed dominated system was not a product of prolonged drought. There is no rainfall data from the Clear Lake area during Stone's observational period. In Sacramento, San Francisco, and Woodland, rainfall in 1872-73 was somewhat below normal. In general, the rainfall in the 1860s and 1870s was high relative to longer run averages (Donley, *et al.*, 1979.) In the 1870s Clear Lake would seem to have been a productive, eutrophic lake, but with scum forming blue-greens a relatively minor component of the algal biomass compared to most years of recent decades.

Farming, animal husbandry, mining, lumbering and tourism had been substantially developed by the 1870s (Simoons, 1952). So, it is conceivable that Clear Lake's water quality had already declined from aboriginal conditions by the 1870s. Unfortunately, we have so far discovered no useful descriptions of the lake from the 1840s or 1850s, and cannot establish any sort of true aboriginal baseline condition.

Other, less detailed, 19th Century descriptions support the hypothesis that Clear Lake was once reasonably transparent and free of major blue-green scums. Pioneer histories of Lake County were written by Menefee (1873) and Palmer (1881). Menefee (1873:224) states that Clear Lake "derives its name from the clarity of its waters." He gives a charming portrayal of Soda Bay, describing the bathing as "fine and exhilarating (sic)."

Palmer's comments on the condition of the lake are slightly more extensive. He attributes the name "Clear Lake" to the "remarkable purity of the surrounding atmosphere", which perhaps can be interpreted as an indirect reference to the water being turbid (p. 92), although elsewhere it is described as "as pure as a crystal," " and "affording fine bathing" (p. 7). There are long passages of praise of the lake's beauty (e.g. p. 165). The problem of the Clear Lake Gnat is forthrightly described (p. 166), suggesting that if noxious scums did occur regularly, they would not have been concealed in the interests of diplomacy.

Fishery biologist George A. Coleman (1930) conducted a basic biological survey of Clear Lake in the first four months of 1925. He also mentions summer and fall visits. The 1924-25 winter was one of average rainfall (29.37 inches at Lakeport- Martin, 1930) after three seasons of drought. In general the early part of the 20th Century was perhaps 10% drier in California than was the last part of the 19th (Donley *et at.*, 1979). Coleman gives a list of phytoplankton genera which is dominated by diatoms, especially S *tephanodiscus* (species not mentioned). *Stephanodiscus* and other diatoms have been most abundant in the recent record during the clear-water drought years, when low sediment yield allowed them to flourish during the cool season (See Chapter 4). Their dominance during a high runoff year is significant. The blue-green *Oscillatoria is* noted as "some-times in open water", but the common winter-spring scum former *Aphanizomenon* of the more recent period was not noted. He reports the large attached algae *Nitella* living on the bottom. In recent decades, high runoff winters have had few diatoms, and the buoyant *Aphanizomenon* has persisted throughout the winter due to its ability to obtain adequate light under turbid conditions as noted above. It is thus no-table that Coleman describes a diatom dominated flora with *Aphanizomenon* absent. Coleman's observations are consistent with the 19th Century reports of relatively clear water. There is every indication that the water was moderately transparent, and that the blue-green scums were not a notable part of the system in 1925.

In the mid 1960s Lake County historian Marion Goeble collected comments on the history of water quality changes from longtime residents of the lake shore. These comments reflect changes during the first half of the 20th Century. The portions most relevant to the algae problem are reproduced in full here. The comments of John Jago, a 60 year old re-tired resort owner, are lengthy and informative (interviewed circa 1967):

> It is very difficult to compare Lake County today with any given time into the past -10, 20, or 30 years ago- but it seems to me that in general, Clear Lake clarity and overall water quality have gradually deteriorated over the 60 odd years that I have lived on or close to its shores.
>
> I have always enjoyed the lake and have spent much time in or on the lake from the time I was a small child; and all my life, even as a child, I have been very observant of the lake, partly because of a scientific turn of mind. As an adult, I have owned property and conducted a business on the lakeshore; and as a member of the Mosquito Abatement District Board of Directors for the past 15 years, I have been constantly aware of water quality problems. My memories are mostly of the lower lake areas as I have lived there most of my life. While most of these memories are very pleasant and I plan to continue living here through my retirement years, still there have always been a few unfavorable conditions -which are the subject of this study.
>
> CLARITY AND WATER QUALITY: To my mind, a gradual decline of water quality in Clear Lake is due principally to the constantly increasing load of nutrients entering the lake each year. For instance, it does not seem to me that Clear Lake was as [A] muddy after heavy rains in years past as it is now under the same conditions. This is probably due to 1) more cultivation of surrounding land, 2) much heavier grazing of range land, and 3) the denuding by forest fires of large areas of brush and grassland which conditions furnish the lake with a yearly crop of fresh soil and other organic material during the rainy season. In the more shallow areas quite a large amount of mud is kept in suspension by the action of waves and boats. In the deeper areas, after the mud and silt brought into the lake by the rains have settled, water clarity is dependent almost entirely on the amount of [B] algae growth.
>
> The [C] organic material picked up by rain water running through thousands of tons of partly-burned and decaying garbage in our garbage dumps, as well as the [D] seepage into streams and into the lake from sewage disposal systems of all kinds most probably affects the water quality adversely. And perhaps most important of all are the nutrients washing into the lake from the hundreds of tons of [E] commercial fertilizers applied to the land.
>
> I have observed that the lake water has better quality during the dry season when there have been sufficient rains the winter before to cause a considerable overflow down Cache Creek. This water coming into the lake during the winter contains less nutriment than the lake water and therefore any overflow helps to dilute the lake water and lower the nutriment level.
>
> A well-known example of what can happen to Clear Lake when there is very little rainfall and no natural flow-through is the situation during World War I. I observed that the water in the Lower Lake was literally black in color at that time, and weed growth was very abundant.
>
> VEGETATION, ETC.: During my childhood and teen years, I remember that many sheltered coves had extensive beds of a very common type of water lily and several other lake weeds. When-ever the lake had been low for several years at a time, the weed beds would become very dense and extend quite far out into the lake. At Lakeshore Beach (now Redbud Park beach) in Clearlake Highlands the weeds extended out for at least a hundred yards and were too dense to row a boat through. This condition has existed several times in recent years also. [I] noticed it in Soda Bay and in other coves along the southwest shore, but I understand that it was prevalent all around the lake.
>
> ALGAE Re: LAKE COUNTY CHAMBER OF COMMERCE WATER QUALITY SURVEY. As a small boy 50 to 60 years ago, I remember that the lake was not really clear, but we could always see down into it for three feet or so. There was al-ways some algae present, in suspension, but as a child I do not remember seeing floating algae or bands of [algal] stain along the shoreline. Algae had become a problem at least by the early 30s, perhaps earlier.
>
> It is accurate to note that algae growth is not consistent each year. During each summer there are several days in a row when we can see down into the lake from our wharf in Jago Bay, as far as 6 to 8 feet. Some summers, this clarity is more frequent than others, particularly if there has been enough

> rain for diluting action and flow-through, the winter before. This past 1967 season is a case in point, when the circulator which we have installed in front of our beach to keep the swimming area free of floating algae did not have to be turned on all season until after Labor Day.
>
> After years of observing and studying the pattern of algae growth, I realized that there are additional factors necessary besides the lack of sufficient rainfall for dilution, flow-through, and a good lake level. One of these important factors is wind. For several years I have noted that algae bloom is always at its worst for several days after a strong wind storm. The wind not only whips nutrients up from the lake bottom by wave action; it also whips air into the water to supply oxygen for algae bloom. I have noted also that after such a crop of algae dies and settles to the lake bottom, we get the clearest water along our pier. There will be no further heavy growth until another wind. I have mentioned this wind factor to other people studying our water quality problems and they have agreed with my conclusions after making observations of their own.
>
> Summer heat is not an essential condition for algae bloom, as many people suppose. I have colored slides made 15 years ago of very heavy algae carpets in late fall, and also in January and February.

Historian Geobel also obtained useful information from Frank A. Sewell, fishing editor, and resident of Lower Lake for 37 years (interviewed circa 1965):

> Clarity of Clear Lake was better a number of years ago, than at the present time, except during years of extreme low water. There were occasionally eruptions in the lake bottom that forced some kind of brick red mineral to the surface which remained in suspension and spread to most of the east end of the lake.
>
> There was more vegetation in the lake 12 to 20 years ago than at the present time. At that time there was a water plant that rooted in 2 to 6 feet of water, sent a stem to the surface where it branched out covering and area of approximately 2 feet. About 12 years ago the condition was so bad that resorts in the area had to hire swimmers to dive down and cut a swath 4 to 6 feet wide from shore to as far as 600 feet out into the lake in order for their guests to get to open water for fishing and boating. One resort owner Bob Todd of Ferndale Resort on Soda Bay built a flat bottom barge with a power cutter of his own make to clear a portion of his waterfront.
>
> In the 37 [roughly since 1928] years I've owned lakefront property, I have seen heavy concentrations of algae during all of that time. However in those days algae only seemed to appear in the summer time and particularly when the weather was hot and humid. In those days there was what we might term dean algae where as now we have nearly as much in the cold months as any other time of the year and the more noxious, turquoise blue floating scum which I have been told is dead algae, the odor of which is nauseating. The ordinary live algae disappears when the water becomes rough and leaves no sign of it on lake bottom or shoreline where as the so called dead algae leaves every object at the water line covered with a foul smelling moss.

Historian Geobel also furnished us with a statement given by Superior Court Judge Benjamin Jones regarding Water Quality Study of Clear Lake for the Federal Water Pollution Control Administration, December 27, 1967. Two key paragraphs are quoted below:

> Before the impounding dam was built at the outlet, there was no stagnant water in Clear Lake. From the time the lake was first known to the earliest settlers up to the building of the dam the water was perfectly clean, Hence, the name "Clear Lake."
>
> Since the impounding dam has been in operation all of the inflow into the lake, together with its impurities is held back until the maximum storage elevation is reached, and only then is the lake drained out from the bottom by opening the gates of the dam by raising them. Any impurities not on the bottom are effectively held back by the screening effect of the upper part of the dam. In other words the current created by the releases is on the bottom and not on the surface of the lake as under natural conditions. The bulk of the pollutants are held back even in those years when the maximum storage capacity is reached. The process is cumulative. Hence the algae, mosquitoes, gnats, dead fish and what have you.

This statement is less useful in speculating about causes rather than citing direct observations, although the implication that water quality declined in the 20th Century is plain. The argument that the operation of the dam has a deleterious effect on water quality is questionable for three reasons: (1) The impoundment operated from 1914 onwards, and the information of Coleman, Jago, and Sewell suggests that algal blooms did not become a serious problem until after 1925. (2) During winter, phosphorus and iron are stored in sediments, and are recycled to the water column in summer. Thus, shifting water outflow from winter to summer as a result of storage for irrigation should have a beneficial effect on water quality. The effect, in principle, is fairly large (see discussion in **Section 9.5).** (3) The dam is 3 to 5 miles from the outlet of Clear Lake, and outflowing water travels down the shallow channel over the Grigsby Riffle. The dam can have very little effect on the depth from which outflowing water is drawn.

The most definitive element in the observations of Mr. Jago and Mr. Sewell is the presence of rooted aquatic vegetation to considerable depths. These remembrances are backed up by photos taken near Lakeport early in the 20th Century, and conform to the descriptions of Livingston and Coleman. The strong competition between algae and rooted vegetation for light means that abundant rooted vegetation is a conspicuous indicator of clearer water, and a relative lack of blue-greens.

Other observations by Mr. Jago regarding high concentrations of algal scums in the Lower Arm after strong winds is are consistent with our findings and implies that floating noxious scum formers were typically blown down into the Lower Arm region and concentrated by prolonged wind activity, although his explanation for this phenomenon is not feasible. See further discussion in Section 3.3.3.

Lindquist and Deonier (1943) made limited observations of water quality in the course of their investigations of the Clear Lake gnat. They made monthly measurements of Secchi depth from January 1939 until November 1941 in the Upper Arm. Although the 1938-39 winter was very dry, and Secchi depths fluctuated irregularly between 1 and 2 feet, the following two winters were wet, and minimum depths during the winter-spring runoff period were about 0.5 ft. On the other hand, in both 1940 and 1941 brief summer clear water periods occurred with transparency reaching 6 feet. These data are very similar to those of the past 23 years, and are consistent with the observations of Mr. Jago.

Garth Murphy (1951) reviewed the history of the Clear Lake fishery, in part based on an extensive examination of early 20th Century newspaper accounts. He also conducted a sustained program of field studies on fish populations in the lake from 1946 until 1950. His observations of water transparency indicate substantial deterioration since Coleman's observations, consistent with Lindquist and Deonier's Secchi measurements around 1940:

> Generally, the clarity of the water of Clear Lake is low. At times the waters of the entire lake are laden with silt borne in by heavy discharges from the tributary streams, particularly in winter and early spring. During the summer the upper shallow end of the lake often becomes turbid, when bottom deposits are disturbed during periods of high Winds. There are large quantifies of phyto- and zooplankton in the water, so that at best visibility usually is limited to a foot or two, even when inorganic turbidity is low.
>
> Early settlers were impressed by the clarity of the waters of Clear Lake. It is unbelievable that they were viewing the lake as it is today.

Murphy attributes the shift to increased nutrient inflow due to sewage, fertilizer use, and general erosion caused by overgrazing, fire, and agriculture.

Murphy also describes the dramatic changes in the fish community, partly as a result of introductions. Trout apparently ceased to be a major part of the sport fish catch in the early 20th Century. The major part of that fishery was steelhead, whose migrations were interrupted by the construction of Cache Creek Dam. Other changes occurred long after the closure of the dam and the major introductions. As late as 1938, splittail and squawfish were abundant in the catches of A.W. Lindquist, whereas they were absent in Murphy's catches in 1946-47. Murphy states that a precipitous decline began after 1942 or 1943, perhaps tied to increased irrigation demand in the summer causing early drying of their spawning streams. It is practically certain that modern irrigation pumping does dry up streams earlier than was the case historically, but accurate data are not available. Stream channel disturbances of other kinds might have played a role. We have discovered no direct evidence tying particular fish population declines to the development of blue-green scums or other water quality changes. However, the loss of stream-spawning fishes, do indicate ongoing major changes in the system at a relatively late date, coinciding roughly with the period over which Jago recalls declining water quality.

Thus, the historical evidence does not support the oft-heard opinion that large blooms of scum formers have always been a natural part of the Clear Lake ecosystem. Rather, it suggests a substantial decline in transparency and increase in blooms of scum formers sometime between 1925 and 1939.

We have used the Mauldin and Geobel archives and other sources to attempt to associate events in the Basin with the hypothesized decline in water quality between these dates. Two unpublished M.A. theses also provide fairly detailed historical accounts. Figure 3.7 is a comprehensive time line of events potentially affecting water quality at Clear Lake.

There are two classic sources of excess phosphorus due to human activities, sewage and soil-erosion products (Goldman and Horne, 1983; Cooke *et al.,* 1993). Both tend to be high in phosphorus relative to nitrogen, and thus to contribute to the sort of nitrogen fixing scum problem that occurs in Clear Lake. (Pollution from groundwater and farm runoff high in nitrogen relative to phosphorus causes a different kind of eutrophication problem and is not further considered here.) As Horne (1975) noted, and our calculations below indicate still hold true (Chapter 5), the sewage waste stream, either that treated via septic systems or that captured by modern treatment plants, is and must always have been a relatively small part of the problem. Erosion is clearly the most important source of excess phosphorus in Clear Lake. What human activities might have generated that excess, and especially what processes might have become important beginning about 1925?

In the Clear Lake Basin intensive development of orchards, vineyards, extensive grazing, and other agricultural activities date far back into the 19th Century. For example, cattle numbers have fluctuated in the vicinity of 10,000 animals in the county, and sheep production peaked at about 50,000 head in the 1880s (Simoons, 1952). Although evolutionary changes certainly occurred in the 1930s, we have been unable to identify any crop trends that would have had a sharp impact in this period. Pear acreage, for example, was rising rapidly in the 1920s to 6,500 acres in 1932, but then fell back to around 4,000 acres for many years, the same as the 1925 level.

The reclamation of Tule Lake for agriculture was complete by 1909, and the reclamation work around Rodman Slough (Robinson Lake) commenced in the 1920s. Flood control improvements were also implemented on Middle and Clover Creeks, including a major channelization project in 1958. Preliminary data suggests that Tule Lake acts as a quite significant settling basin for phosphorus (see Chapter 5). The Middle/Clover Creek system is currently a major source of phosphorus influx into the basin, and probably once deposited a significant amount of its fine-sediment load in the Robinson Lake flood plain and marsh system that existed before reclamation and flood control projects altered the system.

A very significant change in the mid 1920s to 1940 appears to be the advent of powered earthmoving machinery. As the cost of moving dirt, sand, and gravel decreased, disturbances to land, stream channels and lake shore increased dramatically without any modern regulations or erosion controls. It is useful to have in mind a rough figure for how much material one needs to account for to explain a major decline in water quality **(Chapter 5)**. Presently the average annual phosphorus influx into the lake due to human activities is about 75 metric tons. The modern suspended sediment load is about 0.1% phosphorus. Thus, any process capable of making a significant contribution of fine sediment to the lake's waters of about 75,000 tons per annum would help ex-plain a deterioration of lake water quality.

Major road improvement projects were initiated in this period. Highway 20, which follows the shore of the lake for several miles along Oaks Arm, was constructed in the mid 1920s (Simoons, 1952). Fill projects to create dry land and a more usable shoreline probably increased during this period and remained significant for many years (Fredericks, 1969). Rodman Slough was apparently dredged regularly in the 1945 era to build up levees which failed or subsided (Earl Pruitt, personal communication).

The Sulphur Bank Mercury Mine was an open pit operation from 1927 until 1944. Chamberlin *et al.* (1990) were able to document that most of the tailings piles in contact with the lake were emplaced during the 1927-44 period of mining. According to this report, as much as 700,000 m^3 of this material may have eroded into the lake by the date of their study. The phosphorus content of the ore and overburden rock dumped at the mine is not known, but the total annual erosion loss of ca. 50,000 tons per annum is on the right scale to be important

The effects of shoreline dredging and filling are difficult to estimate. Large projects, such as the digging of the Keys subdivision with its 6.5 miles of channels, and the extensive cutting of Windflower Point to create house-pads tend to be late in time, the 1960s. A multitude of smaller dredge-and-fill projects occurred all during the period after 1925. Such operations tended to deliver phosphorus and other nutrients rather effectively to the lake due to proximity and deposition of materials in the shoreline zone where wave action quickly eroded exposed fines into the lake.

In 1920s and 1930s gravel mining moved from scattered barrow pits to centralized operations on the creeks. Zalusky (1992) gives a history of the aggregate mining industry. Unfortunately, no statistics were kept on the details of this movement, or upon production figures until 1965. Current operators believe that early mining in creek channels lacked any control of sediment loss from wash water. However, deposits are deliberately sought for mining which contain only a few percent fines, as handling and washing volumes of material is costly. The potential for such operations to disturb large quantities of very nutrient rich sediment and cause additional erosion losses due to flow **(see Chapter 8)** above and below the operation itself makes them potentially a dominant source of the increased nutrient inflow aside from direct production of fines. Any fines mobilized due to disturbances of creek channels will be moved very efficiently to the lake by high flows by comparison with disturbances off channels.

In the absence of precise studies it is impossible to estimate the quantitative impact of the aggregate industry. We do note that since 1965 the industry extracted about 1 million metric tons of product per annum until the partial moratorium on aggregate

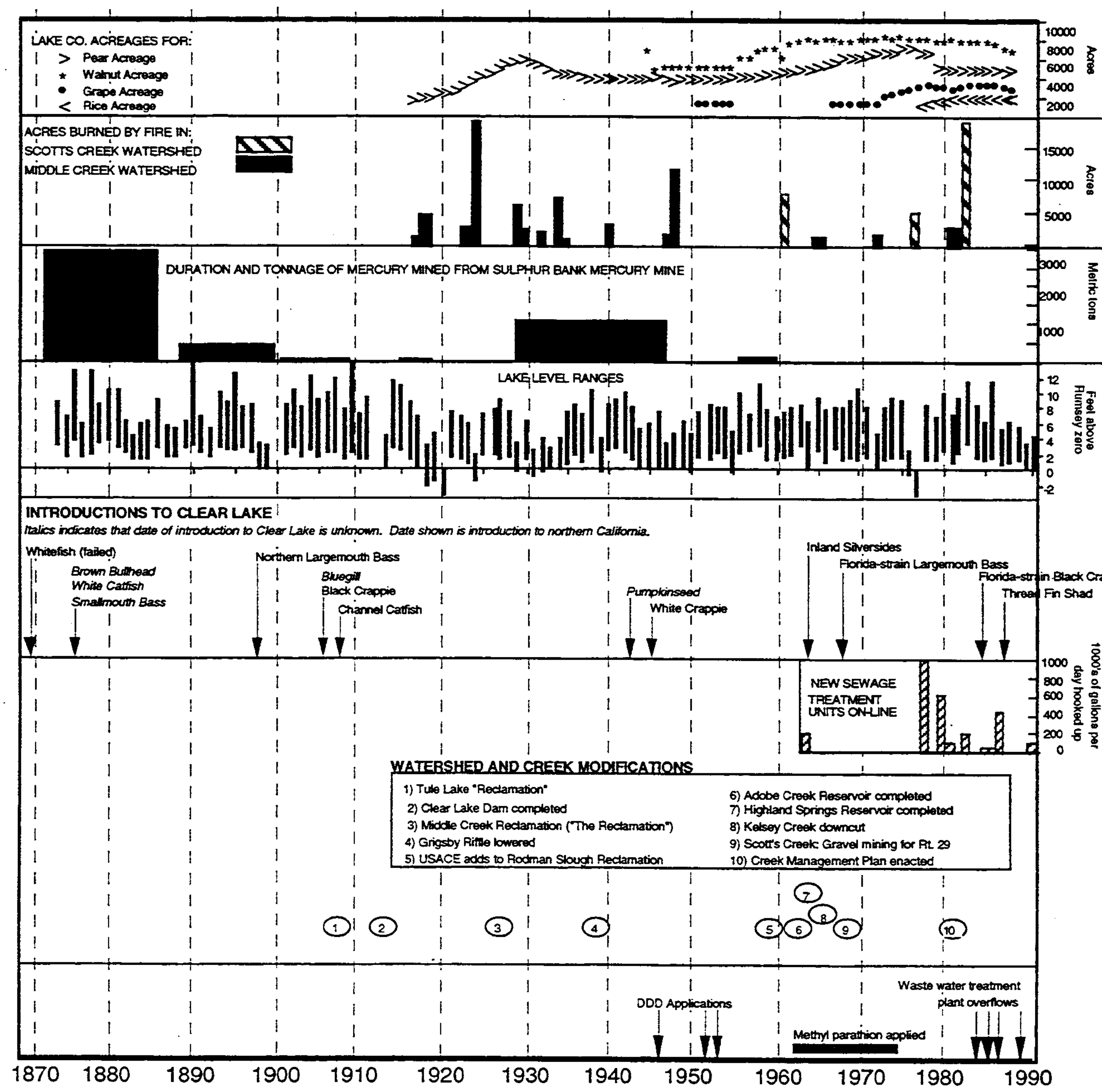


Figure 3.7. Time line showing major events relevant or possibly relevant to changes in lake water quality.

mining in stream channels in 1981, which could conceivably account for fines production on the required 75,000 ton scale.

It is not possible in the scope of this study to provide a true quantitative estimate of the contribution of each of these activities to excess nutrient loads to Clear Lake. Nor, in fact, is it possible to prove beyond doubt that increased land and stream channel disturbance is tightly tied to the change that seems to have occurred between 1925 and 1940. The qualitative facts are nonetheless important preliminary evidence that the scale of these activities, separately or in some combination, is potentially sufficient to explain a decline of water quality after 1925 and continuing on to the present.

33.2 Modern Investigations Of Algal Ecology

The scientific study of Clear Lake's algae began with Charles R. Goldman and Robert Wetzel's (1963) work on algal primary production (phytoplankton photosynthesis) in 1959-1960. They measured primary production with the radiocarbon method approximately monthly for 15 months. There was a moderate spring bloom in 1960, and large mid to late summer blooms in both 1959 and 1960. The measurements demonstrated that the lake was a highly eutrophic system.

Goldman and Wetzel conducted experiments to determine which nutrients limited algal growth by adding various nutrients and mixtures of nutrients to flasks of lake water and tracking changes in growth rate over 5 days. The results indicated that phytoplankton were limited by nitrogen but not phosphorus in summer. *Aphanizomenon* was noted as the bloom species responsible for the intense fall 1959 bloom. They observed very low transparency due to erosion products carried by streams in winter, and to algae in summer. The Secchi disk depth was generally less than 1 m, except for a relatively clear period in June and July 1960, when the Secchi depth reached 2 meters, much as in the last 24 years of the DWR record. Goldman and Wetzel felt that the productivity of Clear Lake had "undoubtedly increased during the present century to its present state of extreme eutrophication," but were unable to give detailed evidence to document changes or their causes. They inferred that the cause of the highly eutrophic state of the lake was some combination of natural drainage, poor land use, agricultural fertilization and sewage.

In 1964 and 1965 the California Department of Water Resources initiated a water quality investigation of Clear Lake (Lallatin, 1966). This report provided a comprehensive review of the knowledge of the applied limnology of Clear Lake as it existed at that time. A total of 31 sampling stations were established, and of these five in the Upper Arm, one in the Oaks Arm and three in the Lower Arm were sampled from one to three times per month from May 1964 to May 1965 for a large number of physical, chemical and biological parameters.

The state of the lake in these years was similar to that a few years later when the continuous DWR record began. Secchi disk depths were very low, read ng a fraction of a meter during runoff events, but with summer clear water peaks up to 2 meters. Phosphate levels were very high, especially relative to nitrogen, in summer. No quantitative data on algae were collected, but in cooperation with Lake County Mosquito Abatement District (LCMAD), basic identifications were made. *Aphanizomenon, Anabaena,* and *Anacystis* (similar to *Microcystis)* were noted. The report recommended authorizing the LCMAD to undertake algae abatement via state legislative action, that the county develop a long range sewage treatment and disposal plan, and that routing Eel River water through Clear Lake be explored. A good deal of work was subsequently done on the Eel River scheme (Kaiser Engineers, 1968; DWR, 1972).

Porcella *et al.* (1970) used microcosm experiments to determine how phosphorus was made available from sediments of different types to algae in the overlying water. The samples tested included two from Clear Lake. One of the important results of this study was that phosphorus extraction from Clear Lake (and other) sediments was highly efficient even when the overlying water was aerated; anaerobic microcosms were only slightly more productive of phosphorus. Phosphorus is thought to be mobilized under anaerobic conditions, but this does not appear to mean that dissolved phosphorus cannot exit sediments that have an oxidized top layer. The results of our work described in Chapter 6 are consistent with this experiment.

Gaonkar (1971) undertook a detailed investigation of the growth characteristics of *Aphanizomenon us*ing a mixture of culture experiments and experiments and observations in the lake. He determined the optimal growth rate as a function of several nutrients, temperature and light. Nitrogen and iron were the most limiting elements, and resulted in the competitive replacement of *Aphanizomenon* by *Anabaena* and *Microcystis* under warm, low nutrient conditions in mid and late summer.

The most extensive investigation of the algae problem was undertaken by the Clear Lake Algal Research

Unit (CLARU) under the direction of Alexander J. Horne. This work was supported by Lake County through the Flood Control and Water Conservation District with a matching grant from DWR. Horne's 1971-75 CLARU Reports and related publications (Horne *et al.*, 1971; Horne, *et al.*, 1972; Horne and Goldman, 1972, 1974; Horne *et al.*, 1979; Horne and Commins, 1987; Wrigley and Horne, 1974; Wurtsbaugh and Horne, 1983) examined the years 1969-1972 in detail. The CLARU work was a field collaboration with DWR, and Lallatin (1975) published a comprehensive report focused on the physical, chemical, and biological investigations of DWR 1968-73. A strong seasonal cycle of most physical, chemical and biological variables was established, based substantially upon measurements taken by DWR (Lallatin, 1975).

Horne's experimental studies documented the importance of the utilization of atmospheric nitrogen by the important scum forming genera *Aphanizomenon* and *Anabaena*. Home and coworkers determined that about 40% of the total nitrogen budget to the lake is supplied by atmospheric fixation. Horne noted a role for high ammonia concentrations in stimulating the irregular appearance of *Microcystis*. His nutrient budgets, based on limited stream monitoring data by DWR and USGS (see USGS, 1973), suggested that creek inflow accounted for over 60% of the lake's ample phosphorus supply, while the sewage waste stream accounted for less than 20%.

Nutrient limitation experiments were conducted by Wurtsbaugh and Horne (1983) from February to October 1975 (see also Horne, 1974). These experiments were basically fertilizer trials, similar to those conducted by Wetzel and Goldman (1963), in which various nutrients and combinations of nutrients were added to flasks (9 experiments) or large polyethylene tubes (2 experiments) of lake water to see which elements most stimulate algal growth and blue-green atmospheric nitrogen use. The use of atmospheric nitrogen by blue-greens was usually enhanced by iron additions and suppressed by additions of fixed nitrogen (nitrate). A small effect of phosphorus addition was noted on one occasion. Algal photosynthesis and biomass accumulation were stimulated most often by nitrate and iron, but on two occasions algal photosynthesis was stimulated by phosphorus, as was biomass accumulation on one date. Dissolved phosphorus levels in 1975 sometimes fell to levels considered potentially limiting, and were generally much lower than the earlier years of Horne's investigations (see below, **Chapter 4**).

These results further demonstrated the important role of nitrogen in limiting algal growth in Clear Lake.

The experiments led to the hypothesis that iron is the primary geochemical limiting factor in Clear Lake because of its ability to limit atmospheric nitrogen uptake. Without the iron limitation, the scum forming populations might often have approximately doubled in biomass before they became limited by phosphorus, since even at the height of the growing season 20-50% of phosphorus remained in the dissolved form.

An extreme sensitivity of Clear Lake blue-greens to copper toxicity was observed by Horne and Goldman (1974), but under other conditions Wurtsbaugh (1983) noted a less drastic effect of copper toxicity.

Based upon his experimental work, Horne advanced several strong hypotheses accounting for factors regulating the seasonal cycle of algal abundance. Winters were characterized by high nitrate, relatively low phosphate, and high inorganic turbidity due to erosion products in runoff. *Aphanizomenon* populations over-wintered in significant numbers likely due to their ability to acquire sufficient light in the turbid waters by floating. This genus then bloomed in the spring, using atmospheric nitrogen after other nitrogen supplies were exhausted. In early to midsummer, blooms collapsed, due to iron exhaustion, and death due to a nutrient-limited inability to escape the surface film. Typically, midsummer was characterized by relatively clear water, until phosphorus (and presumably iron) releases from the sediments stimulated modest late summer to autumn blooms of *Anabaena* and *Microcystis*. The very large supply of sediment released phosphorus relative to available iron resulted in summertime peaks of phosphorus that were far in excess of algal demand. In the 1969-72 period, peak summer dissolved phosphorus levels, often coinciding with blooms, ranged from 50 to nearly 500 ug/L. Limiting levels are generally considered to be around 10 u/L. Variation around this basic pattern was considerable from Arm to Arm and year to year.

Using wastewater diversion or treatment to force phosphorus levels down to levels that inhibit the growth of scum formers is the most common approach to controlling blue-green blooms (Cooke *et* al., 1993; Sas, 1989). Given that little of the phosphorus was in the relatively easy to treat sewage waste stream, Horne considered reducing this element to limiting levels to be impractical. He recommended using aeration/mixing to control blooms in the two lower arms, and trace additions of copper to suppress atmospheric nitrogen fixation in the Upper Arm. The intended effect of aeration was to keep oxygen levels in the deeper parts of the Oaks and Lower Arms high enough to retard recycling (internal loading) of phosphorus, iron, and ammonia. It is well known that anaerobic conditions favor the dissolution and recycling of especially iron and phosphorus, and the release of the ammonia instead of the nitrate form of nitrogen.

Sensitivity to the use of a toxic metal on a large scale prevented any attempt to use copper to suppress blue-green populations (Alex Home, personal communication).

3.3.3 Water Movement Investigations In Relation To Algae

The CLARU and DWR projects measured current speeds in the lake. Wrigley and Horne (1974) used remote sensing to track algal masses and sediment plumes, and measured sustained current speeds ranging from 2 to 15 cm/sec. Lallatin (1975) reports the results of 4 sets of drogue measurements. On each occasion 4 drogues were placed at 5 feet depth and 4 at 10 feet. Three of the series of measurements were done in the Upper Arm, and one in the Oaks and Lower Arms. Consistent with Wrigley and Horne's measurements, these ranged from about 2 to about 10 cm/sec (1 to 7 miles/day). Home (1975) also reports the results of year long series of wind measurements at Buckingham Point. Winds are consistently from the west and northwest at Buckingham Point. Under the influence of these winds, a general drift of surface water to the southeast from the Upper Arm into the Narrows region and the Lower Arm is indicated by the drogue study. Presumably, there was a return current at depth, generally below Lallatin's deepest drogue sets at 10 feet. Wrigley and Horne show that the surface currents are in fact more complex, gyre-like structures.

Under the auspices of a grant from the EPA Center for Ecological Health Research to study multiple stresses, we have been developing a hydrodynamic model and a particle tracking model of Clear Lake. Preliminary particle tracking runs of these models are ongoing_ Initial results indicate that under NW wind conditions, there are relatively high water/particle velocities developed near the Narrows and that NW to SE movement across the surface of the water is compensated by opposite (return) flow along the bottom. In addition, these preliminary results also indicate that there are other complex gyres developed in the Oaks and Upper Arms in response to conditions involving specific types of wind direction, much as Home and Wrigley measured (A. Bale, personal communication).

We interpret these data to mean that the separate Arms of Clear Lake are normally well mixed horizontally on a time scale of a few months. The Narrows has a minimum width of about 1 km and a depth of about 10 meters. If surface currents out of the Upper Arm average 2 cm/sec over a depth of 3 meters, the daily surface flow through the Narrows will be about 5 million m^3/day. The total volume of the lower Arms is a little over 500 million m^3. Thus, it would apparently take about 100 days for the water in the two lower Arms to be completely exchanged with the Upper Arm due to wind driven currents. From the shape of the Narrows region, it seems likely that the Oaks Arm mixes more easily with the Upper Arm than does the Lower Arm. The net drift due to summer outflow down Cache Creek is about 10% of these wind-driven flows.

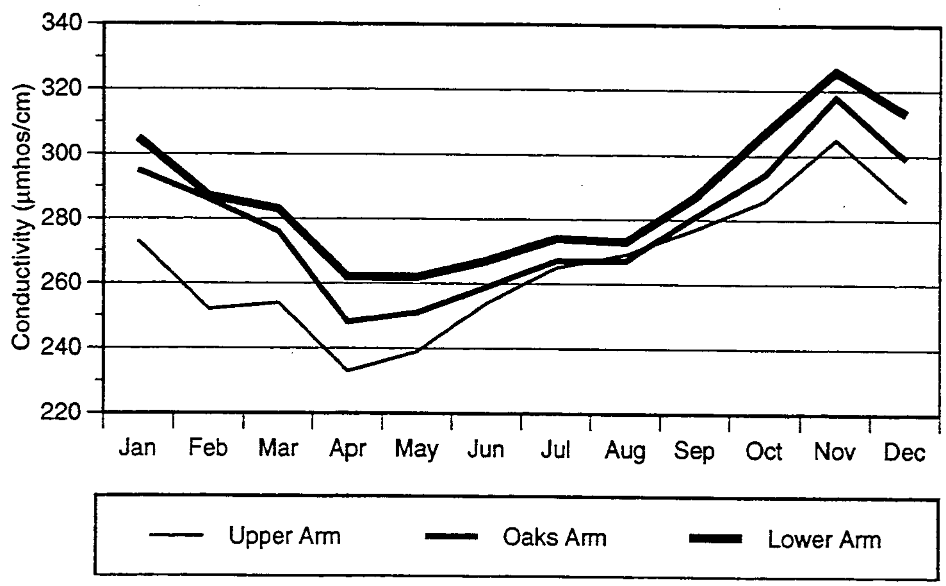
Consistent with somewhat restricted exchange, chemicals with little biological activity like sodium and chloride are slightly concentrated by evaporation in the two lower receiving Arms. DWR's Conductivity measurements in the three arms show that rather significant differences can be maintained (**Figure 3.8**), especially in winter when the lake is diluted with low-conductivity stream water via the major creeks. Since all of the major inflows are into the Upper Arm, it is considerably more dilute on average. For the Oaks Arm, the conductivity difference with the Upper Arm disappears in mid summer, whereas a small, detectable difference between the Upper and Lower Arms persists.

The fact that scum forming algal biomass is usually substantially higher in the two lower arms suggests mechanical concentration by prevailing westerly and northwesterly winds. The buoyant scum formers will be driven into the lower Arms by surface currents, but will tend not to return with the deep return current to the Upper Arm. It is also consistent with differential growth in the two lower arms due to a more gradual increase in nutrients due to the collection of floating particulates or some other effect. Sediment levels of nutrients are also higher. **Figure 3.9** compares the difference in algal counts for the three major buoyant genera *(Aphanizomenon, Anabaena,* and *Microcystis),* and some comparison of non-buoyant genera *(Chlamydomonas, Oocystis, Schroederia)* from the DWR record (also see **Chapter 4** for more detail). These data show that the scum forming genera do have consistently higher populations in the downwind Arms. Results from the UCD hydrodynamic/particle tracking computer model also support the idea of a buildup of floating scum formers in the downwind Oaks Arm, and the stronger isolation of Lower Arm (A. Bale, personal communication). The higher concentration of blue-greens in Oaks Arm relative to Upper Ann are perhaps a more direct result of mechanical concentration, whereas Lower Arm/

Upper Arm differences are more likely related to growth.

A final observation concerning Figure 3.9 is that the suspended genera have higher populations in the Upper Arm. We suppose that deep, nutrient-rich return currents from the other arms of the lake, combined with a differential removal of surface scum formers, tip the balance slightly in favor of suspended non-blue-green species in the Upper Arm.

Horne (1975) discusses the role of buoyancy and vertical mixing by the wind in regulating the *Aphanizomenon* bloom. The success of this species is a consequence of its ability to remain near the surface under turbid conditions, but it is also susceptible to death and decay when trapped at the surface. When surface accumulations occur during the night and early morning, they may be driven below the surface by daytime wind or might have the capacity to actively reverse their tendency to float, or a combination of the two. When this does not occur due to calm conditions, perhaps complicated by nutrient deficiency, the populations die and cause highly obnoxious decaying scums.



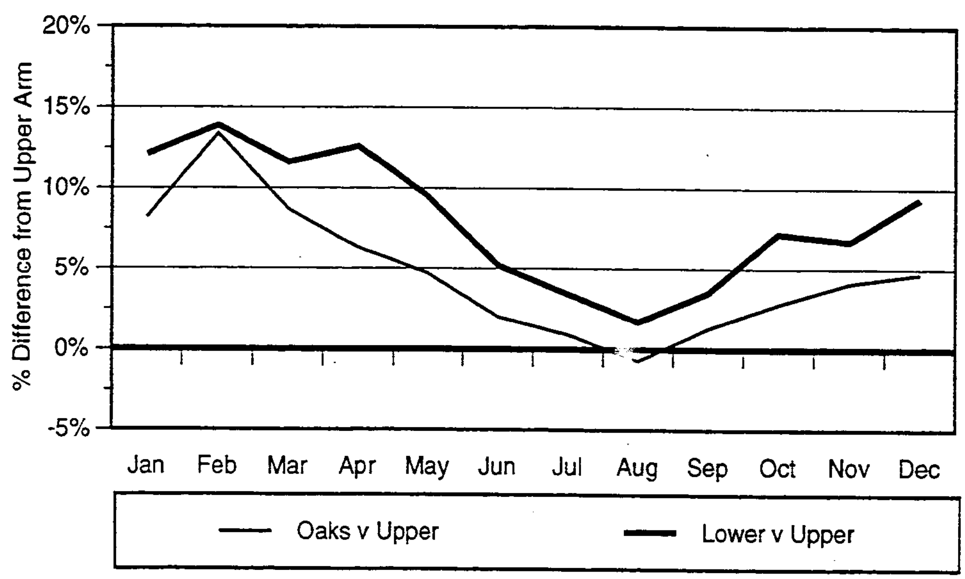


Figure 3.8. The difference in conductivity between the Upper and two lower arms of Clear Lake. Data is averaged over 24 yrs and is taken from the DWR data set. Notice that the Upper Arm conductivity is almost always below that of the other arms of the lake.

The recommendation to aerate the two lower arms was tested in the Oaks Arm in 1975-78 by William F. Rusk (1983) under Horne's guidance. Experimental aeration was conducted on several occasions for periods up to 37 days. Measurements of water quality unfortunately indicated little effect on the critical nutrient concentrations, and dye experiments showed that only a relatively small volume of water was effectively aerated. Rusk noted that low oxygen rather than complete anoxia was sufficient for phosphorus recycling, so large volumes of water would have to be aerated to achieve useful results. Routine use of aeration was not considered cost effective.

3.3.4 Post-CLARU Investigations

In summary, Home and coworkers' CLARU investigations established an excellent basic understanding of Clear Lake algal ecology, but did not determine a practical solution to the scum problem. Unfortunately, the intensive CLARU program was not continued, and subsequent work was quite limited until this study. Fortunately, the DWR monitoring program has continued to the present day.

An exception is the very interesting experiment of Elser and Goldman (1991). They investigated the impact of zooplankton grazing on the phytoplankton of lakes of different productivity. Enhanced zoop-

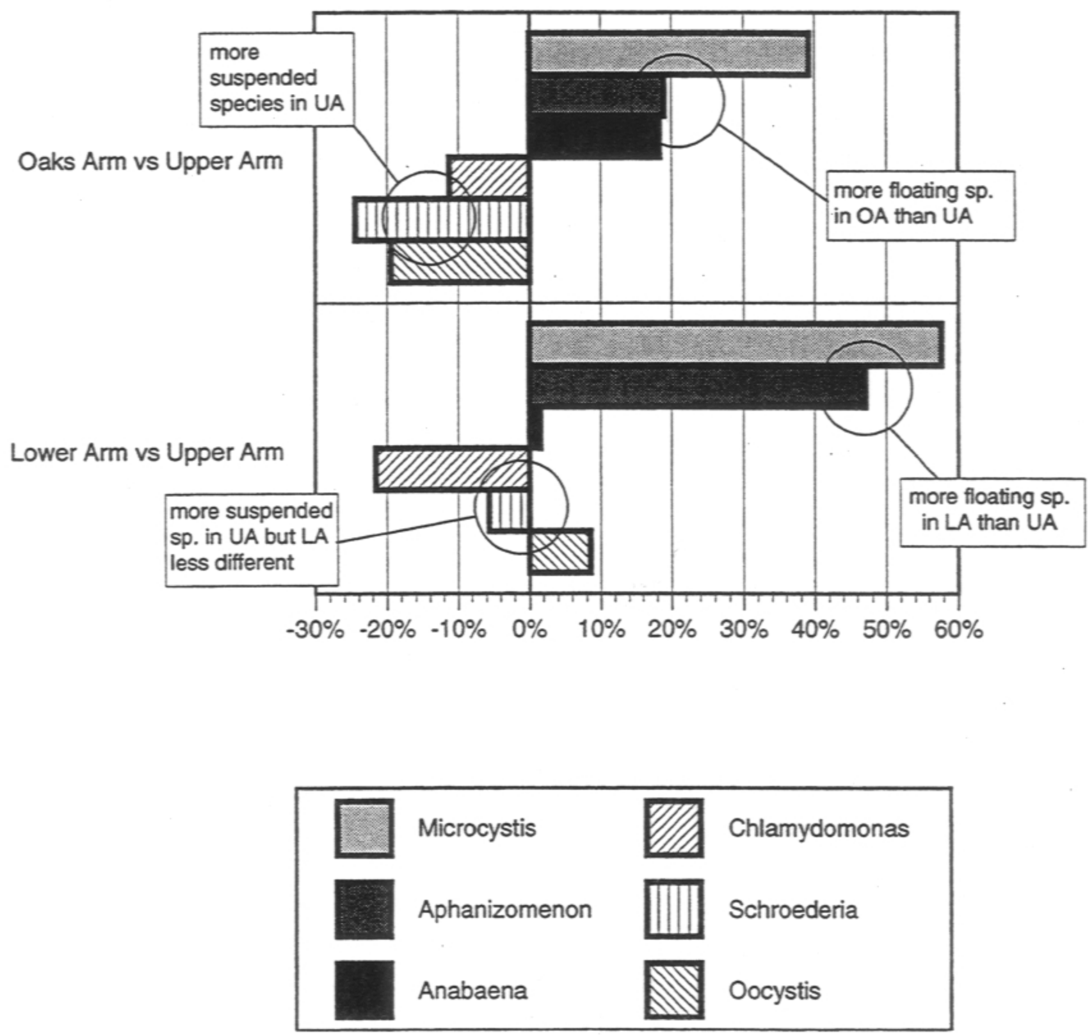


Figure 3.9. The percent difference between the Upper and two lower Arms for selected phytoplankton genera. The numbers represent the average difference of monthly averages pooled over 24 years. In interpreting this figure, notice that positive percent differences indicate that the Upper Arm had the smaller count.

lankton grazing on algae in some systems is capable of substantial improvements in transparency. It is also possible to affect the abundance of zooplankton by manipulating their predators, especially fish. However, it is usually thought that the large cyanobacteria that dominate Clear Lake are not nearly as affected by grazing as smaller more nutritious and nontoxic forms (Haney, 1987; Lampert, 1987). The basic experimental approach consisted of exposing the lake phytoplankton community to a wide range of numbers of grazing zooplankton from below to above natural levels. The impact of grazers was strong in moderately productive Castle Lake, which is dominated by small algae. Practically no effects of grazing could be demonstrated in Clear Lake, as expected from its dominance by large, ungrazable forms. Thus, we expect relatively weak interactions between the algae and higher levels in the food chain in Clear Lake. Fish population changes may affect zooplankton, but the effects should not propagate down to the algae level. Similarly, shifts of algal species composition away from scum forming cyanobacteria should not have large direct effects on zooplankton or fish populations if reductions in total nutrient mass are compensated by a more efficient trophic system.

4 DWR Water Quality Record

P. Neale and C. Woodmansee

4.0 Abstract

The California Department of Water Resources Northern District conducted a water quality monitoring program at Clear Lake from 1969 to the present. Data, usually at monthly intervals, was collected on physical, chemical and biological parameters. In this chapter we analyze this data in order to understand what processes have caused long term patterns in the abundance of scum forming blue-green algae.

The Clear Lake system is highly variable. The influence of runs of wet and dry years has had an especially striking influence on blue-green abundance and many other factors.

Clear Lake has the highest concentration of phosphorus in the summer and fall months, even though most of the phosphorus enters the lake during rainy winters as silt-laden stream flow. Historically, the highest levels of phosphorus in the lake seem to be associated with stream disturbances such as gravel mining, especially associated with events in the very early 1970s, and again in the late 1980s and early 1990s in response to drought. Phosphorus is an important nutrient for algal growth and very high phosphorus levels in 1989 and 1990 led to very large algal blooms.

Limited data for the element iron suggest that it also arrives in stream flows during rains, and that its level in the lake water decreased during recent droughts. Blue-green algae appeared to be limited by low levels of iron in 1991 and 1992, the height of the drought, even though phosphorus levels remained very high. This suggests that the reduction of iron levels in the lake may limit algal growth, but this does not rule out the importance of some other metal or element not measured.

A detailed statistical analysis of the data reveals that the scum forming algal genera *Aphanizomenon, Anabaena,* and *Microcystis* tend to have highest biomasses in drought years. Generic shifts also occur in response to chemical and physical variables, especially those associated with drought. Paradoxically, the highest and lowest years of blue-green biomass have both occurred in drought years when phosphorus levels have reached very high levels. Unfortunately, measurements of iron were made infrequently and using two different methods, and a statistical test of the iron hypothesis is not possible.

That there is some statistically reliable dependence of blue-green biomass and dominance on phosphorus is encouraging. It is likely that reducing phosphorus concentrations to levels prevailing in the lowest years of record and lower will have a substantial impact on the prevalence of scums, independently of the behavior of iron.

4.1 Overview

The California Department of Water Resources (DWR) has conducted water quality studies at Clear Lake since the early 1960s. The record is continuous from May 1968 until the present, although phytoplankton were not reported until 1969. Data on physical conditions, major ion and nutrient chemistry, and phytoplankton abundance were collected approximately monthly (data are missing in winter in some years, etc.) from one centrally located station in each of the three arms of the lake. The last report on these data was by Lallatin (1975). As of this writing, the DWR data record is available through 1992 for nutrient data and through 1991 for phytoplankton data.

Other important long-term records on Clear Lake include lake level data from the Lake County Flood Control and Water Conservation District (LCFCWCD), data on aquatic dipterans and other lake parameters collected by the Lake County Mosquito Abatement District (LCMAD), weather data from the U.C. Hopland Field Station, and many other miscellaneous sources. A report by the Database Sub-committee of the Basin Integrated Resource Management Committee (*Why et al.* 1992) provides a key to such information. In **Figure 3.7** we presented a time line which helps to focus comparisons of critical events in Clear Lake's history with changes in the level of productivity as reflected in the DWR data-base record analyzed here.

4.2 Methods

4.2.1 DWR Field Methods

The locations of the three DWR stations were noted on **Figure 3.1.** Station Clear Lake 1 is in the Upper

Arm in about 7.2 m of water. Clear Lake 3 is in about 9.5 m of water in the Lower Arm. Clear Lake 4 is in 12 m of water in the Oaks Arm. The locations were centrally located within the basins and originally defined by triangulation points using prominent landmarks on shore. Normally all stations were visited on the same day or subsequent days. On about 5% of dates bad weather or other difficulties prevented data from being taken at all three stations. Data were not collected during the winter months from 1971-72 until 1976-77.

Water samples were taken with a plastic Van Dom bottle at 2 to 3 meters depth intervals at each station. Several measurements were taken in the field based on these water samples. Initially, pH was determined using a Hellize comparator; but during the past few years, was determined with a Hach 1 meter calibrated against pH 7.0 and 10.0 standard buffer solutions. Turbidity was determined using a Hach nephelometer, and alkalinity by titration with H_2SO_4 to the 4.5 endpoint. These measurements were made according to the procedures described in Standard Methods (1989). Samples for mineral analysis were stored in plastic bottles after being filtered through a 0.45 micron membrane filter. Samples for nutrient analysis were stored in plastic bottles, placed on ice for transport, and frozen. Samples for minor element determinations were stored in acid washed plastic bottles, and acidified with concentrated HNO_3. These samples were shipped to DWR's Bryte Water Quality Analysis Laboratory. Phytoplankton samples were collected in glass bottles preserved with Lugols solution (Standard Methods 1989).

Temperature and oxygen were measured at one meter intervals with a YSI submersible probe. The oxygen electrode was calibrated at each station against an Azide modification of the Winkler Method (Standard Methods 1989). The temperature sensor was checked against ASTM thermometers at the laboratory.

Field data were entered on a standard data sheet designed for Clear Lake use.

Over the 24 years that these samples were collected, consistency of methods was carefully maintained by thorough training of new personnel, and cross-checking when there were minor methods changes (instrument replacement, following revised procedures in successive editions of Standard Methods, and other modifications).

4.2.2 Chemical Water Quality Data

Water samples were analyzed in part by the DWR Chemical Laboratory in West Sacramento, California (Bryte Laboratory), for micronutrient, mineral and metal content, beginning in 1969. Of particular interest to this study are their methods for measuring total phosphorus (total-PO_4 P), dissolved phosphorus (ortho-PO_4 P) and iron (Fe).

DWR field crews collected water samples at surface, depths and bottom. Only surface and bottom samples were shipped for analysis. Each collection was divided into four sample containers, two of which were filtered on-site through 0.45 micron filters. These filtered and unfiltered samples were then transported to DWR for storage (refrigeration) and eventual shipment to Bryte. Samples at Bryte lab were either refrigerated up to 48 hours after arrival, or else were kept frozen until analysis. The type of samples collected and shipped are as follows:

Surface Samples (approx. 30 cm):
1.0 qt Filtered, mineral
0.5 pt Filtered, acidified mineral
0.5 pt Filtered, frozen nutrient
0.5 pt Unfiltered, frozen nutrient
2.0 pt Unfiltered, acid washed and acidified minor elements, 3x per yr

Bottom Samples:
Nutrients only

Mineral and heavy metals samples were stored in boxes until transported to Bryte. Nutrient samples were stored frozen. Samples were transported via car to Bryte within 30 days of collection with lab-submitted sheets listing analyses to be performed. The lab has used different methods for determining total-phosphorus, dissolved-phosphorus, and Fe over the years of this study, as the lab has kept up with changing instrumentation, etc. A detailed outline of methods used and dates of use is presented below.

For Total Phosphorus methods (unfiltered water):

4/1/70 to 9/23/75
DIGESTION-Refer to Standard Methods 13th edition, p. 525. This is a sulfuric acid/nitric add digestion.
MEASURE-S.M. p. 530. A molybdate colorimetric measurement after extraction with organic solvent.

9/23/75 to 4/20/78
DIGESTION-same as above.
MEASURE-same as above, except without the extraction.

4/20/78 to date

(Changed to EPA method 365.4) DIGESTION-Sulfuric acid with Hg catalyst MEASURE- Molybdate, ascorbic acid colorimetric automated.

For Dissolved Phosphorus methods (filtered water):

MEASURE-same as above for all dates, except that 4/20/75 to date, changed to EPA method 365.1; molybdate, ascorbic acid colorimetric automated method.

For Iron methods:

4/5/77 to 4/18/80, dissolved method using filtered water:
Refer to Standard Methods 13th edition, p213.
(This is a MIBK-APDC extraction used for low levels of Pb and Cd, and also used for Fe.)

9/16/80 to date, total recoverable method based on unfiltered water.
Switched to EPA method 236.2; furnace AA.

For Nitrate (filtered water, includes nitrite if present):

2/68 to 6/79
Brucine colorimetric, Env. Sci. and Tech. June 1967.

6/79 to present
Cadmium reduction colorimetric, equiv. to EPA 353.2.

For Ammonia (filtered water):

1959 to 5/79
Distillation, nesslerization colorimetric equiv. to EPA 350.2.

1/79 to present
Phenate colorimetric automated. EPA 350.1.

Organic N (unfiltered water):

1959 to 5/78
Sulfuric Acid Hg catalyst digestion, distillation, nesslerization colorimetry; equiv. to EPA 351.3.

5/78 to present
Sulfuric Add Hg catalyst digestion, phenate colorimetric automated; EPA 351.2.

All nutrient concentrations are expressed as elemental (e.g. Ammonia Nitrate/l).

DWR has a policy for checking the accuracy of Chemical Laboratory methods through the use of blanks, spiked samples, and split samples in accordance with its QA program. We have no reason to suspect that any changes in chemical methods since 1968 qualitatively affected the scientific conclusions of this report. The unfortunate exception is iron, where a significant methods change in 1980 complicates interpretation.

4.2.3 Enumeration And Biomass Calculation

Phytoplankton samples were enumerated under 200 magnification using a Sedgwick-Rafter cell. The sample bottles were inverted several times and a pipette of the well-mixed sample was delivered into the cell (volume =1 ml). Two transects were counted. The forms counted were identified to genus using standard reference guides:

How to Know the Freshwater Algae (Prescott) Algae of the Western Great Lakes Area (Prescott) Guide to the Common Diatoms of Water Pollution Surveillance System Stations (USDI)
Freshwater Biology (Edmundson)

Counts were recorded on hardcopy paper data sheets, and were recently (1992-93) entered into a computer database as described below.

Biomass of each genus was determined by measuring several cells or colonies of all the abundant forms (remeasuring if different values were obtained), and approximating volumes by combinations of simple shapes like cylinders and spheres. These methods closely follow those recommended in the various editions of Standard Methods. Biomass data were missing for some dates, particularly during a two-year period from mid 1979 until mid 1981. We have estimated the biomass factor for these missing periods by using averages from nearby years. In some cases, biomass per unit count (cell or colony) varies substantially, so the biomass estimate has a correspondingly large uncertainty in these cases. How-ever, the variation in biomass between genera is generally greater, so any reasonable biomass correction tends to make the data more comparable across genera and time.

4.2 .4 Computer Entry of Data

The long-term data provided by DWR was transcribed from paper data sheets into an electronic computer database for the purposes of analysis. To the extent that the original data sheets were accurate records of field and lab observations, the databases have been found to be accurate as well. During en-try, summation fields allowed a cross check against summations performed by hand when the data sheets were drafted.

Entry error rates were checked by checking a portion (about 10%) of the database against photocopies of random original data sheets provided by DWR. A few key-entry errors (one number substituted for a neighbor on the keyboard, transpositions of numbers) were found. The error rate was 4/2,300 entries checked. None of these errors led to a qualitative change in the entry.

We have also checked all data that are apparent outliers (extreme values) in the data series. Since the data were collected at multiple depths and multiple stations, there are internal consistency checks for the realism of outlier values. To date, extreme values have been shown to be duplicated at multiple depths, stations, and/or dates. There have been no outliers attributed to errors in data entry or other simple errors in procedure. No scientifically important conclusions depend upon a few extreme values in any case.

4.3 Basic Patterns

Over the 24 years of the DWR record, substantial changes have occurred in Clear Lake. The physical/chemical data and the phytoplankton biomass data show some outstanding trends, often associated with the dramatic changes that occurred after significant reductions in gravel mining in the early 1970s and in response to the prolonged drought that extended from 1986-92.

4.3.1 Physical/Chemical Data

In Chapter 3, some basic descriptive data were presented for the Upper Arm station over the entire period over which the DWR data were collected, providing general trends in: (1) minima/maxima for water pH, alkalinity, hardness, sodium, sulfate, chloride, and boron, (2) annual averages for turbidity, and (3) monthly averages for temperature and electrical conductivity.

A more complete record of 24 years of monthly values for temperature, pH, and water clarity (Secchi disk readings) are provided in **Figure 4.1A** (Upper Arm station), **Figure 4.1B** (Lower Arm) and **Figure 4.1C** (Oaks Arm). These plots provide documentation of the variability in these parameters over the DWR study period, indicating both the range of values and some indication of extreme years.

Temperature in Clear Lake has shown moderate consistency both within and among years throughout the DWR data collection period. The temperature plots on **Figure 4.1** look as if the years from ca. 1971-1976 had both higher temperatures and lower variability than other years, but this is an artifact of the lack of data collection during the winter periods of those early years in the record. Secchi disk readings (a measure of water clarity) and pH both increased dramatically during the latter drought years from ca. 1989-92 (see **Figure 4.1**). Secchi disk readings also had unusually high readings in the years from 1985-86.

The DWR data set on water column phosphorus shown as monthly means (pooled for the same month over the entire time series) for each arm in **Figures 4.2** and as an annual time series (from monthly averages) for each arm in **Figures 4.3.** Note that both figures show how variable phosphorus is in the timing and extent of peaks in different years. As with the weather, normal (= average) years are actually exceptional.

These data are readily interpretable in terms of the processes of internal and external loading from our own detailed studies reported below in **Chapters 5 and 6**. On a seasonal basis, the external phosphorus load arrives almost entirely with the winter flood flows and is stored in the sediments due to the insolubility of phosphorus compounds under cold, oxygenated conditions. Phosphorus peaks sharply during the late summer due to the internal loading process (defined here as the recycling of nutrients from sediments to the water column). On a long-term annual basis, the DWR data set shows a very large variation in summer peaks of dissolved and total phosphorus between years, reflecting internal loading over the years since 1968. Internal loading was apparently very high from 1968 until 1973. Thereafter, a decline set in, and from 1974 until ca. 1986 summer total phosphorus peaks were 1/4 to 1/2 of the earlier period.

Presumably, the drop in summer internal loading after 1973 is closely related to winter external loading, although it is clear from the very high internal loading during the recent drought that internal and external loading do not have to be closely related. The drop in winter turbidity **(Figures 3.5 and 4.4)** after 1971-2 supports the hypothesis that sediment loading was very high in the first years of the record. The lack of winter data for several years in the early 1970s, however, may exaggerate the apparently sharp drop in 1972-3 in **Figure 3.5.** In our own data, roughly 20% of the total phosphorus in the surface sediments

Continued on page 12

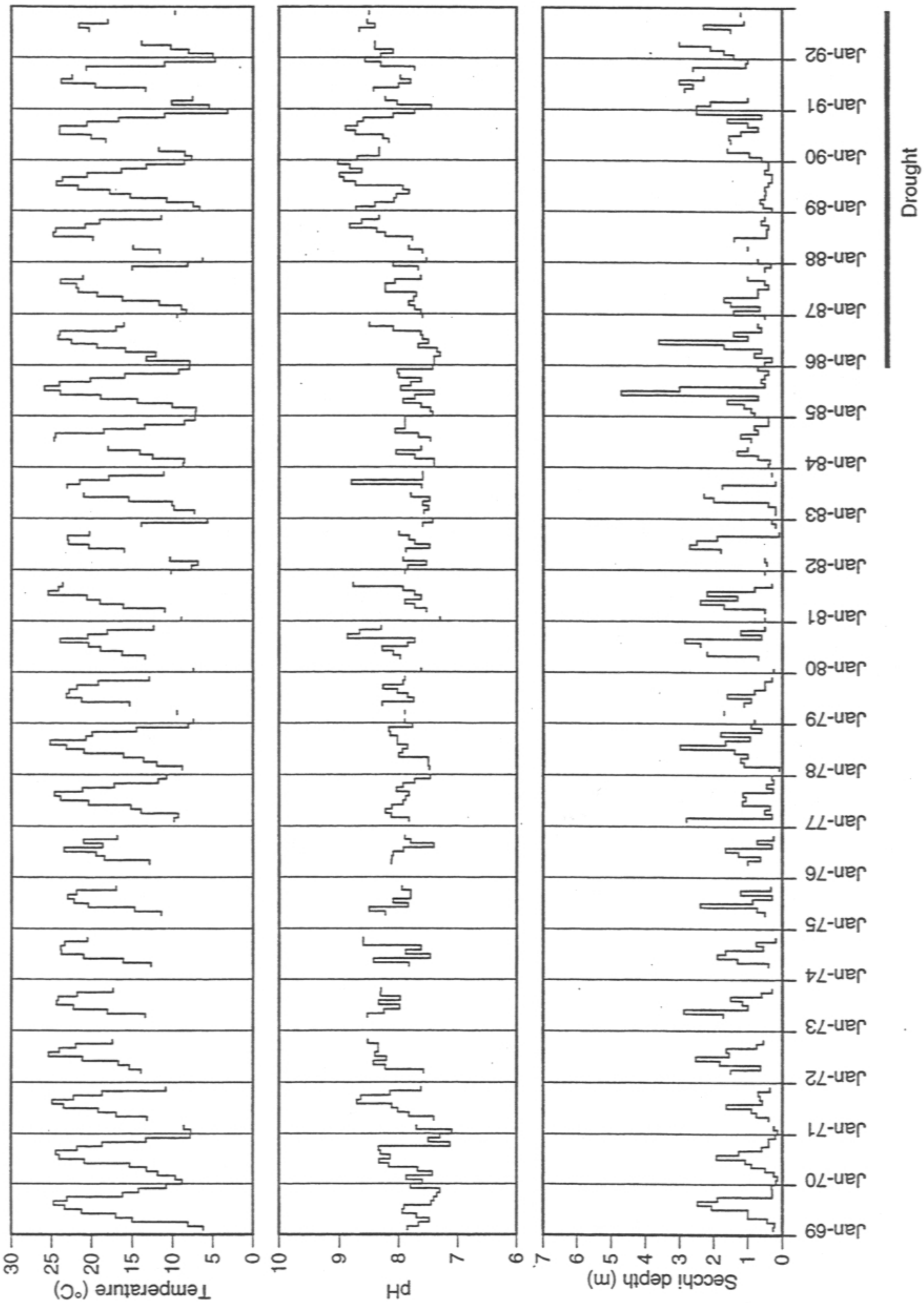


Figure 4.1A Upper Arm Time series data (from monthly averages) for temperature, pH and Secchi disk readings from the DWR data set.

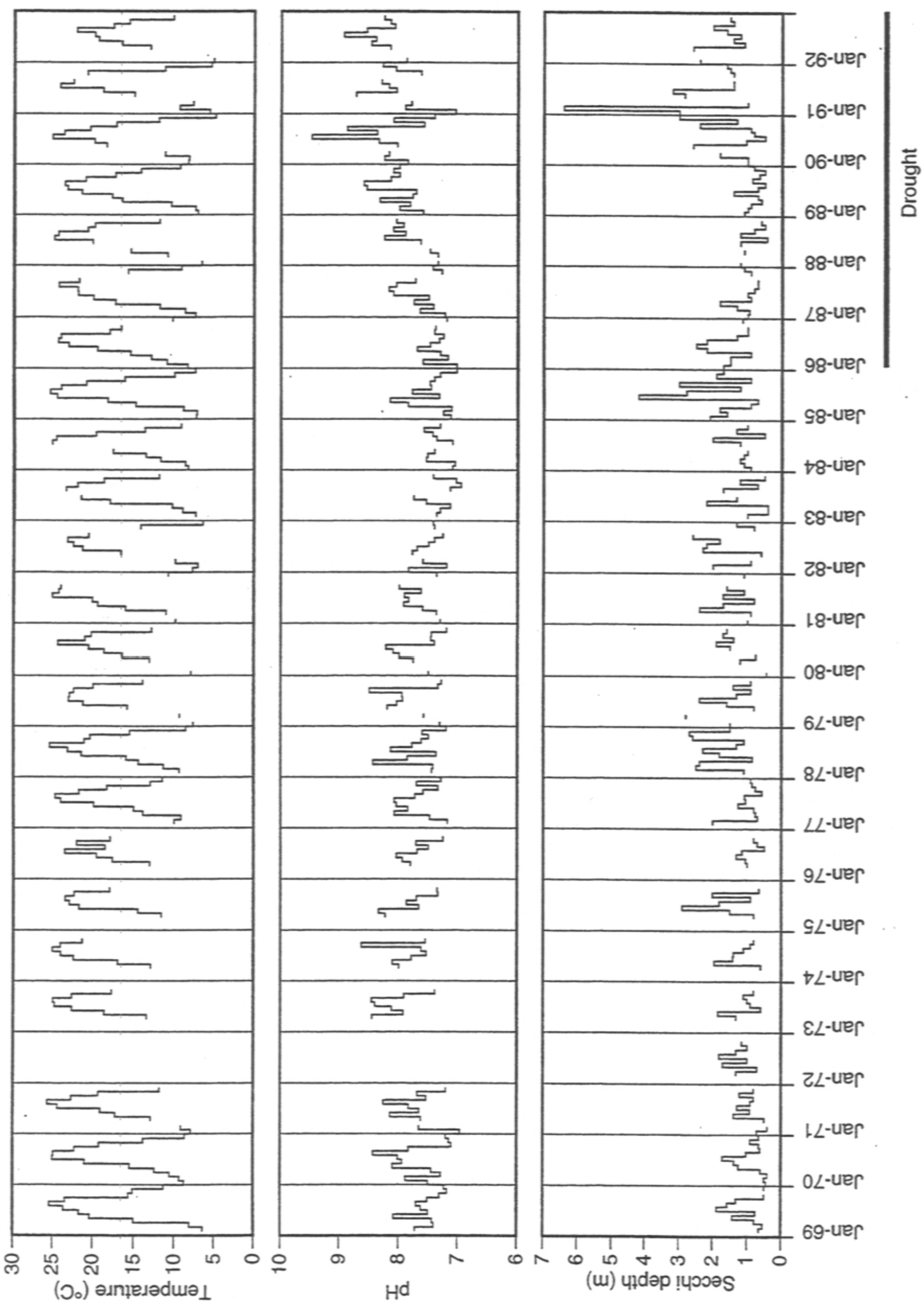


Figure 4.1B Lower Arm time series data (from monthly averages) for temperature, pH and Secchi disk readings from the DWR data set.

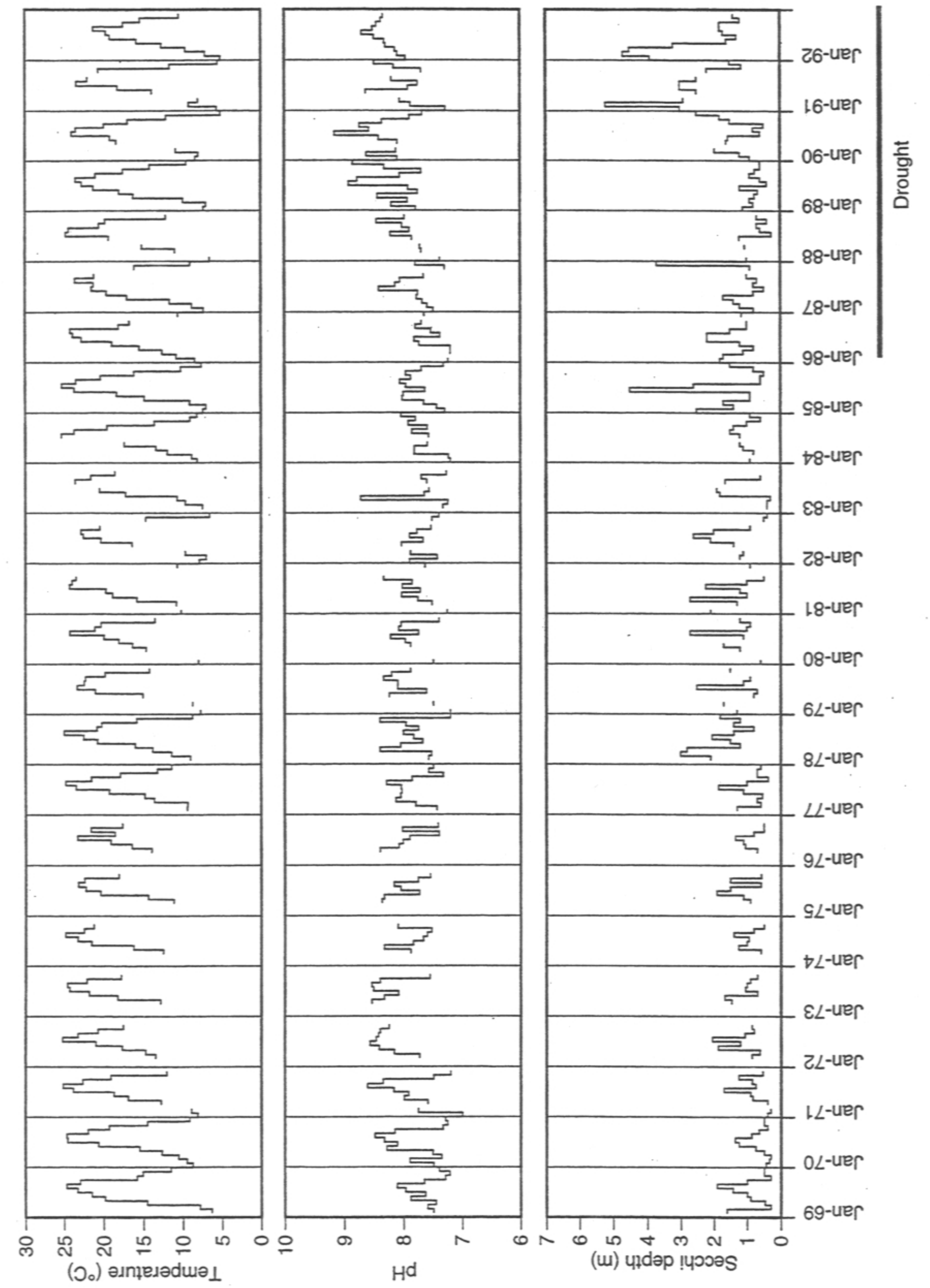


Figure 4.1C Oaks Arm time series data (from monthly averages) for temperature, pH and Secchi disk readings from the DWR data set.

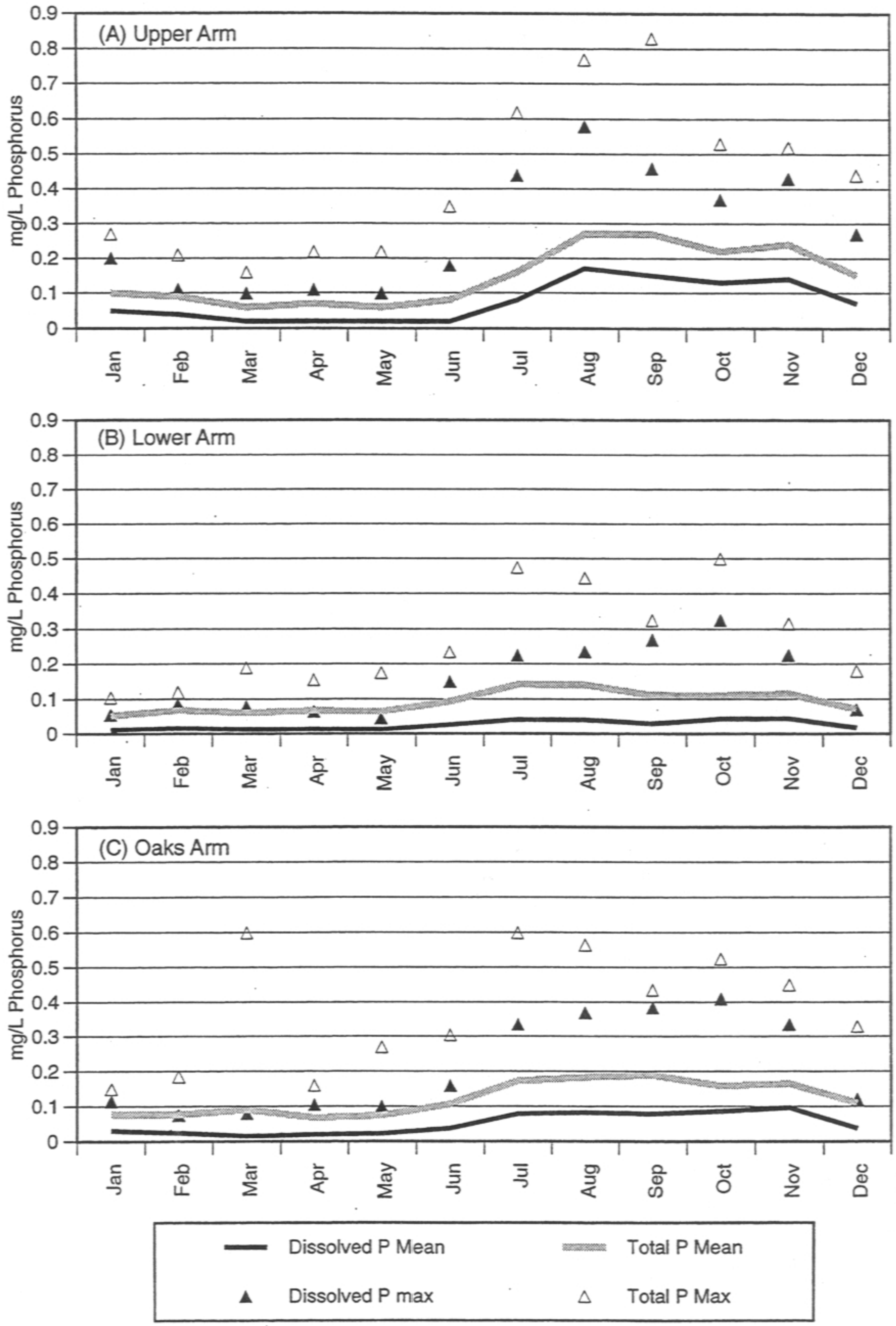


Figure 4.2 Pooled monthly means and maxima for dissolved and total phosphorus from the DWR data set for (A) the Upper Arm, (B) the Lower Arm and (C) the Oaks Arm.

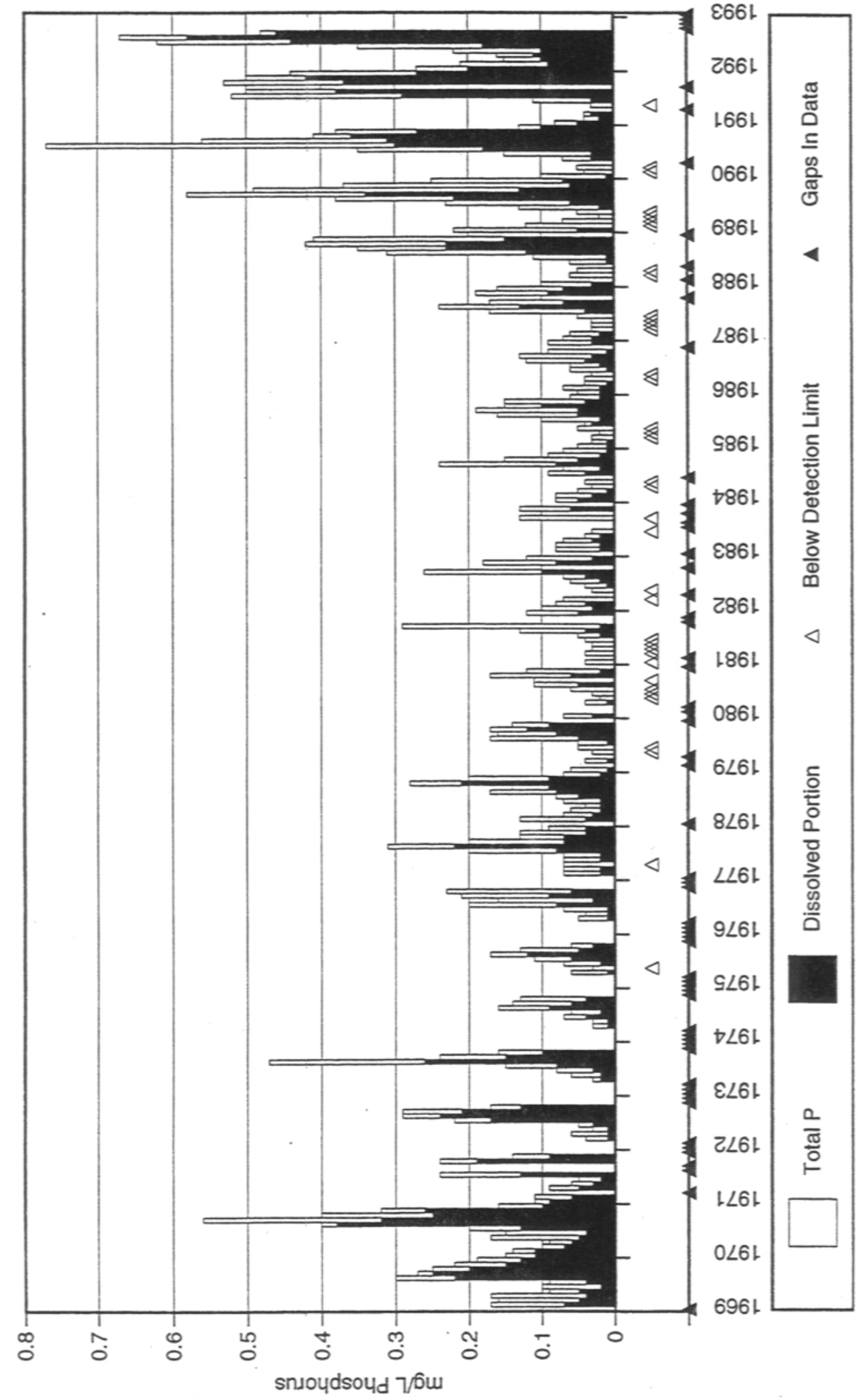


Figure 4.3A. Time series for phosphorus for the DWR Upper Arm station.

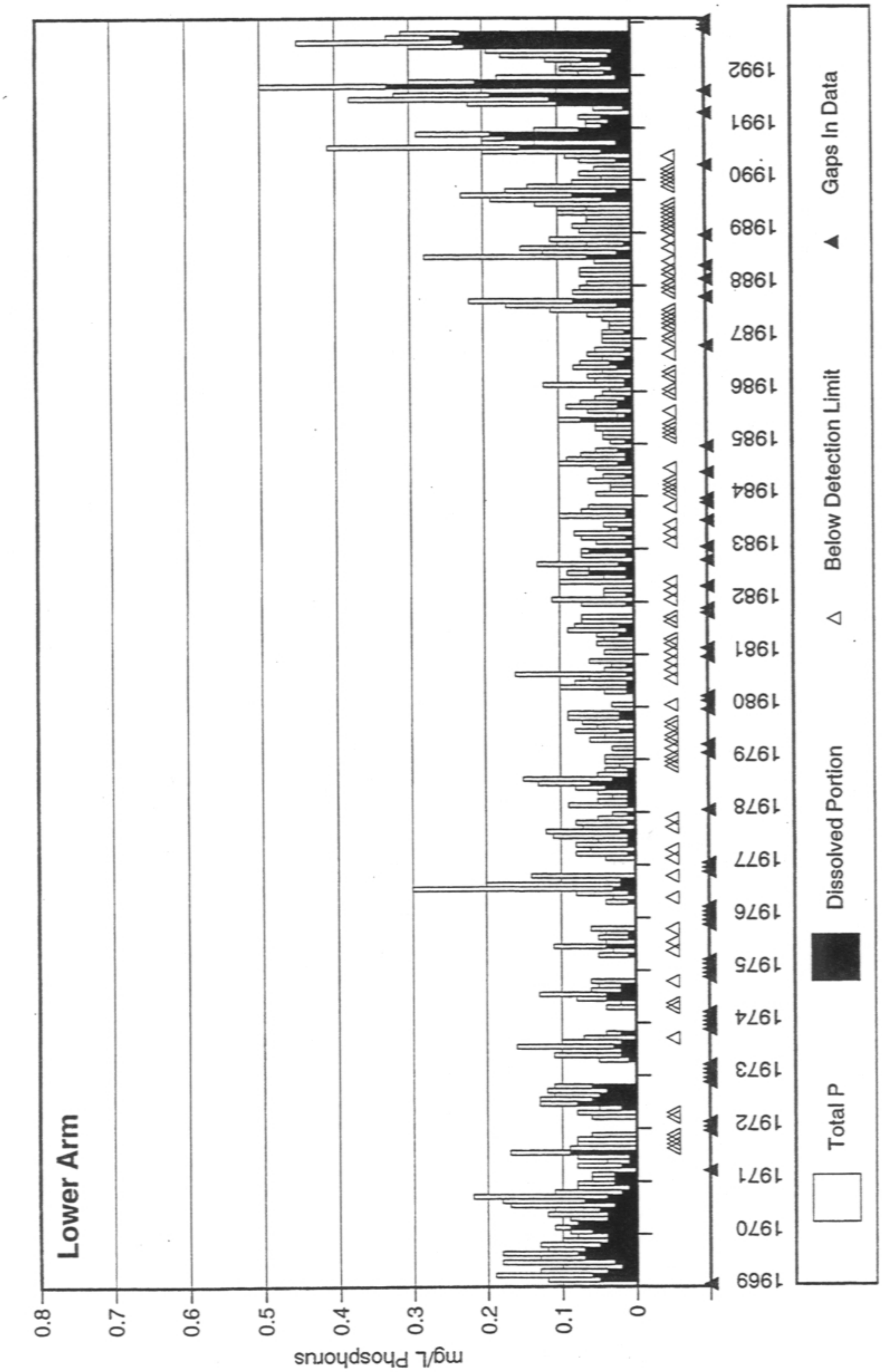


Figure 4.3B. Time series for phosphorus for the DWR Lower Arm station.

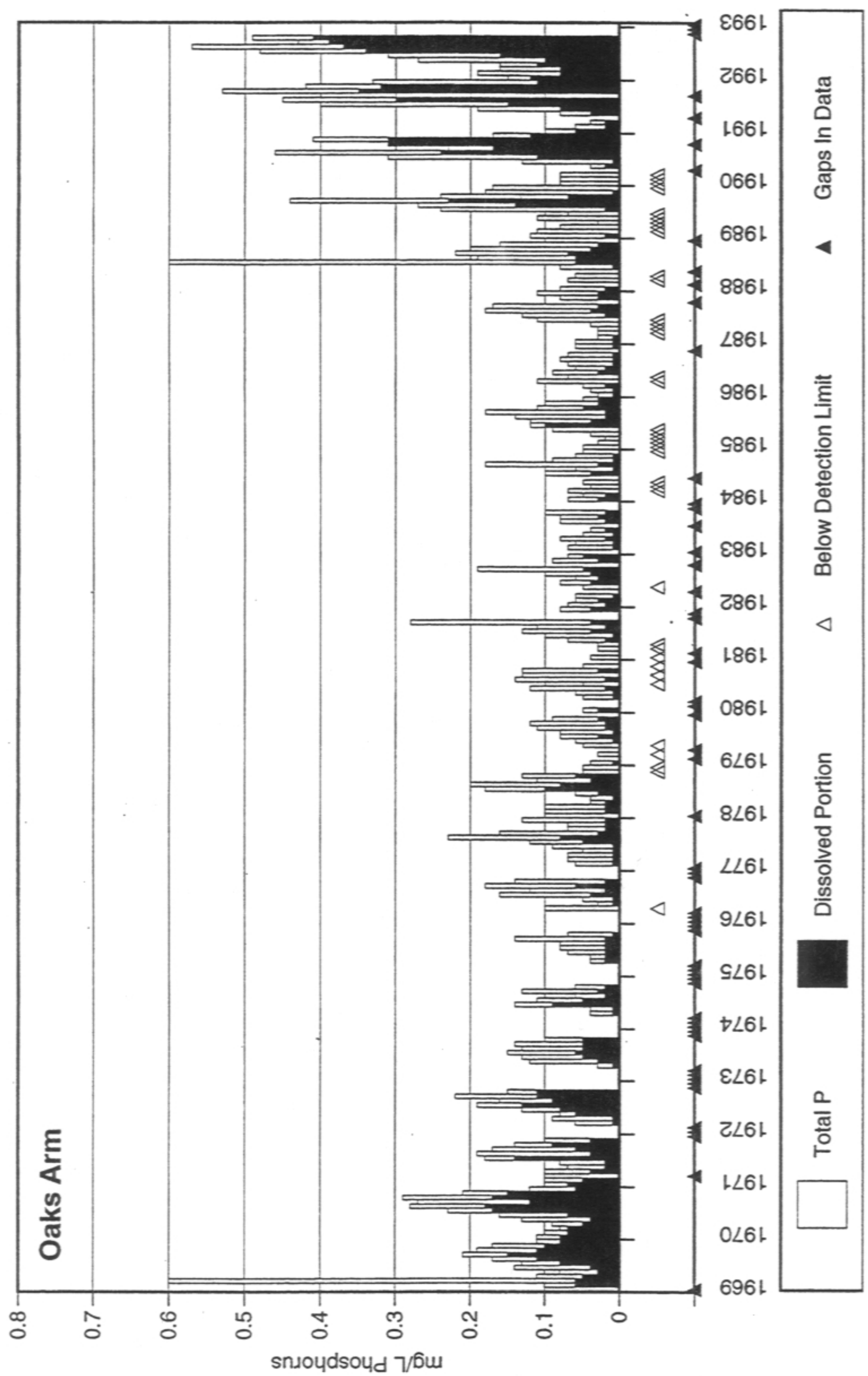


Figure 4.3C. Time series for phosphorus for the DWR Oaks Arm station.

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Continued from page 4

was mobilized each summer and appeared in the overlying water (**Chapters 5 and 6**).

The high phosphorus levels in the lake during the 1969-73 period was plausibly a result of stream channel erosion due to gravel mining and other activities. A major episode of gravel mining occurred in Scotts, Kelsey and Adobe Creeks for the construction of the Route 29 freeway segment from Kelseyville to Lakeport in 1970-72. The most pronounced degradation of Kelsey Creek was near Main St/Merritt Road (degradation of 8' during the 1960s, 5' during the 1970s, and 2' of aggradation during the 1980s) while the Dorn wells indicate degradation of only 4' near the Dorn Crossing during the 1960s, and stability since then. Kelsey Creek may have downcut substantially near the lake in the same period due to the opening of the channel at its mouth. The drop in phosphorus leading to the lower levels during the middle 1974-86 period appears to reflect the cessation of very high loading from the worst effects of these disturbances.

The decline of summer phosphorus peaks from 1972-74 and the rise from 1986-92 indicates that the sediment reservoir of phosphorus averages over a few years, but not a few decades. 1971-2 was the last very turbid winter recorded until 1983, and lake summer total phosphorus peaks appear to have reached equilibrium with the new regime of external loading by 1974. The rise over the recent drought period was likewise rapid.

For both monthly averages and annual averages, there is a high degree of internal consistency between data from each of the three arms of Clear Lake. However, the Upper Arm (where the bulk of the external loading sources originate from stream input) does exhibit the highest levels of both dissolved- and total-phosphorus (Figures 4.3). Significantly, dissolved-phosphorus levels were in the range where inhibition of blue-green growth (which is expected below about 0.010 mg/l, marked as below limit of detection in Figures 4.3) was likely occurring for much of the year during much of the period from 1975 to 1990. This trend is especially marked in the Lower Arm. The very large surplus dissolved phosphorus concentrations observed by Lallatin and Horne existed for shorter periods and generally were on a much reduced scale in these years. Under such conditions, improvements in water quality through aggressive control of phosphorus supplies seem more plausible than they did under the conditions observed by Lallatin and Horne.

Other variables besides phosphorus show decade scale variation. **Figures 4.4** show variation in dissolved nitrogen (nitrate and ammonia), electrical conductivity (a measure of total dissolved minerals), and turbidity. Nitrate declined from peaks in the late 1960s to early 1970s but varied more irregularly than phosphorus through the period. Winter turbidity declined sharply after 1971, especially in the Upper Arm, apparently reflecting lower inputs of sediment from streams. Subsequently, high runoff years still caused significant winter turbidity peaks in the Upper Arm. Over the whole record there has been a slow tendency in the direction of lower turbidity in all seasons. Electrical conductivity clearly reflects the long term "cycles" of wet and dry years. (See also Section 3.1, Figure 3.4).

Dramatic changes occurred during the drought years from 1987 to 1992. Internal loading of phosphorus increased in this period until levels surpassed those of the 1969-72 period **(Chapters 5** and **6).** Phosphorus remained at potentially limiting levels in spring and early summer from 1986-89, but brief excesses of phosphorus developed during late summer **(Figure 4.3).** In 1991 and 1992, there appeared once again huge concentrations of dissolved phosphorus in excess of phytoplankton demand, as in the Horne-Lallatin period. 1991 and 1992 initiated the present clear-water regime in the lake, and the low phytoplankton biomass cannot utilize the available phosphorus in any season, presumably due to iron and nitrogen limitation **(Chapter 7).**

The limited total recoverable iron data **(Figure 4.5)** from the DWR data set suggests falling concentrations of this element during the recent drought period, but patterns are hard to interpret due to the infrequent sampling. Iron levels were generally consistent with the earlier work of Horne (Goldman and Horne, 1983:156) for the dissolved method (1977-spring 1980). Subsequent measurements of total recoverable iron largely reflect particulate iron of unknown availibility to plankton. As far as they go, they suggest severe iron limitation in the latter two years of drought, neglecting one high iron measurement in Lower Arm in 1992. The possible mechanism for limiting iron availability during the drought are discussed in **Section 5.3.11**. The relatively high pH and phosphate in Clear Lake are expected to reduce the solubility and availability of iron to algae. Clear Lake, and more generally lakes in semi-arid regions, tend to have low dissolved iron levels (Goldman and Home, 1983: 153). Our own 1992-3 data on soluble iron **(Table 7.2)** are very similar to those of Horne, ranging from 5 to 50 μg/l, while our total iron measurements are consistent with the recent measures of DWR (70-800 μg/l.

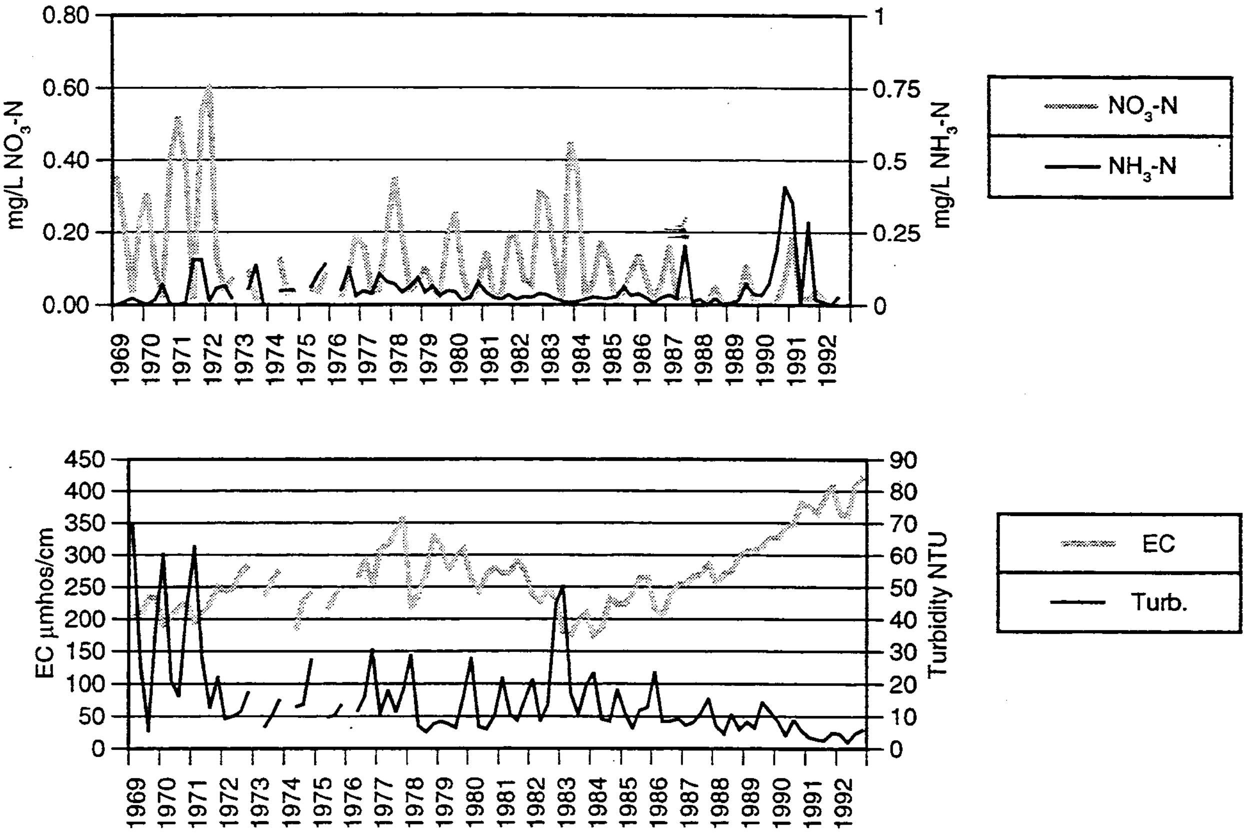


Figure 4.4A. Quarterly averages of inorganic nitrogen species, electrical conductivity, and turbidity for the Upper Arm.

As the conductivity data show, inorganic substances were concentrated during the drought by an aver-age factor of about 2 times **(Figure 3.4).** Summer peaks of total phosphorus are very noisy, but the average increase is twofold or more, as it is for a few other substances in the DWR record, such as boron. Conductivity mostly reflects compounds like sodium chloride and calcium carbonate with a fairly simple response to evaporative concentration. For elements as complex and dynamic as phosphorus and iron, the hypothesis of evaporative concentration is much too simple. A further consideration of the issue of drought levels of phosphorus is reserved for Chapters 5 and **6.**

Inorganic nitrogen cycles out of phase with phosphorus, with peaks of both nitrate and ammonia usually in the winter**. Figures 4.3, 4.4** and **4.6** illustrate these patterns. Peaks of organic nitrogen occur in summer, coincident with the season of algal biomass maxima (most organic nitrogen is presumably in phytoplankton biomass). Winter nitrate maxima show a pattern of very high values in the early high phosphorus period (1969-72), lower and rather variable levels during the second period from the mid 1970s to mid 1980s, and rather average levels during the recent drought. The wet 1992-93 winter showed unusually low winter levels of nitrate. The early years of the drought (1987-88) show higher levels of nitrogen, culminating in a very sharp spike of ammonia in winter 1990-91 in the aftermath of a large *Microcystis* bloom. Limited nitrogen fixation due to low iron levels is the probable reason for low total mass of avail-able nitrogen during the later drought years.

Major improvements in the sewage treatment system (increased sewage hookups and land disposal of effluents) in the late 1970s (see **Figure 3.7**) are not detectable in the phosphorus record. As with mass-balance calculations (Horne 1975; **Chapter 5** below)

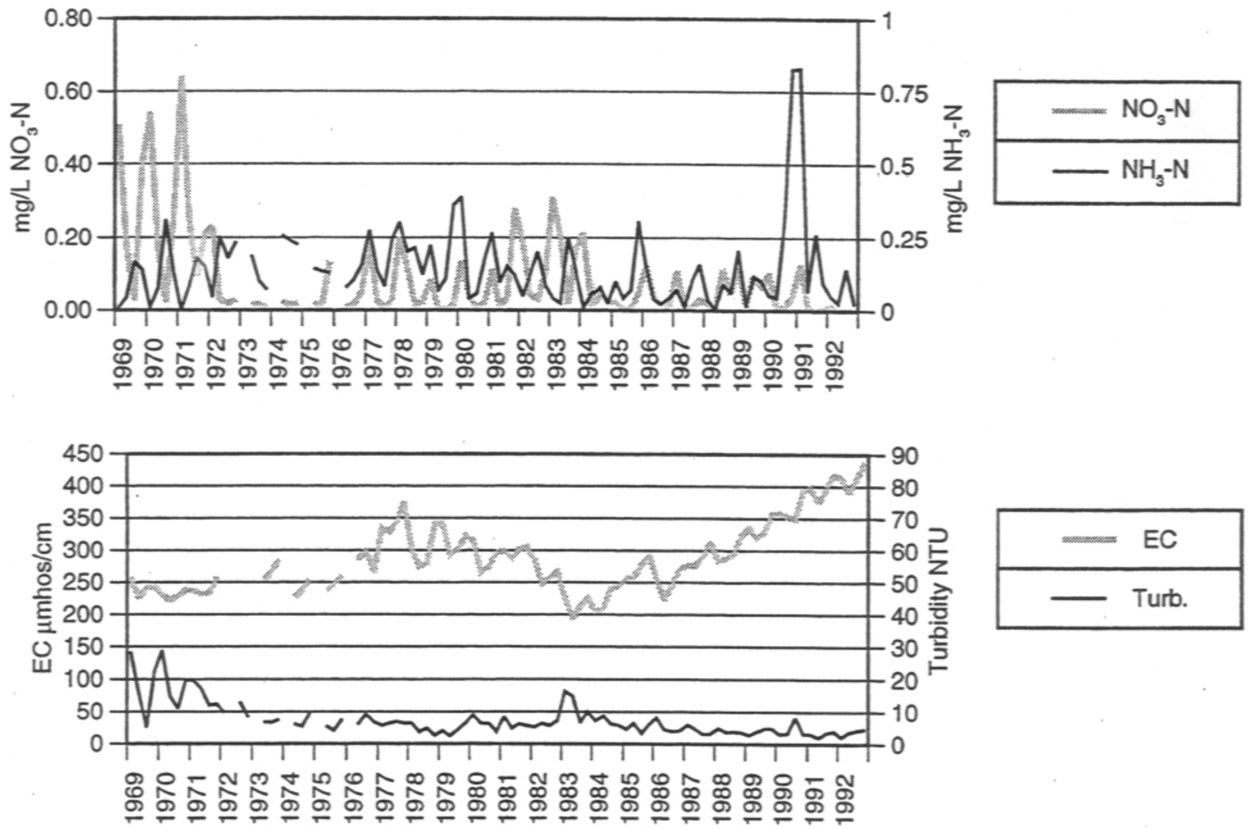


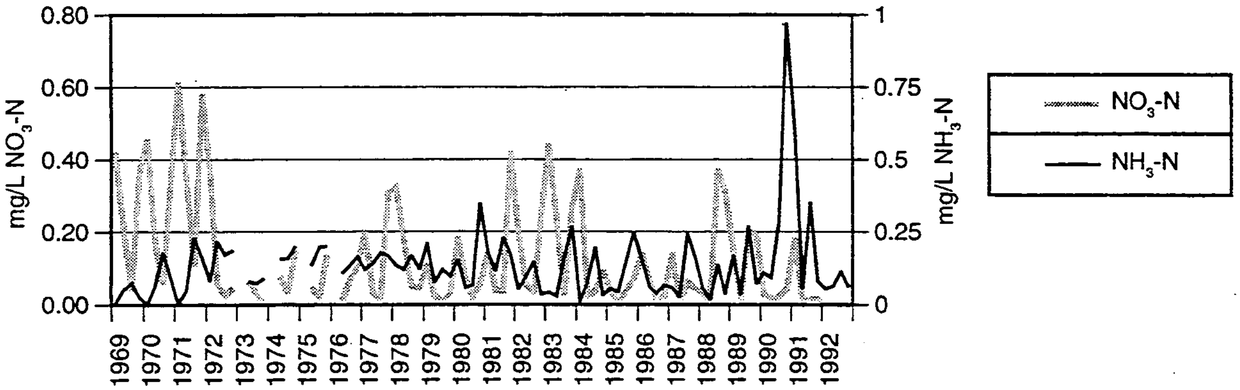
Figure 4.4B. Quarterly averages of inorganic nitrogen species, electrical conductivity, and turbidity for the Lower Arm.

the statistical record indicates the relative unimportance of the sewage waste stream as an influence on lake-wide phenomena. However, in localized, especially enclosed, situations (such as the Clear Lake Keys) sewage input may significantly influence the local nutrient load.

Thus, the long-term data record is generally consistent with the hypothesis Home (1975) derived from his budget estimates. On a lake-wide basis, non-point sources of erosion products are a more important cause of high phosphorus loads than is sewage. Phosphorus apparently increased in the recent drought period due to a lack of dilution by outflow and net losses from the sediments (see **Chapter 5.3).** The storage of winter flood waters and their release during the summer irrigation season tends to accelerate the flow of phosphorus out of the lake during years of significant draw-down in summer. Phosphorus levels are highest in the water column during summer due to internal loading and are much lower in winter, as loading from creeks rapidly precipitates to the sediments. As a result, summer water releases are about twice as efficient at draining phosphorus (and probably iron) out of the system as winter flood flows into Cache Creek (see **Figure 4.3B).**

4.3.2 Phytoplankton Data

The common (see **Section 4.4** for definition) phytoplankton genera that were identified from the DWR database are listed in **Table 4.1.** The relative and absolute abundance of the major scum-forming algae *(Anabaena, Aphanizomenon* and *Microcystis),* as well as other phytoplankton such as diatoms, green algae and flagellates, has varied over 5-10 year periods. **Figure 4.7** summarizes the data for total biomass, broken down into the relative contribution of blue-green and non-blue-green genera. **Figures 4.8** pro-



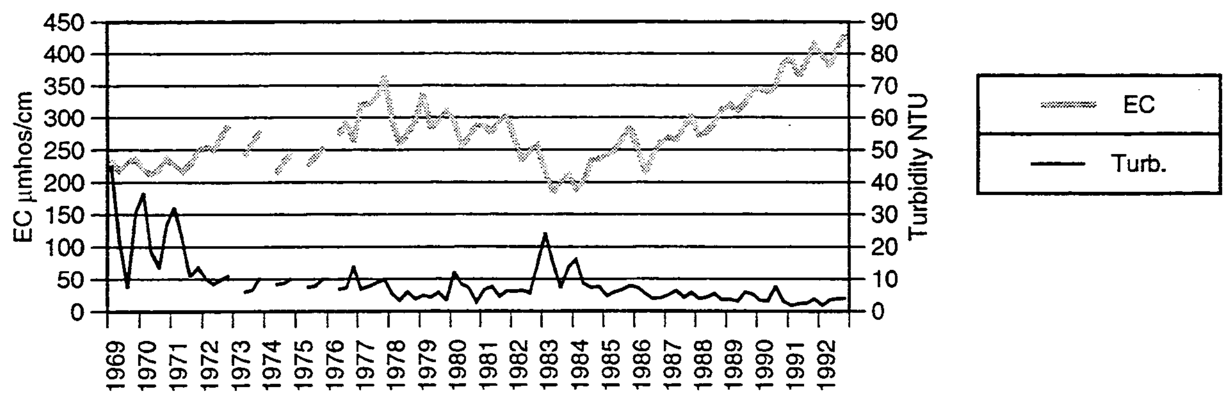


Figure 4.4C Quarterly averages of inorganic nitrogen species, electrical conductivity, and turbidity for the Oaks Arm.

vide a time series of monthly variation in biomass for the three common scum-forming phytoplankton over the entire 23 year period for which the DWR phytoplankton data record has been analyzed. Data were available from DWR for 1992, but sample sizes were too small on most dates to give statistically reliable data after April. The major blue-green dominant, *Gleotrichia,* was not detected, probably due to the small amounts of water counted in the summer and fall months. Hence, we have not used the 1992 data. **Figures 4.9** provide a comparable time series of biomass data for other, non-scum-forming, phytoplankton by major groups. The biomass data are average values over all depths throughout the entire water column at each site for each month.

These data show similar temporal trends but also with some pronounced differences between the three arms of the lake. For example, very few *Microcystis* were documented in the Upper Arm for most of the data set until the very large bloom in 1990. However, there were some noticeable peaks in the mid 1970s in both the Oaks Arm and the Lower Arm with the largest blooms recorded by DWR occurring in the Lower Arm occurring in 1991 rather than 1990. Despite the offset of one year in timing of these events from one arm to another, the physical/chemical conditions generated by the later years of the drought caused significant population explosions of this blue-green. *Aphanizomenon* in general showed similar trends. It was the most important scum former for most years up to 1985, when it suddenly became much less important. The Lower Arm once again most often exhibits higher levels of this genus than does the Upper Arm. This is likely an artifact of the prevailing winds blowing blue-greens entrained in surface waters to the Lower Arm of the lake. *Anabaena* was never responsible for extreme blooms

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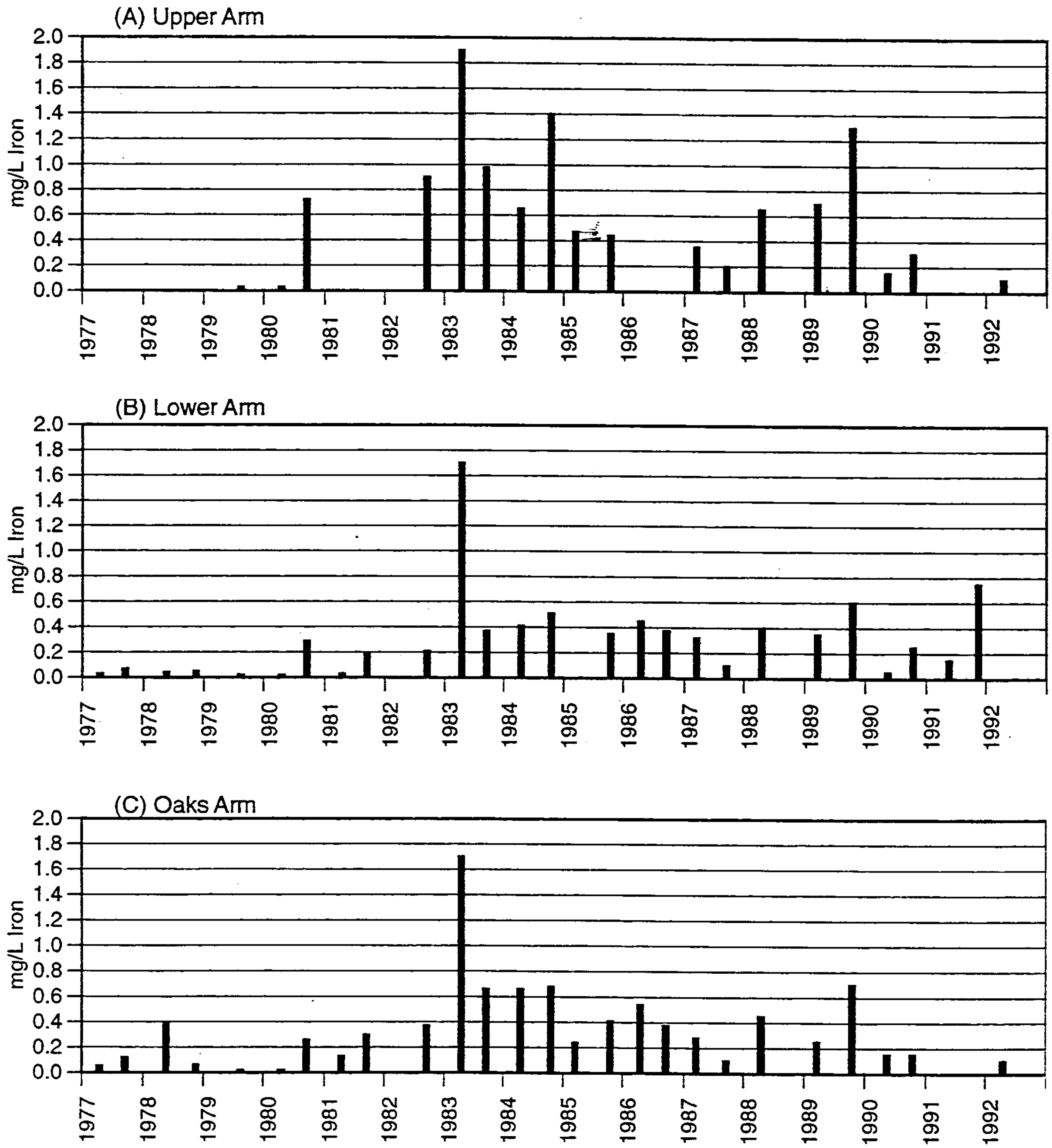


Figure 4.5. Limited water column iron (Fe) measures for the three arms of Clear Lake; (A) Upper Arm, (B) Lower Arm and (C) Oaks Arm. Method switch from dissolved total between the early and late samples in 1980.

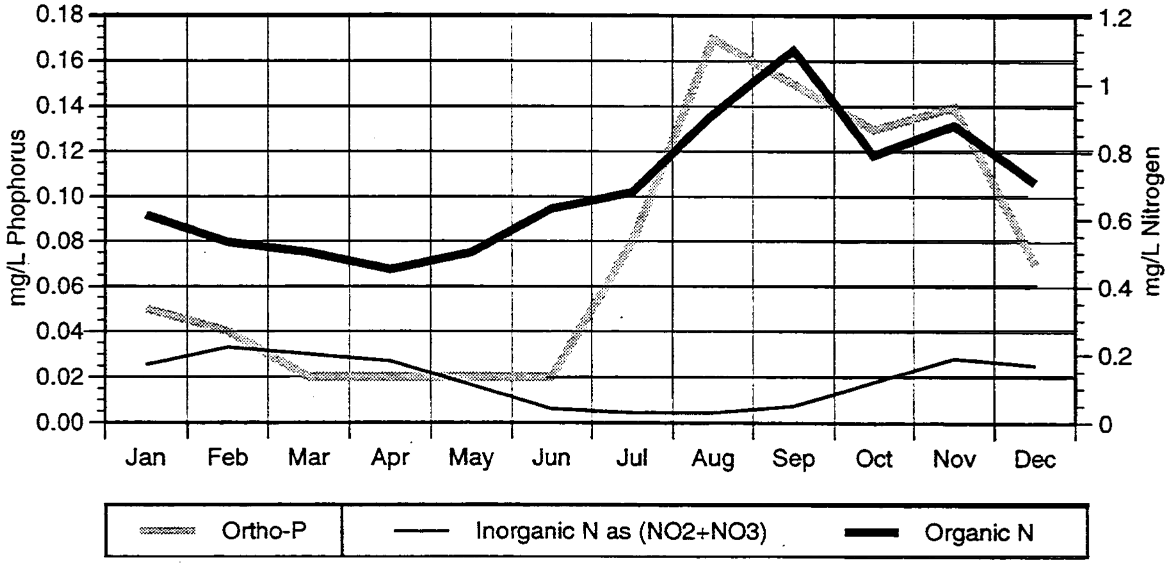


Figure 4.6. Performance of nitrogen and phosphorus as monthly means over 24 years for the Upper Arm station.

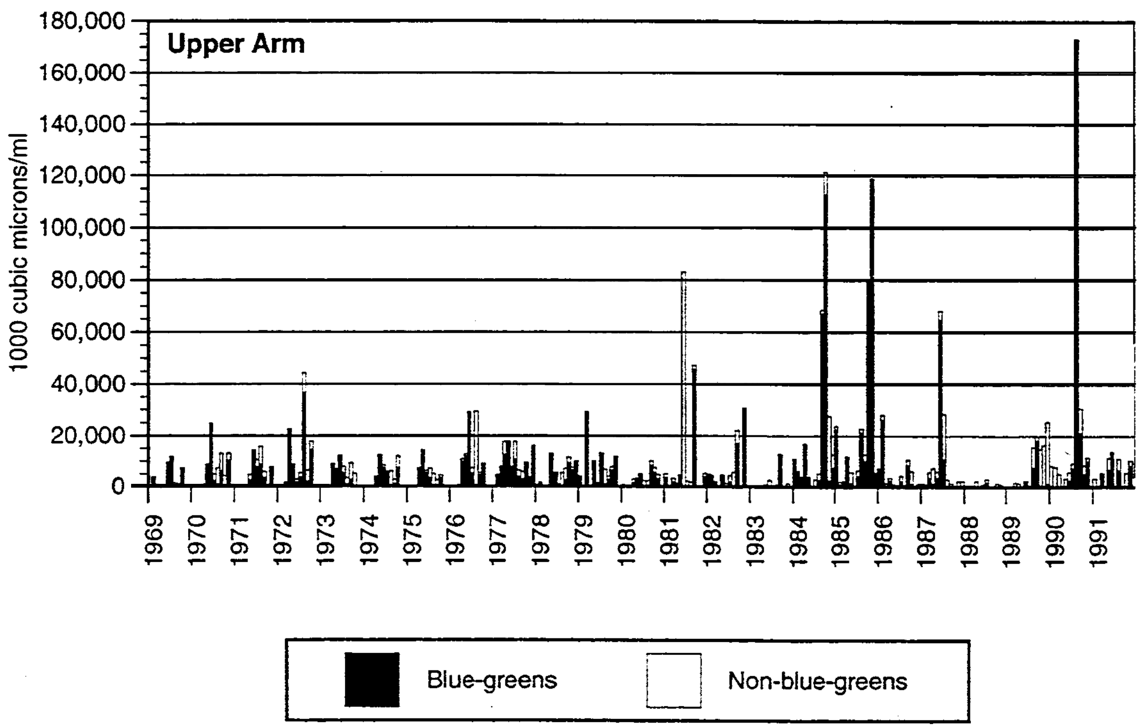


Figure 4.7A. Total blue-green and non-blue-green algae by month for Upper Arm. Values are averages over the water column for each month. The height of the bar is the sum total of all algae.

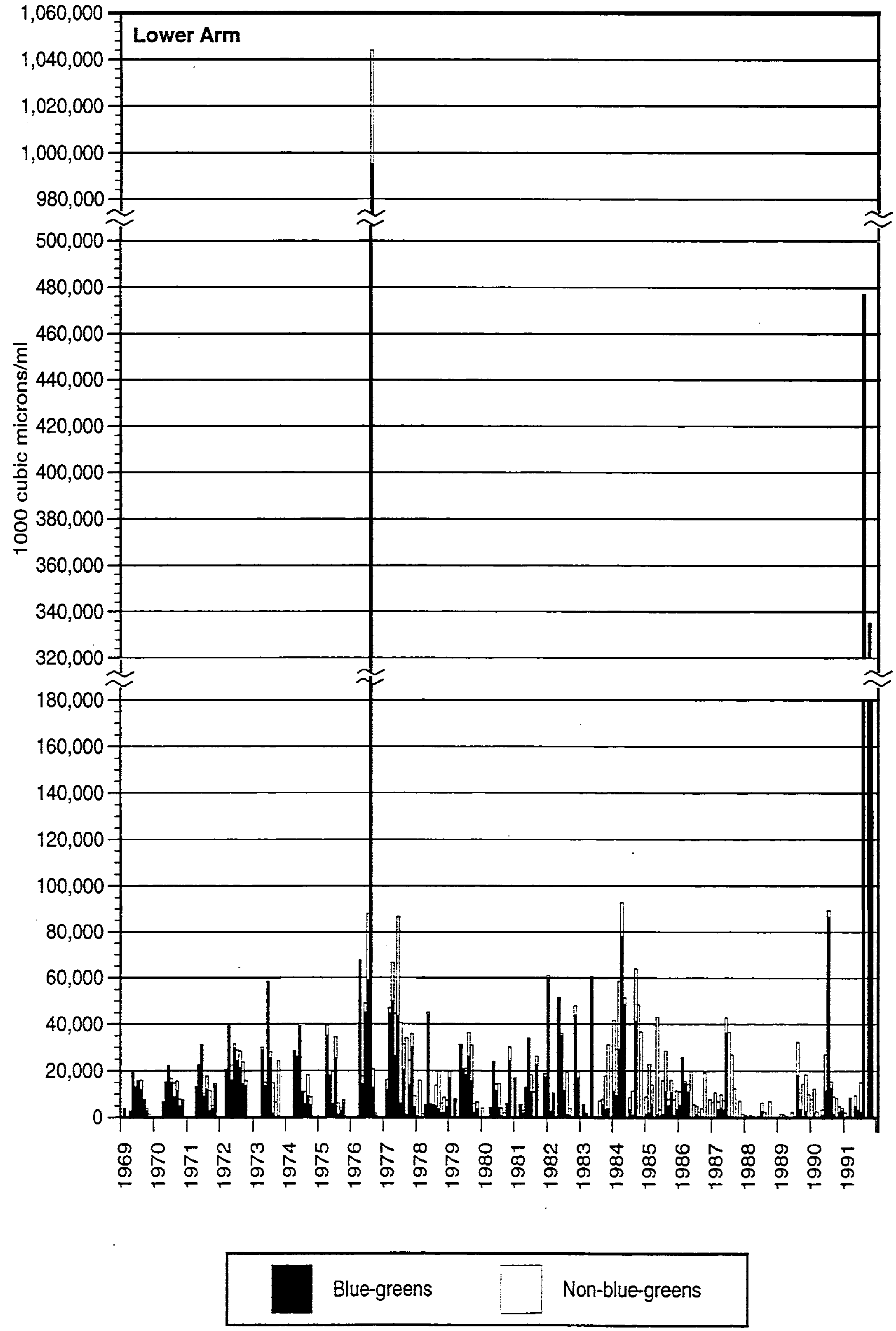

Lower Arm
1,060,000
1,040,000
1,020,000
1,000,000
980,000
500,000
480,000
460,000
440,000
420,000
400,000
380,000
360,000
340,000
320,000
180,000
160,000
140,000
120,000
100,000
80,000
60,000
40,000
20,000
0
1000 cubic microns/ml
1969
1970
1971
1972
1973
1974
1975
1976
1977
1978
1979
1980
1981
1982
1983
1984
1985
1986
1987
1988
1989
1990
1991
Blue-greens
Non-blue-greens

Figure 4.7B (facing page). Total blue-green and non-blue-green algae by month for Lower Arm. Values are averages over the water column for each month. The height of the bar is the sum total of all algae. Because of scaling problems, the figure is broken into three regions. The lower range is comparable to the full range on the other figures in this set.

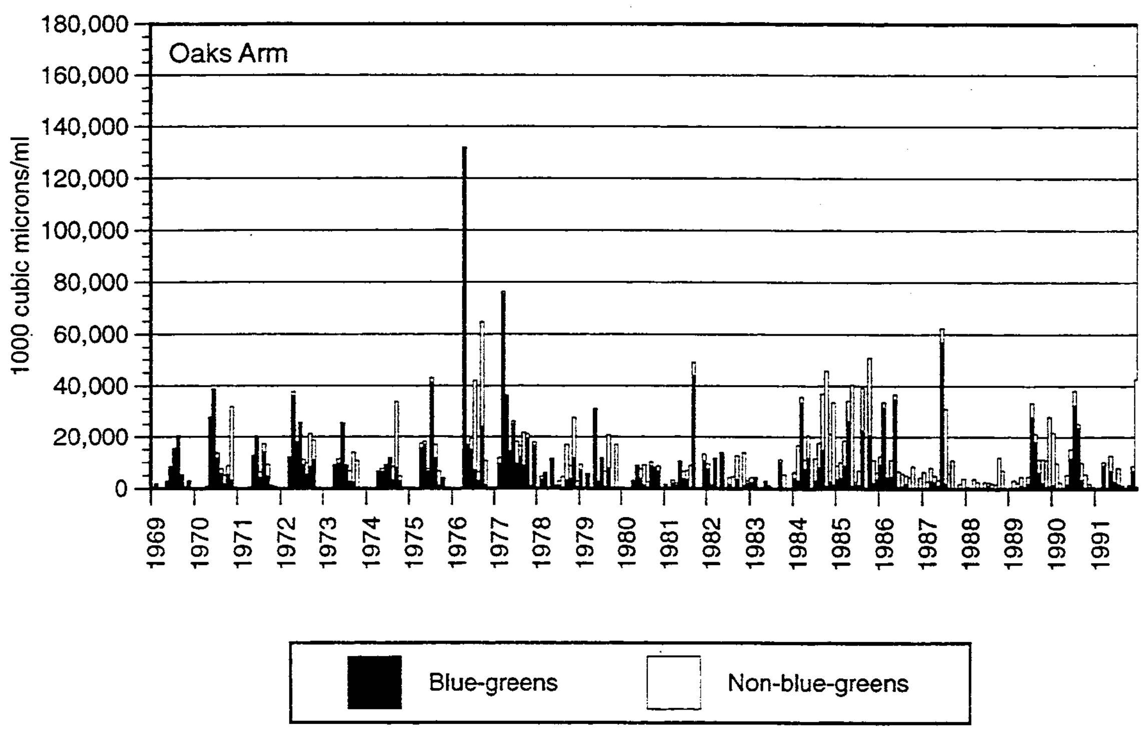


Figure 4.7C. Total blue-green and non-blue-green algae by month for Oaks Arm. Values are averages over the water column for each month. The height of the bar is the sum total of all algae.

Group	Name	'69	'70	'71	'72	'73	'74	'75	'76	'77	'78	'79	'80	'81	'82	'83	'84	'85	'86	'87	'88	'89	'90	'91
B-G Algae	Microcystis	•	•	••	••	•	•	•	••	•	••	•		•	•	•	•	•			•		•••	••
	Aphanizomenon	••	••	••	•••	••	••	••	••	••	••	••		••	••	••	••	•••	••	••			••	••
	Anabaena	•	••	••	••	••	••	••	••	••	••	••		•••	••	••	•••	••	••	••	••	••	••	•
	Oscillatoria	•	•	•	•			•	••	•	•	•		•	••					•			•	
	Lyngbya																•	•	•	••				
	Chroococcus	•	•		•				•	•	•						•	••	••		•		•	
Green Algae	Chlamydomonas			•					•	•	•	•						•	•	•	•	••	••	••
	Schroederia	•			•				•	•	•	•		•	•	•	•	•	•	•	•	•	•	•
	Oocystis	•	••	•	••	••	••	••	•	•	••	••		•	•	•	•	•			•	•	•	•
	Scenedesmus	•		•	••		•	•	•	•	•	•		•	•	•		•	•	•	•	••	•	•
	Sphaerocystis							•	••	•	•	•		•	•		•	•				•	•	•
	Ankistrodesmus	•		•					•	•	•	•		•	•		•		•	•	•	•	•	
	Crucigenia	•			•			•	•	•	•	•		•					•		•	•	•	
	Pediastrum		•	•	•				••	•	••	•		•	•		•	•	•	••		•	•	•
	Closteriopsis		•							•				•			•		•	•		•	•	
	Coelastrum	•	•	•	•	•	•	•	•	•	•			•		•				•			•	
	Closterium									•				•	•		•	•	•		•	•	•	
Diatoms	Cyclotella	•	•	••	•		•	•	•	•	•	•		••	•	•	••	•	••	••	•	••	••	•
	Melosira	•	•	••	••	••	••	••	•	••	•	••		••	••	••	••	••	••	••	••	•	••	••
	Coscinodiscus	•	•	•	••	••	•		••	••	•										•	•	•	
	Stephanodiscus	•					•	•	•	•	••	•			••	•	••	••	••	••	•	•	••	••
	Fragilaria				•						•	•		•	•	•	•	•	•		•	•	•	
	Synedra		•	•															••	••		•	•	•
	Asterionella										•	•			•			•		•	•	•	•	
Flagellates	other Flag.						•					•			•	•	•	•						
	Rhodomonas																	•	•		••	••	••	•
	Cryptomonas	•	••	••	••	••	••	•	••	••	•	•			•			•	••	••	•	••	•	••
	unid. sm. cryptophyceae	••	••	••	••	•	•	••	••	•	•	•		••	••	••		•	•					•
	Trachelomonas	•	•	•			•	•				•		•		•			•	•		•	•	
	Dinobryon	•		•		•								•	•	•		•	•	•		•		
	Ceratium						•			•	•	•		•	••		••		••	•		•		•

Table 4.1. "Common" algal species identified from the DWR Upper Arm data set, and their occurrances during the long-term study. Note that there is no biomass data in the record for 1980. Abundances are approximated as; (•) present, (••) abundant and (•••) very abundant.

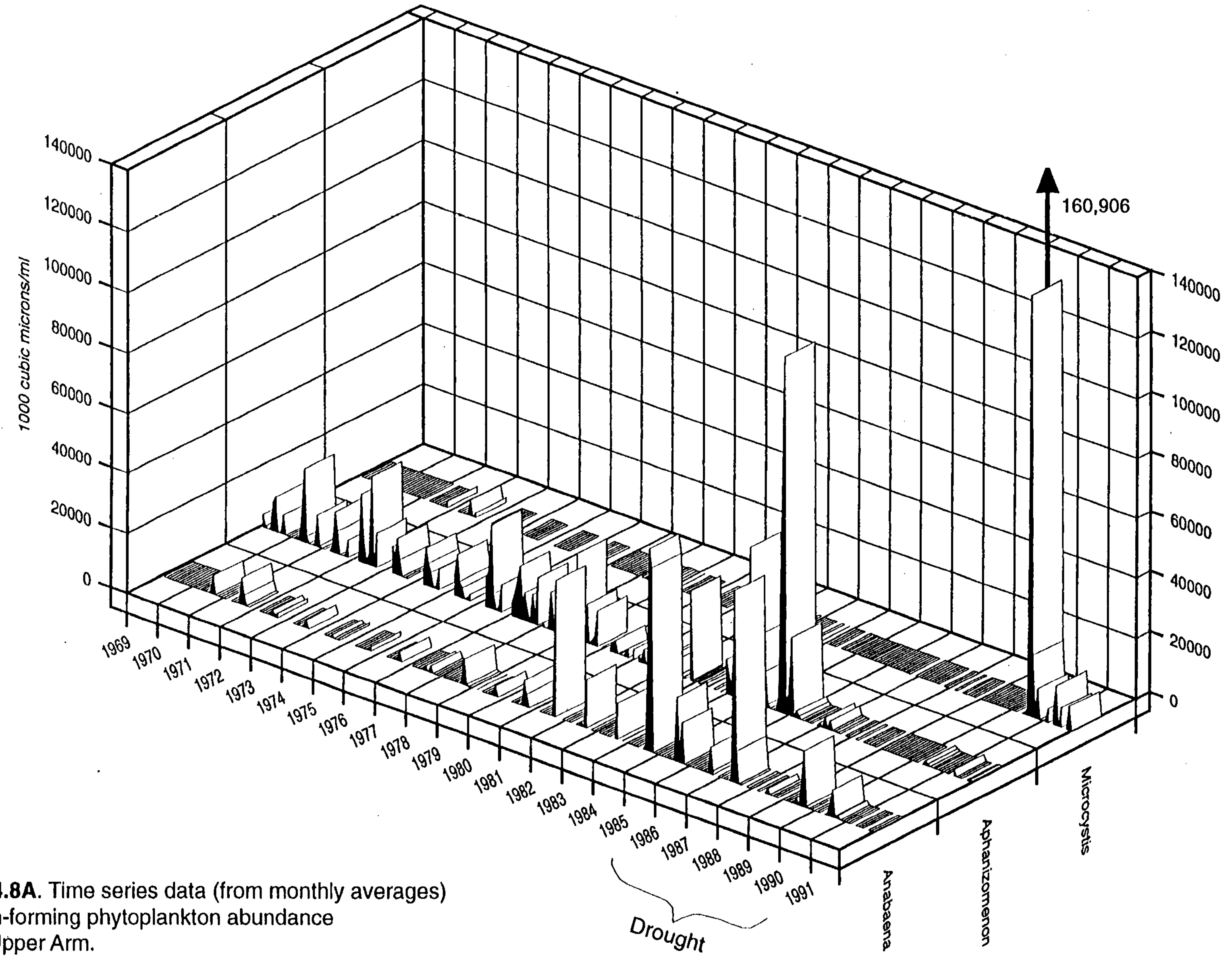


Figure 4.8A. Time series data (from monthly averages) for scum-forming phytoplankton abundance for the Upper Arm.

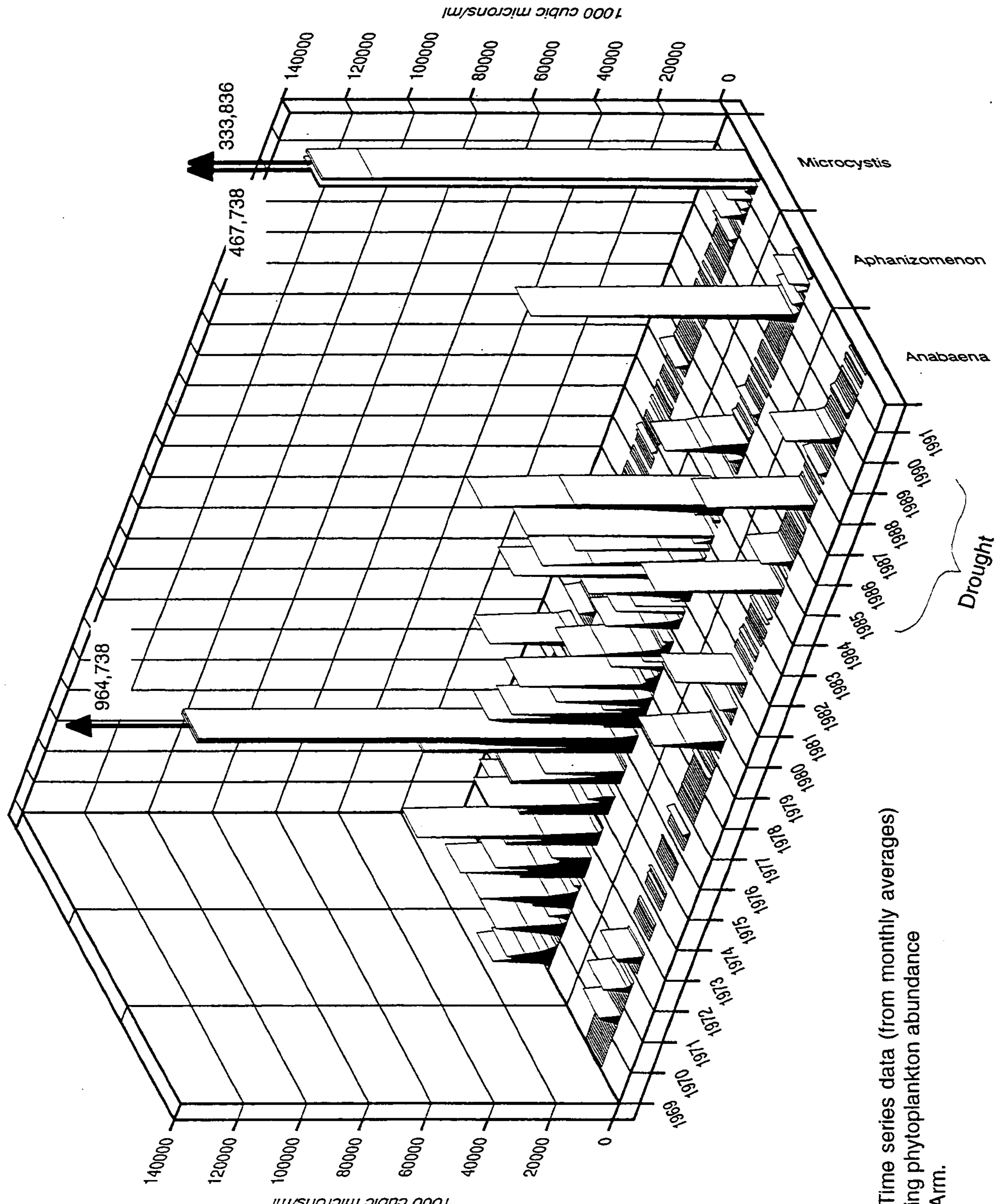


Figure 4.8B. Time series data (from monthly averages) for scum-forming phytoplankton abundance for the Lower Arm.

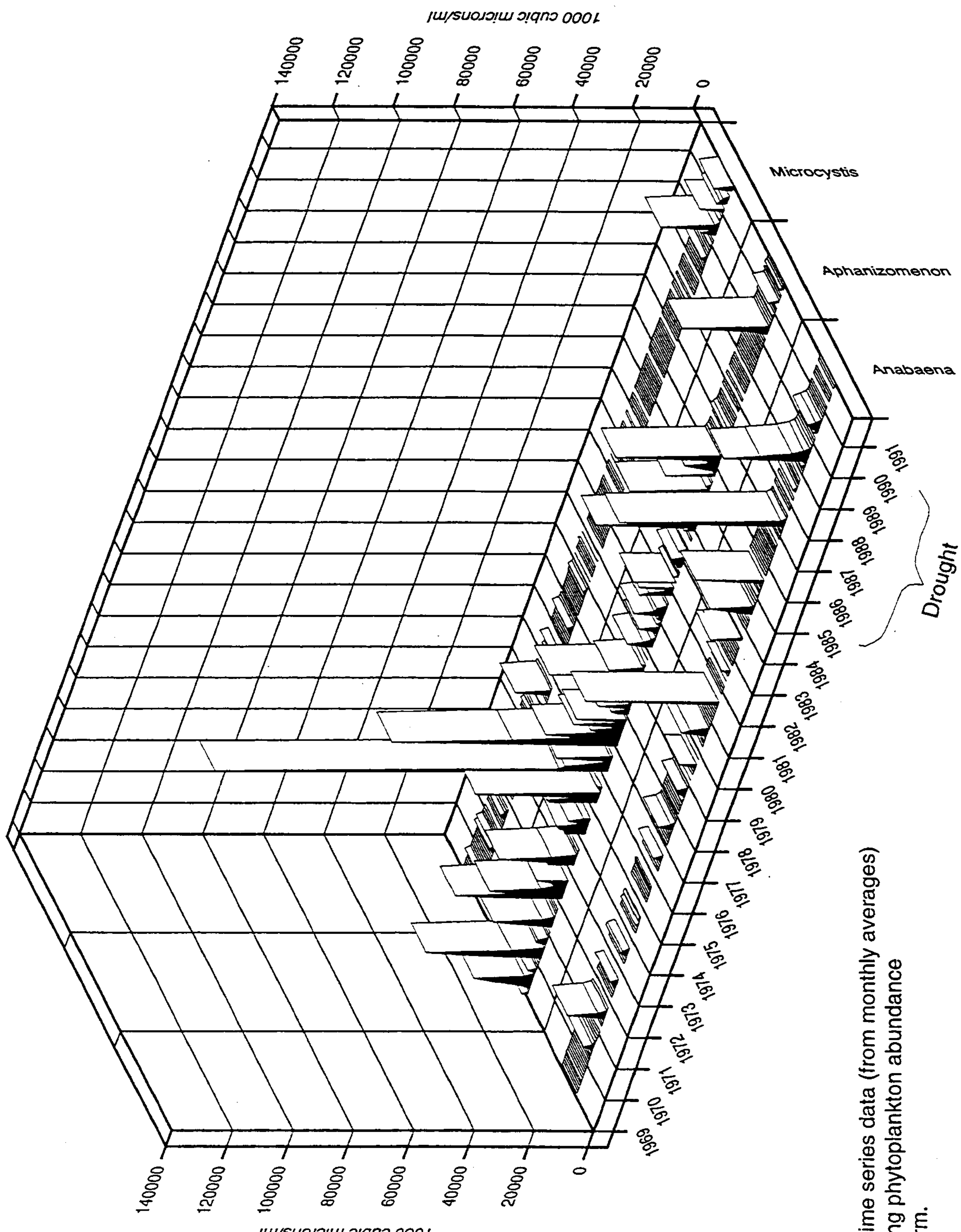


Figure 4.8C. Time series data (from monthly averages) for scum-forming phytoplankton abundance for the Oaks Arm.

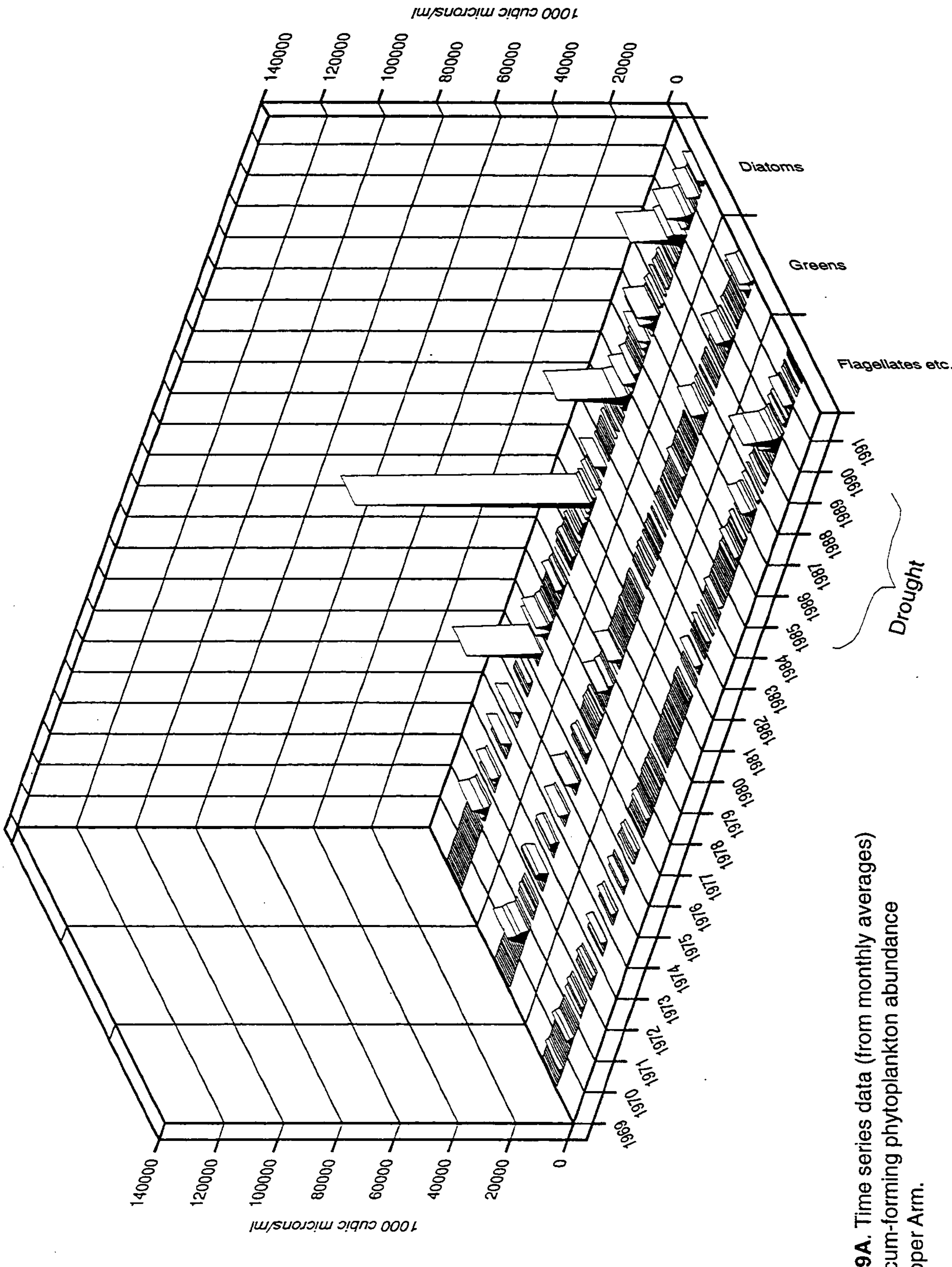


Figure 4.9A. Time series data (from monthly averages) for non-scum-forming phytoplankton abundance for the Upper Arm.

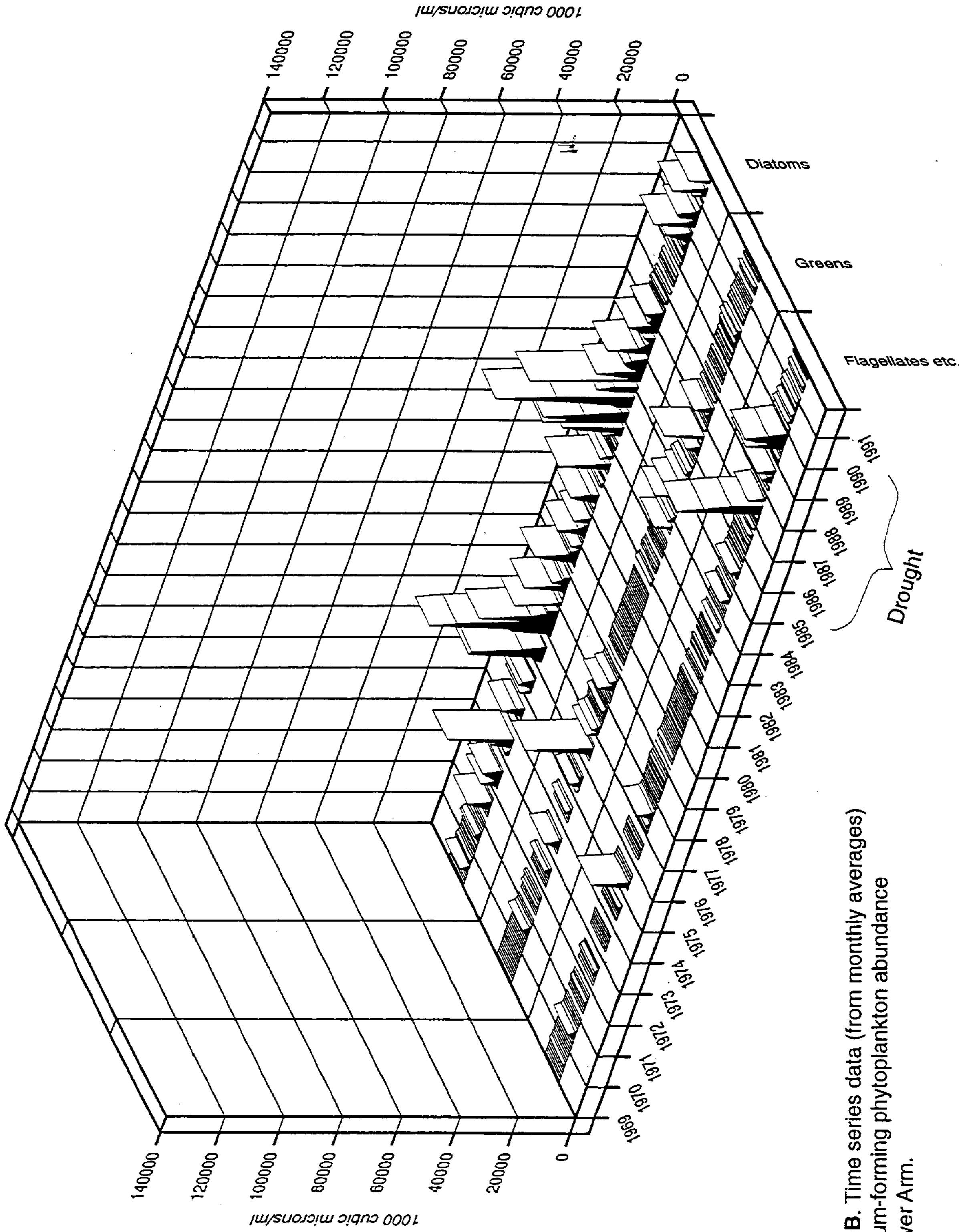


Figure 4.9B. Time series data (from monthly averages) for non-scum-forming phytoplankton abundance for the Lower Arm.

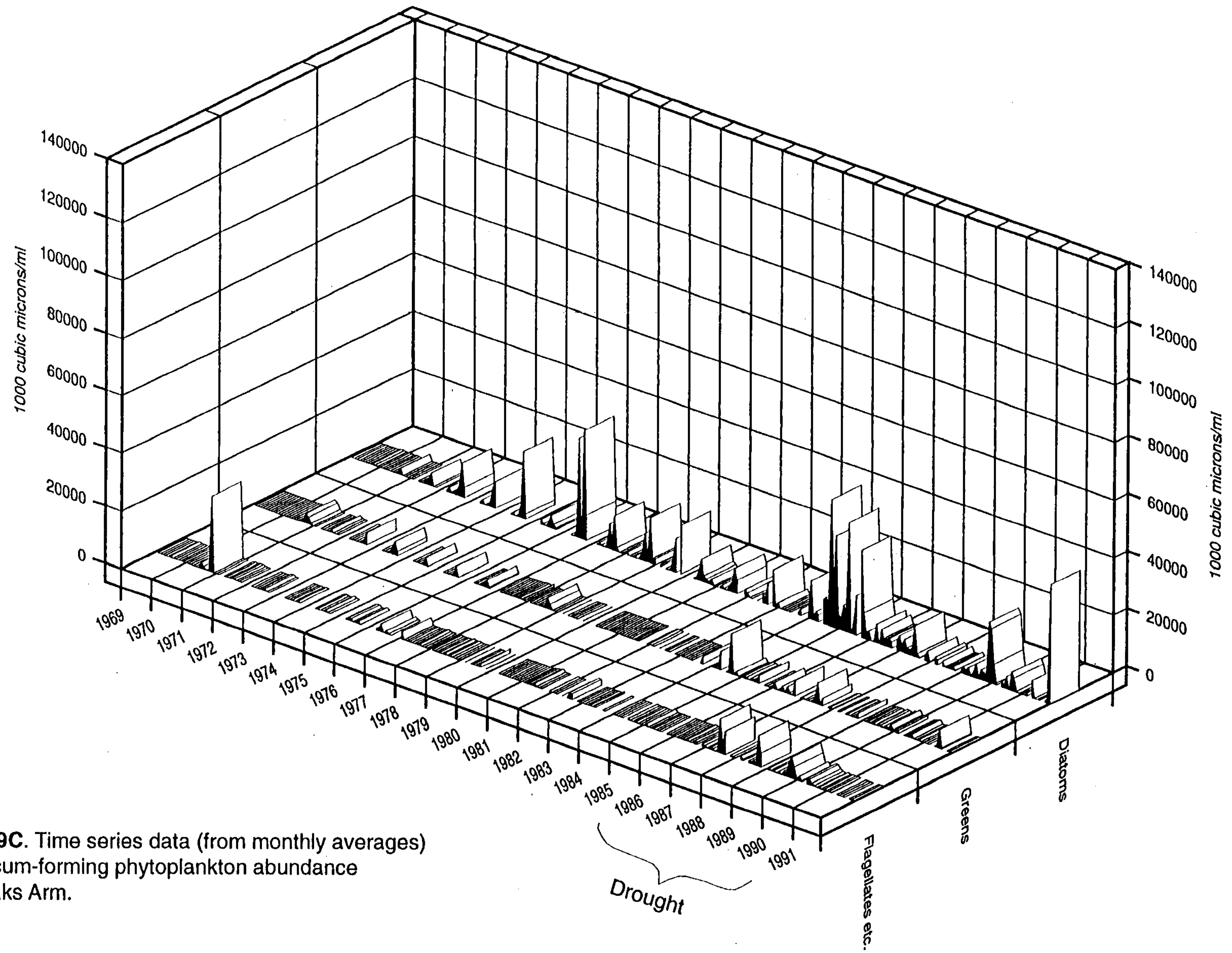


Figure 4.9C. Time series data (from monthly averages) for non-scum-forming phytoplankton abundance for the Oaks Arm.

Continued from page 15

on the scale of *Microcystis* and *Aphanizomenon* in any arm, but was a common component of the midsummer scums throughout the record, becoming relatively much more important after 1980 before diminishing as the drought progressed. These trends sup-port the evidence provided in **Chapter 3 (Figure 3.9),** which compares relative abundances of floating (scum-formers) and non-floating (non-scum formers) algae in the Upper Arm with both the Oaks Arm and the Lower Arm. We believe that wind is the primary factor in driving more buoyant genera from the Up-per Arm into the Oaks Arm, but environmental differences may contribute significantly in explaining higher levels in the more isolated Lower Arm.

During the mid-1970s to mid-1980s period, *Aphanizomenon* blooms tended to occur in summer and fall, and even late fall, quite unlike the spring pattern observed by Horne. In general, the importance of *Anabaena* increased somewhat relative to *Aphanizomenon*. The lower phosphorus levels of the mid-1970s to mid-1980s period appeared to curtail the spring bloom, but there were still annual episodes of ample phosphorus in summer and fall, permitting significant *Aphanizomenon* and *Anabaena* blooms.

During the later drought years, large blooms of the scum formers occurred, especially in the summer and fall of 1989 and 1990, supported by the large internal load of phosphorus. The culmination of this trend was the very large (and highly visible) bloom of extremely buoyant *Microcystis* which occurred in the late summer and fall of 1990. This bloom caused serious air quality problems and even restricted boat passage in some regions of the lake (e.g. Horseshoe Bend and the end of the Oaks Ann outside the Keys subdivision). In addition to the biomass data, There is chemical evidence for the bloom in the very high pH (Figure 4.1) and organic nitrogen found during the bloom period. Very high levels of ammonia during the winter 1990-1 (Figure 4.4) also apparently

Later in the drought, Clear Lake water became extremely transparent compared to the rest of the record (beginning in 1991, see **Figure 3.5),** and had low bio-mass of all scum formers, despite a very large excess phosphorus supply. Genera never before recorded from the lake became dominants (e.g. *Gleotrichia* since 1992, identified by PJR, verified by Norman Anderson, personal communication). Submerged water-weeds (dominated by *Potamogetons)* grew abundantly in shallow water **(Chapter 8).**

Our hypothesis is that drought conditions eventually reduced the iron release from the sediments sufficiently to severely inhibit blue-green growth (see Section 5.3.11). This hypothesis is consistent with the measured fall of iron in the very limited DWR data (see Figures 4.5), with the bioassay response of phytoplankton to iron in the CLARU studies, and with our own investigations **(Chapter** 7). Phosphorus concentrations increased dramatically during the drought due to internal loading **(Chapters** 5 and 6), and scum-formers certainly were not limited by this element. Unfortunately, the lack of detailed study of either iron dynamics or phytoplankton responses to nutrients until the very end of the 1991-92 clear water period mean that this hypothesis cannot be tested as thoroughly as is desirable. For example, some other important metal such as molybdenum could also be limiting. The data on iron were too limited to include in the following statistical analysis, a serious handicap if the iron limitation hypothesis is correct.

4.4 Statistical Analysis Of DWR Water Quality Data Set

A statistical analysis was performed on the 24 year (1969-1992) DWR data set for physical/chemical data and the 23 year (1969-1991) DWR data set for plank-ton. This analysis focused on describing the trends in phytoplankton community dynamics and the relationship of these changes to nutrients and other environmental variables. The patterns of

For the purposes of analyzing the variability in both physical/chemical and biological data as a function of time throughout the year, and to analyze relation-ships between physical/chemical and biological data, seasonal averages were computed from each of the monthly sampling dates for each of four seasons of the year. The four seasons were defined as: Winter = Jan, Feb, Mar; Spring = Apr, May, Jun; Summer = Jul, Aug, Sep; Fall = Oct, Nov, Dec. For the biological data logarithmic averages were used and all biomass values (not missing values) were increased by one to eliminate values of zero (for which the logarithm is undefined). A subset of common genera were selected for further analysis based on the following criteria: (1) non-zero yearly average in at least half of the time series (11 of 23 years) or (2) occurrence in less than half of the record but exhibited a clear peak or (if near start/end of the record) a clear declining/ increasing sequence. The minimum sequence length to. qualify for selection was 5 years. The selection procedure identified 31 common genera, most of which were chosen because of the first criteria **(Table 4.1).** The estimated biomass of each genus was used in the analysis because phytoplankton cells and colonies vary enormously in size, even within a genus.

In order to reduce the extreme variability from one quarter to another, the data were further smoothed by a seasonal adjustment. The seasonal pattern of each genus was computed as the average over all 23 quarters in the data (grand seasonal mean) and then each averaged seasonal value was normali zed to the grand seasonal mean. Therefore, the variations being analyzed are the deviations from the "typical" seasonal pattern in any one year. Again, logarithmic averages were used for the biological data. The seasonal adjustment was only partially effective at smoothing the physical chemical data, mainly because the "typical" seasonal pattern changed so much from one time to another (e.g. in drought vs. nondrought years). Additional smoothing techniques were therefore used to further decrease the contribution of seasonal variability in the physical/ chemical data. A robust filtering method, locally weighted least case inter-seasonal variation for each variable. The relative magnitude of the component scores for the physical/chemical variables is a reflection of the degree to which their variations are correlated, either directly or inversely. That is, when one variable is increasing or decreasing in magnitude, another variable (or set of variables) is also changing in roughly the same magnitude (regardless of the direction of change). The first combination of variables (Principal Component 1) is the most significant statistically defined explanation of variability in the data. Principal Component 2 is the next best combination of variables that explains as much of the remaining variability as possible in the data not explained by Principal Component 1. Using similar logic, Principal Component 3 represents yet another combination of variables that explains as much of the still remaining variability in the data. Conceptually, one hopes to explain as much variability as possible with the least number of Principal Components. Theoretically, there can be 20-30 "components", but after the first few they generally account for very little of the variation in the data. We have chosen to use only three Principal Components in our data presentation.

Informally, Principal Component 1 gives the "big picture," Component 2 is a "major wrinkle," and Component 3 is "something interesting."

Physical/ chemical Principal Component 1 most likely represents a "drought factor," " Principal Component 2 most closely approximates a "yearly temperature factor," and Principal Component 3 appears to reflect "major differences between arms." The loading scores that measure the influence of each variable on the computed Principal Components are shown Figures 4.10, and the time course of each Component is shown in Figures 4.11.

The PCA Loading Scores in **Figure 4.10** show that Principal Component 1 is negatively correlated to turbidity and nitrate (which vary together) and positively correlated with the variation in electrical conductivity, alkalinity, total and dissolved phosphorus. This first

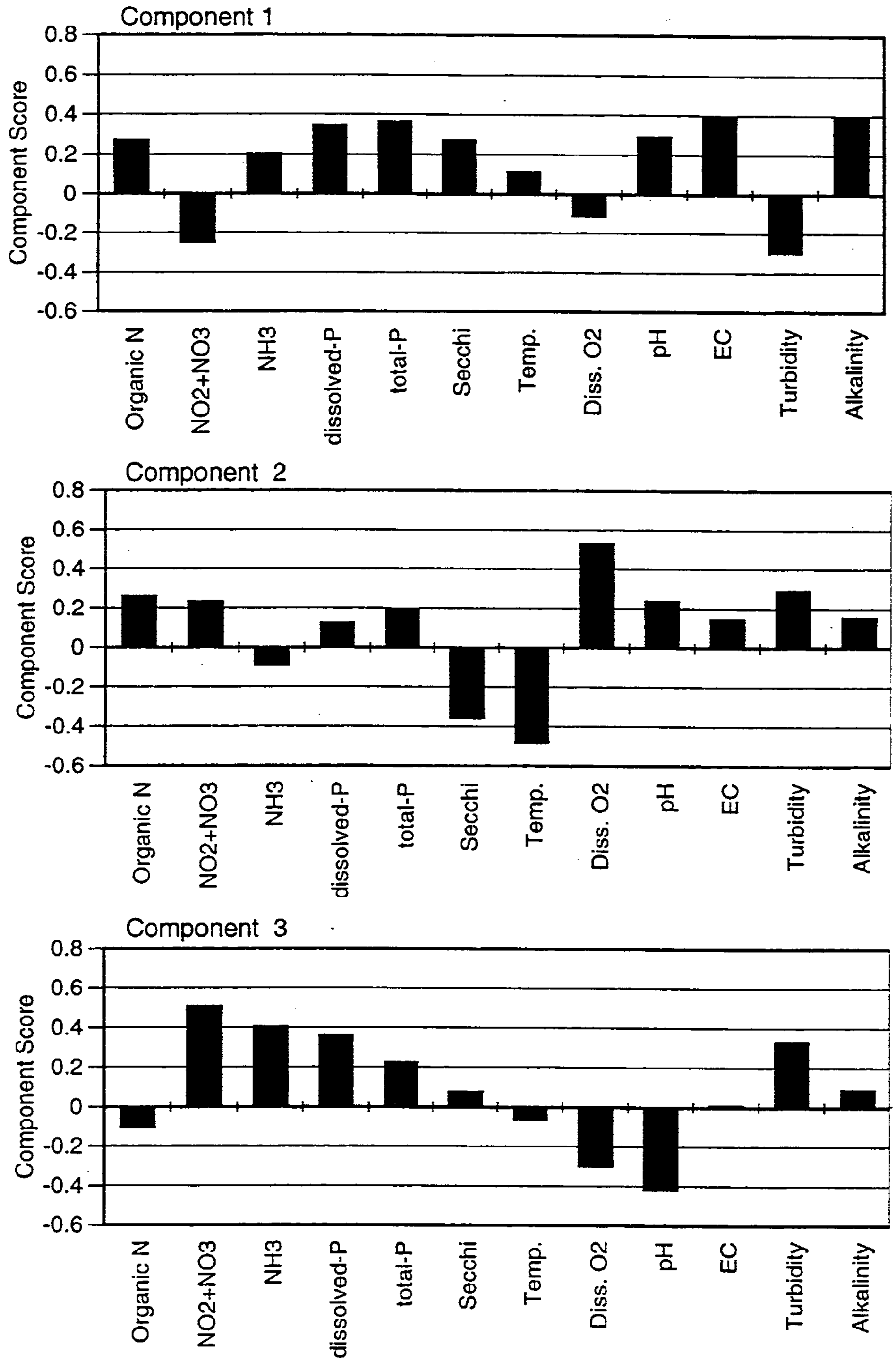


Figure 4.10B Lower Arm Principal Component loading scores for physicaUchemical parameters.

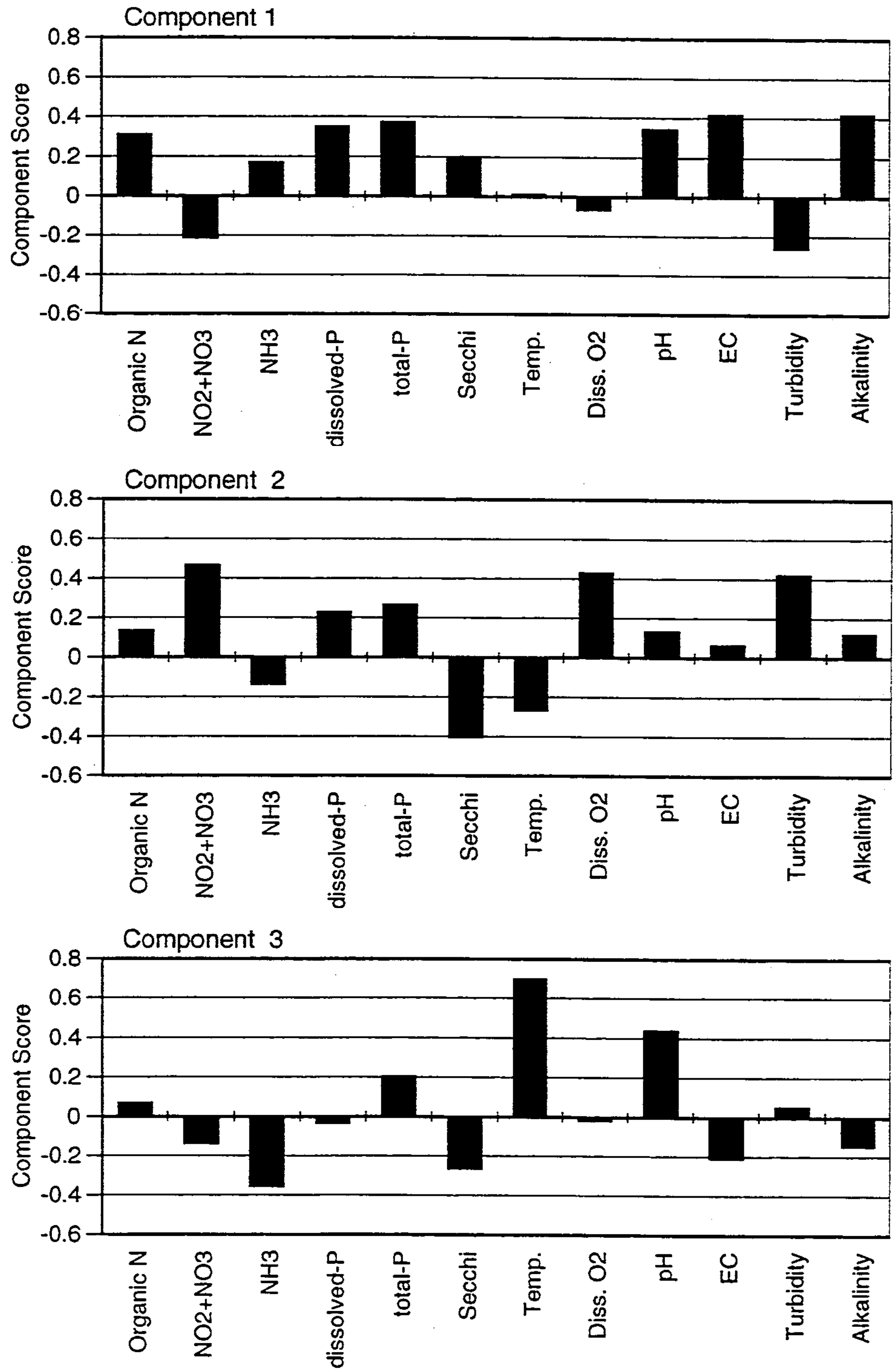


Figure 4.10B Lower Arm Principal Component loading scores for physicaUchemical parameters.

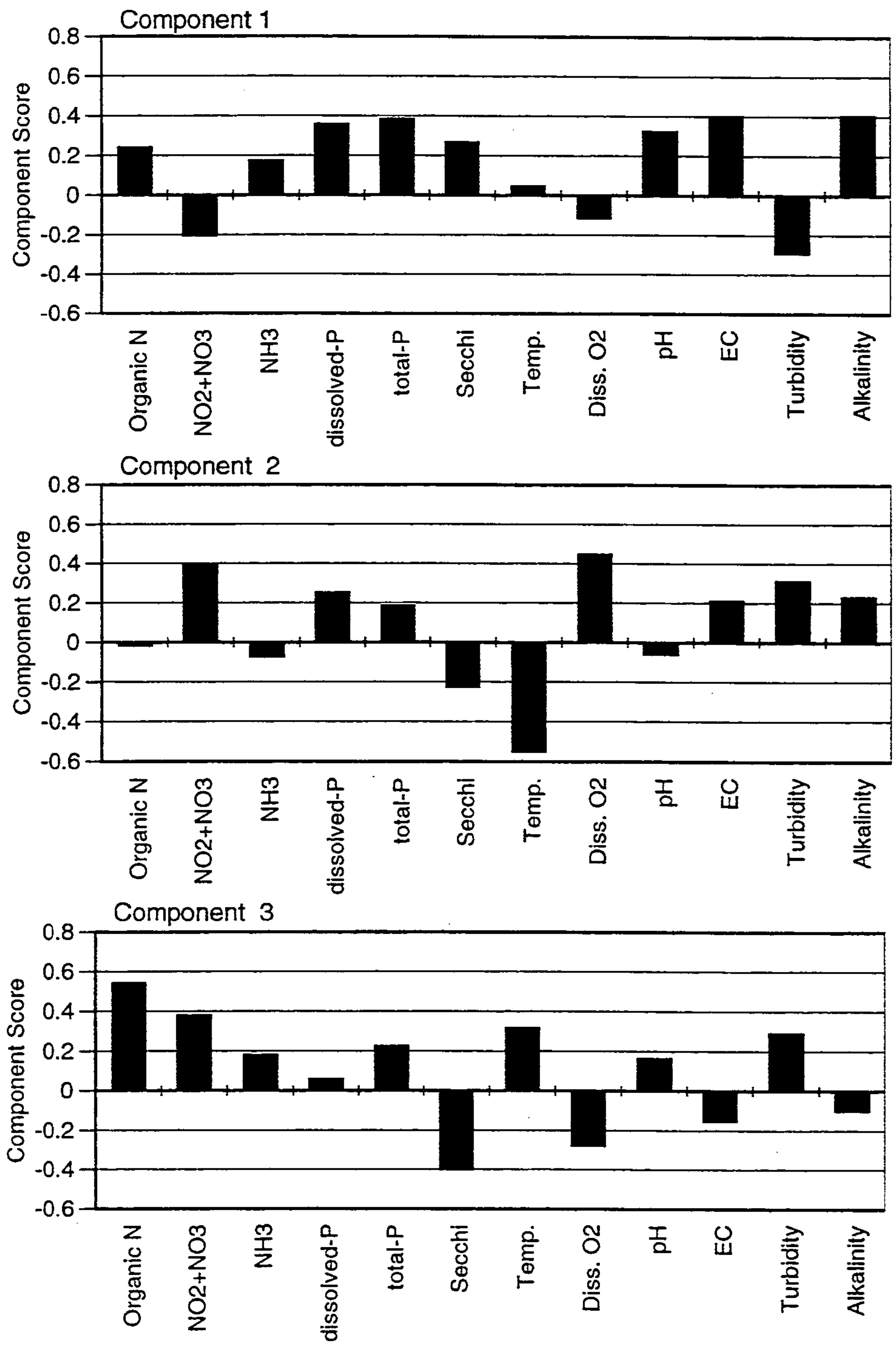


Figure 4.10C. Oaks Arm Principal Component loading scores for physical/chemical parameters.

significant "driver" for physical/chemical variation in Clear Lake appears to be chemical variations associated with dilution during runs of high-rainfall years and evaporative concentration due to runs of drought years, with the recent long run of dry years producing the most significant physical/chemical signal in the water column for the entire 24 year data set.

The next most significant amount of the remaining variability is accounted for by Principal Component 2, which is negatively related to temperature and Secchi depth and positively related to variations in dissolved oxygen and to a lesser extent turbidity, pH, organic nitrogen and nitrate. 19-23% of the total variation is associated with Principal Component 2. This Component was high in the late 1960s and early 1970s, and again at the very end of the record. It rather clearly reflects the influence of warmer and cooler years, the next most significant "driver" of physical/ chemical variations in Clear Lake after drought effects are removed from the data. Since oxygen is more soluble in water at cooler temperatures, the inverse relationship between the two variables is to be expected.

Principal Component 3 accounts for a relatively small amount of the total physical/chemical variation (11-12%), and the pattern of loadings is quite different in the Upper Arm compared to the two lower arms, which are rather comparable to each other. Nitrate and ammonia loadings in the Oaks Arm, however, resemble those in the Upper Arm, not the Lower Arm. As we have noted before, each arm is somewhat unique due to partial isolation (especially of the Lower Arm), and the accumulation of the buoyant scum formers in the downwind arms (especially Oaks Arm). Component 3, reflects these differences; it is an "Arm factor."

In total, the first three Principal Components account for 73-82% of the variation in physical and chemical measurements. The physical/chemical properties of the lake have been dominated by a few simple patterns over the 24 year record, especially the drought "cycle."

4.4.2 Principal Component Analysis Of Phytoplankton Data

We have analyzed the variability in these data with PCA using both annual and quarterly averages. The two averaging methods give very similar results, and, as with the physical/chemical data, we report only the quarterly data.

The Principal Component loading scores for the quarterly biomass data of each genus for the first three Principal Components is shown in Figure 4.12, and the quarterly time series analysis for each factor is plotted in Figures 4.13. Biological Principal Component 1 most likely represents a "taxonomic factor," Principal Component 2 most closely approximates a "biomass factor," and Principal Component 3 may reflect biological differences between arms induced by wind patterns that tend to concentrate more buoyant forms in the two lower arms or other "between arms" differences.

The PCA loading scores for Principal Component 1 for the quarterly phytoplankton biomass data (Figures 4.12 and **Table** 4.2) show that the blue-greens *Aphanizomenon, Microcystis* and *Oscillatoria* strongly co-occur, while a collection of diatoms, cryptomonads, some green algae and the scum former *Anabaena* occur together. The genera loading positively on this component tend not to co-occur with those loading negatively. The genera associated with Component 1 are very similar in each arm, and Component 1 accounts for 37-39% of the total variation in the biomass data. **Figures 4.8 and 4.9 and Table** 4.2 all indicate that Principal Component 1 is likely representative of a "taxonomic component." That is, when *Aphanizomenon is* abundant, a collection of other genera including *Anabaena* are scarce, and vise-versa. Most notably, Component 1 is very low in the early *Aphanizomenon* dominated years of the record, and notably high in the late 1980s when diatoms *(Fragilaria, Cyclotella, Synedra, Melosira* and usually *Coscinodiscus)* cryptomonads *(Trachelomonas* and *Rhodomonas)*, and greens *(Chlamydomonas, Scenedesmus, Ankistrodesmus)* dominated the flora. Table 4.2 also provides a quick heuristic overview (for Principal Component 1) of the patterns of co-occurrence of various genera, emphasizing the ways in which specific taxa tend to group similarly. Note that this factor has a lake-wide pattern, with similar loadings in all three arms.

Component 2 in the PCA accounts for 13% of the variation in the Upper Arm and 20-21% in the two Lower Arms. The loadings of genera on this Component are also similar in the two lower arms, but some-what different in the Upper Arm. The abundant diatom and green as well as blue-green genera tend to have large positive loadings on this Component. The patterns of quarterly variation have a general similarity in all three arms. Principal Component 2 seems to reflect an overall "biomass factor," as can be seen by comparing **Figures** 4.13 with **Figures** 4.7.

Phytoplankton Principal Component 3 accounts for 10-12% of the variation. Once again, the pattern of

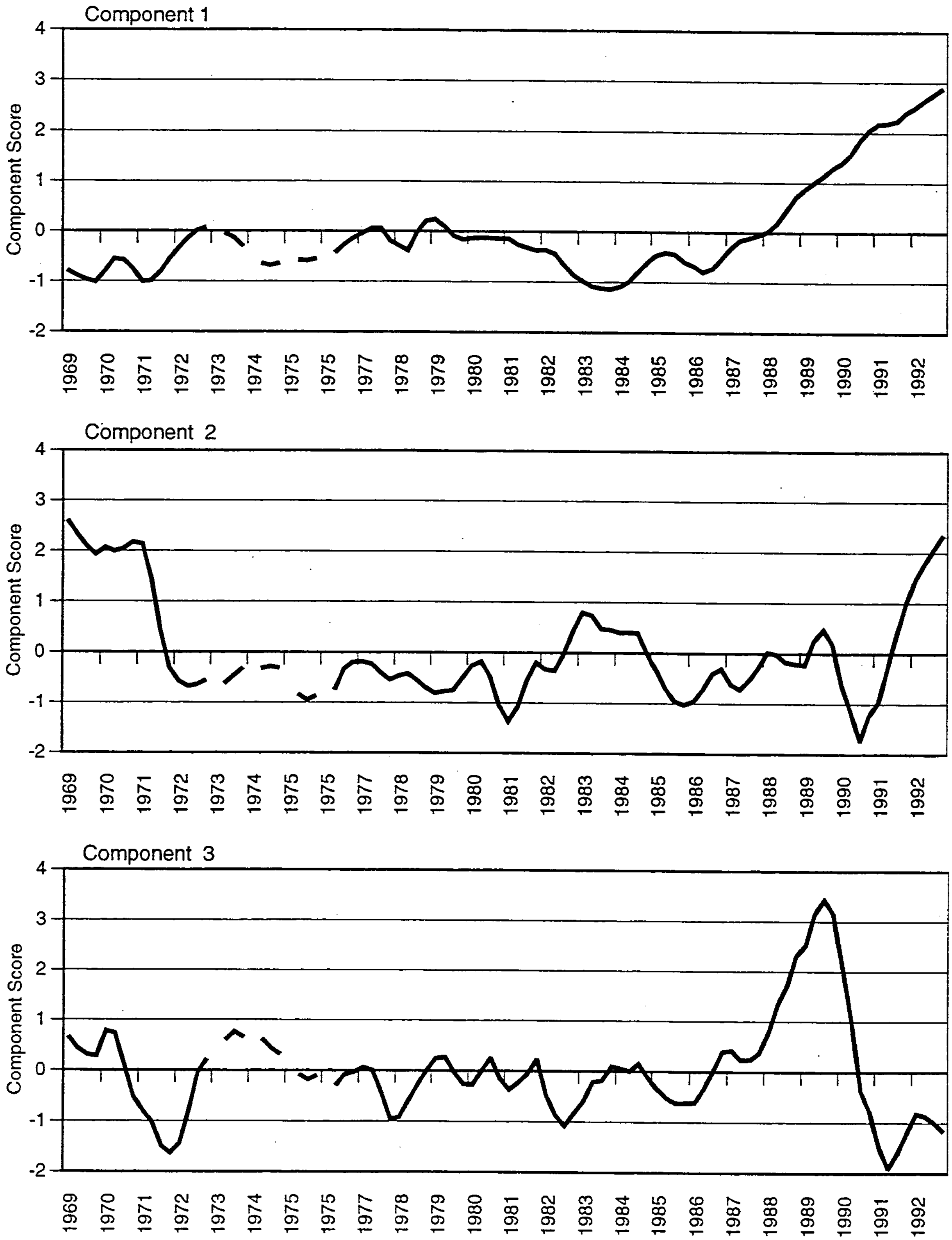


Figure 4.11A Upper Arm Principal Component loading scores for time series data (using smoothed quarterly averages) for physical/chemical parameters.

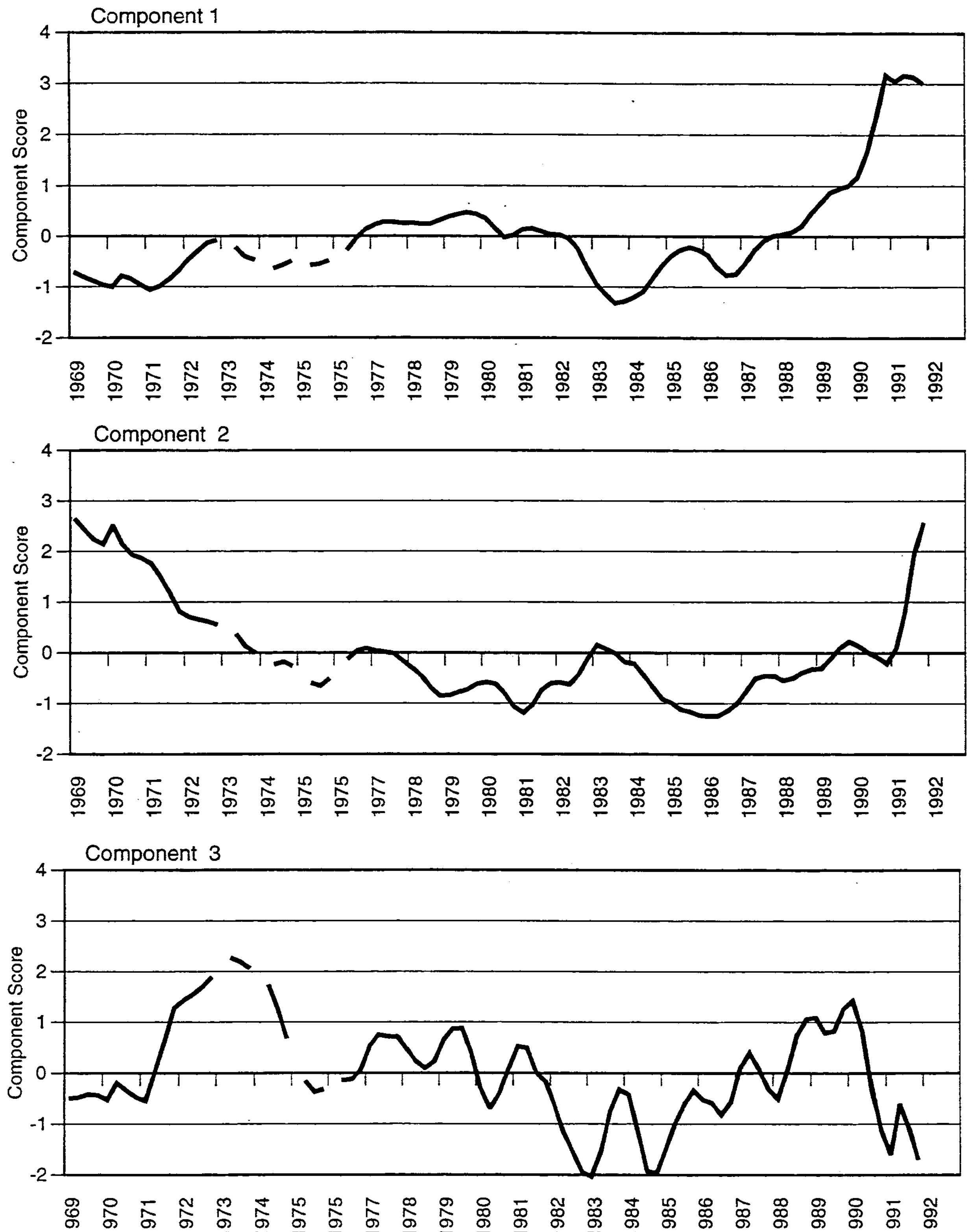


Figure 4.11 B Lower Arm Principal Component loading scores for time series data (using smoothed quarterly averages) for physical/chemical parameters.

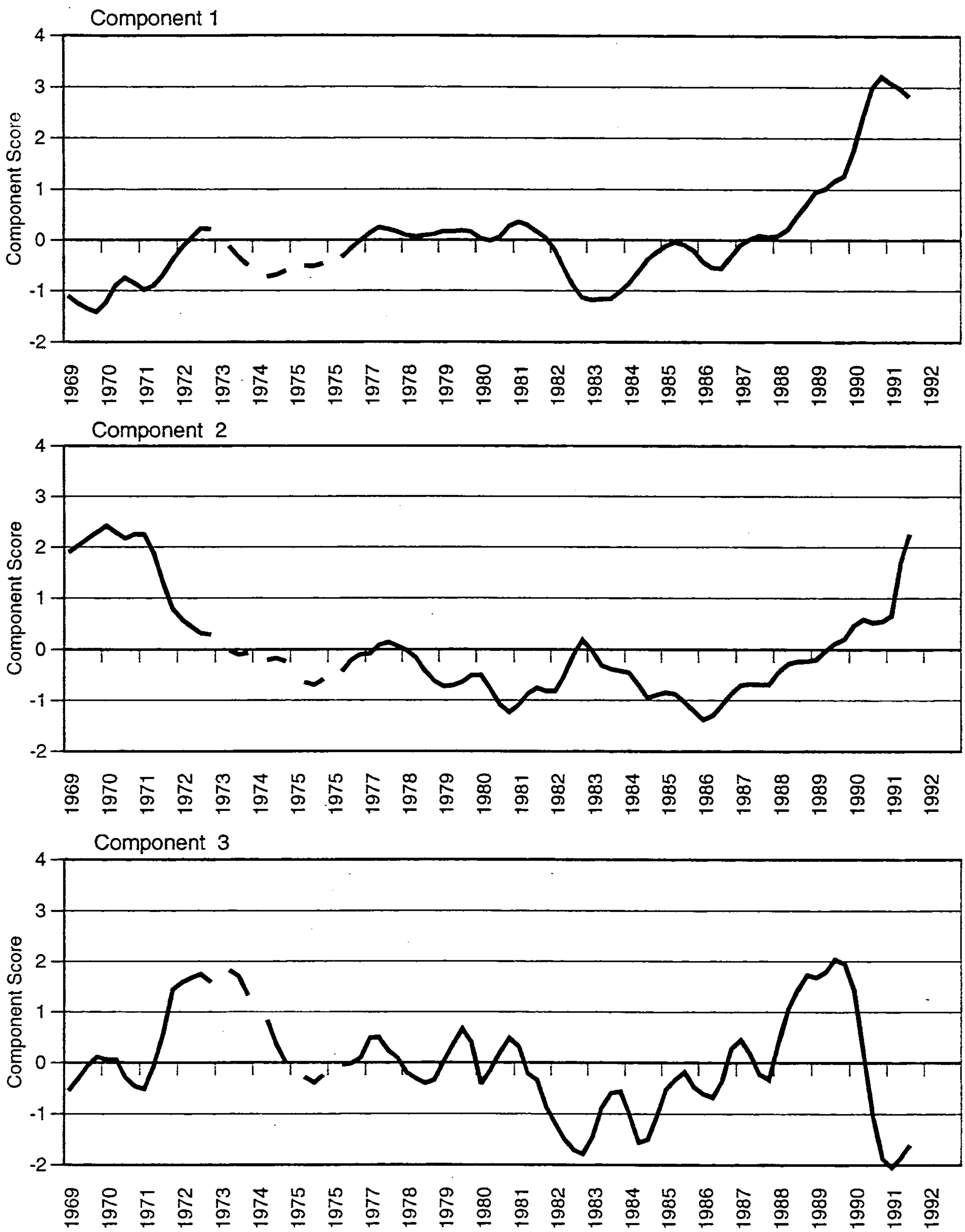


Figure 4.11C Oaks Arm Principal Component loading scores for time series data (using smoothed quarterly averages) for physical/chemical parameters.

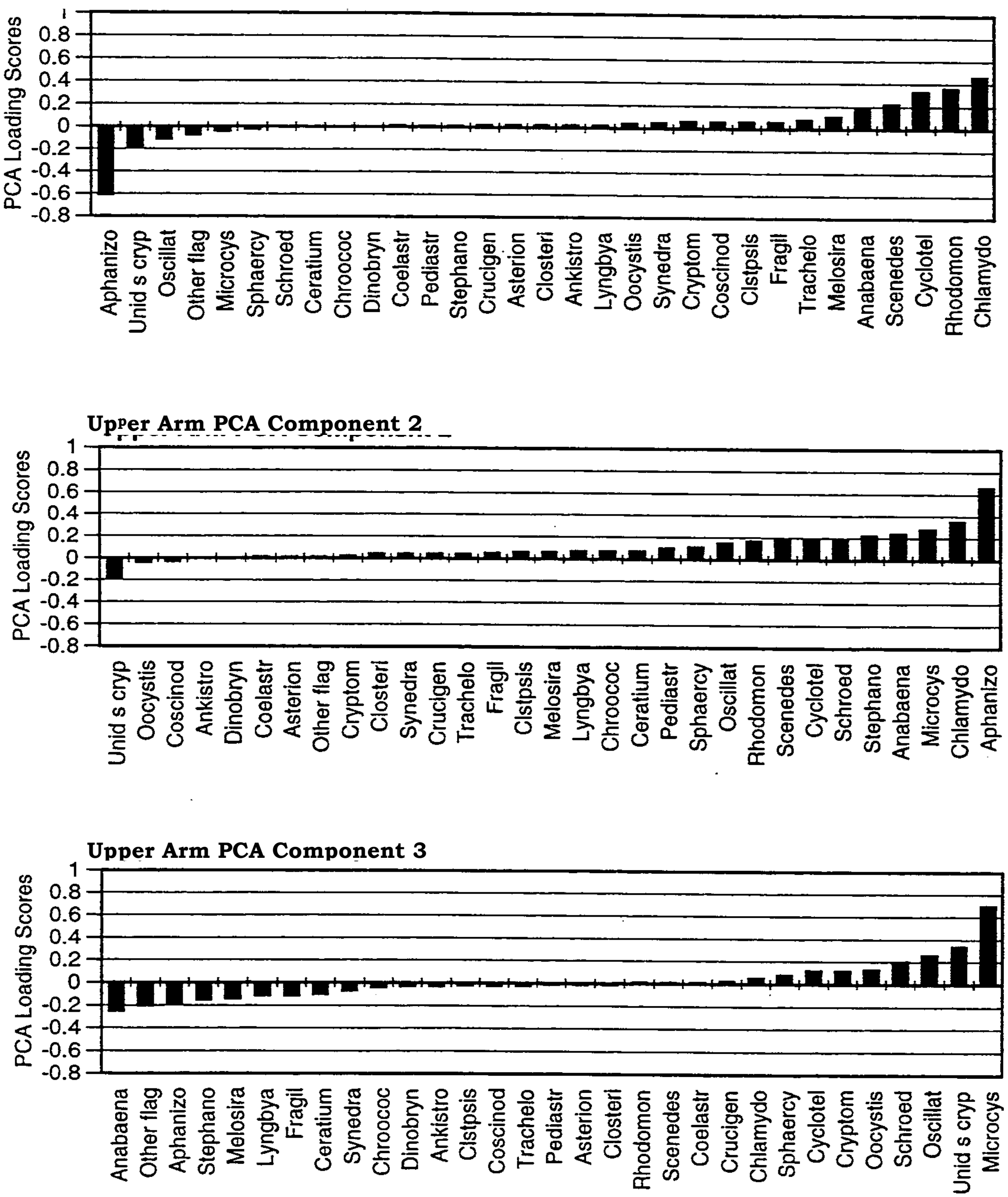


Figure 4.12A. Principal Component loading scores for Upper Arm phytoplankton.

loadings is somewhat similar in the two lower arms, and somewhat more different in Upper Arm. One interesting difference is that *Anabaena* loads positively onto Component 3 in the two lower arms, but negatively in the Upper Arm.

The total amount of variation accounted for by the first three components is 63-71%, slightly less than for the physical/chemical data, but again indicating strong patterns in the phytoplankton biomass data. Unfortunately, the three important scum formers *(Aphanizomenon, Microcystis* and *Anabaena)* do not all load with the same sign over the years of the DWR record. The blue-greens *Aphanizomenon* and *Microcystis* (together with *Oscillatoria,* a non-scum forming blue-green) vary together. However, *Aphanizomenon* and *Microcystis* tend to be replaced by *Anabaena* during other years. The drought brought

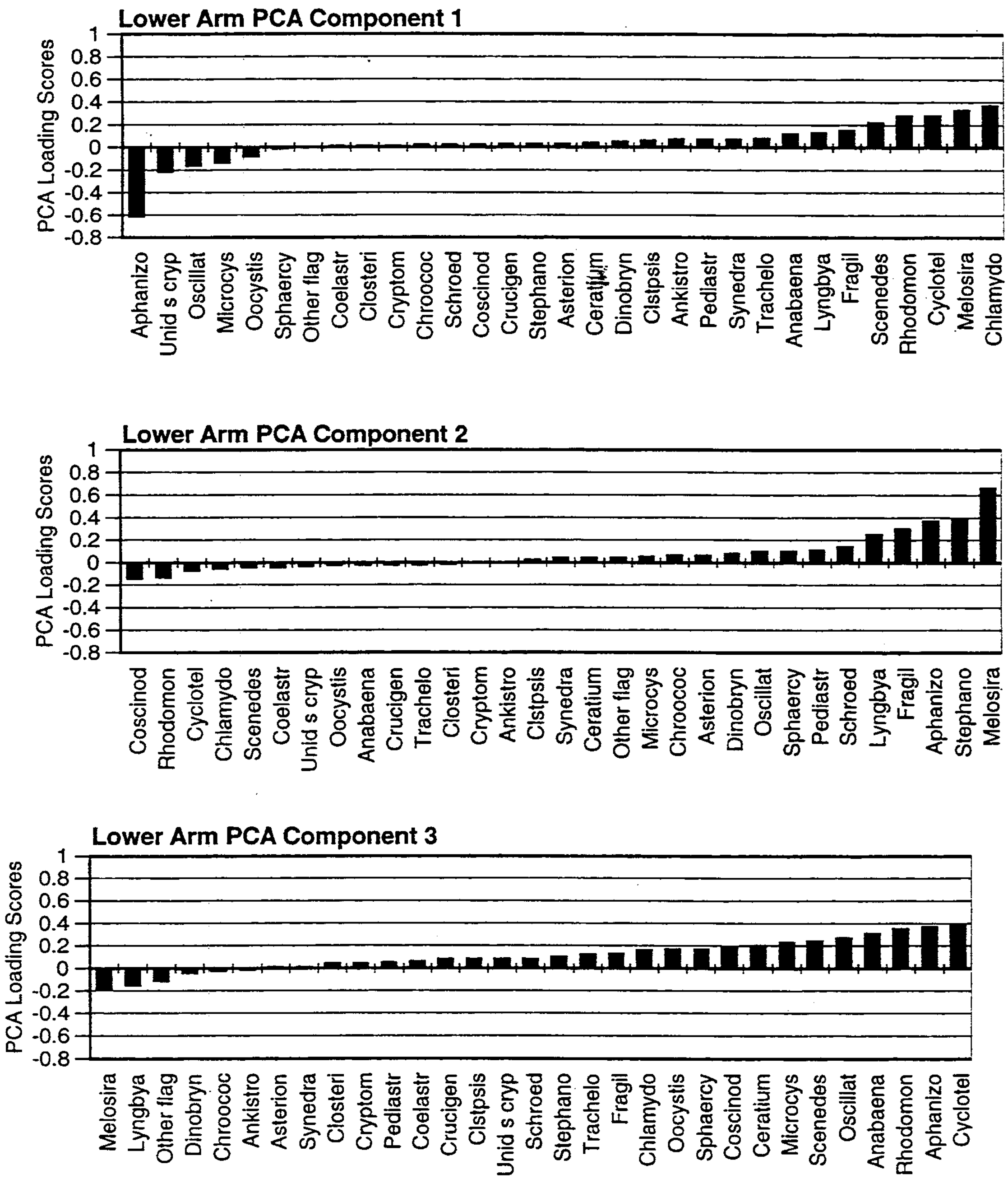


Figure 4.12B. Principal Component loading scores for Lower Arm phytoplankton.

a shift to a flora highly favoring *Anabaena* and *Lyngbya* (also see Table 4.2). This shift changed abruptly during the latter years of the drought (1990-1991), when there was an enormous production of *Microcystis* and *Aphanizomenon* again. The trend toward clearer winters, exaggerated under drought conditions (Figure 3.5), has tended to increase the abundance of non-blue-greens, especially diatoms, in winter. However, only in 1988 was the summer season substantially free of blue-greens. See **Figures** 4.7 and 4.8.

Anecdotally, the recent pattern of dominance of the phytoplankton by *Gleotrichia* reinforces the conclusion that the lake trades blue-green dominants but rarely showing a summer flora dominated by *any* other group. Under the relatively very clear but very high phosphorus conditions from 1992 to the present,

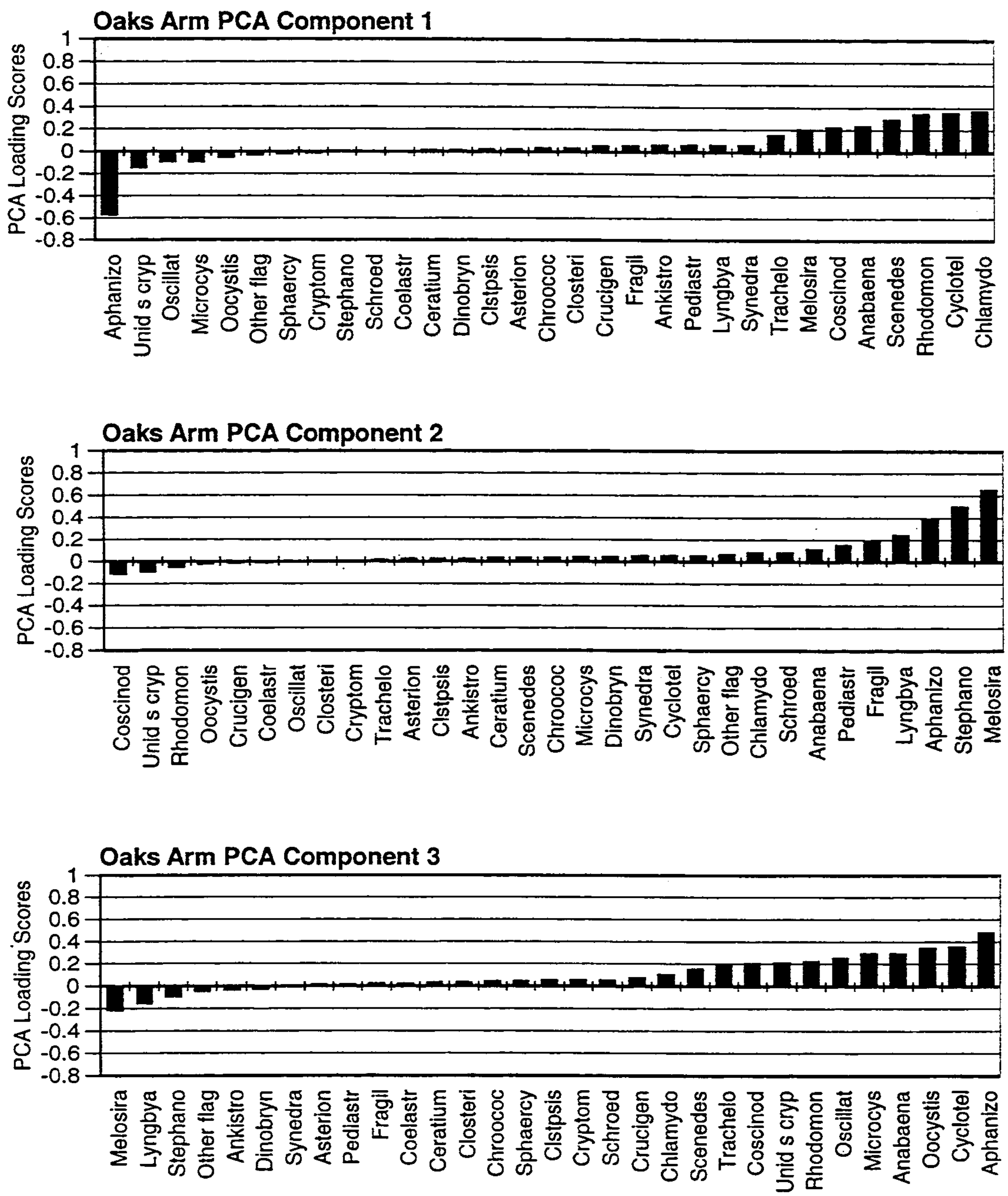


Figure 4.12C. Principal Component loading scores for Oaks Arm phytoplankton.

a new nitrogen-fixing scum forming blue-green has come to dominate the summer phytoplankton, and the historically important genera have become quite scarce. What is significant under all conditions since 1969 is that the low ratio of nitrogen to phosphorus in Clear Lake has favored a scum forming dominated phytoplankton flora in virtually all summer seasons, and frequently in other seasons as well.

4A.3 Canonical Correlation Analyses

The next step of our analysis was to identify linkages between biological and physical/chemical variation. The question asked was, do patterns of variation in physical/chemical factors drive the changes in the phytoplankton community in arty strong way?

Taxa		Influence on PCA Component 1								
		Negative			Neutral			Positive		
		CL-1	CL-3	CL-4	CL-1	CL-3	CL-4	CL-1	CL-3	CL-4
Blue-Greens	Aphanizomenon	••••	••••	••••						
	Oscillatoria	••	••	••						
	Microcystis	•	••	•						
	Chroococcus				•	•	•			
	Lyngbya				•				••	•
	Anabena							••	••	••
Greens	Sphaerocystis	•				•	•			
	Oocystis		•	•				•		
	Closterium				•	•	•			
	Coelastrum				•	•	•			
	Schroederia				•	•	•			
	Crucigenia				•	•				•
	Pediastrum				•				•	•
	Ankistrodesmus				•				•	•
	Closteriopsis						•			
	Scenedesmus							••	••	••
	Chlamydomonas							•••	•••	•••
Diatoms	Stephanodiscus				•	•	•			
	Asterionella				•	•	•			
	Coscinodiscus					•		•		••
	Synedra							•	•	•
	Fragillaria							•	••	•
	Melosira							••	•••	••
	Cyclotella							•••	••	•••
Flagellates	uniden. crypto.	••	••	••						
	other flagellates	•				•	•			
	Ceratium				•	•	•			
	Dinobryon				•	•	•			
	Cryptomonas					•	•	•		
	Trachelomonas							•	•	••
	Rhodomonas							•••	••	•••

Table 4.2A Influence of various taxa on PCA Component 1 for the three sites. The number of dots indicates the magnitude of the influence, more dots indicating a greater deviation from neutral.

A general indication of such a linkage was obtained by computing simple correlations between physical/ chemical variables and the Principal Components of the phytoplankton abundance data set. Several of these correlations are significant, e.g. phytoplankton Component 1 (the more taxonomically related component) correlates somewhat with organic nitrogen (r = 0.32 to 0.75) and total phosphorus (r = 0.11 to 0.48) but is inversely correlated with turbidity (r = -0.37 to -0.44). Phytoplankton Component 2 (related more to biomass) correlates well with electrical conductivity (r = 0.46) in the Upper Arm, but patterns in the other arms are not very similar.

To obtain a more complete picture of the correlation between physical and biological variations, the relationship between the phytoplankton principal components and the physical/chemical data was ana-

Taxa		Negative CL-1	Negative CL-3	Negative CL-4	Neutral CL-1	Neutral CL-3	Neutral CL-4	Positive CL-1	Positive CL-3	Positive CL-4
		Influence on PCA Component 2								
Blue-Greens	Microcystis					•	•	••		
	Anabena					•		••		•
	Oscillatoria						•	•	•	
	Chroococcus						•	•	•	
	Lyngbya							•	••	••
	Aphanizomenon							••••	•••	•••
Greens	Chlamydomonas		•					••		•
	Oocystis				•	•	•			
	Closterium				•	•	•			
	Coelastrum				•	•	•			
	Ankistrodesmus				•	•	•			
	Crucigenia				•	•	•			
	Closteriopsis					•	•	•		
	Scenedesmus					•	•	••		
	Sphaerocystis						•	•	•	
	Pediastrum							•	•	•
	Schroederia							••	•	•
Diatoms	Coscinodiscus		•	•	•					
	Cyclotella		•				•	••		
	Synedra				•	•	•			
	Asterionella				•		•		•	
	Fragillaria				•				••	••
	Melosira							•	••••	••••
	Stephanodiscus							••	•••	•••
Flagellates	uniden. crypto.	••		•		•				
	Rhodomonas		•				•	••		
	Cryptomonas				•	•	•			
	Trachelomonas				•	•	•			
	Dinobryon				•		•		•	
	other flagellates				•	•				•
	Ceratium					•	•	•		

Table 4.2B Influence of various taxa on PCA Component 2 for the three sites. The number of dots indicates the magnitude of the influence, more dots indicating a greater deviation from neutral.

lyzed using Canonical Correlation Analysis (CCA). CCA identifies components of like-acting variables but differs from PCA in seeking pairs of derived components (one from each data set) that are highly correlated. In the analysis of the Clear Lake time-series, the goal was to provide possible links between the physical/chemical water quality data and the modes of biological variation already described for the phytoplankton time-series, for example cyclical changes in generic composition or years of average higher or lower biomass. Therefore we "constrained" the analysis by identifying the biological part of each canonical correlation as one of the phytoplankton principal components already defined in **Section 4.4.2, and** then let the CCA procedure find the combination of physical and chemical variables that gave the best correlation with each these biological components.

Chapter 4: DWR Water Quality Record and Related Long Term Data

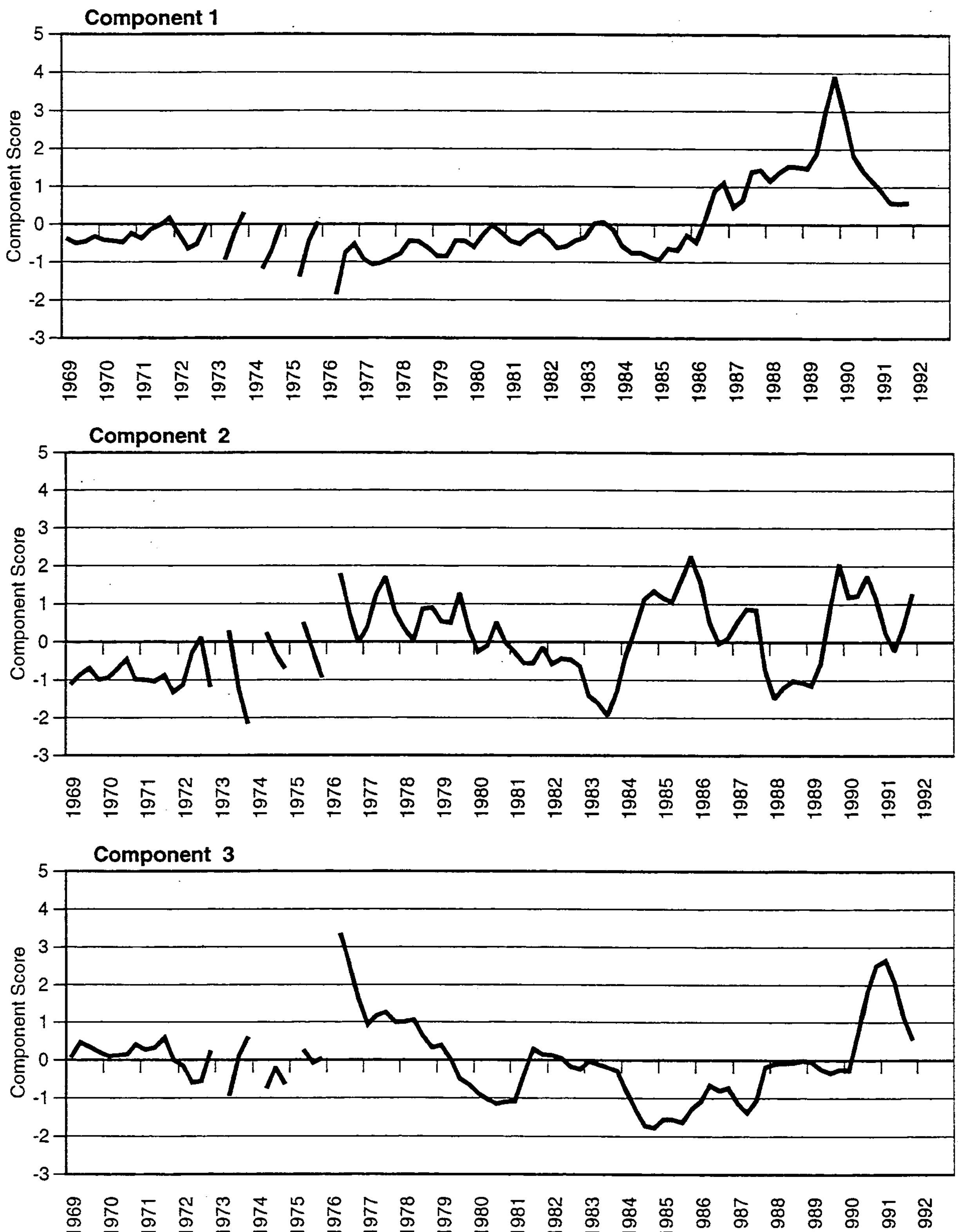


Figure 4.13A Upper Arm Principal Component loading scores for phytoplankton time series data (using seasonally adjusted quarterly averages.)

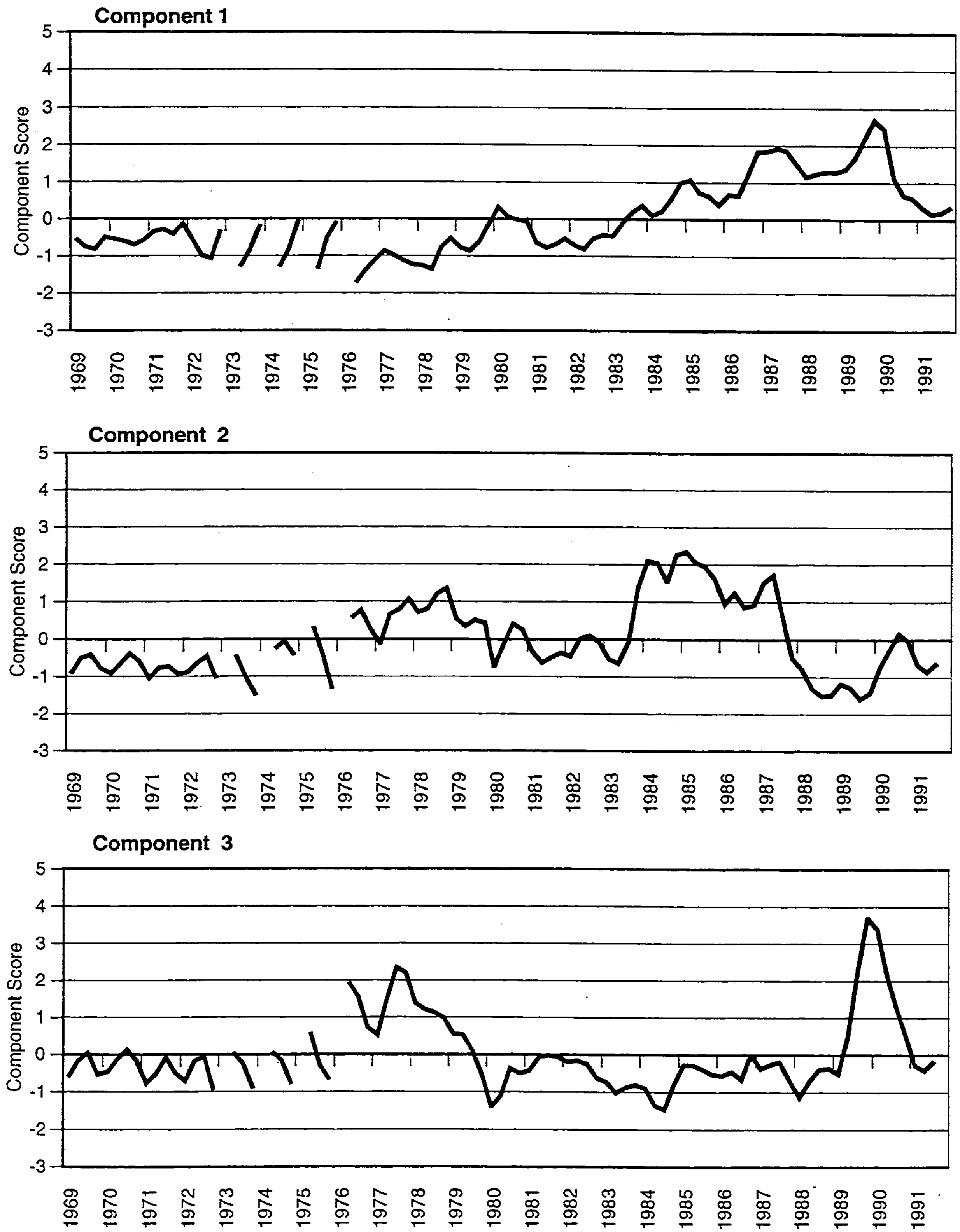


Figure 4.13B Lower Arm Prinicipal Component loading scores for phytoplankton time series data (using seasonally adjusted quarterly averages.)

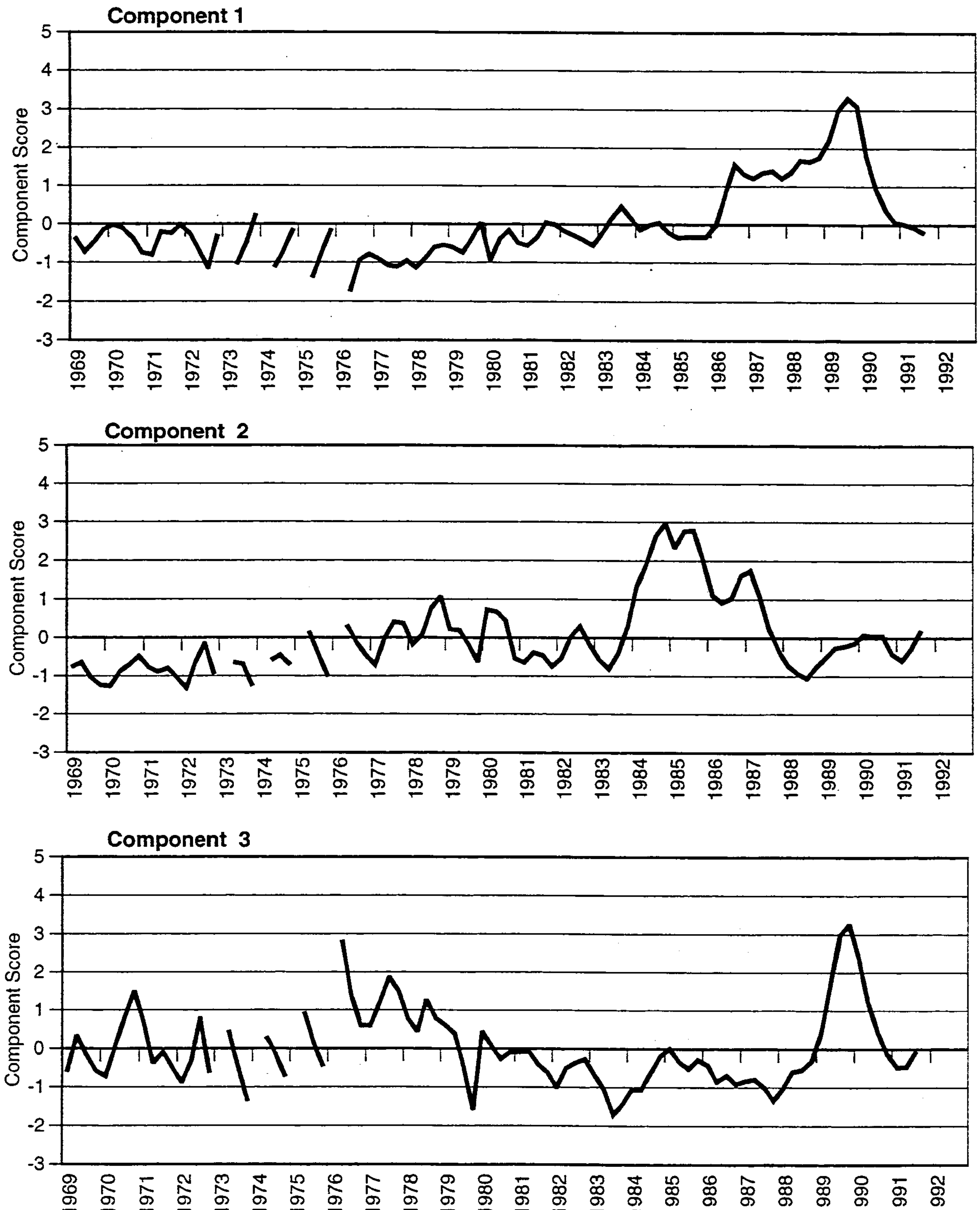


Figure 4.13C Oaks Arm Prinicipal Component loading scores for phytoplankton time series data (using seasonally adjusted quarterly averages.)

The procedure derived Canonical Components of the physical/chemical variables that had high correlation with phytoplankton Component 1, with the over-all correlation between the phytoplankton Principal Component and the physical/chemical canonical variate (the Canonical Correlation) ranging from 0.79 to 0.91 over the three basins. The pattern of loading on the physical/chemical components are similar between basins **(Figure** 4.14). The phytoplankton Principal Component 1 (again, most strongly related to taxonomic differences) is related to a physical/chemical component that positively weights total phosphate concentration and organic nitrogen but negatively weights turbidity, orthophosphate, ammonia and electrical conductivity. A negative score on component 1 occurs during *Aphanizomenon* dominated years while positive scores occur when greens and diatoms and *Anabaena* are dominant. The CCA results suggest that higher turbidity and a large difference between dissolved and total phosphate concentrations (much dissolved phosphorus relative to that taken up by algae) are characteristic of *Aphanizomenon* years. Lower turbidity and lesser amounts of surplus phosphorus are characteristics of *Anabaena/green/diatom* years.

The Canonical Correlation between the phytoplankton Principal Component 2 (mostly reflecting variations in biomass) and a second physical/chemical Canonical Component was lower but still moderately significant, ranging from 0.66 to 0.82. Phytoplankton Component 3 had Canonical Correlations from 0.68 to 0.78. The Canonical loading scores for Components 2 and 3 varied substantially from arm to arm for all of the physical/chemical variables tested, but it is unclear why such patterns exist. The statistical success of the CCA was quite good, in that it is clear that phytoplankton patterns are related to physical/chemical patters, albeit in complex ways.

4.4.4 Multiple Regression Analyses

We tested the possible value of several physical/chemical variables as predictors of phytoplankton variation, in particular the occurrence of the scum forming blue-greens, using a multiple regression analysis. Even if Clear Lake's summer phytoplankton has been dominated by scum forming blue-greens since 1969, are there specific conditions when this biomass is lower rather than higher?

For this analysis we focused on yearly variation in scum-forming blue-green phytoplankton biomass for the Upper Arm. The genera analyzed were: *Microcystis*, *Aphanizomenon*, *Anabaena* and *Oscillatoria*. Yearly geometric mean biomass (cell/colony volume) was calculated as follows: (1) The monthly average for each genus was increased by 1 (to eliminate zero values). (2) Geometric means (i.e. antilog of mean log) were calculated over the year from which 1 was subtracted. The total cell volume was summed over yearly means for the four blue-greens (Total BG) and for all cells. Then % blue-green biomass was computed as 100 * Total BG/Total biomass. Yearly averages for physical/chemical data were computed as simple arithmetic means over the year.

Plots of total blue-greens versus the main control-ling variables identified in the canonical correlation analysis (electrical conductivity, total-phosphate, alkalinity, turbidity) showed two main groups of points - one group of years in which blue-green biomass was strongly correlated with physical/chemical variables, and another group of years lacking such correlation. The former group tended to be years with high proportion of blue-greens, the latter had low proportion of blue-green biomass, frequently because of large biomasses of other algae in winter. **Table 4.3** shows how the years were divided by this criteria.

Generally, a high percent blue-green year is also a high biomass year, but the distinction is not perfect, especially for certain years with much lower biomass (e.g. 1983 and 1986). The years identified as "High" blue-green years **(Table 4.3B)** had an aver-age of 86% contribution from blue-greens, whereas the "Low" blue-green years have an average of 46% contribution from blue-greens.

For the high blue-green years, the following regression equations resulted:

1. Total Blue-greens biomass = 42.4 * EC - 7668, R^2= 0.68
2. Total Blue-greens biomass = 34 * EC + *45724* * (dissolved P) -7546, R^2=0.85
3. Total Blue-greens biomass = 27 * EC + 49750 * (total P) -8881, R^2=0.86
4. Total Blue-greens biomass = 266 * ALK - 81 * EC - *7067*, R^2=0.95

These are two well defined groups of years, years when blue-greens grow well and their growth is related to water quality variables, and other years when blue-greens do poorly irrespective of water quality variable measured by DWR. Some unmeasured variable or variables must govern the growth of blue-greens in the low relationship years, perhaps dissolved iron. The percentage of blue-greens is correlated with total phosphorus. Of the measured water quality variables, it is the best predictor of blue-green dominance. However, the strength of the relationship is low (R^2 = 0.24). Note that total phosphate had the highest loading on Canonical Component 1, the taxonomic composition variable. The regression equation is:

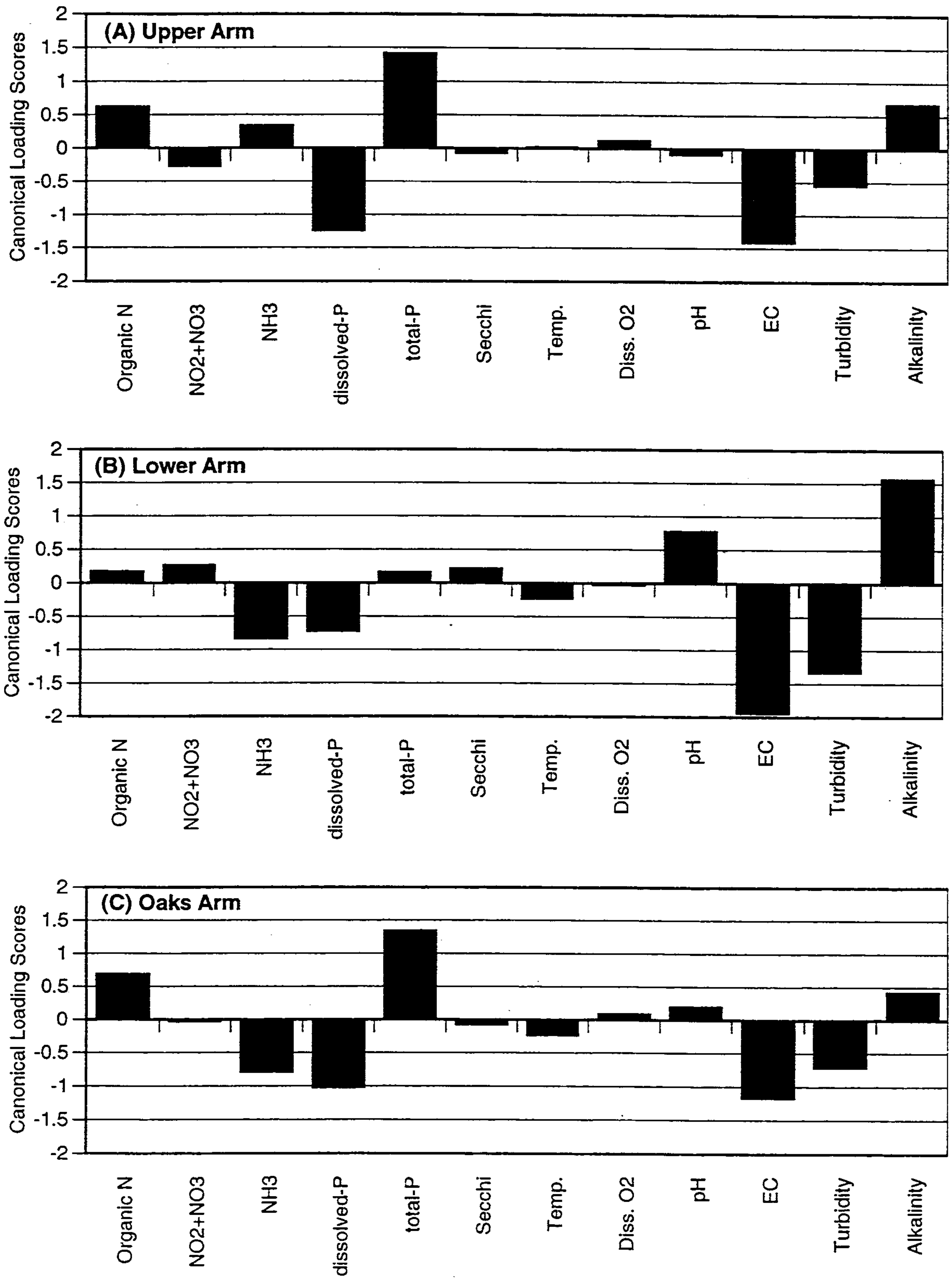
$$\% \text{ blue-greens} = -232.5 * \text{totPO}_4 + 98$$

4.4.5 Interpretations/Conclusions Based On Statistical Analyses

The results from the DWR data from both the Principal Components Analysis and the Canonical Correlation Analysis show that significant relationships exist between the interannual variation of biological and physical/chemical characteristics of Clear Lake. However, the dependency of total scum forming biomass, which is what we wish to control, best correlates with the drought index electrical conductivity, and this correlation only holds in some years. There is also some dependence on phosphorus, again best in the "High" subset of years blue-greens, and some dependence of blue-green dominance on phosphorus.

There are likely two reasons for these results. First, the low summer ratio of total nitrogen to phosphorus means that the summer phytoplankton has a perennially important nitrogen fixing component. The lowest phosphorus years did cause a shift from *Aphanizomenon* to *Anabaena* dominance, but was not sufficient to eliminate midsummer blooms. The lowest blue-green year on record, 1988, had ample dissolved phosphorus to support a late summer and fall blue-green population. Second, the lack of routine measurements of iron probably means that the element most directly controlling scum former abundance is missing from the statistical exercise. No other single physical/chemical factor acts as a surrogate for iron, but the high overall explanatory power of the CCA and multiple regressions perhaps suggests that a combination of physical/chemical variables do so.

The existence of some relationship between phosphorus and blue-greens is encouraging. If phosphorus concentrations in the lake can be lowered to the vicinity of the lowest years of the DWR record or lower, blue-green biomass and dominance should be substantially reduced, regardless of whether iron (or other unmeasured control variables) are reduced as well.


(A) Upper Arm
(B) Lower Arm
(C) Oaks Arm
Canonical Loading Scores
2
1.5
1
0.5
0
-0.5
-1
-1.5
-2
Organic N
NO2+NO3
NH3
dissolved-P
total-P
Secchi
Temp.
Diss. O2
pH
EC
Turbidity
Alkalinity

Years Having a High **Correlation** for Biomass **and the Variables**	**Variables**			**Total Blue-Green Biomass**	**Blue-Green Biomass as Percent of Total Biomass**
	Total Phosphorus	Alkalinity	Electrical Conductivity		
1972	0.14	127.8	261.4	5,496.0	98.4
1974	0.09	100.0	211.6	3,357.8	97.1
1975	0.09	104.1	229.5	1,951.3	92.0
1976	0.13	127.2	275.0	4,893.4	96.4
1977	0.13	154.2	333.0	6,643.9	98.3
1979	0.09	129.4	299.3	3,700.8	97.5
1982	0.10	103.8	236.4	1,351.7	90.7
1983	0.09	84.2	189.7	44.9	36.9
1984	0.09	93.9	205.7	933.8	83.3
1985	0.08	111.7	248.6	2,457.9	97.9
1986	0.07	99.0	226.2	207.2	56.8
Average	*0.10*	*112.3*	*246.9*	*2,821.7*	*85.9*

Years Having a Low Correlation for Biomass **and the Variables**	**Variables**			**Total Blue-Green Biomass**	**Blue-Green Biomass** as **Percent** of **Total Biomass**
	Total Phosphorus	Alkalinity	Electrical Conductivity		
1969	0.18	127.0	221.2	269.8	73.3
1970	0.23	na	210.0	477.2	79.2
1971	0.13	129.2	219.1	96.1	49.0
1973	0.17	117.5	252.8	302.6	67.1
1978	0.12	116.1	269.5	429.3	81.7
1981	0.08	122.1	279.2	728.6	46.4
1987	0.11	117.4	267.8	114.7	20.7
1988	0.20	126.0	274.3	10.3	3.9
1989	0.24	141.0	314.4	79.2	15.2
1990	0.27	158.1	351.1	203.8	32.2
1991	0.28	176.8	386.4	111.6	33.5
Average	*0.18*	*133.1*	*276.9*	*256.7*	*45.7*

Table 4.3. Statistical features of high-blue-green versus low-blue-green years. Groups were determined by the degree of correlation to the variables.

5 External Nutrient Loading

S. Why, T. Smythe, C. Vaughn and B. Lamphere

5.0 Abstract

The external phosphorus load to Clear Lake averages about 160 MT/yr. By far the largest load is erosion products from non-point sources. Groundwater is apparently a negligible source, and wastewater, even counting overflows in high water years, is a comparatively small source. Greatest losses are to the sediments, with losses to the outflow averaging about 25 MT/yr. Exchanges between sediments and water are also large terms in the budget. All terms are highly variable, with large loads, high sediment storage, and high outflow during wet years, and very much lower loads in low runoff years. In low load years, the sediments can act as a large source of phosphorus. Tule Lake removes from half to two thirds of the sediment load of Scotts Creek, indicating the important role that wetland loss has played in the eutrophication of Clear Lake. Iron loading and cycling is, unfortunately, difficult to evaluate. A model of phosphorus cycling under reduced loading conditions shows that load reduction will likely lead to a loss of phosphorus from the water column over time. While sediment storage correlates strongly with creek loading, the relationship becomes weak with low loading. As a result, this term introduces significant uncertainty to the model.

5.1 Introduction

To understand the dominance of scum-forming blue-green algae in Clear Lake and to control their growth, we need to understand how the nutrients enter and leave the lake. Most treatment options depend upon lowering the nutrient budget of a lake so that the total algal crop is reduced. Usually, strategies are aimed at making phosphorus the limiting nutrient because of the high requirement of blue-greens for this element relative to nitrogen. Most scum forming blue-greens fix atmospheric nitrogen, and thus have a competitive advantage over more desirable algae when the concentration of nitrogen is low relative to phosphorus.

Home's (1975) nutrient budgets for Clear Lake were based upon a rather limited series of stream water quality data. His estimates for total phosphorus and nitrogen are shown in Table 5.1. Subsequent sampling has been even more limited. Water quantity gauging has also been less than comprehensive. With the assistance of Hopland Research & Extension Center, the UC team collaborated with Lake County Flood Control and Water Conservation District (LCFCWCD) in 1991 to collect water quality samples from gauged streams during storm events during the 1991-92 and 1992-93 runoff years. Initial analytical efforts focused on phosphorus, but as it became clear from bioassays that iron limitation remains important, that analysis was added.

5.2 Water Budget

Nutrients such as phosphorus and iron which have no important gaseous compounds can be carried into a lake by surface water flow, ground water flow, and rainfall and dustfall directly on the lake surface. In eutrophic systems with reasonable-sized drainage basins rainfall and dustfall all contributions can usually be neglected. Losses of nutrients are principally in

Nitrogen

	Gains (tons N)		Losse (tons
River inflow	+410	River outflow	-375
N fixation	+450	Denitrification	-100
Rainfall	+90	Sedimentation	-695
Groundwater	+130	Fish harvest	-10
Domestic wastewater	+115		
Insect Control	+1		
	+1196		-1180

Phosphorus

	Gains (tons P)		Losse (tons P)
River inflow	+130	River outflow	-40
Groundwater	+30	Sedimentation	-125
Domestic wastewater	+40	Fish harvest	-2
Insect Control	+2		
	+202		-167

Table 5.1. Phosphorus budgets for Clear Lake from Home (1975), adapted from Lallatin 1972.

the outflowing water and permanent burial in sediments.

The case of nitrogen is complicated by its gaseous form. Nitrogen fixers can obtain nitrogen from the air, and sediment bacteria can use nitrate to fuel their metabolism, returning nitrogen to the atmosphere in the process. Nevertheless the water budget related flows are important for this and similar elements like sulfur and carbon.

Estimating external loading of nutrients depends upon data on (1) water flows into and out of the system and (2) the nutrient concentrations in those waters. Thus, the hydrologic budget is critical to the nutrient budget.

5.2.1 Historical Water Budget

Chamberlin *et al.* (1990) provide a summary of the water budget based on historical data. These data are quite limited and contain what we believe to be errors, primarily in the estimation of flows from ungauged tributaries, direct precipitation and lake evaporation. We here recalculate the budget based on somewhat different assumptions.

Although some streamflow gauges have been operated continuously for long periods, the best records are for lake level and outflow. DWR and USGS operated a fair number of gauges on inflowing streams, but only for limited periods of time, usually a decade or less in the 1945-1979 period. Only the gauge at Kelsey Creek above Kelseyville, operational since 1945, has both a long and continuous record. Middle Creek has had continuous gauging since 1959, with additional gauging from 1949-1959 about 4 kilometers upstream of its present location. The gauge was relocated when the Middle Creek levee project was constructed. Scott's Creek has gauging data from 1948-1968 at the old Scott's Valley Road bridge. When the bridge was replaced, the gauge was moved about 4.7 kilometers downstream to Eickhoff Road. Figure 5.1 shows the periods of record for the stream gauges. Streamflow varies strongly by watershed, season and year, so data of limited coverage in time and space degrade the water budget estimate appreciably. On the positive side, weather events are strongly correlated across the basin, so that flows in the various creeks are strongly correlated (Figure 5.2). The largest source of uncertainty in the historic water budget stems from the fact that only 47.6% of the drainage basin has ever been gauged. At the same time, run-off per unit area ranges from annual averages of 3.7 (Burns Valley Creek) to 21 (Kelsey Creek) liters/sec/km^2. Thus, large ungauged areas translate into significant uncertainties. A majority of the gauges are for the creeks which drain the mountainous areas of the County. The mountainous areas tend to experience significantly higher precipitation, and have shallow soils combined with steep terrain, resulting in rapid surface runoff. Much of the ungauged area is the valley floors which are able to store significant quantities of water in the alluvial soils for later release to plants, low water creek flows and groundwater pumping. Consequently, the unit flows of creeks draining the valley floor are significantly lower, as evidenced by Burns Valley, which has a drainage area consisting of predominantly rolling hills and broad alluvial valleys. Comparison of the Kelsey Creek flows measured at the USGS gauge (12.9 km from Clear Lake with a 94.8 km^2 drainage area) and the DWR gauge (2.7 km from Clear Lake with a 143 km^2 drainage area) during the period when data exists for both gauges (9 years, 1981-89) shows that the downstream gauge has an average annual flow of only 98% of that of the upstream gauge, although its drainage area is 51% larger. Only the months of December, January and February have greater average annual flows at the downstream gauging station (Table 5.2). Caution should be used when interpreting this data, as the 9 year average flow at the USGS gauge is only 84% of the 45 year average, indicating that 1981-1989 was a drier than normal period, which results in a greater than average groundwater recharge: creek flow ratio. Although the effects of groundwater recharge may be more pronounced on Kelsey Creek than other creeks which monitor flows below the primary re-charge areas, such as Middle and Scott's Creeks, a reduced unit runoff (liters/sec/km^2) can also be expected on these creeks at their mouth. Using this phenomenon in combination with average precipitation maps (DWR, 1966), the unit runoff for ungauged areas of the basin are rather lower than the Chamberlin *et al.* estimations. **Table 5.3 summa**rizes our estimates of average stream flow, which can be compared with Chamberlin, *et al.* Table 4-8, based on extrapolations proportional to area.

Another source of uncertainty is the estimate of evaporation from the lake surface. Chamberlin *et al.* estimated evaporation from evaporation pan data, which tend to overestimate lake evaporation because small pans are warmer than large lakes. The estimate of 1.52 m/yr (60 in/yr) appears high. During years when there was no release through the dam, *1977* and 1990, a water level drop of 0.942 m (3.09 ft) and 0.936 m (3.07 ft) was observed, respectively, for the summer. Similar summer water level drops have been observed at Highland Springs Reservoir. LCFCWCD (1987) estimated the potential evapo-transpiration from Clear Lake at 1.05 m/yr (41.2 in/yr), which corresponds more favorably with observed water level

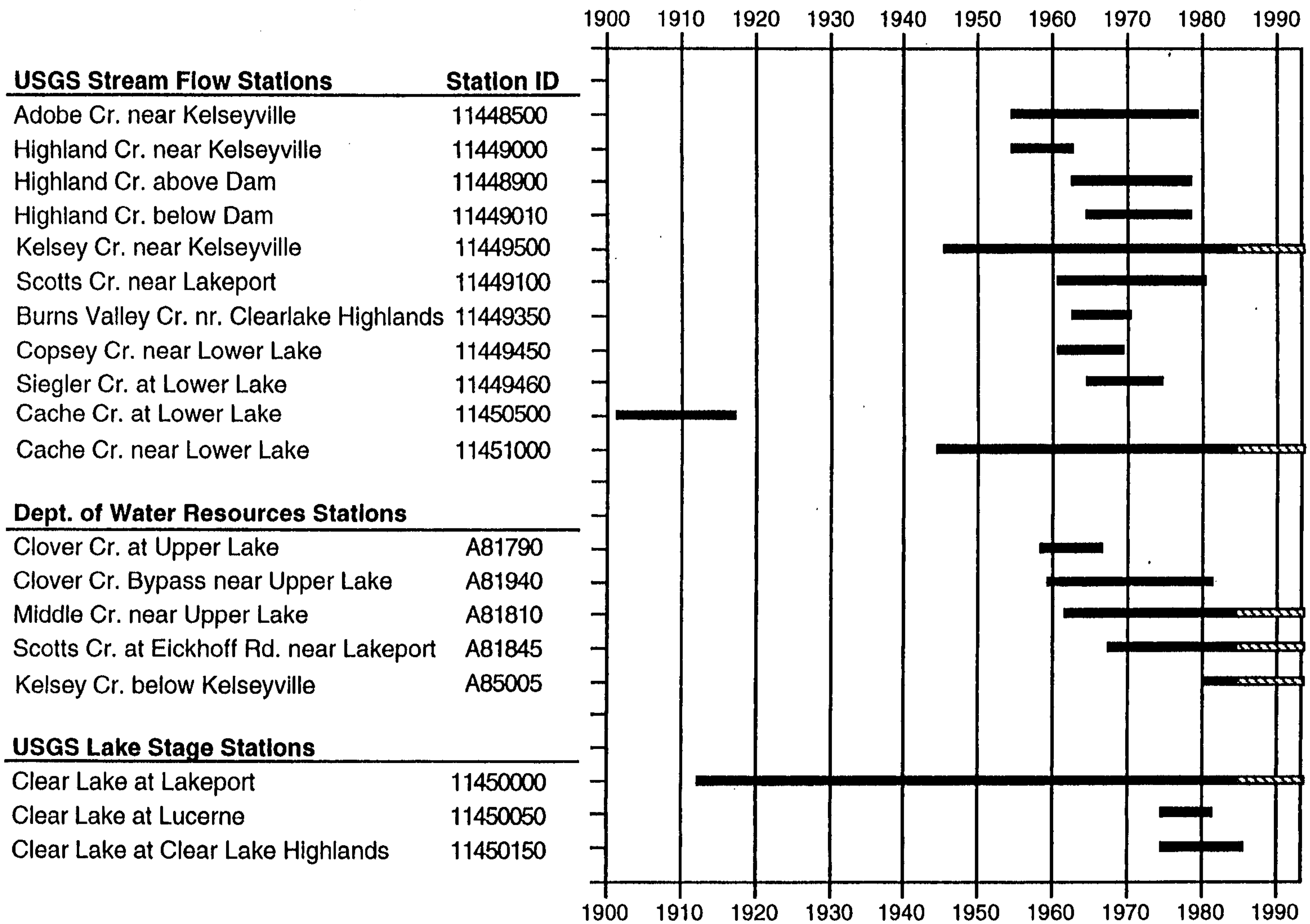


Figure 5.1. Periods of record for USGS and DWR streamflow and lake stage stations from Chamberlin *et al.* (1990).

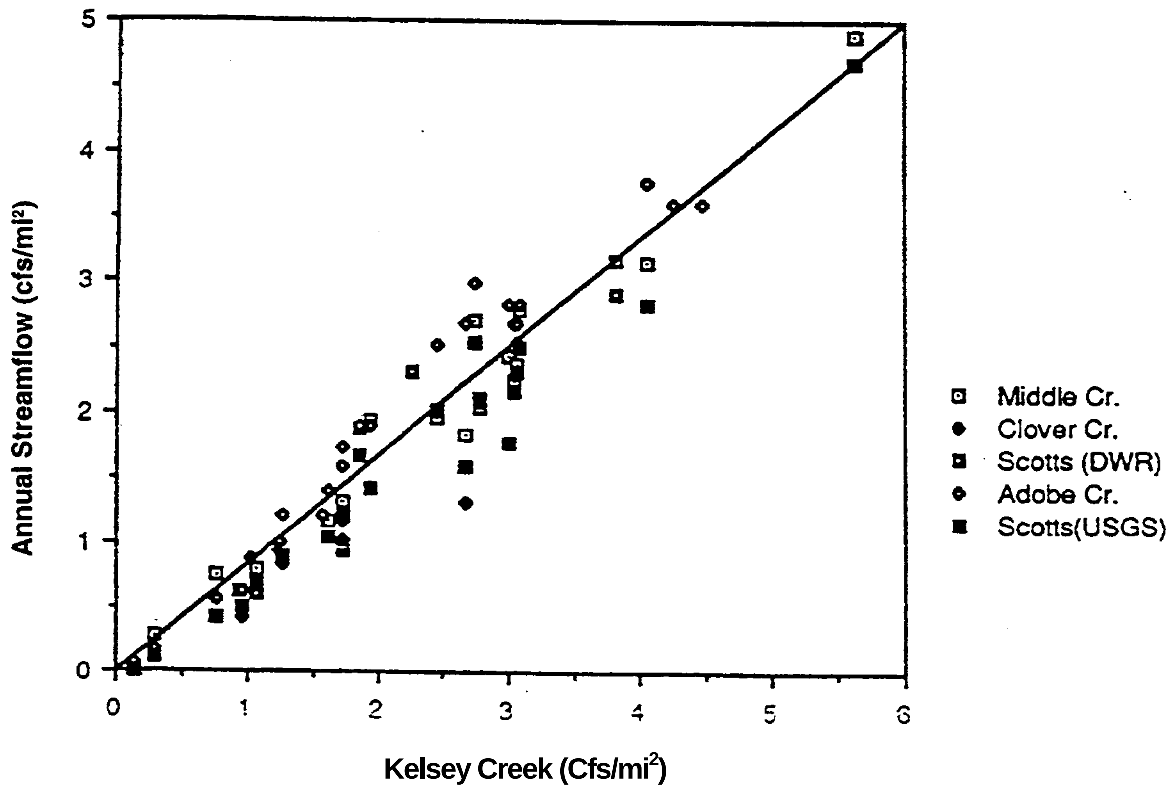


Figure 5.2. The correlation between average annual streamflow in 5 creeks with limited gauging compared with the well-gauged Upper Kelsey Creek. From Chamberlin et al., (1990).

Period	1947-92	1981-82, 1984-89, & 1992		Flow Ratio of Downstream to Upstream
	Upstream Gauge (1000 m³)	**Upstream Gauge (1000 m³)**	**Downstream Gauge (1000 m³)**	
October	932	429	49	0.11
November	3,588	5,066	4,664	0.92
December	9,627	8,863	9,085	1.03
January	14,253	7,426	7,773	1.02
February	14,025	12,840	13,799	1.07
March	11,220	10,887	10,707	0.98
April	5,804	5,425	5,319	0.98
May	2,176	1,316	925	0.70
June	858	596	129	0.22
July	407	296	9	0.03
August	259	173	0	0.00
September	272	228	0	0.00
Annual total	**63,421**	**53,545**	**52,459**	**0.98**

Table 5.2. Comparison of average monthly measured flows on Kelsey Creek.

Sub-Basin	Tributary	Drainage Area (sq.	(%) (note 1)	Average Annual Streamflow (cfs)	(cfs/sq. mi.)	(10^3 ac-ft/yr)
Upper Arm	Clover Creek	25.2	7.3%	30.0	120	
	Middle Creek	48.5	14.0%	91.0	1.90	
	Scotts Creek (note 2)	55.2	15.9%	89.0	1.60	
	Highland Creek (note 3)	14.2	4.1%	22.0	1.50	
	Adobe Creek	6.4	1.8%	12.0	1.90	
	Kelsey Creek	36.6	10.5%	76.0	2.10	
	ungauged	161.3		46.4%	1.70 (note 4)	
	Total	*347.4*	*100%*	*594.0*		*430.0*
The Narrows	ungauged	3.7		6.0	1.70 (note **4)**	
	Total	*3.7*		*6.0*		*4.0*
Oaks Arm	unqauqed	20.2		34.0	1.70 (note **4)**	
	Total	*20.2*		*34.0*		*25.0*
Lower Arm	Bums Valley	4.4	152%	1.5	**0.34**	
	ungauqed	24.3	84.8%	8.0	0.34 (note 5)	
	Total	*28.7*	*100%*	*9.5*		*7.0*
Clear Lake	gauged	190.5		47.5%		
	unqauqed	209.5		321.		
	Total	*400.0*	*100%*	***643.5***		*466.0*
(downstream)	Siegler Creek	13.2	34.1%	11.0	0.83	
(note 6)	Copsey Creek	12.5	32.3%	14.0	1.10	
	unqauqed	13.0	33.6%	12.0	0.96 (note 7)	
	Total	*38.7*	*100%*	*37.0*		*27.0*
All Streams	*Total*	*(note 8)*		*381.0*		*276.0*

1 % of total sub-basin drainage area.
2 Average of USGS Station 11449100 and DWR Station A81845.
3 Highland Creek below Highland Creek Dam (USGS Station 11449010).
4 Based on an area-weighted average of the unit streamflows in gauged Upper Arm sub-basins.

Table 5.3. Estimated average annual tributary streamflow by sub-basin.

changes. This also corresponds to observed pan evaporation rates at Lakeport (ranging from 0.927 to 1.19 m/yr (36.50 *to 46.77* in/yr) and the rate of 1.38 m/yr (54.25 in/yr) at Finley 1 SSE. These locations near the lake are probably cooler and more humid, and hence have evaporation rates closer to lake values than more distant stations. They also have longer records. Therefore, for the purposes of this calculation, direct evaporation from Clear Lake is assumed to be 1.07 m/yr (42 in/yr).

The direct precipitation of water onto the lake surface of 0.813 m/yr (32 in/yr) also appears to be overestimated by Chamberlin *et* aI., based on the average annual rainfalls determined by DWR (1966). Using a weighted average based on these lines of equal precipitation over the lake itself, the average precipitation on the lake surface is 0.686 m/yr (27 in/yr).

Agricultural and municipal/industrial water use directly from Clear Lake is small when compared to groundwater usage. LCFCWCD (1987) estimated the 1990 water demand for the Clear Lake Basin at 82.9 x 10^6 m^2/yr (67,250 ac-ft/yr). Of this water demand, approximately 10.8 x 10^6 m^2/yr (8,744 ac-ft/yr) is municipal/industrial demand pumped directly from the lake. Estimation of agricultural water pumped from the lake is much harder to quantify. Essentially all the agricultural water utilized in Tule Lake Reclamation and in the reclamation area adjacent to Rodman Slough is pumped directly from the lake. There are also several large tracts of land at the north end of Big Valley which are irrigated with lake water. Assuming an application rate of 0.91 m/yr (3 ft/yr) and an irrigated area of 1093 hectares (2,700 acres), agricultural water consumption is approximately 9.95 x 10^6 m^2/yr (8,100 ac-ft/yr).

Table 5.4 summarizes our estimate of the long-term annual water balance, which can be compared to Chamberlin *et al.* Table 4-9.

5.2.2. Groundwater Inflow

The Chamberlin *et al.* and earlier water budgets have assumed that groundwater inflow is zero. While the qualitative argument is a good one, we here provide a discussion of the available data and a more formal

Component	Symbol	Annual Contribution (1) (10^3 m^3/yr)	(m/yr) (2)	Remarks
Tributary Streamflow	Q(t)	+355,060	+20	see Table 5.2
Downstream Streamflow	Q(ts)	+33,450	+0.2	see Table 5.2
Direct Precipitation	P`A(s)	+121,450	+0.7	Precipitation from DWR (1966) x lake surf. area
Releases from Dam	Q(r)	(-310,220)	(-1.8)	see Table 5.2
Evaporation	E*A(s)	(-188,900)	(-1.1)	LCFCWCD (1987) x lake surf. area
Extraction	Q(e)	(-20,800)	(-0.1)	This document
Groundwater	Q(gw)	+1,355	+0.0	High estimate, this document
Change in Storage	deltaS	(-8,605)	+0.0	

1 Positive values are inputs and negative values are outputs.
2 Based on a surface area of 178 km^2 (68.4 mi^2 or 43,776 acre)

Table 5.4. Long-term annual water balance for Clear Lake.

quantitative estimate. Horne (1975) suggested that groundwater flow was a significant source of nutrients, so the zero assumption does require documentation. Improving upon the "practically zero" estimate would require expensive studies, and it is important to be sure that it is indeed sound.

5.2.2.1. Physical Setting

Groundwater within the Basin occurs in unconsolidated alluvial deposits, within the fractured sedimentary and metamorphic rock of the Franciscan Assemblage, and within the Clear Lake volcanic deposits. Significant information is available for the major alluvial deposits. However, there is very little information available for the groundwater in the minor alluvial, fractured rock and volcanic aquifers.

Four LCFCWCD studies (1967, 1970, 1978 and 1987) give information on the ground waters adjacent to Clear T Lake. Approximately 84 percent of the surface water enters the Upper Arm, with the largest, most productive, groundwater basins also occurring around the Upper Arm. The Big Valley and Upper Lake groundwater basins border the Upper Arm on the north and south sides. The Scott's Valley Basin is separated from the lake by 1.6 km (1 mi.) of terrace deposits on the west side of the Upper Arm. Detailed groundwater studies have been completed for these three basins. There are several small alluvial deposits on the east side of the Upper Arm, however, significant groundwater resources are not associated with these deposits.

Schindler Creek is the primary drainage in the Oaks Arm, but significant aquifers are not associated with this creek or any of the other alluvial deposits bordering the Oaks Arm.

Burns Valley Creek is the largest drainage directly discharging to the Lower Arm. Unfortunately it has not been studied in detail, and its connection to Clear Lake is unknown. The Lower Lake Basin is recharged by Seigler, Copsey and Herndon Creeks. The downgradient edge of this basin is located on Cache Creek between Clear Lake and the Clear Lake Dam. It may contribute water to Clear Lake when the dam is not discharging.

Much of the perimeter of the lake is directly bordered by the fractured Franciscan Formation and the Clear Lake Volcanics, both of which are known to contain groundwater. There is very little information on associated groundwater levels and water quality. Numerous natural springs occur around and under the lake. These springs are generally associated with the many fault lines around the lake and have highly variable water quality. Some flow and water quality data is available from testing performed by the United States Geological Survey (USGS) in the 1960's and 1970's for surface springs (Berkstresser, 1968; Thompson *et al.*, 1981a and b), but no data is available for subsurface springs.

5.2.2.2. Hydrologic Contribution

An understanding of the geomorphic processes which formed the Clear Lake Basin is critical to understanding the groundwater contribution to Clear Lake. Clear Lake is located in a highly faulted area associated with the San Andreas Fault system. The Main Basin, Konocti Bay and Big Valley Faults are the most predominant faults in the formation of Clear Lake (Sims *et al.*, 1988). Vertical movement along the faults have approximately equaled the sedimentation rate, resulting in lacustrine deposits of up to 200 meters under the Upper Arm of Clear Lake. The vertical movement and sedimentation has resulted in burying of the downstream ends of most of the aquifers which lead toward Clear Lake, with the lake bottom primarily of thick silt and clay deposits. A discussion of each of the major groundwater sources and their relative hydraulic contribution follows.

Big Valley: Paleographic reconstruction indicates the north end of Big Valley was inundated by ancestral Clear Lake for 220,000 of the last 475,000 years, resulting in lacustrine deposits with interbedded alluvial and floodplain deposits several hundred feet thick near the lake (LCFCWCD, 1967). Much of the water bearing deposits are overlaid with 10 to 30 meters (33 to 98 ft.) of blue clay and other fine grain deposits. Two USGS sediment cores collected in 1973 showed 10 meters of mud overlying sand and gravel deposits approximately 2-3 km (1-2 mi.) north of Quercus Point (Sims *et al.*, 1988).

Semiannual and monthly groundwater level monitoring by the LCFCWCD has indicated there is an average hydraulic head of about 0.3 meter (1 ft.) for 3 months of the winter. Assuming an average permeability of 1 x 10^{-6} cm/sec for the overlying clay layer (blue clays associated with the Cache formation have tested with permeabilities of less than 1 x 10^{-7} cm/sec [Steve Zalusky, personal communication]), a transmission area of 1,035 hectares (4 mi^2), the groundwater contribution is calculated using Darcy's Law (Bear, 1979) (Figure 5.3).

This calculated contribution is a high estimate. Due to lower head differentials under the lake bottom, the contribution is probably on the order of 40,000 to 80,000 m^3/yr (32 to 65 ac-ft/yr).

Additional contribution occurs during extended wet periods when the recent alluvial deposits are saturated from surface recharge and discharge at the lake shore, usually where channels have been dug back from the natural shoreline. It is very difficult to measure these flows, however, the reports of these flows only occur during wet winters, and are anticipated to be less than half of the calculated underflow.

Upper Lake Valley: Paleographic reconstruction of Clear Jake (Sims *et al.*, 1988) indicates that ancestral Clear Lake inundated the southern end of the Upper

$$V = t \times A \times k \times (h2-h1)/T$$

where:

V = flow volume, in m3
t = time, in seconds
A = area of transmission, in m2
k = permeability, as m/sec.
(h2-h1) = piezometric head differential, in meters
T = thickness of media, in meters

For this application:

$$V = (90 \times 86,400) \times (1,035 \times 10^4) \times (10^{-8}) \times (0.3/1.0)$$

V = 241,445 m^3/year (= 196 ac.)

Figure 5.3 Darcy's Law applied to Clear Lake conditions.

Lake Basin for much of the last 250,000 years, resulting in lacustrine deposits up to 30 meters thick overlying isolated lenses of the aquifer (LCFCWCD, 1978). Sediment cores in the north half of the Upper Arm collected by the USGS in 1973 and 1980 do not indude any sand or gravel layers within 20 meters (66 ft) of the bottom of Clear Lake. Evaluation of well logs has indicated the groundwater basin is confined by overlying clay deposits over most of the valley floor and is only unconfined in the upper areas of the valley. Semiannual monitoring of groundwater data by the LCFCWCD since early 1960 indicates only two feet of hydraulic head of the Upper Lake Basin near Bloody Island to Clear Lake (unpublished data). A calculation similar to that done for Big Valley was not done due to the lack of good water level data. Due to the very low hydraulic head and up to 3 km (2 mi.) of fine grained lacustrine deposits upstream of Clear Lake, the groundwater contribution is anticipated to be approximately half of the Big Valley contribution, or on the order of 20,000 to 40,000 m^3/ yr (16 to 32 ac-ft/yr).

Scott's Valley: The older terrace deposits between Scott's Valley and Clear Lake are probably delta deposits when Scott's Creek flowed directly into Clear Lake when the level of Clear Lake was several hundred feet higher. The terrace deposits are of variable permeability and are estimated to contribute less than 246,660 m^3/yr (200 ac-ft/yr) and probably closer to 24,660 to 37,000 m^3/yr (20 to 30 ac-ft/yr) per year (LCFCWCD, 1970).

Remaining Alluvial Basins: As mentioned above, there is very little additional information available for the other small groundwater basins surrounding the lake. Because the total area and volume of the remaining groundwater basins are significantly less than the Big Valley Basin, and their interface with Clear Lake is probably similar to the larger basins, their combined input is probably significantly less than 246,660 m3/yr (200 ac-ft/yr).

Fractured Rock and Volcanics: Total storage in the rock surrounding the lake is probably significantly less than the major groundwater basins, and the outcrops into the lake are probably covered with a layer of lacustrine sediment, restricting their inputs to seeps and springs. It has been theorized that the Volcanics serve as one of the primary recharge areas for the deep, geothermal hot water and steam supplies underlying the Geysers steamfield (McLaughlin

A significant contribution from these rock formations may be by springs within the lake. Correlations have been made with the locations of natural springs and fault zones, within the major aquifers and the lake. Groundwater quality samples in the major groundwater basins have shown significantly higher boron, sulfide and iron levels in deep wells adjacent to known and inferred fault lines (Department of Water Resources, 1957). Mapping of the springs in and adjacent to Clear Lake show a much higher frequency of the springs along the known fault lines and adjacent to known calderas (Norman Lehrman, personal communication). There has been significant controversy whether the observations of bubbles within Clear Lake are from springs or gas vents. It is likely that both springs and gas vents exist on the bottom of the lake.

Observations by divers of the large upwelling between Henderson Point and Kono Tayee found that bubbles appeared to come out of solution between the lake bottom and the lake surface (Norman Lehrman, personal communication), indicating at least some of the vents are springs with significant quantities of dissolved gases. No measurements or estimates of flow for the springs beneath the lake are available. Consequently, it is very difficult to estimate the contribution of groundwater to Clear Lake from these sources. It seems reasonable to assign such sources a total contribution of less than 500,000 m^3/ yr (406 ac-ft/yr).

5.2.2.3 *Cumulative Groundwater Contribution*

Based on the above calculations and assumptions, the groundwater contribution to Clear Lake is less than 1,326,000 m^3/yr (1,075 ac-ft/yr), and probably on the order of 220,000 to 450,000 m^3/yr (178 to 365 ac-ft/yr). The average annual inflow to Clear Lake is estimated to be 514,000,000 m^3/yr (417,000 ac-ft/yr) (LCFCWCD, 1987). Therefore, the groundwater inflow is probably less than 0.25 percent, and probably on the order of 0.04 to 0.09 percent, of the surface water inflow. This confirms the previous assumption that groundwater inputs are not significant to the hydrologic budget of Clear Lake.

5.2.2.4 Groundwater Nutrient Contribution

A limited amount of data exists for groundwater quality, and the available data shows that groundwater quality is highly variable, even within the same

Nutrient	Low mg/L	High mg/L	**Average** mg/L
Nitrate, as NO_3	0.00	33.0	8.78
Soluble Phosphate	0.00	0.9	0.33
Total Phosphate	0.03	1.9	0.59
Iron	<0.01	21.0	4.93

Table 5.5. Summary of groundwater quality measurement.

Data used for this analysis is from Berkstresser (1968), Thompson *et al.* (1981a and b), unpublished data from DWR, and raw water analyses from several public **water supplies. Table 5.5** shows minimum, maximum and average concentrations observed for nitrate, phosphate and iron.

As could be expected, elevated nitrate concentrations were observed near intensive cattle operations, unsewered residential developments, near public wastewater plants and in unconfined aquifers. Nitrate concentrations in wells located within 2 km of each other varied by as much as 145 times. Nitrate concentrations above the primary drinking water standard of 45 mg/L as NO_3 were not observed.

Phosphate is not normally tested for in groundwater, therefore only 8 phosphate levels from Saratoga Springs, the Sulfur Bank mine, and Seigler Springs (Berkstresser, 1968) were used. The high phosphate concentrations were in highly mineralized springs, with concentrations varying between individual springs at a site by as much as 30 times.

Iron concentrations were highly variable throughout the Clear Lake Basin. The available data indicates higher iron concentrations are observed along the major fault lines, as noted by DWR (1957). The highest observed iron concentration of 86 mg/ L was from the spring-fed Herman Pit at the Sulfur Bank Mine (Thompson, 1981a). Consistently high iron concentrations were observed in the geothermal mineral springs. Iron concentrations observed in the unconsolidated aquifers were generally low, however, concentrations of up to 1.8 mq/L were observed.

Table 5.6 provides an upper bound estimate of the groundwater contribution of nitrate, phosphate and iron to Clear Lake. Flows for this balance are the upper limits provided in the Hydrologic Contribution section. Unconsolidated aquifers were assumed to contribute an average of 7.3 mg/L NO_3 (average of 43 wells, 10 of which were above 10 mg/l), 0.07 mg/L phosphate (Springtime creek concentrations during clear water, Stephen Why, personal communication), and 1 mg/l iron. The rock aquifers were assumed to contribute 8.8 mg/L NO_3 (average of all 50 samples), 0.59 mg/L PO_4 (average of 8 total phosphate samples), and 7.0 mg/L iron (an average of 10 shore-line springs). It is probable that the nutrient input is 25 to 50 percent of the amounts calculated in Table 5.6. Even so, it appears that a very small percentage of the Clear Lake nutrient budget come from groundwater sources.

The main uncertainty here is the form of iron when it enters the lake system. As high iron concentrations from various springs around the lake indicate, ground water can mobilize large amounts of iron. It is conceivable that iron rich ground waters, associated with fractured volcanic rock and supplying subsurface springs discharging directly into the lake, could provide a significant part of the iron budget. The uncertainties of the iron budget itself (see Section 5.3.11) are very large due to limitations of both monitoring data and basic knowledge about the behavior of iron.

We can attempt a calculation to indicate the magnitude of the source that would be necessary to have an impact: In principle, the blue-green algal iron requirement is about 1% of their phosphorus requirement (Brand, 1991). Given an internal loading of phosphorus ranging from roughly 150-500 metric tons (MT) during the peak of the summer release a source of only 1.5-5 MT of highly available iron would satisfy this requirement. The probable actual supply of iron in typical years is probably on the low side of this range, as surplus dissolved phosphorus due to iron limitation usually occurs at the high internal loading rates (See Chapter 3). Based on the conservative estimate of 3.5 MT of iron provided by natural springs within the lake, this could have a significant affect on the growth of blue-green algae. Unfortunately, because the form of this iron is unknown, and may not be readily available for biological up-

Source	Row (m^3)	C Concentration (mg/L)			Metric tons		
		NO_3	PO_4	Fe	NO_3	PO_4	Fe
Big Valley	241,445	7.3	0.07	1	1.76	0.017	**0.24**
Upper Lake	120,722	7.3	0.07	1	0.88	0.008	0.12
Scotts Valley	246,660	7.3	0.07	1	1.80	0.017	**025**
Remaining Alluvium	246,660	7.3	0.07	1	1.80	0.017	**025**
Fractured Rock & Volcanics	500,000	8.8	0.59	7	4.40	0.295	3.50
Total Contribution	—	—	—	—	10.64	0.354	**4.36**

Table 5.6. Estimated groundwater nutrient contributions.

take, its impact is unknown. The oxidation of iron from the soluble ferrous to insoluble ferric form is very rapid under oxidizing conditions, which is why it tends to becoming limiting.

5.3 External Phosphorus Loads and Losses for Years 1992-93

5.3.1 Introduction

Surface water flows are the major means of external phosphorus loading to Clear Lake, given the minimal estimates from groundwater. This section de-scribes our measurement program and results for the two years of this study for which we measured phosphorus concentrations in gauged streams in order to estimate phosphorus loading. Since substantial inflows only occurred in 1992-93, the data is essentially limited to one year.

Measurements made from topographic maps of subwatershed areas (Table 5.7) show that 92% of the 458 mi^2 of land in the Clear Lake Basin is connected to the lake by surface water flows consisting of creeks, streams, drains or direct run-off. The remaining 8% are Konocti volcanics and isolated Thurston, Borax and Boggs Lakes, which lack surface flow into the lake.

Most phosphorus in natural systems is in various insoluble forms. The bulk of the element entering Clear Lake comes adsorbed to clay-sized suspended particles during winter storm flows. As surface water and rainfall come into contact with unprotected soils, particles of soil with adsorbed phosphorus are detached and transported by sheet flow run-off, to rills and into stream channels (Gillett et al, 1978). As a result, the adsorptive capacity of clay particles causes them to be a major supply route for phosphorus as soil particles are moved by erosion from the watershed into the lake.

Goldman and Wetzel (1963), and Home (1975) pointed out that Clear Lake's blue-green scum problem was due to excessive loading of nutrients from the watershed. Horne estimated Clear Lake's seasonal nutrient cycles, as well as nutrient loads within the system, from analysis of brief DWR (Lallatin, 1975) and USGS (1973) creek data and detailed 1968-73 water quality data on Clear Lake.

Horne calculated that in an average year creeks would load 130 MT of phosphorus into the lake (Table 5.1), representing over 60% of the 200 MT entering the lake annually. He considered improved management of waste water loads, calculated as 40 MT or 20% of the total external load in **Table 5.1, to** be the only practical means of reducing phosphorus input. However, he also concluded this would be insufficient to reduce levels of phosphorus in the lake to a point that would effect the blue-greens due to a large surplus of dissolved phosphorus during the algal bloom season (See Chapter 6 for discussion of phosphorus recycling from sediments.)

The DWR data from 1973-92, outlined in Chapter 4 above, show that soon after these conclusions were made, phosphorus levels in the lake dropped to limiting levels more frequently, and suppressed growth of blue-greens from 1973-86 more frequently as well.

To gain a basis for determining if management of phosphorus loading is possible, a study exploring the processes underlying external phosphorus loading from creeks was initiated in the winter of 1991-92. This turned out to be the last year of a six-year

Sub-watershed	**Sq.**	Acres	**Sq. km**	**Hectares**
Middle	50.07	32,045	130	12,968
Clover	28.12	17,997	73	7,283
Robinson	328	2,099	9	850
Lyons	3.66	2,342	10	948
Scotts	103.77	66,413	270	26,876
Forbes	4.65	2,976	12	1,204
Lakeport-N	4.94	3,162	13	1,279
Todd Road	1.58	1,011	4	409
Manning	12.00	7,680	31	3,108
Rumsey Slough	2.07	1,325	5	536
Rodman Drainage	8.09	5,178	21	2,095
Adobe & Highland	28.00	17,920	73	7,252
McGaugh Slough	7.14	4,570	19	1,849
Nice to Oaks	37.65	24,096	98	9,751
Bums Valley	10.65	6,816	28	2,758
Sulphur Bank	2.48	1,587	6	**642**
Borax *	2.54	1,626	7	658
Molesworth	5.05	3,232	13	1,308
Below Grigsby/Copsey	24.75	15,840	64	6,410
Seigler	11.73	7,507	30	3,038
Andersen Marsh to State Park *	14.51	9,286	38	3,758
Thurston *	19.35	12,384	50	5,012
Boggs '	1.38	883	4	357
Cole	25.75	16,480	67	6,669
Kelsey	44.71	28,614	116	11,580
Watershed Total	*457.92*	*293,069*	*1,191*	*118,598*
**Sub-surface drainage*	*37.78*	*24,179*	*98*	*9,785*
Total surface drainage (Watershed Total - Sub-surface drainage)	*420.14*	*268,890*	*1,092*	*108,813*

Lake Surface	**Sq. miles**	**Acres**	**Sq. km**	**Hectares**
Upper Arm	42.78	27,379	111	11,089
Oaks	6.03	3,859	16	1,563
Lower	12.63	8,083	33	3,274
Sub Total Lake	*61.44*	*39,322*	*160*	*15,925*
Grand Total (all surfaces)	***519.36***	***332,390***	***1,350***	***134,618***

Table 5.7. Area measures for Clear Lake basin sub-watersheds. Lake surface estimated from USGS maps, which assume a full lake level at 5.76 Rumsey at a lake elevation of 1326'.

drought, (23 inches of rain), so the watershed study was continued through the 1992-93 wet season. Higher discharge and flows during the second win-ter (40 inches of rain) enabled significant comparison of phosphate loadings under more typical climate conditions.

The mass balance calculations in this chapter and the next are aimed at understanding phosphorus bud-get of the lake under post-1973 conditions. The accu-mulated DWR data, and water quality and quantity information collected in the course of this study dur-ing the 1991-92 (dry) and 1992-93 (wet) winters, are used to recalculate the external phosphorus budget. This information is supplemented in the next chapter by an analysis of the internal load of phosphorus from the lake sediments during the growing season. Our objective is to estimate the magnitude of erosion control necessary to make a sufficient reduction in the phosphorus supply to limit scum-forming blue-green algal biomass. How much phosphorus would have to be removed to make a significant difference? How rapidly will losses of phosphorus down Cache Creek and to burial in the sediments exhaust the internal store of phosphorus in the bottom sediments?

Some data are also presented on iron, but as noted in Chapter 3 and elsewhere a mass balance for this element is impractical. Recall that iron does behave very generally like phosphorus in its movement into the lake on *clay* particles and its emission from the sediments during the summer. With very great caution, the phosphorus budget can be treated as a shadow budget for iron. The reasons for caution are that iron is more rapidly converted to insoluble forms than is phosphorus and that total iron (almost all in solid or insoluble suspended forms) is very abundant relative to plant needs. The chemistry of availability of the two elements is different, and those for iron poorly understood.

5.3.2 Site Selection, Discharge and Water Sampling Methods

For the calculation of external loads being delivered from the watershed into Clear Lake, two factors must be measured to estimate the mass of phosphorus flowing into the lake: (1) the rate of discharge of water down each creek and (2) the average concentration of phosphorus in the creek water. Integrated over the whole winter season, a total tonnage of phosphorus delivered to the lake is calculated.

Continuous discharge data into Clear Lake was derived by DWR from measurements of creek height electronically recorded every 15 minutes at gauging stations on Middle, Scott's, and Kelsey Creeks:

Middle Creek at Rancheria Road, above Hwy. 20
Scott's Creek at Eickhoff Road in Scott's Valley
Kelsey Creek above Soda Bay Road.

Two additional gauges are operated by USGS, one on Kelsey Creek in the canyon above Kelseyville, and the other on the Cache Creek outlet to Clear Lake near Lower Lake. Attempts were made by LCFC&WCD to gauge flows on Clover and Adobe Creeks during the study period. Other gauge locations have also been historically operated (Figure 5.1).

The three extant DWR stations provide reliable discharge data for an estimated 390.6 km^2 (151 mi^2), incorporating much of Middle and Kelsey, and 54% of the Scott's Creek watersheds. These figures were determined from independent topographic map measurements of watershed areas shown in Table 5.7. Area measurements were made from the ridge line surrounding a watershed, down to the perimeter of the lake or confluence with another drainage area., which differs from Table 5.3 where Chamberlin *et al* reported surface areas above and below gauges. On Scott's Creek, for example, the watershed area above th the gauge is 143 km^2 or 55.2 mi^2 (Table 5.3), but the total watershed down to the confluence with Middle Creek at Rodman Slough is 269 km^2 or 104 mi^2 (Table 5.7). Similar differences exist for other gauged watersheds.

The three DWR gauges incorporate 36% of the surface-draining Clear Lake watershed (45% to below Tule Lake on Scott's Creek). The Clear Lake drain-age was measured as 1,086 km^2 (420 mi^2) to above the Cache Creek dam, excluding 160 km^2 (62 mi^2) of Clear Lake surface, and 98 km^2 (38 mi^2) of subsurface draining Konocti volcanics and Borax Lake; totaling 1,345 km^2 or 520 mi^2 (Table 5.7).

In the calculation of external loads discharge quantities for the ungauged portion of the watershed, or sub-watersheds, were extrapolated from the gauged watersheds. Differences effecting discharge and phosphorus loading do exist between watersheds, for example in topography, rainfall, vegetation, soils, geology, land use, and in-channel features. But given that dominant basin-wide storm events under saturated conditions produce similar runoff conditions throughout the watershed, discharge patterns will be very similar from stream to stream (Figure 5.2). Figure 5.4 shows such similarities in 1991-92 discharge for Kelsey, Middle and Scott's Creeks.

The discharge factor is a function of surface area, rainfall intensity, and sub-basin characteristics such as soil type and slope, as described **in Section** 5.2. A major question though is the potential difference in phosphorus loading per unit runoff between watersheds. Would particular creeks have higher rates of erosion and phosphorus concentrations (higher external loading factors)? It therefore became important to sample as many of the creek systems as possible to determine such differences.

Routine water sampling sites were established as close as possible to the lake for representative measurements of phosphorus. Also, given the importance of discharge, and the error inherent to extrapolation, ideal sites were close to where the creeks were gauged.

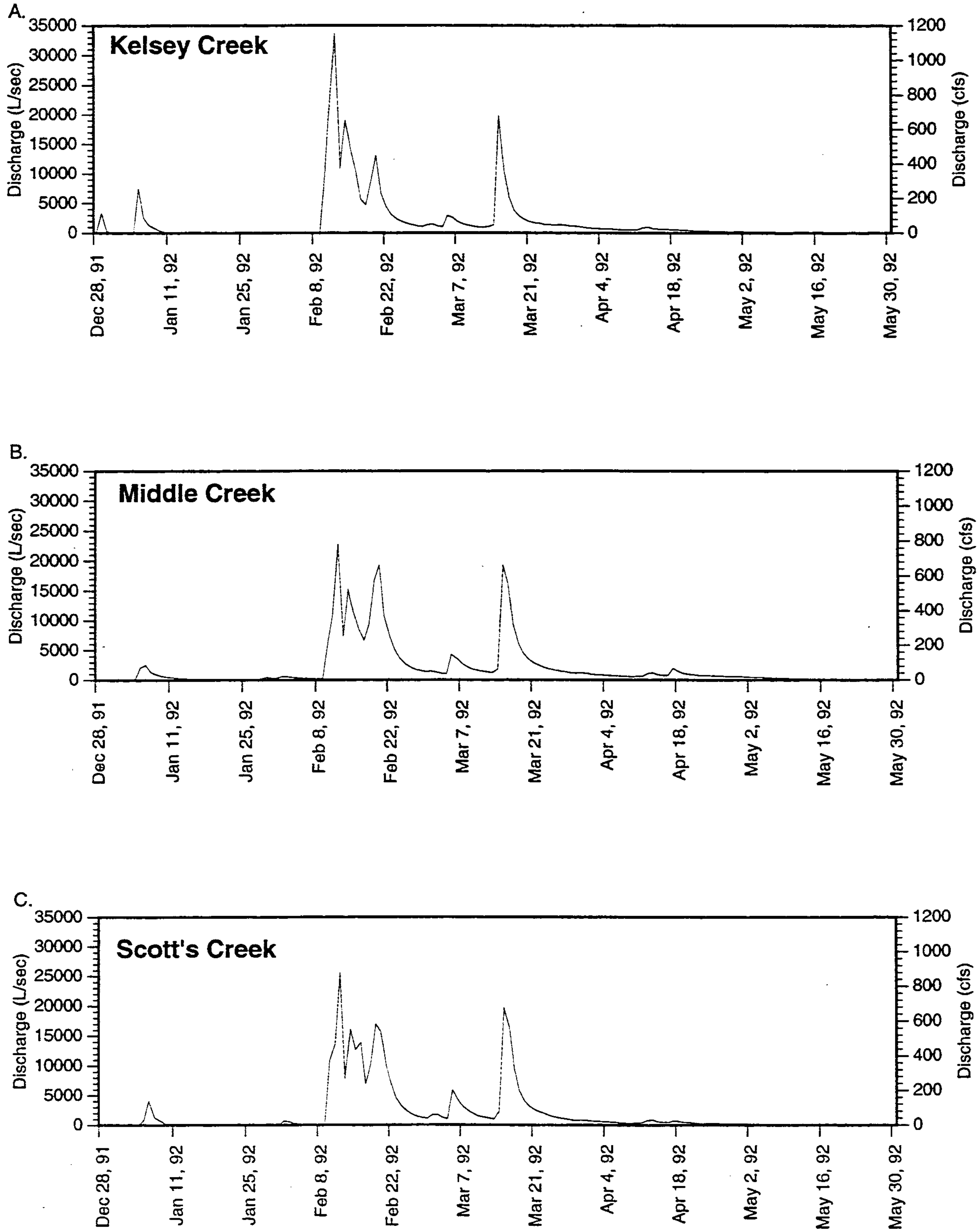


Figure 5.4. Comparison of daily discharge rates during the 1991-92 water years for three creeks, (A) Kelsey Creek, (B) Middle Creek, and (C) Scott's Creek.

These criteria were best met on Kelsey and Middle Creeks. There the DWR gauges and this study's sampling sites were 2.7 and 7.9 km (1.7 and 4.9 miles) respectively, above Clear Lake (Zalusky, 1992, LC Aggregate Resource Management Plan aerial photographs) at Soda Bay Road on Kelsey Creek, and at Highway 20/Rancheria Road on Middle Creek

On Scott's Creek, the DWR Eickhoff Road gauge is 23.7 km (14.7 mi.) above the lake, so discharge had to be extrapolated to water sampling sites downstream, and to below Tule Lake. Water chemistry sites were established above and below Tule Lake on Scott's Creek, to assess the effect of a major sedimentation basin.

To understand the contributions of individual watersheds, sampling sites were also established on ungauged creeks, namely Manning, Adobe, Lyon's, Robinson, Clover and Morrison; occasional samples were obtained from Cole, Schindler, Burns Valley and Seigler Creeks. Donna Lowdermilk of Lucerne generously took samples from Morrison Creek, and on a very wet day in January LCAQMD staff helped sample North shore creeks and city drains.

To address the contribution from watersheds supporting wastewater treatment plants, Lyon's Creek was routinely monitored. Wastewater treatment plants, such as the Northwest Plant at Lyon's Creek, take advantage of the phosphorus sorption capacity of soils, and the use of cattle to graze the grass and remove nutrients. But if the sorption is imperfect or if erosion of soils within the treatment area occurs, such systems may be a significant source of phosphorus. As a comparison, samples were simultaneously taken from neighboring Robinson Creek which has a similar watershed, but no wastewater treatment plant and very little cattle grazing.

On most creeks, bridges provided the best access for water sampling and creek height measurements. Sampling was organized on a storm event basis during both winters, and attempts were made to sample over different discharge stages. As many creeks as practically possible were sampled during an event. On a number of occasions particular sites were repeatedly sampled within an event to attempt to estimate sediment load during the course of events in case phosphorus tends to move disproportionately in the early or late part of storm *flow* pulses.

Water sampling followed recommended USGS methods (USGS, 1977). These methods are aimed at obtaining a representative sample from a stream in which the distribution of coarser suspended particles may be complex. On a pilot level, low *flows* were initially sampled using a US DH-48 wading sampler with attached 473 ml glass jar and variable diameter nozzle. At higher flows integrated depth samples were obtained from bridges using a US D-74 sampler suspended from a Stevens Reel mounted on a Bridge Boom. But ultimately more homogeneous dissolved nutrients and particles of less than 0.062mm (silts, clays and colloids) were targeted and sampled by surface grab sample.

When sampling many creeks during an event, only one water sample was generally taken per site. Variation within and between sites was explored periodically by taking triplicate samples.

At each creek sampling site the height of the water surface below a selected reference point on the bridge ramp or handrail was routinely measured; termed direct level (USGS, 1977) or in this study bridge height. The same reference point was always used, and a rating curve for each site was established by plotting discharge against water level relative to the bridge reference point.

Instantaneous discharges to construct the rating curve were periodically measured by standard USGS *(1977)* methods for cross sectional velocity metering, using either a Price AA or *Pygmy* current meter attached to either a wading rod or Stevens Reel, and an Aquacalc 5000 Datalogger to calculate an integrated discharge value.

The 1991-92 data on daily releases from the lake at the Cache Creek gauge near Lower Lake, were obtained from annual USGS Water-Data Reports. For 1992-93, preliminary data was obtained from Yolo FC&WCD (personal communication Christy Barton, Assistant General Manager). The average monthly concentration of phosphorus in the Lower Arm was multiplied by discharge in acre-ft to calculate external monthly losses of phosphorus from the lake.

5.3.3 Water quality analyses

At each creek site water samples were stored in acid washed, and double rinsed 500 ml polyethylene bottles. Standard field measurements of water temperature, conductivity, pH and dissolved oxygen were taken using standard methods, together with notes on weather, creek state, water color and condition (for example, the presence of clays and organic material).

Total suspended and dissolved solids (total solids or TS), total dissolved solids (TDS), and by subtraction total particulate matter, were determined by oven drying filtered (Whatman #40) and unfiltered samples at 103-105 °C for 24 hours (Standard Methods, 1989). Oven-dried aluminum dishes were weighed empty, full, and then reweighed once dry and having cooled for a further 24 hours in a dessicator. Due to the large number of samples per storm event, replicates were not generally included. Since analytical errors are small relative to differences between storm events and creeks, and since all important conclusions are based upon averages over time and space, replication of samples is not efficient.

Dissolved phosphorus was measured on returning to the lab or within 24 hours after refrigeration, using a Hach DR/ 1A colorimeter and an EPA approved ascorbic acid method adapted from Standard Methods (Hach Procedures Manual for Portable Colorimeter DR/1A,1988). In an external lab comparison on May 11 1992, there was good agreement between the Hach field method and duplicated water samples analyzed for dissolved phosphate at Hopland Field Station by direct colorimetry (Murphy and Riley, 1962).

During the 1992-93 winter, total phosphorus was routinely determined on unfiltered creek samples at Hopland using a perchloric acid digestion (Sherman, 1942) prior to phosphate determination.

For preliminary investigative purposes dissolved plus suspended iron concentrations were also determined at Hopland by flame atomic adsorption spectrophotometry. Refrigerated some 2-3 weeks before analyses could be performed, decanted water samples were acidified to 1% of concentrated nitric acid just before analysis, and directly aspirated.

5.3A External loading results

Figure 5.4 gives daily DWR discharge data for 1991-92 (DWR, 1992) and Figure 5.5 shows hourly discharge data for 1992-93 (personal communication Jane Stewart, DWR) for Kelsey, Middle and Scotts Creeks. There were some very significant discharge differences between water years.

In early 1992 the lake rose from 0.13 to 1.36 meters (0.43 to 4.52 feet Rumsey) by April. Between October 1991 and May 1992 $3.9x10^6$ m^3 (3,178 ac-ft) discharged from the lake down Cache Creek (USGS, 1992). There were only three minimal storm events generating flows greater than $1.5x10^4$ 1/ sec (500 cfs) in the three major creeks. The plots show the highest flow, experienced on February 11 on Kelsey Creek at Soda Bay Road, peaked above $3.3x10^4$ l/sec (1,100 cfs). Total rain for 1992 was 58 cm (23 in.) at Lakeport.

In contrast, 1993 flows crested above $3.0x10^4$ L/sec (1000 cfs) during more than 11 separate storm events. The lake rose from 0.16 to 2.53 meters (0.52 to 8.44 feet Rumsey) in late February, with 2.5x108 m^3 (200,726 acre-ft) exiting through Cache Creek between January and April. The maximum discharge experienced was on January 22nd with $1.7x10^5$L/ sec (5500 cfs) flowing through Kelsey Creek. Rainfall for the 1993 water year was 100 cm (40 in.) at Lakeport.

For each of the three major creeks, during the 1993 runoff season (water year), total phosphorus concentrations and total suspended solids are plotted against discharge **(Figures 5.6** and 5.7). Finally, **in** Figure 5.8 all total phosphorus measurements are plotted against total solids. Total solids is a measure of everything that is either dissolved in, or being carried in suspension by, the water. Recall that the sampling method is a grab sample, so total solids is the suspended sediment fraction less than 0.062 mm (silt and clay).

Relationships between the parameters in **Figures 5.6,** 5.7 and **5.8** are strong. For each of the three major creeks, a doubling of discharge caused a doubling in both total phosphorus **(Figure 5.6),** and total solids (Figure 5.7) concentrations. Creeks vary in loading rate of phosphorus and total solids, as indicated by the different slopes in Figures 5.6 and 5.7, evidence that the erosion process differs between watersheds.

The rate of run-off or discharge is a critical factor in the process of erosion and the delivery of phosphorus to the lake. A doubling of velocity enables water to carry 32 times more soil, and to transport particles 64 times larger; the total erosive power is 4 times greater (Gillett *et al.,* 1978).

For the majority of natural creeks total phosphorus loads correlated strongly with total solids indicating that (total) phosphorus is strongly associated with erosional processes. Lyons Creek samples do not fit on the general regression line of total phosphorus against total solids (Figure **5.8** and 5.9). The Lyons Creek watershed is impacted by wastewater effluent spray fields of the Northwest Regional Treatment Plant and intensive cattle grazing. The impacts on the watershed by wastewater treatment facilities are discussed in Section **5.3.6.**

The range of total and dissolved phosphorus and total solids concentrations measured during 1992-93 was considerable. The highest total phosphorus concentration sampled was a single value of 4.8 mg/L observed in a muddy roadside stream flowing into Soda Bay on 20th January, the period of highest winter discharge. During the three largest storm events

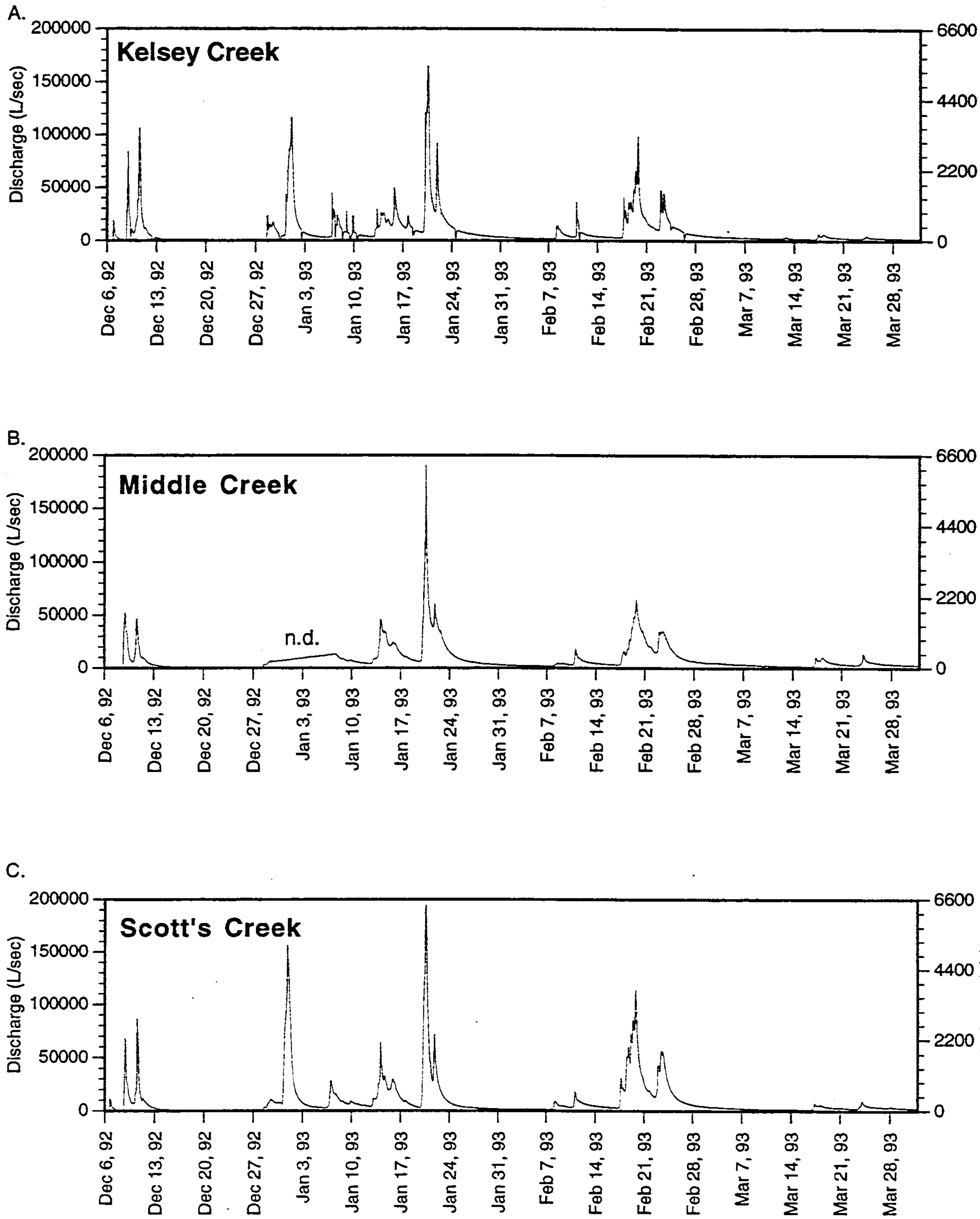


Figure 5.5. Comparison of hourly discharge rates during the 1992-93 water years for three creeks, (A) Kelsey Creek, (B) Middle Creek, and (C) Scott's Creek. "n.d." indicates a period of missing data.

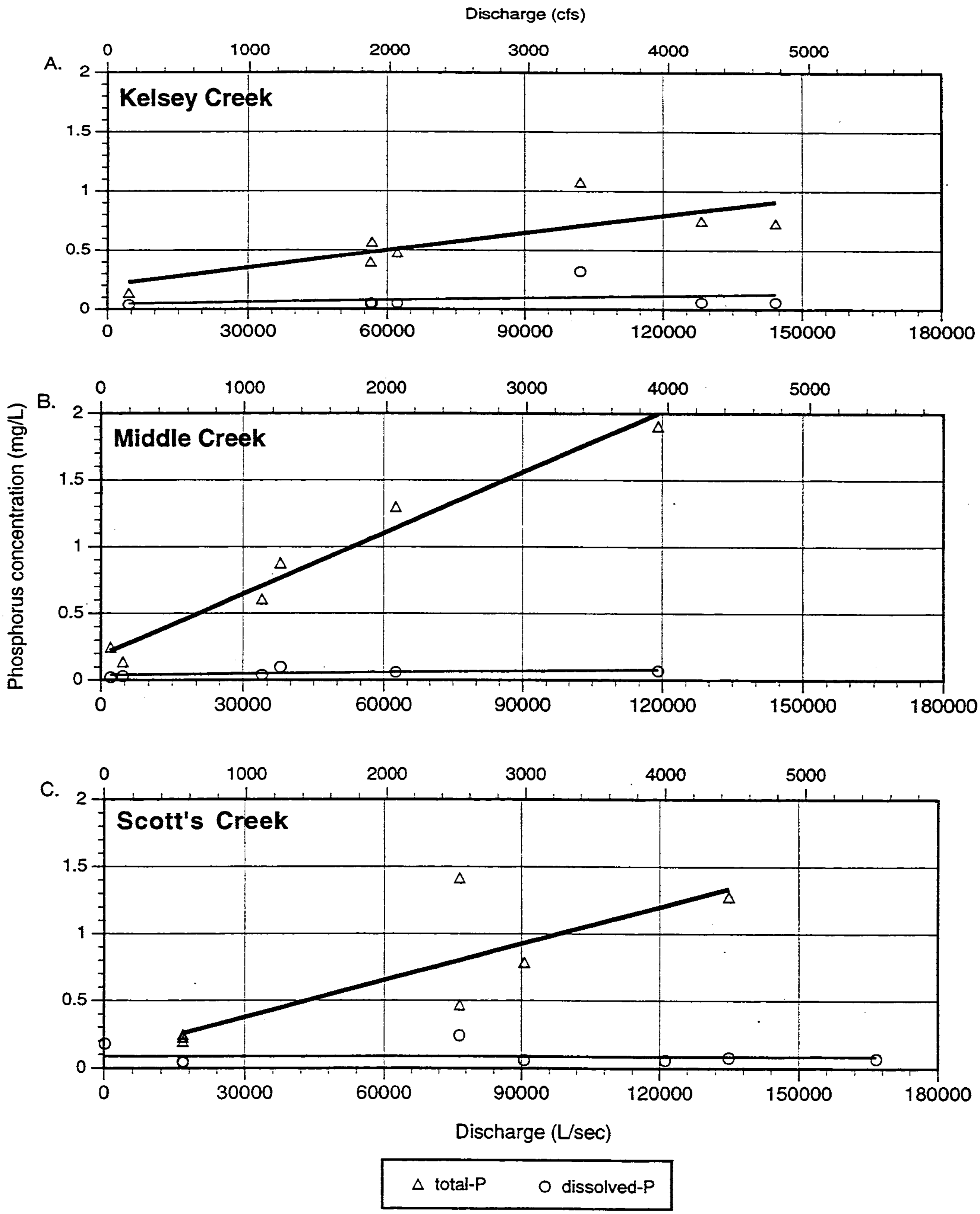


Figure 5.6. Total and dissolved phosphorus plotted against discharge for three creeks, (A) Kelsey Creek, (B) Middle Creek, and (C) Scott's Creek.

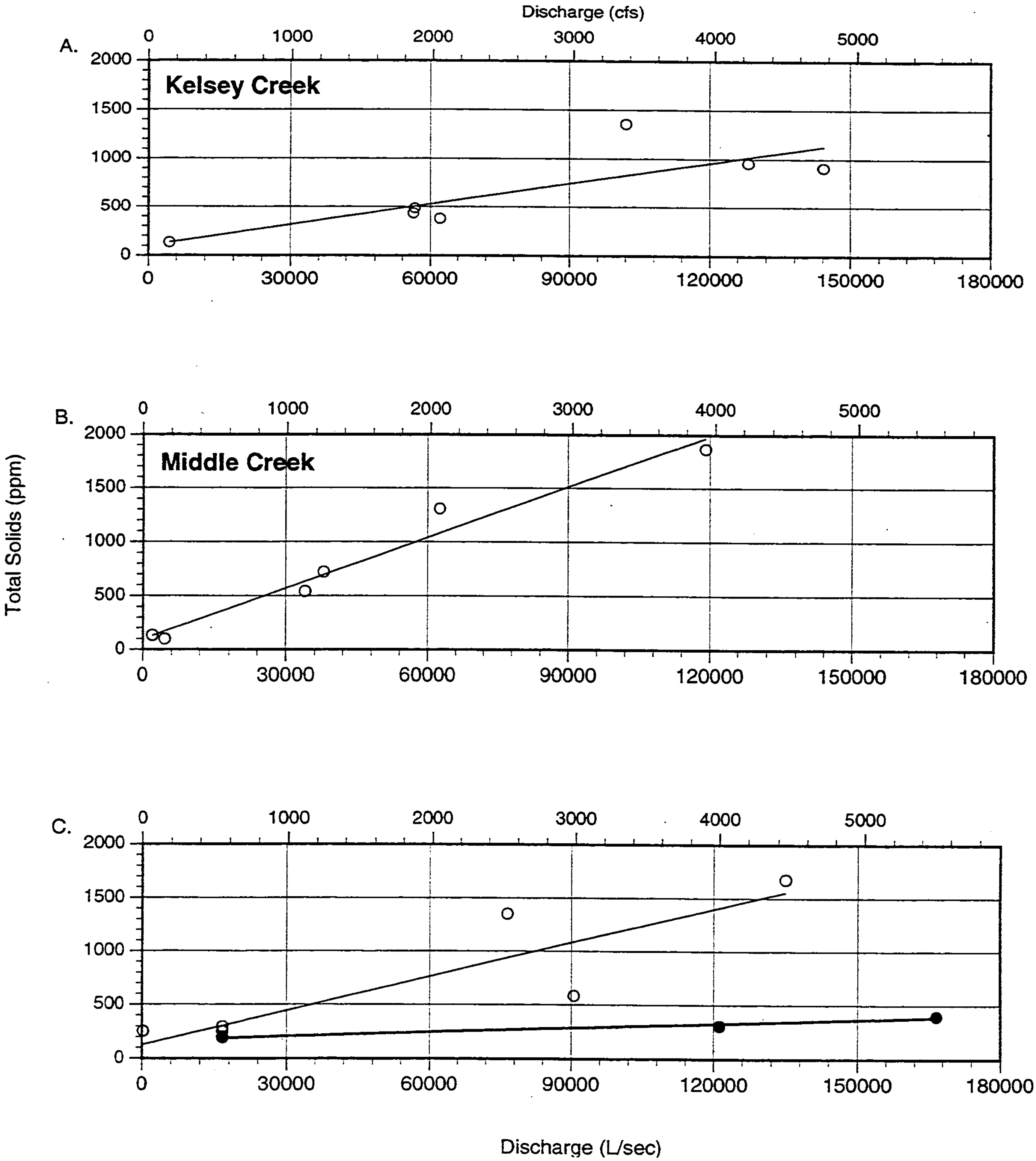


Figure 5.7. Total solids plotted against discharge for three creeks, (A) Kelsey Creek, (B) Middle Creek, and (C) Scott's Creek. The Scott's Creek chart shows data taken above (0) and below(•) Tule Lake.

around December 10th, January 1st and 20th, above-

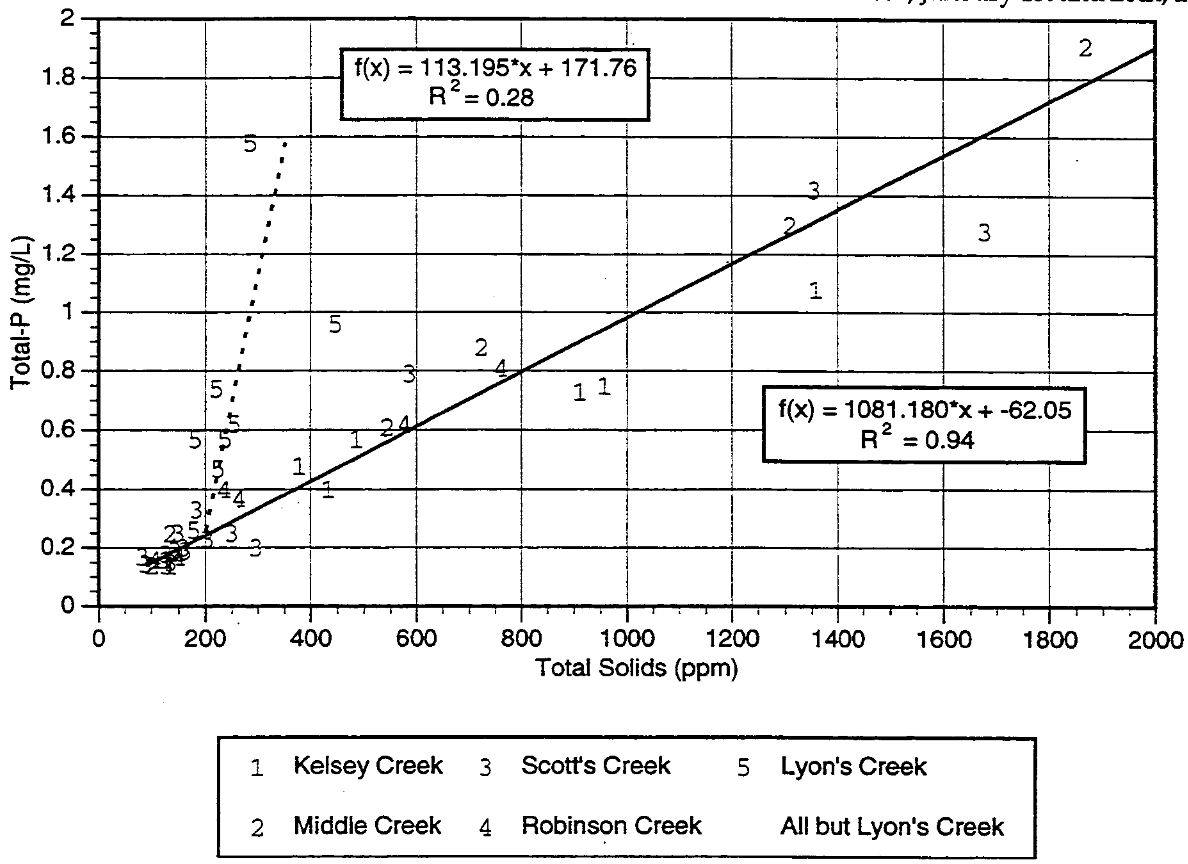


Figure 5.8. Total phosphorus plotted against total solids for major creeks. The solid regression line fits data for Kelsey, Middle, Scott's, and Robinson creeks, whereas the dotted line is for the Lyon's Creek data series only.

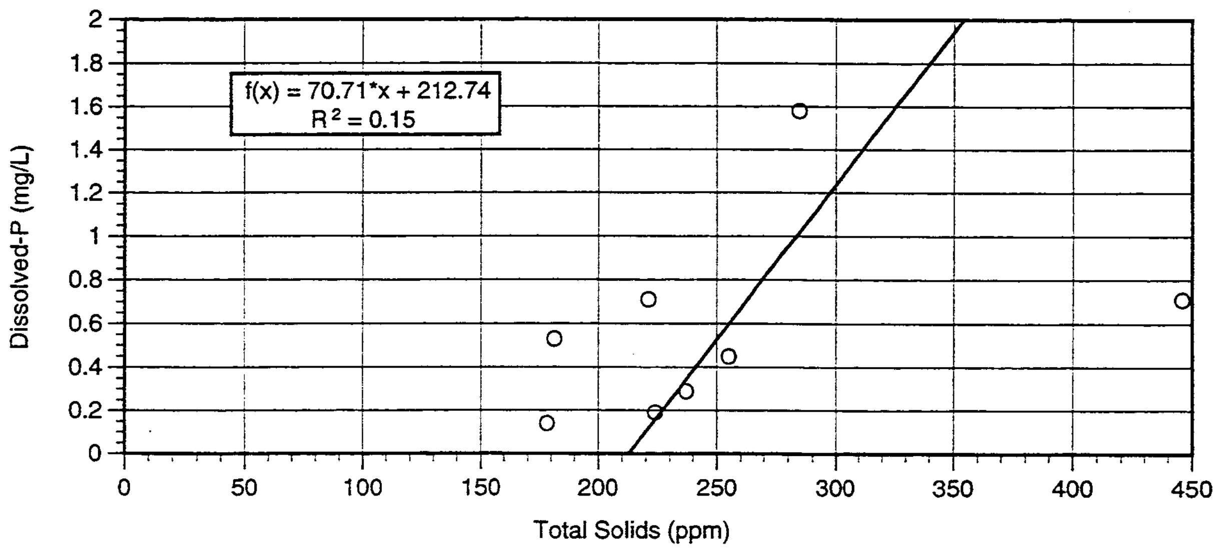


Figure 5.9. Dissolved phosphorus plotted against total solids for Lyon's Creek.

average samples of between 1 and 3 mg/L were taken from major creeks, including Middle (n=2), Scott's (n=2), Kelsey, Manning, Lyon's (n=2) and Morrison (n=3), and also from roadside drains in Lakeport on Scott's Valley Road and Hill Road where slips occurred during the January 20th event. Some 60 other creek water samples taken from ungauged creeks and roadside drains around the Clear Lake basin between December 2, 1992 and 1 April, 1993 showed lower total phosphorus levels ranging from 0.1 to 0.8 mg/L.

It is an interesting to note for comparison that total phosphorus concentrations in the lake during the December 1992 through March 1993 winter period averaged 0.23 mg/l. Creek concentrations were generally higher. Once in the lake, external total phosphorus loads settled quickly into the lake sediments. During this winter period dissolved phosphate in the lake dropped from 0.13 to 0.04 mg/l. Although this followed the typical pattern, the 1992-93 winter had unusually high levels of both dissolved and total phosphorus, but especially dissolved phosphorus.

For dissolved forms of phosphorus, stream sample concentrations were highest during the periods of treated wastewater discharge at the end of February 1993. During this time, Lyon's Creek ran at 1.6 mg/L dissolved phosphorus. At other times in the winter, levels ranging between 0.8 and 0.4mg/l were measured in Lyon's (n=4), Burns Valley, and Morrison Creeks, as well as in Todd Road Drain, and roadside flows on January 20 in Lakeport's Konocti Avenue, Lakeport Boulevard, and D Street. Many of these higher concentrations of dissolved phosphate appear to be associated with wastewater overflows or other urban sources. As might be expected, all major creek systems showed lower concentrations of dissolved phosphate, between 0.4 and 0.01 mg/l, due to large dilution and sorption of phosphate onto the heavy suspended load.

Total solids measurements are useful in characterizing sediment delivery to the lake. During the main event of winter on January 20th, roadside drains, associated with landslides, showed the highest concentrations of total solids (eroded silt and clay), between 3500 and 5000 ppm. During the same period, steep and erosive Morrison Creek was measured at 2800 ppm total solids, and the large $6.1x10^4$ l/sec (2,000 cfs) flows going through Middle Creek at Highway 20, and Scott's Creek at Hendricks Road were observed carrying 1800 ppm total solids. During the three high flow events of December 10th, January 1st and 20th, Morrison, Kelsey, Scott's, Middle and Manning had total solids concentrations between 800 and 1500ppm. All other sampled creeks ran below 600ppm total solids, and after January 23 nothing was measured above 400ppm total solids, even with two fairly significant events towards the end of February, perhaps indicating most of the erosion in the basin and transportation to the lake had taken place during December and January.

Clear Lake's outlet, Cache Creek, is the main route by which phosphorus is discharged **from the system. Table** 5.8 shows the creek's monthly release profile, total phosphorus concentrations, and loading for 1991-93.

Discharge of water via Cache Creek during the course of this work can be broken into five periods **(Figure** 5.10). Through the winter of 1991-92 the lake had risen to only 1.4 meters (4.54 feet Rumsey) by April. In the 8 months up to May 1992 some $3.9x10^6$ m^3 (3,178 acre-ft) was released from the lake. Under the Solano Decree draw-down schedule, Yolo County Flood Control released about $4.4x10^{6}$ m^3 (36,107 ac-ft) for irrigation purposes from June through October 1992. During a third period at the end of 1992 prior to the big 1993 rains, releases were back to a minimum of $0.4x10^6$ m^3 (363 ac-ft) over the two months.

In January 1993 the lake filled rapidly, and on January 25 at 2.1 meters (7.10 feet Rumsey) the dam was opened under the Gopcevic Decree. This decree regulates winter lake levels and the release of flood water such that, if the rains have been sufficient, Clear Lake will be full at 7.56 feet Rumsey (2.3 meters) on March 15 each year. According to preliminary figures made available by Yolo FCWCD, around $2.5x10^8$ m^3 (200,726 ac-ft) was released before May 1993, and from then until August 1993 a further $1.4x10^8$ m^3 (116,970 ac-ft) was discharged for the irrigation season.

Monthly total phosphorus concentrations of the Lower Arm of Clear Lake are also shown in **Table** 5.8, and are considered representative of the concentrations of total phosphorus leaving through Cache Creek. The loadings are discussed below.

5.3.5 Calculation of external loads of phosphorus

The discharge versus concentration curves **(Figures** 5.6 and 5.7) give expected concentrations of total phosphorus and total solids at any flow rate. The slope of these relationships are shown in **Table** 5.9 (cfs, the standard nomenclature, have been converted to m^3/sec for calculation purposes). The slope is called the External Loading Factor (ELF), and is a good index of the sediment yield status of a watershed. The flux of phosphorus into the lake increases

Date		Discharge Ac-ft	Discharge m^3	Total-P mg/L	Total-P MT
Oct	1991	747	921,798	0.50	0.461
Nov		1,170	1,443,780	0.30	**0.433**
Dec		584	720,656	0.18	0.130
Jan	1992	-	181,398	0.07	0.013
Feb		72	88,848	0.10	0.009
Mar		126	**155,484**	0.08	0.012
Apr		149	183,866	0.08	0.015
May		183	225,822	0.13	0.029
Jun		10,970	13,536,980	0.14	1.895
Jul		14,560	17,967,040	0.31	5.570
Aug		6,540	8,070,360	0.33	2.663
Sep		1,990	2,455,660	**0.35**	0.859
Oct		2,047	2,525,998	**0.48**	1212
Nov		240	296,160	027	0.080
Dec		123	151,782	**0.34**	0.052
Jan	1993	49,911	61,590,174	025	15.398
Feb		64,533	79,633,722	024	19.112
Mar		74,956	92,495,704	024	22.199
Apr		11,326	13,976,284	0.18	2.516
May		28,572	35,257,848	021	7.404
Jun		28,368	35,006,112	0.12	4201
Jul		34,596	42,691,464	026	11.100
Aug		25,434	31,385,556	na	na

Source: *1991/92 Q-*data *USGS Ca-92-4; 1992/93 Prelimin. YCFCWCD 1991/92 Total-P DWR CL4; 1992/93 Why/Vaughn*

Table 5.8. Cache Creek discharges and total phosphorus export, 1991 to 1993.

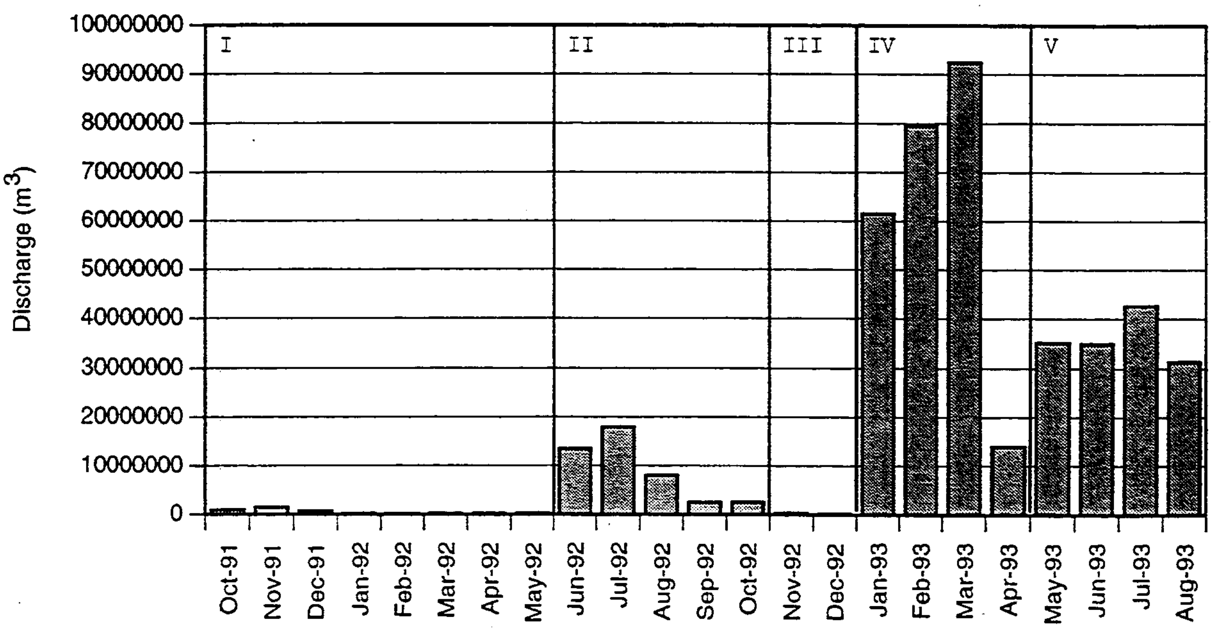


Figure 5.10. Cache Creek discharge profile, 1991 to 1993.

faster than flow rate because each cubic foot carries proportionally more phosphorus at higher flows due to its higher power to cause erosion, maintain particles in suspension; and ultimately to deliver them to the lake; at low water streams are clear and in flood they are muddy.

A similar regression relating total phosphorus to total solids in Figure **5.8** shows that for all creeks measured, 1 gram of total solids yields approximately 1 mg. of total phosphorus. So, every MT of silt and clay entering the lake puts in 1kg of phosphorus.

Table **5.9** shows that during 1992-93, based on the ELF, Middle Creek watershed sediment yield potential was around 2.5 times that of Kelsey Creek Middle Creek at Highway 20 also had approximately 1.4 times the potential of Scott's Creek above Tule Lake. Without Tule Lake, Middle and Scott's Creek sediment yield potentials were fairly similar. Assuming a steady state inflow and outflow for Tule Lake at high flows, this 2.6 km^2 (1 mi^2 or 640 acre) natural sedimentation basin reduces the loading factor for Scott's Creek by an estimated 70% due to the deposition of upstream erosion products (see Section 5.3.8 for details). As a result the sediment yield potential of Middle Creek is about 5 times greater than Scott's Creek Middle Creek may also be more erosive since all of the natural floodplains have been destroyed or are non-functional, while Scott's Creek has extensive depositional flood plains between Eickhoff Road and Tule Lake.

The total solids loading for the whole 1993 winter period from the three major creeks, or 45% of the watershed, is calculated by multiplying each hour's discharge squared by the ELF for total solids in Table 5.9, then totaling the results for each creek, and multiplying by the number of seconds per hour, and ac-

In the calculation of total load, the discharge is squared, since the calculation has two parts. In the first part it is necessary to work out the expected concentration of total solids or total phosphorus for the particular known discharge. Thus, the discharge is multiplied by the ELF to give the expected concentration. Recall that ELF is the slope in the graph of Discharge (Q) vs. total solid concentration [TS] (Figure 5.7), so QxELF=[TS]. In the second part of the calculation, the mass of TS passing the sample point, per second ,has to be calculated. So, Qx[TS] will equal the mass of TS passing per second. These two calculations can be combined, so that the mass of TS per second = QxELF(xQ) = ELFxQ^2. This mass is then multiplied by the number of seconds per hour, to give the mass per hour. Each mass per hour is then summed for the winter season to calculate the total winter load.

The Scott's Creek calculation is further complicated, since flows are gauged at Eickhoff Road, which represents only 146 km^2 (56 mi^2) or 54% of the Scott's Creek watershed. Below Tule Lake the total Scott's Creek watershed area was measured as 270 km^2 (104 mi^2). Total solids loadings above Tule Lake were calculated by multiplying the Eickhoff Road discharge data by the loading factor measured above Tule Lake (Table 5.9), and then extrapolating this figure from 146 to 270 km^2 of watershed. This gives a theoretical figure of Scott's Creek loading to Clear Lake, without taking Tule Lake into consideration. The problem of calculating the effect of Tule Lake is discussed further in Section 5.3.8.

Table 5.10 shows the calculated sediment and phosphorus loading from the three gauged and measured creeks. These loadings must be extrapolated to the entire watershed. If we assume that the three measured creeks together are representative of the whole watershed, there are two methods of extrapolation. (In both cases the extrapolation uses Scott's Creek

Entering CL	ELF (Total-P) $g\ m^{4}\ sec$	ELF (TS) $g\ m^{-6}\ sec$
Middle Creek	0.0185	19.4
Kelsey Creek	0.0071	8.5
Scotts Creek:		
above Tule Lake	0.0124	14.1
below Tule Lake	0.0035	4.2

Table 5.9. External loading factors (ELF) for the discharge of total phosphorus and total solids into Clear Lake from gauged watersheds from 1992 to 1993.

above Tule Lake to represent that portion of the watershed.) We can suppose that sediment yield is (1) proportional to the estimated area of ungauged watershed (Table 5.10A) or (2) proportional to the estimated flows from ungauged watersheds (Table 5.10B), based on data from Table 5.3. Recall from Section 5.2 that the ungauged watersheds are, on average, at low elevation and are generally less steep that the three larger gauged watersheds. Thus they probably have lower rainfall and lower runoff per unit area than the measured streams and thus less potential for erosion. On the other hand, human disturbances from grazing, road building, agriculture, urbanization, gravel extraction and the like are concentrated on the lower, gentler parts of the basin, and nutrient loads per unit runoff are likely to be higher in the lower elevation watersheds. Comparing the results of the two assumptions gives an indication of the degree of uncertainty in loading estimates due to limited data. With the limited data available, it is probable that the 1993 erosion rate was in the range of 219,000 to 311,000 MT.

Since 1993 was a wetter than average year, loading will also average less. We have used the 1993 ELF and past discharge data to estimate loading since 1969 **(Table 5.15 in Section 5.3.9).** The mean annual loading using the two different extrapolation methods is estimated to range between 110,000 and 156,000 MT (see Section 5.3.9 for details).

An additional method of estimating sediment yield is by calculating the historic sedimentation rate based on sediment rate studies on the deep water cores in the Upper Arm taken by the USGS in 1973 and 1980 (Sims, *et* al., 1988: 38-9). For the last 10,000 years, the sedimentation rates for these cores have averaged 0.65 mm/yr (density corrected to 2.65 g/cc) and consisted of primarily fine grain sediments. However, sedimentation rates are not uniform across the lake. Deposition will be negligible in shallow water due to wave resuspension, and sediment will be disproportionately accumulated in the deeper waters. The magnitude of this sediment focusing effect is not known for Clear Lake, but a factor of 2 enhancement is a reasonable guess. Assuming a uniform sedimentation rate for fine grain sediments across the entire Upper Arm (surface area of approximately 111 km^2), about 96,000 MT/yr of fine grain sediments have been deposited over the last 10,000 years ($1.11*10^{12}$

A			Method 1		
	Area km2	Area %	TS (metric tons)	total-P (metric tons)	total-P (MT/km^2)
Middle Creek	129.7		51,378	52	0.40
Kelsey Creek	115.8		24,264	24	021
Scotts Creek					
w/o Tule Lake			103,531	104	0.39
w/ Tule Lake	268.7		31.113	31	0.12
Total, gauged	514.2	47%	106,755	107	0.21
Clear Lake Basin	1,101.9		311,250	311	028

B	Method 2				
	Average Flow (L/sec)	Average Flow %	TS (metric tons)	total-P (metric tons)	total-P (MT/L/sec)
Middle Creek	2,576		51,378	52	0.020
Kelsey Creek	2,007		24,264	24	0.012
Scotts Creek					
w/o Tule Lake	2,986		103,531	104	0.035
w/ Tule Lake	2,986		31.113	31	0.010
Total, gauged	7,579	61 %	106.755	107	0.014
Total. ungauged	4.752	39%	112,341	112	0.024
Clear Lake Basin	12,331		219,096	219	0.018

Table 5.10. Annual phosphorus loading for Clear Lake, 1993.

cm^2 * 0.0325 cm/yr * 2.65 g/cc * 10^{-6} MT/g). About 79% of the lake's inflow is to the Upper Arm. If sedimentation in Lower and Oaks Arms scale proportionately to inflow, the total sediment load would be 122,000 MT/yr. This estimate corresponds to a natural erosion rate before human disturbance.

Goldstein and Tolsdorf (1994), using a modeling approach, estimate that approximately 189,000 MT of fines currently are eroding from the basin annually, and that rates without human disturbance would be 50% of this figure.

These estimates for sediment loading agree within the wide uncertainties dictated by limited data. For simplicity, in calculations of phosphorus loading that follow, we assume the area-based estimate of 311,000 MT of sediment for 1992-93 (corresponding to 311 MT of total phosphorus). Further reducing uncertainty in the loading calculation will depend upon obtaining longer and more complete monitoring data.

5.3.6 Wastewater Contribution to 1993 external Loads

The contribution of wastewater to the nutrient budget of Clear Lake has been presumed small. Home's (1975) budget estimated that domestic wastewater contributed 40 MT/yr of phosphorus. The construction of zero discharge treatment plants in the 1970s using sprayfields for the disposal of secondary-treated waste should have reduced this figure still further. Studies of the Lyon's Creek drainage, the site of the disposal area for the Northwest Treatment Plant, are used here to evaluate this hypothesis. We also discuss possible contributions from septic system leakage and exfiltration from sewage systems.

5.3.6.1 Wastewater Treatment Plants

Wastewater is treated at the plant and the secondary treated effluent is sprinkled on rolling hill land, which is used as cattle pasture. The sprayfield is drained by Lyon's Creek During the winter when the fields are saturated with water, the treated waste is stored in a reservoir until the dry season. In theory, no waste leaves the site, though the heavily fertilized and grazed spray field can hardly be expected to behave exactly like an unirrigated drainage. However, phosphorus tends to be immobilized in soils, and thus sprayfield application should be a quite efficient treatment method.

The difficulty with the current systems is that in high rainfall years the winter storage capacity for secondary treated effluent is exceeded. There are four treatment plants around the basin that operate similarly to the Northwest Plant., the Southeast Plant at Clearlake, the Clearlake Oaks Treatment Plant, and the Lakeport Treatment Plant. During this study the Northwest Treatment Plant winter storage reservoir began overflowing into Lyon's Creek and Clear Lake after February 20, 1993 (personal communication Steve Brodnansky, Special Districts, Martin Winston, Environmental Health, and Paul Marshall, Regional Water Quality Control Board). Previously, the reservoirs at all four plants overflowed to the lake during 1982,1983, and 1986; all except Lakeport did so in 1993. The SE plant also had overflows in 1979, 1980, and 1984 (personal communication Steve Brodnansky, Special Districts). In each year discharges continued from six to 12 weeks (personal communication Paul Marshall, Regional Water Quality Control Board). The Southeast plant discharges to Burns Valley Creek, Lakeport to Manning Creek through the Todd Road drain, and the Oaks plant goes directly to the lake.

Table 5.11 shows the concentrations of both total and dissolved phosphorus in Lyon's Creek during 1992-93. Results for Robinson Creek, the control watershed without treatment plant, are also presented. Mean concentrations of total and dissolved phosphorus in Lyon's Creek were 2 and 7 times (respectively) more concentrated than flows in Robinson Creek Concentrations more than doubled at the end of February when the reservoir overflow dominated flows according to our sample of March 1. **Figure 5.8** shows the difference between total phosphorus load as a function of discharge for Lyon's Creek in comparison to other creeks. **Figure 5.9** shows the Lyon's Creek discharge of dissolved phosphorus as a function of total solids. Some combination of spraying wastewater application and cattle grazing elevates especially dissolved phosphorus in Lyon's Creek relative to Robinson Creek

Since neither of these creeks are gauged, discharge figures are not available, but it is possible to estimate the total run-off based on surface area comparisons with Scott's, Middle and Kelsey Creeks. Some differences might be expected since Lyon's is saturated earlier on due to irrigation, and once hydraulic overload is reached reservoir discharge is being caused by Inflow and Infiltration (I&I) water being pumped from other watersheds. Creek discharge is therefore likely to be higher than estimated from the gauged watersheds. However, other complexities such as land elevation and subsequent rainfall can counteract this underestimation of discharge.

To estimate the phosphorus contribution from the wastewater treatment plants, the contribution must be broken down into two parts, the runoff from the

Sampling date	Total Phosphorus (mg/L)		Dissolved Phosphorus (mg/L)	
	Lyon's Creek	Robinson Creek	Lyon's Creek	Robinson Creek
1218/92	0.96	0.81	0.71	02
12/10/93	0.62	0.4	0.45	0.08
12/31/93	0.57	0.37	0.29	0.07
1/20/93	0.47	0.62	0.19	0.08
2/20/93	0.57	0.17	0.53	0.04
3/1/93	1.58	0.16	1.58	0.04
Mean	0.8	0.42	0.63	0.09

Table 5.11. Concentration of phosphorus in Lyons Creek, site of the Northwest Wastewater Treatment Plant watershed, and Robinson Creek control.

irrigation fields and the overflows from the storage reservoirs. The Northwest Regional Treatment Plant irrigates 22 km^2 (540 acres) of grazing land and overflowed approximately 874,000 m^3 (231. million gallons) between February 17 and May 4, 1993. The Southeast Regional Treatment Plant irrigates 0.99 km^2 (244 acres) and overflowed 454,000 m^3 (120 million gallons) in 1993. No sampling below the Southeast plant was conducted.

With the very limited data available, the contributions from the Northwest Regional Treatment Plant are estimated as follows. First, we estimate the contribution from spray fields during the non-overflow periods using the data **in Table 5.11** excepting 1 March 1993, which was taken during the overflow period. Since Lyon's Creek, area of 9.6 km^2, and Robinson Creek, area of 8.5 km^2 (sampled as a control), are not gauged, their runoff must be estimated. As with the Calculation of External Loads, Section 5.3.5, the loading from runoff will be calculated using two methodologies, surface area and average flow.

Method 1, applying the average loading factor for the entire basin from **Table 5.10 to** Lyon's Creek, yields an estimated load from Lyon's Creek of 2.7 MT. During the period when winter flows were only carrying water from the spray fields (plus non-sewage related loads), total phosphorus concentrations were elevated by 35% in Lyon's relative to Robinson Creek, leading to an estimate of about 1 MT of additional phosphorus due wastewater spraying.

Method 2 assumes that the flow in Lyon's creek is identical to the estimated ungauged Upper Arm contribution to runoff of 3.71 l/sec/k^{m2}. Assuming that without the treatment plant, Lyon's and Robinson Creeks would have the same total phosphorus concentration, Lyon's Creek should carry 0.60 MT of phosphorus. Applying the same 35% elevation factor, this method estimates the wastewater derived component of Lyon's Creek flows at only 02 MT.

Method 1 overestimates load because the low-elevation Lyon's Creek drainage will tend to have quite low runoff relative to the basin as a whole. Method 2 will tend to be an underestimate because it does not take into account the importation of water into Lyon's Creek drainage by the sewage treatment system. On average, soils and groundwater in the Lyon's Creek drainage will be more saturated due to summer importation and irrigation than would be the case without the treatment system, and less rainfall will be stored in the system. Without actual flow data, it is difficult to narrow these upper and lower bound estimates.

The loading due to overflows of the storage reservoir can be calculated by multiplying the measured phosphorus concentrations and the amounts overflowed. While only two Lyon's Creek samples were collected during this overflow period, only the March 1, 1993 sample was collected during low flows and consists primarily of treated overflows. The 1.58 mg/ 1 concentration appears low for secondary-treated wastewater. Typical levels range from 6-15 mg/l total phosphorus (Table 12-3, Metcalf & Eddy, 1979). This indicates that some of the phosphorus is probably removed from the treated wastewater stream within the reservoir (by adsorption onto sediments) and/or the stream system above the sampling location. Using this as the average concentration for the entire overflow period leads to an estimated phosphorus loading of approximately 1.3 MT (0.874x1.54).

Therefore, it is estimated that the Northwest Regional Wastewater Treatment Plant added approximately 1.5- 2.3 MT of total phosphorus to Clear Lake in 1993. Insufficient data is available for the other treatment plants around the lake to estimate their contribution, although it is reasonable to assume a similar contribution, increasing the total phosphorus loading to 3 to 6 MT.

When compared to the total 1993 phosphorus loading of 219 to 311 MT, the addition of 3-6 MT does not appear to be significant. The major uncertainty in this conclusion is due to the fact that sewage phosphorus is disproportionately in dissolved and otherwise available forms, whereas much of the contribution from erosion products may not participate actively in the biological cycling of phosphorus. If we assume that as little as 1 /3 of the total phosphorus load from streams is available, the contribution from treatment plants is still comparatively small, though not negligible.

The Central Valley Regional Water Quality Control Board has issued a Cease and Desist Order that overflows stop by September 1, 2001. Without overflows, the increased total phosphorus concentrations due to spray disposal of treated effluent would probably be on the order of 1 MT/yr. Thus, the sewage treatment plant modernization at Clear Lake, when complete, will constitute a 98% improvement relative to the contribution estimated by Horne (1975).

Collection lines may leak considerable amounts of sewage between households and the treatment plant. The same leaks that permit infiltration of the systems at high lake level may permit discharge to the lake in the summer. At the present time, there are no data on which to base a quantitative analysis of this source. Major leaks are likely to result in high counts of fecal coliforms. According to Martin Winston (personal communication), there are no data suggesting major leaks, although high counts in confined channels are perhaps attributable to minor leakage.

5.3.6.2 Loading From Septic Leachfields

An unknown percentage of lakeside residences around Clear Lake still depend upon septic systems for waste disposal, despite considerable extension of sewer systems in the last two decades (Martin Winston, personal communication). In principle, septic leachfields are much like land disposal systems and should effectively retain phosphorus and iron in the leachfield. However, most septic systems at Clear Lake were constructed long ago, and do not meet modern standards. Two studies, one of Jago Bay and one of Soda Bay (Questa Engineering Corp., 1990a, 1990b) document that failure and leakage of septic systems is common, particularly at the steep Jago Bay location. However, studies of fecal coliform counts and nitrate concentrations in shallow waters in these areas failed to demonstrate a significant impact of wastewater loading from septic systems.

It is informative to attempt a worst-case calculation for septic leachfield loading. Reckow and Chapra (1983) cite 0.3-1.8 kg/capita/yr for phosphorus input from septic waste fields. Since most lakeside residents are summer homes, we might figure the lower number is most reasonable. If there are 2,000 lakeside homes unsewered, with an average of three residents per home , the contribution from unsewered residences might be ca 2MT/yr. If we assume that dissolved phosphorus rich leach fields are entirely available and that stream total phosphorus is 1/3 available, then streams deliver ca. 50 MT of avail-able phosphorus in an average year. Under these assumptions, septic leachfields might contribute 4% of the available phosphorus. This calculation, together with the largely negative findings of the Questa studies, suggests that septic leachfields are, at worst, a relatively small source of phosphorus.

5.3.7 1992 and 1993 Losses to Discharge from Cache Creek

Cache Creek external loading releases during 1991-92 and 1992-93 are calculated by multiplying the mean monthly lake concentrations of total phosphorus (DWR data) with the volume of water released via Cache Creek (Table 5.8).

The estimated loads of total phosphorus lost from the lake for the period of our measurements are given in Table 5.12. 311 MT of total phosphorus entered Clear Lake during the 1993 water year, and 83 MT were released (as of July). This represents a direct loss of 27% of the input. as a result, a net gain of approximately 228 MT of total phosphorus occurred. The loss of total phosphorus during the 1992 water year was much less at just 12 MT, since water was not available for irrigation releases due to the low levels in Clear Lake at the end of the drought.

5.3.8 The Role of Tule Lake

The preliminary analysis in Section **5.3.5** of the effect of Tule Lake on the water quality of Scott's Creek, indicated that 70% of the suspended sediment and

Period from	to	Total Phosphorus (metric tons)	Purpose of water release
Oct-91	May-92	1.1	Minimal/Environmental
Jun	Oct-92	122	Yolo irrigation
Nov	Dec-92	0.1	Minimal
Jan-93	Apr-93	59.2	Flood release
May	Jul-93	22.7	Yolo irrigation
1992 water year		12.1	(Oct 91 - Sep 92)
1993 water year (partial)		83.3	(Oct 92 -Jul 93)

Table 5.12. 1991-93 Cache Creek releases of total phosphorus.

total phosphorus load may be deposited or filtered-out before flowing into Clear Lake. Here we analyze the effects of Tule Lake in detail because of implications of this finding for the selection of a strategy to control phosphorus influx into Clear Lake.

Tule Lake is a 26 km^2 (1 mi^1) natural wetland alongside Highway 20, through which Scott's Creek passes before joining Middle Creek and entering Clear Lake via Rodman Slough. Tule Lake is presently used for wild rice and other seasonal agricultural practices including sheep pasture, and was used historically for commercial green bean farming. A low flow by-pass channel passes around the basin to the south, enabling some low flow control. Spring-summer drainage and pumping takes place when Clear Lake has dropped sufficiently. The upper reaches of Tule Lake have an elevation of 398 meters (1327feet) above Mean Sea Level, similar to the elevation of high water on Clear Lake. The outlet to Tule Lake is 224 km (1.4 mi.) above the confluence of Scott's and Middle Creek (Zalusky, 1992).

Using sampling methods (and constituent analysis) described above, water flowing into Tule Lake was sampled during 1993 from a bridge across Scott's Creek at a point just above the small outlet to Blue Lakes. This Blue Lakes site is 2.7 km (1.7 mi.) above Tule Lake, and provided optimal access to Scott's Creek at all but the highest flows. Discharging water samples were taken from Tule Lake at the actual outlet control structure, accessible up to medium flows. Low by-pass flows could be sampled from the bypass structure when Tule Lake was filling. But high outflows could only be sampled for safety reasons from the Highway 29 bridge a kilometer below the outlet. This site is subject to dilution effects from Clear Lake at low flow, but these are considered negligible at high flows. Nevertheless the Highway 29 bridge was only used when the Tule Lake outlet was inaccessible for practical safety reasons.

Scott's Creek discharge is monitored continuously at the DWR Eickhoff Road gauging station located 12.8 km (8 mi.) above Tule Lake near the lower reaches of Scott's Valley, incorporating a 146 km2 (55 mi^2) drain-age area (see Table 5.7). At the Blue Lakes bridge sampling site above Tule Lake the drainage area was measured as 195 km^2 (75 mi^2), increasing to 224 km^2 (86 mi^2) of watershed at the upper limit of Tule lake (Flood Insurance Study, 1990, 216 km^2 this study). The outlet of Tule Lake includes 270 km^2 (104 mi^2) of the total Scott's Creek watershed.

Table 5.13 shows the results of total phosphorus and total solids analysis of water samples taken above and below Tule Lake on six occasions between December 10,1992 and March 11, 1993. **Figure 5.4 gave** the discharge profile for Scott's Creek at the Eickhoff Road DWR gauge under the different storm events of the 1993 water year. Four of the first six major storm events were sampled, including the January 20 main event of winter.

Table 5.13 also shows the chronology of sampling relative to peak flows within storm cycles. Thus, for example, during the December 10, 1992 storm event a sample was taken above Tule Lake at the Blue Lakes bridge at 1630 hours, and according to the Eickhoff Road gauge, Scott's Creek peaked half an hour later at Blue Lakes at 1700 with a flow of $9.4x10^4$ L/sec (3100 cfs). This means that in this case the sample above Tule Lake was taken half an hour before peak flow, when the suspended sediment and total phosphorus load was approaching a maximum for that event. The outlet was sampled the next day on De-

Blue Lakes - Estimated Flows					Tule Lake samples					
Date	Time of Sampling	Flow (L/sec)	Time of Peak Flow	Peak Row (L/sec)	Date	Time	total-P (mg/L) Blue Lake	total-P (mg/L) Tule Lake	Total Solids (ppm) Blue Lake	Total (ppm) Tule Lake
12/10/92	16:30	87,143	17:00	93,930	12/11/92	12:00	1.42	0.33	**1354.4**	182.8
12/31/92	1524	90,173	23:00	166,044	111/93	8:40	0.79	0.47	586.1	308.1
1/16/93	15:45	22,149	1:00	33,936	1116/93	16:00	0.25	0.25	249.9	195.3
1/20/93	12:10	144,319	17:00	209,070	**1/21/93**	12:20	128	**0.55**	1677.6	4021
3/2/93	1520	5,090	-	-	3/2/93	15:45	0.14	0.19	127.4	142.4
3/11/93	15:10	2,273	-	-	3/11/93	14:30	02	0.25	1582	**155.5**
[Mean]							0.68	**0.34**	692.27	**231.03**
(Reduction								0.5		**0.33**

Table 5.13. 1992-93 measurements of the effect of Tule Lake on Scott's Creek water quality, comparing total phosphorus and total solids in water samples taken from above Tule Lake (at the Blue Lakes bridge) and below Tule Lake.

cember 11, 1992 at 1200, some 19 hours after peak flows entered Tule Lake. There had been no 1993 outflow from Tule Lake prior to this. The timing of other samples relative to peak flows, and the discharge at Blue Lakes and below Tule Lake are also shown in Table 5.13. The final two samples in March were routine rather than event-based.

The extrapolation of flows to the Blue Lakes bridge above Tule Lake was based on the following calculation, following those in the Flood Insurance Study (1990). There is an estimated 2 hours, 2 minutes delay between peak flow at Eickhoff Road and Blue Lakes. Discharge quantities also increase down stream, but not proportionally to flood volume. The Flood Insurance Study (1990) gives a relationship between drainage area and peak discharge for 10, 50, 100 and 500 year events on Scott's Creek. This non-linear relationship was extrapolated for the lower flows encountered during this study. The resulting estimates are shown in Table 5.13. Samples during the four main events were taken near peak discharges.

Thus in quantifying the sedimentation effect of Tule Lake, defined as sediment input minus outlet loading, an estimate for 1993 loading above Tule Lake was approximated by combining sampled concentrations and the discharge profile extrapolated from Eickhoff Road data to the Blue Lakes bridge. Discharge from Tule Lake is a more complex problem. We did not attempt to reinstall and operate the old DWR gauge below Tule Lake because flows are affected by lake level as well as discharge rate, and in the past led to unreliable data. Extrapolation from flows into Tule Lake to the discharge below the lake is possible, but complex and assumption laden.

Some information on peak discharge is available in the 1990 Flood Insurance Study, where for a ten-year event a 2.6x105 L/sec (8500 cfs) inflow will exit Tule Lake at about 2.3×10^5 L/sec (7750 cfs). Under a 50 year event 3.9×10^5 L/sec (12,750 cfs) of inflow will be reduced to 3.1×10^5 L/sec (10,300 cfs). The difference is less at lower flows, approaching equality. But this relationship also depends on the volume of water stored in Tule Lake, since the outflow will be considerably less if the lake has yet to fill. Under a 10 year event early in the season, Tule Lake will take over 12 hours to fill to an average depth of 3.9 meters (13 feet) and will then exhibit the peak flow discharge characteristics of the Flood Insurance Study, assuming constant inflow. At this point there will be 10 million m^3 (8000 acre-ft, or 350 million cubic feet) of water stored in Tule Lake.

When Tule Lake is full, any significant sustained event of less than a 10 year event will result in equal rates of peak inflow and outflow. Under such infrequent equilibrium conditions it is possible to accurately estimate discharge below Tule Lake. These conditions were probably reached during the December 31, 1992 and January 1, 1993 sampling events. But more typically, when Tule Lake is either filling, slowly discharging, or events are not sustained, the outlet discharge profile would be significantly different from the profile above.

Nevertheless, as a first approximation, equilibrium conditions across Tule Lake were assumed, and Figure 5.11 shows total phosphorus and total solids concentrations for sites above and below Tule Lake, plotted against extrapolated discharge at the Blue Lakes bridge. Slopes for these graphs were earlier defined as the External Loading Factors (ELF's) for the particular watershed or location. Our estimated ELF's

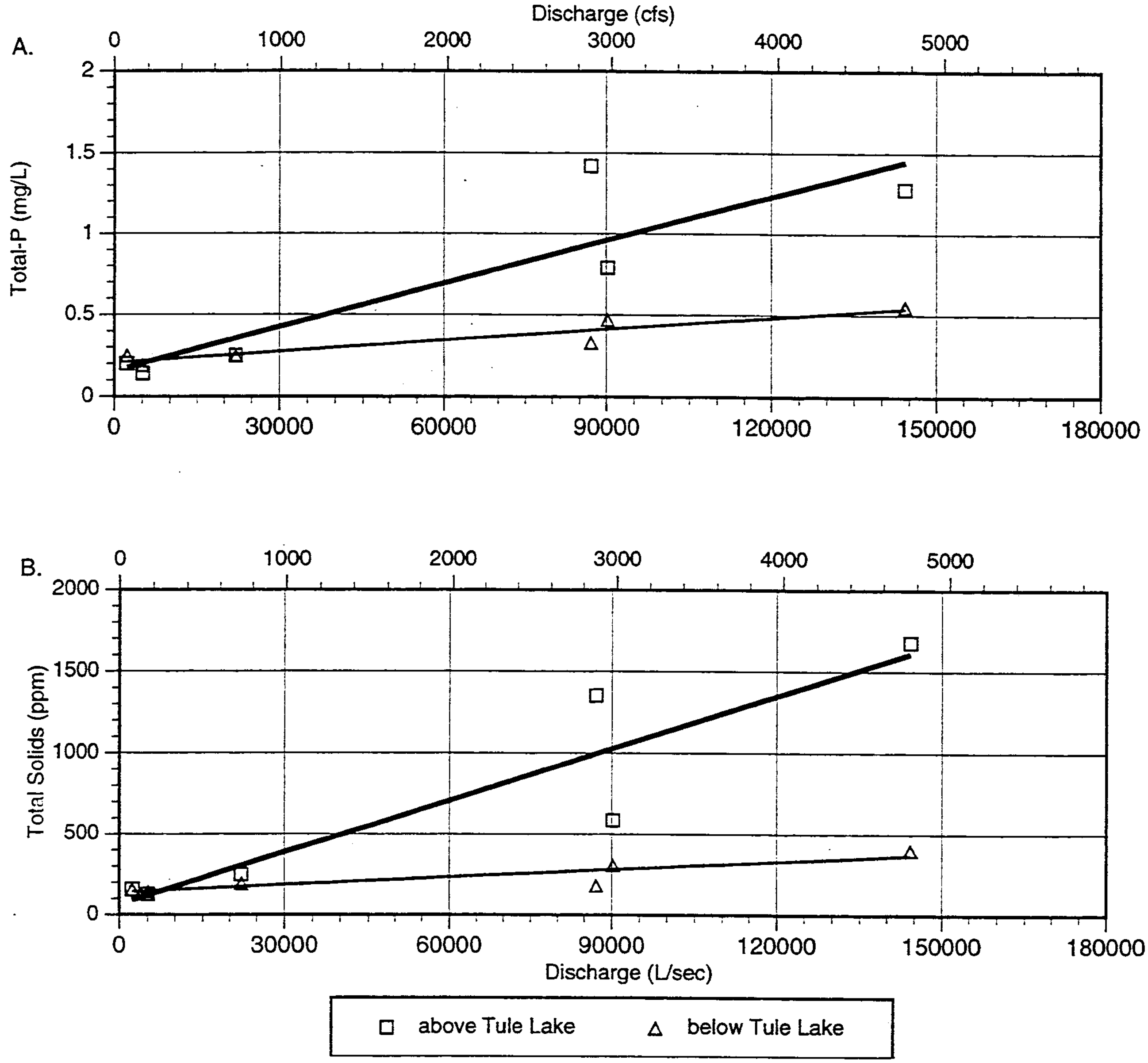


Figure 5.11. Total phosphorus (A) and total solids (B) above and below Tule Lake, against Scotts Creek discharge.

for total phosphorous above and below Tule Lake are 12.4 and 3.2 mg/m⁶/sec respectively.

Calculations based on these ELF's under assumed equilibrium conditions across Tule Lake, result in the concentration of total phosphorus being reduced by 70%. In 1992-93, as Scott's Creek water traveled across Tule Lake and discharged into Clear Lake, the total phosphorus loading to Tule Lake of 104 MT was reduced to 31 MT **(Table 5.10;** extrapolation based on surface area). This loading reduction ratio is incorporated in the next section's estimation of historical external loading.

Since equilibrium conditions do not occur across Tule Lake, there remains considerable uncertainty in this estimate. Alternative methods of estimation give an idea of this uncertainty. Two methods are: (1) direct comparison of sampled concentrations irrespective of discharge rate (ignoring implicit relationship between discharge and concentration), and (2) a comparison of expected removal at peak discharge for the events we sampled. As **Table 5.13** shows, the mean reduction of total phosphorus is 50% and the mean reduction of total solids 67%. In neglecting the importance of flow rate to loading, the mean reduction is a minimum estimate. Table 5.14 compares the mean reduction with the reduction expected at peak flows from the relationship between flows and loads depicted in Figure 5.11. At peak flows, 72% of total phosphorus and 79% of total solids are estimated to be removed by Tule Lake. The equilibrium calculation is closer to the peak flow estimate than to the simple mean estimate because high flows contribute disproportionately to loading. The limited data available strongly suggest that Tule Lake is a highly efficient system for retaining total phosphorus, as our basic estimate of 70% removal in 1992-3 indicates.

5.3.9 Summary Phosphorus Budget

The whole-lake phosphorus budget for Clear Lake is a useful way to summarize our knowledge of the nutrient dynamics of the system. Conceptually, the budget is the balance of all external loads and losses from the lake. Stream inflow of total phosphorus is the largest inflow term, and we neglect the small, uncertain contributions from other sources such as sewage in this section. There are two major losses, outflow down Cache Creek, and burial in the sediments. Finally, the water column content of total phosphorus can change substantially from year to year through exchange with the sediment. In algebraic terms:

Creek Loading - Cache Creek Outflow - Lake Water Change - Sediment Storage = 0

The stream ELF's developed from the 1992-93 data can be used in conjunction with the stream flow gauging record to model the historical record of stream loading for 1968-1991. For consistency with the daily DWR averages available for earlier years, the 1993 loading versus discharge curves were re-computed using average-daily data. DWR measurements are originally taken every 15 minutes, and the data are generally published as average-daily discharges. Kelsey Creek data were only available for 1981-93 water years.

We computed the monthly losses of total phosphorus from the lake through Cache Creek, from 1969-93 using monthly DWR Lower Arm total phosphorus concentrations, multiplied by USGS published release data for Cache Creek. The difference between external creek input and Cache Creek loss provides an estimate of the net annual quantity of phosphorus remaining in the lake.

Changes in lake water column total phosphorus were computed using the DWR data (Figure 4.2). From summer peak levels in one year to summer peak levels the next, the lake concentration of total phosphorus often increases or decreases substantially. At each summer peak concentration in the Upper Arm, the surface total phosphorus figure was multiplied by the volume of water in the lake calculated from the area capacity curve and lake level at the time of sampling, to give the mass of phosphorus.

We estimate the loss of total phosphorus to sediment burial as the amount needed to balance the other three terms. There is no simple method to check this estimate, so its large value may indicate the inclusion of large errors in the estimation of the other terms.

Table 5.15 and Figure 5.12 summarize the budget model estimates for the historic budget. Several features stand out. First, all terms vary very substantially from year to year. Most of these variations are associated with drought and flood. Flood years are years of high rates of loading, but also high rates of storage in sediments, and high losses to Cache Creek outflow. Drought years have little loading and little loss down Cache Creek. Given the hydrology of flood and drought, and the non-linear dependence of stream loading on flow, the patterns of loading to discharge are inevitable. Less obviously, sediment

Sample Date Above TL	Below TL	Row Blue Lakes (L/sec)	Loss total-P	Loss Total Solids	Peak Flow Blue Lakes (Usec)	Loss total-P peak	Loss Total Solids peak
12/10/92	12/11/92	87,143	77%	87%	93,930	77%	87%
12/31/92	1/1/93	90,173	40%	47%	166,044	76%	86%
1/16/93	1/17/93	22,149	0%	22%	33,936	56%	56%
1/20/93	1/21/93	144,319	57%	76%	209,070	77%	85%
3/2/93	3/2193	5,090	-36%	-12%	-	-	-
3/11/93	3/11/93	2273	-25%	-7%	-	-	-
Mean			50%	67%		72%	79%

Water Year	Creek Input (metric tons)	Cache Creek Outflow (metric tons)	Lake Water Change (metric tons)	Sediment Change (metric tons)	Lake Level Fall low feet-Rumsey)	Lake Level Spring high feet-Rumsey)
1969	256	105	-340	491	1.41	8.63
1970	299	58	142	99	2.19	9.9
1971	168	23	-155	300	**1.36**	7.47
1972	10	4	-25	31	1.6	4.44
1973	136	33	46	57	0.65	**7.45**
1974	322	34	-147	435	1.31	8.78
1975	145	18	-7	134	1.71	8.37
1976	**3**	0.24	117	-114	1.68	2.24
1977	0.0009	0.04	-37	37	-0.55	t
1978	275	16	9	250	-3.33	7.48
1979	24	6	-85	103	1.57	6.77
1980	230	17	50	163	1.63	9.17
1981	23	10	63	-50	2.7	6.94
1982	318	64	-18	272	1.39	721
1983	565	67	-108	606	2.47	10.34
1984	193	30	80	83	3.73	7.86
1985	17	1	-24	40	1.75	6.09
1986	485	20	-65	530	1.46	10.17
1987	19	9	120	-110	1.54	4.85
1988	50	12	**45**	-7	0.84	5.88
1989	44	7	204	-167	1.05	5.28
1990	6	2	145	-141	1.47	**3.34**
1991	19	8	-101	112	0.32	3.97
1992	26	12	123	-109	0.48	4.51
1993	*311*	*83*	-170	224	0.52	8.48
Mean	**158**	**26**	**-6**	131	**1.24**	**6.62**

Table 5.15. Annual phosphorus budget for Clear Lake, as calculated changes in sediment/water total phosphorus for the period 1969 to 1993, shown as gains and losses.

storage is also related to stream discharge. Years of high loading are also years of high sediment storage. Much of the phosphorus that enters the lake in flood years enters the sediments but does not enter the water column during the following summer. During low loading years, substantial amounts of phosphorus are released from sediments. This tendency is especially marked in the recent long drought, and undoubtedly accounts for the large rise in total phosphorus concentrations in the lake in the late 1980s and early 1990s (Figure 4.3).

If the flow-based, rather than area-based, method is used to estimate creek inflows, the loading estimate decreases by about 30%. However, using daily average hydrograph data to compute load has an effect of almost the same magnitude in the same direction (in 1992-93 using daily averages rather than hourly reduced estimated load by 25%). Moreover, the qualitative pattern of variation is not sensitive to changes in estimated loading on the order of differences between Methods 1 and 2. Estimating sediment storage and release as a residual as we have done collects unknown and unmeasured effects in this term. It is interpretable as true exchange with the sediments only if it is larger than the plausible uncertainties and errors in the calculation, which it is. Estimated sediment storage of phosphorus in high water years remains large relative to all other loss terms, and release from sediments is clearly important in some drought years. Thus, exchanges of phosphorus with the sediment pool are very significant, as we discuss in more detail in the next chapter.

On average over the past 25 years, the estimated annual external loading by tributaries was 158 MT of total P. Of this, 26MT were released from the lake, and 131MT deposited in the lake sediments. Water column levels change from year to year, but are near zero on average. The main 'loss' of phosphorus from the Clear Lake ecosystem is thus sediment burial, not outflow to Cache Creek. During extreme droughts at least 30% of surficial sediment phosphorus may be recycled.

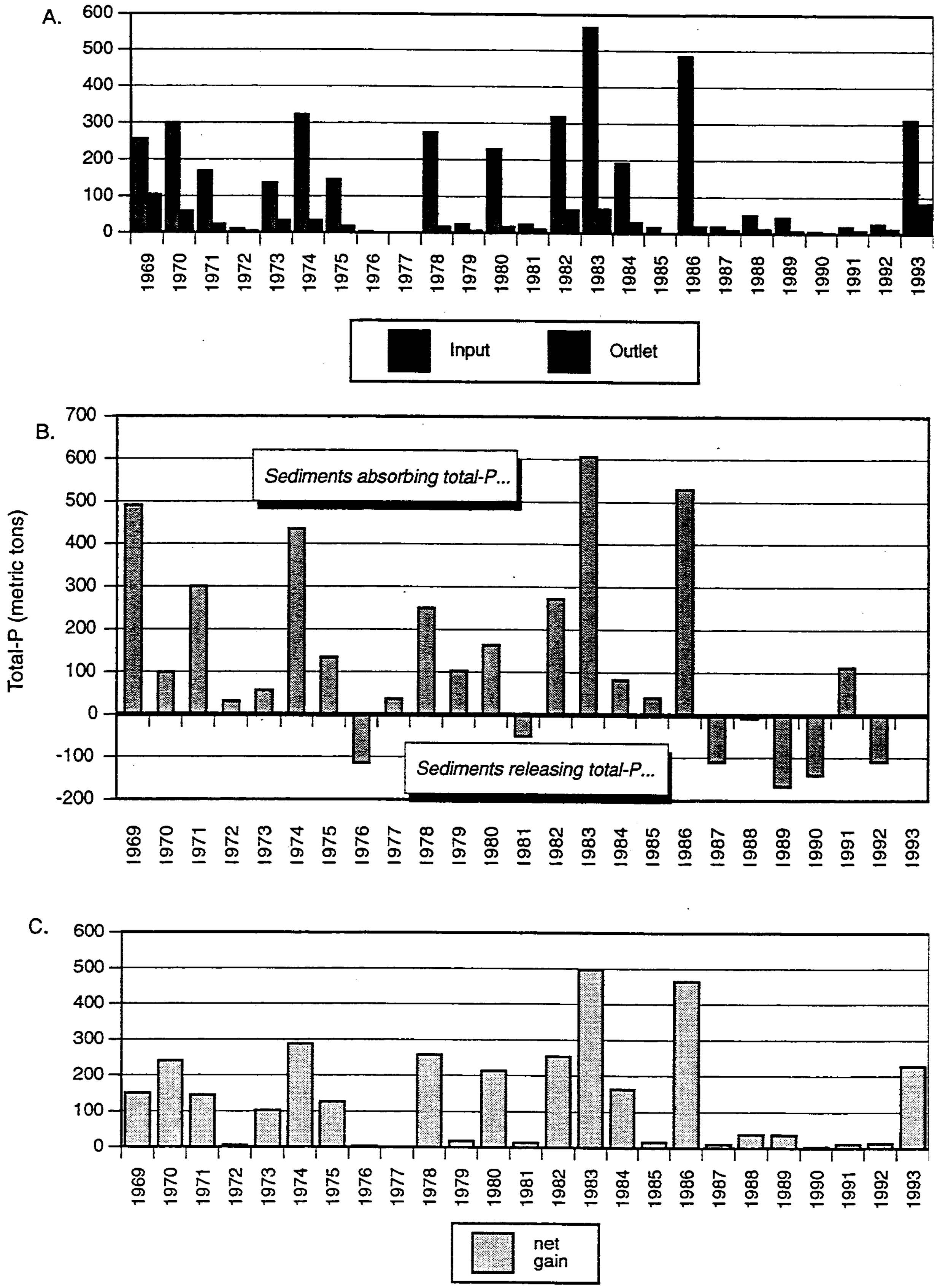


Figure 5.12. Clear Lake external loading for total phosphorus in metric tons. (A) shows stream input and Cache Creek output, (B) shows sediment gains or losses and (C) shows net gain to the lake system as a whole.

The latest cycle of flood and drought illustrates how the budget changes as a function of inflow and outflow volumes. 1982 through 1986 was a 5 year period of large external input and sediment burial of phosphorus (Figure 5.12B). Using the data from Table 5.15 we estimate that 1,578 MT of phosphorus entered the lake during this 1982-6 high rainfall period, 182 MT was lost through Cache Creek, 65 MT remained in the water at the end of the period, and 1,331 MT or 84% was buried in the sediments. What is not clear is how much of this phosphorus may have been permanently buried. Figure 5.12B and Table 5.15 show that, during the next 6 years of drought, substantial phosphorus was released from the lake sediments back into the water column during the summers, suggesting that by the end of the drought in mass balance terms one third of the 1982-86 1,331 MT buried (or from some other sediment pool) was recycled back into the water column. At the peak total phosphorus level in 1992, as much as 680 MT of total phosphorus were present in the water column, compared to a peak of 230 MT in 1987.

During 7 out of 11 low rainfall years since 1969 (76, 81, & 87-92) sediment recycling or release has exceeded external inputs. During the dry years of 1972, 79,85 and 91 tributary inputs were also low, but were exceeded by inter-annual losses of phosphorus from the water column, resulting in a net gain by the sediments.

The mean budget described in the last row of Table 5.15 shows some interesting differences from the Horne-Lallatin budget in Table 5.1. The groundwater term in the Horne/Tallatin budget is much too high, and the domestic wastewater term is now much lower. The other terms are fairly close, given the uncertainties of inherent in the calculation.

5.3.10 Modeling of Phosphorus Budget with Load Reduction

Goldstein and Tolsdorf (1994) estimate that stream sediment loads could be substantially reduced by best management practices. Additional gains from the restoration of wetlands are feasible (see Section 9.3), for a total reduction of about 50% of present loads. How much difference would such a reduction make in lake concentrations of total phosphorus? The statistical relationships in the historic data can be used to build a basic mass balance model that estimates the effects of load reduction. Let us suppose that best management practices had been implemented instantly in 1968, together with the restoration of major wetlands to achieve this 50% reduction. What might have happened?

The model uses the general equation for the mass balance of phosphorus shown in Section **53.9** and divides the previous year's water mass of total phosphorus plus water change in total phosphorus by average annual lake level to obtain an estimate of average annual water phosphorus concentration. Beginning with an initial mass of phosphorus in the water 208 MT in October 1968, the model steps forward year by year from 1%9 to 1992 with the assumption that creek loading was half of its estimated value in Table 5.15.

The Cache Creek outflow term of the model is a function of the annual average phosphorus concentration at the Lower Arm DWR station. We estimate the Lower Arm concentration by first using the model to estimate lake average concentration. Based on a regression between lake average concentration and Lower Arm concentration from the 23 years of DWR data, the Lower Arm concentration should be $0.666{*}[P]_{w}$h**ole** lake +0.671($P<0.001$, $R^2=0.85$). This value multiplied by the annual discharge from the Clear Lake dam gives the annual Cache Creek outflow mass. Because a disproportionate amount of discharge occurs at periods of high phosphorus concentration in summer irrigation outflows but also in some years in high flow winter events, outflow was adjusted by the ratio of the sum of monthly outflow to annual outflows to correct for complications introduced by seasonal variations in discharge and phosphorus concentrations.

Sediment storage is estimated in the model using its strong regression with creek inputs ($P<0.001$, $R^2=$ 0.73). While the slope of the regression is highly significance, the intercept has a standard error of ±30 MT, making it the least certain term in the model. The use of sediment data rather than a correlation would almost certainly improve the precision of this term, but no long-term sediment data are available. Accordingly, Table 5.16 shows results for runs of the model with "mean," "optimistic," and "pessimistic" assumptions about the intercept of the sediment storage/creek loading relationship. The "optimistic" scenario adjusts the sediment storage term one standard error upwards, and the "pessimistic" one standard error downwards.

With these assumptions, the "mean" scenario predicts that if external loading were reduced to half its historical level beginning in 1969, the water column phosphorus levels would be virtually unchanged over the actual measured values. Even over a favor-able series of years, which lacked the recent severe drought, the model estimates that average annual water levels would not decline under the best-estimate condition. By contrast, the "optimistic" scenario

Water Year	MEAN MODEL Creek Input *(MT)* Historical / 2	*Sed Storage* Estimate	*Outflow Estimate (MT)*	Est *water* [P] (mg/L)	Total-P load *in water* (MT)	Flux *of* P *in water (MT)*	*REAL Ann. avg. (P)* ***(mg/L)***
1969	*128.0*	*98.8*	*95.6*	0.124	168.3	*48.0*	
1970	*149.5*	*123.7*	*77.5*	*0.106*	135.5	***32.8***	*0.18*
1971	*84.0*	*48.0*	***45.8***	*0.107*	*121.4*	*14.1*	*0.12*
1972	*5.0*	***-432***	12.6	0.136	133.6	*-12.1*	*0.10*
1973	*68.0*	***29.5***	*52.1*	*0.107*	164.8	**-312**	*0.12*
1974	*161.0*	*137.0*	*87.1*	*0.095*	*143.3*	21.5	*0.07*
1975	*725*	34.7	***55.5***	0.112	114.0	***292***	*0.08*
1976	*1.5*	*-47.3*	*0.5*	0.145	**1352**	*-21.1*	*0.09*
1977	*0.0*	-49.0	*0.2*	0.185	*183.4*	***-48.2***	*0.13*
1978	137.5	109.8	*68.0*	0.137	211.0	*-27.6*	*0.09*
1979	*12.0*	*-35.1*	***452***	*0.183*	*181.8*	*29.2*	*0.08*
1980	115.0	*83.8*	*55.7*	*0.132*	*199.9*	*-18.1*	*0.07*
1981	11.5	***-35.7***	***38.5***	0.174	*174.8*	25.1	*0.08*
1982	159.0	*134.6*	*74.7*	*0.112*	184.8	-10.0	*0.09*
1983	*282.5*	*277.3*	104.4	*0.083*	**135.4**	***49.4***	*0.08*
1984	***96.5***	*62.5*	*69.1*	*0.090*	*90.4*	*45.0*	*0.07*
1985	*8.5*	*-392*	***3.3***	0.110	101.3	*-10.9*	*0.07*
1986	*242.5*	231.1	***40.6***	0.064	110.8	*-9.5*	*0.07*
1987	*9.5*	*-38.0*	13.1	*0.126*	*86.2*	*24.6*	*0.08*
1988	*25.0*	-20.1	*14.0*	*0.099*	***125.4***	*-39.2*	*0.13*
1989	*2.0*	-46.7	*16.9*	*0.137*	*156.7*	*-31.3*	*0.20*
1990	*3.0*	-45.5	*3.7*	*0.176*	*197.8*	*-41.1*	***0.22***
1991	*9.5*	*-38.0*	*11.9*	*0.209*	*237.7*	*-39.9*	*0.22*
1992	*13.0*	***-34.0***	14.1	*0.238*	*279.7*	*-41.9*	*0.30*
Avg. all	*74.9*	*37.5*	*41.7*	*0.133*	*1572*	*-2.6*	*0.119*
Avg. w/o drought	*96.4*	*62.3*	*51.5*	*0.122*	*149.4*	*5.9*	*0.094*

Water Year	**PESSIMISTIC MODEL** *Sediment storage one* SE *smaller* than *mean* Est *water* [P] (m^{g}/L)	Total-P load *in water* (MT)	Flux *of P in water* (MT)	**OPTIMISTIC MODEL** *Sediment storage one* SE *larger* than *mean* *Est water [P] (mg/L)*	Total-P load *in water* (MT)	Flux *of P in water* (MT)	***REAL*** *Ann. avg. [P]* ***(mg/L)***
1969	*0.14*	*190.1*	*28.0*	0.108	146.5	*62.0*	
1970	*0.13*	*172.6*	*17.5*	*0.077*	*98.4*	*48.2*	*0.18*
1971	*0.15*	170.3	*2.3*	*0.075*	*72.5*	*25.9*	*0.12*
1972	*0.20*	*196.8*	*-26.5*	*0.072*	*70.3*	*2.2*	*0.10*
1973	*0.17*	*254.3*	*-57.5*	*0.048*	*75.2*	*·4.9*	*0.12*
1974	*0.16*	*235.9*	*18.5*	*0.033*	*50.7*	***24.5***	*0.07*
1975	*0.19*	194.2	*41.6*	*0.035*	***33.8***	*16.8*	*0.08*
1976	*024*	*227.8*	*-33.6*	*0.047*	*42.5*	*-8.6*	*0.09*
1977	*0.31*	*305.6*	*-77.7*	*0.062*	61.2	-18.7	*0.13*
1978	*024*	363.1	*-57.5*	*0.037*	*58.8*	*2.3*	*0.09*
1979	*0.31*	312.5	*50.6*	*0.053*	51.0	*7.8*	*0.08*
1980	*023*	351.4	*-38.9*	*0.031*	*48.4*	*2.6*	*0.07*
1981	*0.30*	*305.8*	*45.6*	0.045	*43.8*	*4.6*	*0.08*
1982	*020*	*334.9*	*-29.1*	0.019	34.7	*9.1*	*0.09*
1983	0.16	*267.2*	*67.7*	*0.001*	*3.6*	31.1	*0.08*
1984	*0.18*	182.3	*84.9*	*0.001*	*0.0*	*3.6*	*0.07*
1985	021	*193.4*	-11.1	*0.014*	*10.3*	-10.3	*0.07*
1986	*0.14*	231.0	*-37.6*	-0.007	*0.0*	10.3	*0.07*
1987	*025*	*172.8*	*58.2*	0.013	*5.0*	*-5.0*	*0.08*
1988	0.19	*237.2*	-64.4	*0.015*	*18.8*	-13.8	*0.13*
1989	*025*	*285.2*	*-48.1*	*0.029*	*32.9*	-14.1	*0.20*
1990	*0.31*	*350.9*	*-65.7*	*0.044*	*49.3*	-16.4	***0.22***
1991	*0.36*	*415.4*	*-64.5*	*0.057*	*64.5*	*-152*	*0.22*
1992	*0.41*	*483.8*	*-68.4*	*0.068*	*79.9*	-15.4	*0.30*
Avg. all	*0.226*	*268.1*	-11.1	*0.040*	*48.0*	*5.4*	*0.119*
Avg. *w/o* drought	*0.204*	*249.4*	*-0.7*	0.041	*50.1*	11.6	*0.094*

Table 5.16 The terms of the model budget for phosphorus for Clear Lake with stream phosphorus input halved relative to the estimates of actual loads shown in Table 5.15. The estimated water total-P under each of the three versions of the model should be compared to the "real" column (right), based on DWR measurements.

predicts that average annual concentrations would decline to zero by 1983 and remain quite low throughout the drought. The "pessimistic" predicts that reducing external loading would *increase* the amount of phosphorus in the lake to catastrophic levels, by making the sediment a large net source of phosphorus in the long-term.

The explanation for this behavior of the model is simple. The sediment acted as a quite significant net source during the recent extreme drought, while in wetter years it acts as a major sink as we have seen. Since we used a regression based upon historic creek loading and sediment storage, our model halved-load years tend to act like drought years. Under the model, many more years have sediment release than under the actual conditions 1969-92. This assumption may be realistic or not, depending upon how sediments react under a real regime of reduced phosphorus load but a normal water budget. Another way of putting it is that the model sediments act as an infinite source of phosphorus under low-loading conditions, and the chemistry of sediment water interactions remains as under drought. In fact the lower load due to erosion control would take place without drought induced chemical changes like high alkalinity and conductivity, and sediments have not stored an infinite amount of phosphorus to release during low load years. For these reasons the explosive increase in phosphorus in the "pessimistic" scenario is presumably quite unrealistic. At some point the sediment potential for further internal loading would be exhausted. The "optimistic" scenario would occur if the data from the drought years are slightly pessimistic with regard to sediment storage and release of phosphorus at low loading levels. That is, if the stored phosphorus in the sediments is relatively limited, the sediments will stop acting as a source a few years after loading is reduced.

The model scenarios essentially calibrate our level of understanding of the behavior of Clear Lake sediments. As is noted in Section 9.3, experience in other lakes is also a poor guide as to the behavior of sediments under reduced loading. Some lakes appear to have modest amounts of available phosphorus stored in sediments and respond rather quickly to reduced loads. Limited data from other shallow lakes suggest that total phosphorus levels in Clear Lake sediments put it in the range of the "optimistic" scenario. There is no known lake which as actually behaved as in the "pessimistic" scenario, although lakes with sediment total phosphorus several times higher that Clear Lake often do behave as in the "mean" scenario. We emphasize, model aside, that there are real uncertainties in how Clear Lake will behave under reduced sediment load. Without statistical experience derived from actual years of high water flow with lower sediment loading, we cannot more accurately forecast the sediment behavior from monitoring data. Without a better experimental understand of Clear Lake sediments, we cannot predict from first principles.

5.3.11 Iron Cycling

The data available on iron are quite limited, due mainly to methodological limitations (see **Chapter** 3). Most compounds of iron are extremely insoluble, and-le total iron in sedimentary material is very high, on the order of 15% or more on a dry weight basis, compared to 0.1% for phosphorus. Exchanges among the many pools that make up total or total dissolved iron are not firmly linked by experimental evidence to phytoplankton uptake capabilities. Any interpretation of iron data is fraught with difficulty.

The problem is illustrated by our measurements of the iron load in streams. Figure **5.13** shows the discharge versus supernatant iron for Kelsey, Middle, and Scott's Creek. The total iron would be enormously higher because the suspended load will be about 15% iron. just taking account of the dissolved fraction flux is as high or higher than for phosphorus at all but the highest flows.

Since iron is known to be limiting in the lake even though it is required in much lower quantities than phosphorus, almost all of this iron load must be permanently stored in the sediments. There is no known chemical method that will allow us to evaluate the gross load in terms of an available versus unavailable fraction in either the dissolved or solid pools.

The large amounts of total iron in the lake sediments indicate that the recycling processes that produce an internal load of iron are probably even more critical than in the case of phosphorus. Again the complexity of iron chemistry bedevils attempts to understand the internal iron cycle. As discussed in **Chapter** 4, the roughly twice yearly iron data collected since 1977 by DWR are weakly consistent with the hypothesis that available iron storage in the sediments was eventually exhausted during the drought years, limiting nitrogen fixation and allowing the growing excesses of dissolved phosphorus observed drought years 1986-92.

Goldman and Horne (1983:156) show the change of dissolved iron for one year in Clear Lake. This shows winter levels of about 0.03 mg/l, and summer levels less than 0.01 mg/l. These levels are quite low by comparison with other lakes surveyed in Goldman and Home. The later DWR data are sometimes consistent with very low values such as these, but other

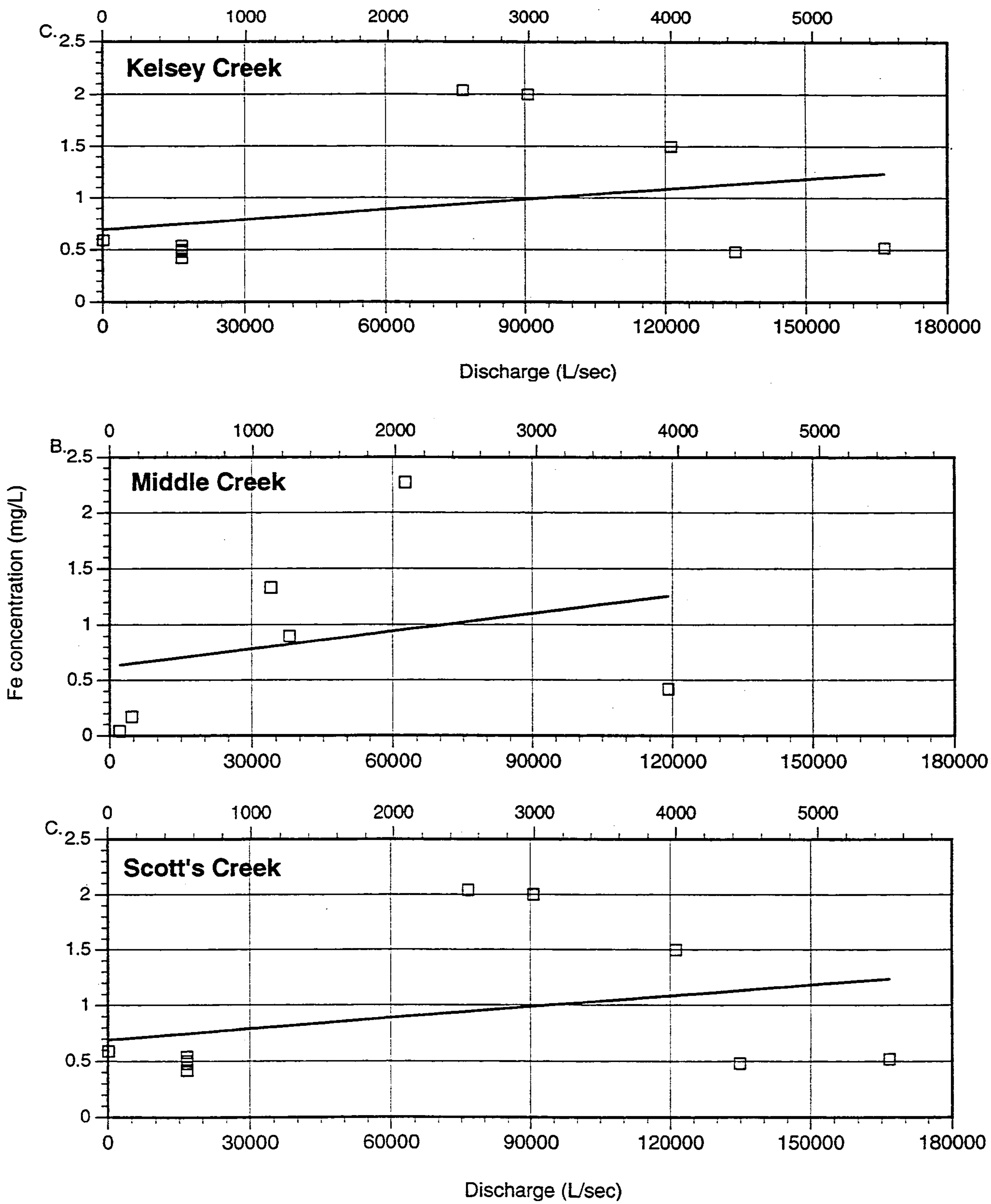


Figure 5.13. Supernatant iron plotted against discharge for (A) Kelsey Creek, (B) Middle Creek and (C) Scott's Creek.

sistent with very low values such as these, but other years show quite high iron, in the range of 1.0 mg/l (see Figure 4.5). These differences reflect differences in method rather than real changes.

Our measurements of supernatant dissolved and suspended iron in Clear Lake water from August 1992 until April 1993 (Figure 5.14) show that iron concentrations were high, around 0.5 mg/l, at the mouths of creeks and during creek flow periods, namely at Rodman Slough delta, Kelsey Creek and in Soda Bay (Kelsey Creek inflow) in midwinter 1993. The Up-per Arm site in the well mixed part of the lake showed a gradual increase to 0.15 mg/l at this time. Other sites throughout the lake showed concentrations of less than 0.15 mg/L from August 1992 to April 1993. Results for August 1993 showed iron to be at below detectable limits of 0.03 mg/L, implying a period of relative limitation. Unfortunately, the type of iron being measured is not necessarily that which is available for algal growth, soluble iron, as already discussed. Also, this particular method has a lower detectable limit of 0.03mg/l, whereas soluble concentrations being reported above by Horne and Goldman (1983) peaked at 0.03mg/l. (See also iron data collected in connection with bioassays in Table 7.2)

A method is required that can measure the low levels of soluble iron required by, and apparently often limiting to, algal growth. Such levels need to be monitored over a number of different years and climatic variables.

One hypothesis holds that an inverse relationship between iron and phosphorus due to phytoplankton uptake exists. When phytoplankton are phosphorus limited they may allow iron to accumulate, whereas when they are iron limited, phosphorus accumulates. Testing this hypothesis is complicated by the sparse iron data, and by the competing chemical hypothesis that iron and phosphorus should be positively correlated because they are released from the same minerals under anoxic conditions in the sediments. One might expect a positive correlation between the dissolved phase of the two elements early in the summer release cycle as they come out of the sediments together, followed by a negative correlation during the resulting bloom, as one of the elements becomes limiting while the other stabilizes at non-limiting values. No statistical relationship could be detected in the limited data we have between soluble iron levels and any measure of phosphorus or phosphorus limitation.

Brezonik (1994) has recently reviewed the chemistry of iron in natural waters. Reaction rates from the reduced ferrous to oxidized ferric state are strongly affected by pH, as is the redox potential at which ferric iron is the stable species at equilibrium. The half life of a ferrous ion may be as short as 10 seconds at pH 8. pH rose in Clear Lake during the drought (see Figure 4.1). Thus it is plausible that high pH drove the ferrous-ferric transition deeper into the sediments on average, and increased the rate at which ferrous iron was oxidized to ferric once it did escape the sediments. The limitation of phytoplankton by iron may thus be more extreme during the latter part of the drought. Since dissolved organic matter tends to chelate ferrous iron and protect in from rapid conversion to the ferric state, there is a positive feedback potential as well. The less phytoplankton biomass, the less chelation capacity, and the more rapid the conversion of iron to the unavailable ferric form.

The cycling of iron in Clear Lake is unfortunately full of open questions. We do not believe that practical answers to these questions will be forthcoming in the next few years, although scientific understanding of iron geochemistry is a scientific hot topic at the moment (e.g. Brand, 1991).

5.4 Conclusion

The data collected give cause for cautious optimism about the prospects of reducing phosphorus levels in Clear Lake by reducing loading. While years vary greatly in levels of external loading, sediment storage, and outflow, years have on average input 158 MT from creeks, lost 26 MT from outflow, and lost 131 MT to sediment burial (Table 5.15). A model estimating phosphorus levels in Clear Lake if external loading had been halved beginning in 1969 shows that a decrease in external loading would plausibly decrease lake phosphorus levels sharply, especially if the recent drought does not recur, and if the quantities of stored available phosphorus are relatively modest. The uncertainty in the model comes primarily from the sediment storage estimation, which due to the lack of long-term sediment data must be indirectly calculated. The "mean" scenario predicts essentially no change even with a 50% reduction in loading and the "pessimistic" scenario a dramatic increase. Both seem to depend upon an unrealistically large pool of stored available phosphorus, but the scenario of a very slow response to decreased loading cannot be ruled out on the basis of present data. We discuss the implications of these contrasting scenarios for treatment alternatives in Chapter 9.

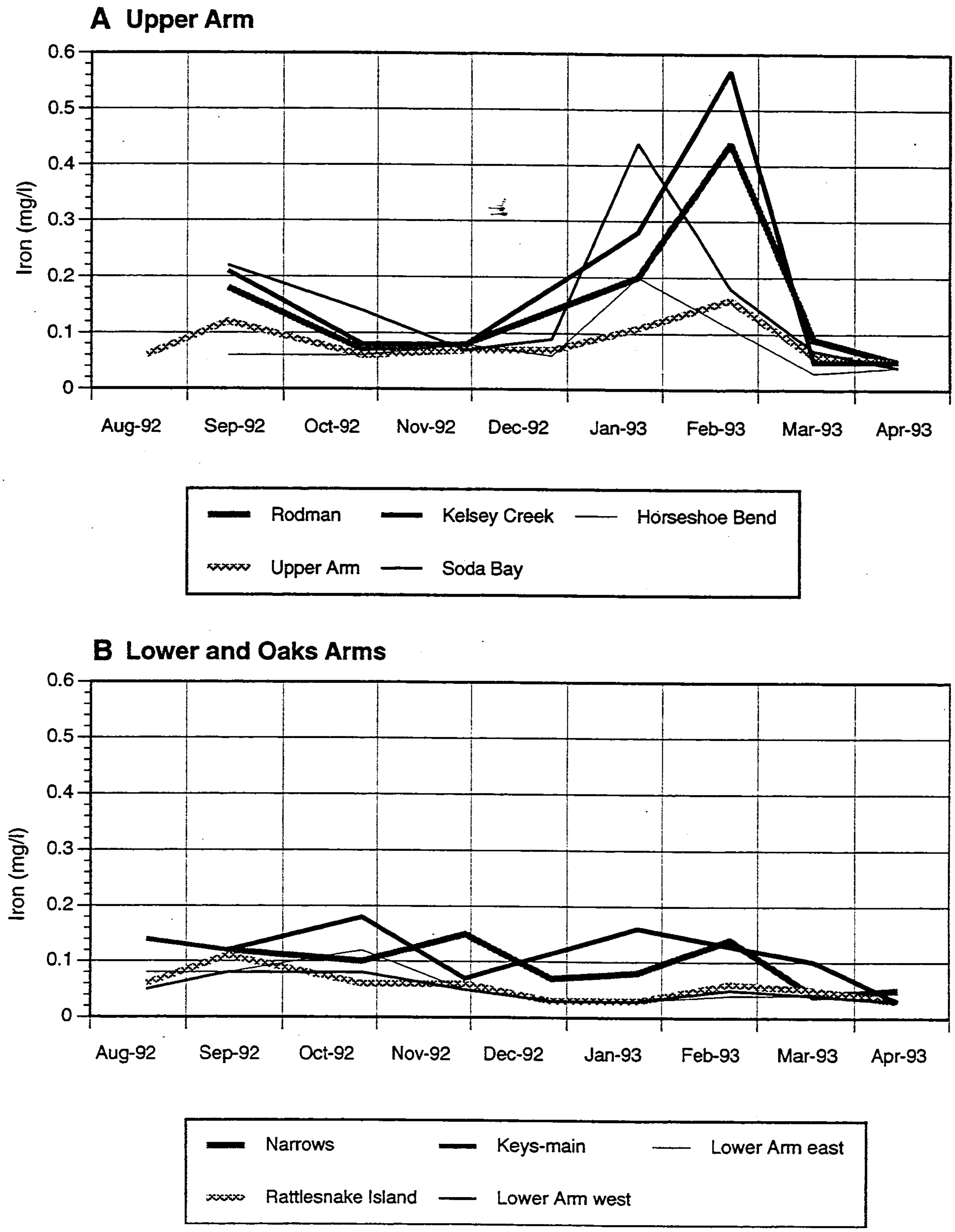


Figure 5.14. Clear Lake surface iron (mg/L) for (A) Upper Arm and (B) Lower and Oaks Arms.

6.0 Abstract

Clear Lake, like many shallow, productive lakes, absorbs most of its external load of phosphorus into its sediments during winter and recycles it into the water column in summer and fall. This regenerated phosphorus is termed the' "internal load." An important question for management purposes is the size of the internal store of phosphorus in sediments that can contribute to the nutrition of algae even after the external supply of nutrients from the basin is reduced. Clear Lake has about 3,500 metric tons (MT), 3,860 short tons, of phosphorus in its upper 10 cm (4 inches) of sediment.. Of this total about 500 MT has cycled into the water during the recent drought-influenced years. Under more normal conditions from the mid-1970s to mid-1980s, only perhaps 100-200 MT cycled. Most of the active phosphorus is contained in the iron/ aluminum-bound fraction, of which about half of the total amount stored in the top 10 cm of sediments in spring recycled to the water in summer and fall of 1992 and 1993. The amount of actively cycling phosphorus during drought conditions was 3 times the external gains and losses in more typical years, like 1974- 86. If the pools of active phosphorus behave according to simple dilution principles, decreases in the external load will cause a reasonably prompt response in lake nutrient levels. If average losses to the sediments initially remain at about 130 MT/yr, losses down Cache Creek at 25 MT/yr, and the external load is substantially reduced to, say, half current averages of 160 MT/yr, the loss of actively cycling phosphorus will be rapid. If the sediment storage term varies with load, more of the burden of diluting the sediment-stored load will fall on the Cache Creek outflow, slowing the response to reduced inputs. If the sediments act as a source of phosphorus after load reduction, as in the drought years, response to load reduction could be very slow.

6.1 Background

As we discussed in previous chapters, the bottom sediments of the lake forms a large reservoir of phosphorus and iron with an active seasonal cycle. These two nutrients adsorb onto the sediments from the water column in winter and spring, and are emitted from the sediments in summer and fall to create an internal load of recycled nutrients. The seasonal cycles of phosphorus, iron and nitrogen (together with physical factors like light and temperature) in turn drive the seasonal cycle of algal growth. As a result, control strategies for scum forming blue-green algae must limit the amount of phosphorus and/or iron recycled into the water column during the warm summer period to be successful.

Feedback loops between scum formation by blue-green algae and emission of phosphorus and iron from sediments may be important. The scum formers are not eaten by zooplankton and therefore tend to eventually decay on the bottom, lowering oxygen tensions at the bottom of the water column and in the sediments. The low oxygen concentrations in turn favor the release of phosphorus and iron into the water column, which fuels more blue-green algal growth.

Chapter 5 quantified the external creek loading and the annual releases from the lake, and constructed a mass balance model for phosphorus cycling over 25 years. The mass balance exercise showed that the lake sediments may act as both short and longer term sources and sinks of phosphorus. Within the annual cycle, as seen for 1991- 94 in **Figure 6.1,** phosphorus load from creeks and the pool of phosphorus in the water column are stored in the sediments in winter and recycled in summer. The long-term results from 1969-92 are depicted in Figure 4.6, and show that considerable quantities of sediment phosphorus can be recycled when, as in the recent drought, external loading is low. Such recycling may slow the response of the lake to controls on the external load, complicating control strategies. The objective of the measurements in this chapter is to understand this recycling process in more detail than is possible through the whole-lake mass balance model. We measured the phosphorus contained in four sediment fractions as a first step to understanding the chemistry of the recycling process.

The seasonal cycle of lake phosphorus levels seems paradoxical at first glance (Figure 6.1A). Dissolved and total phosphorus levels are highest during the summer when there is no external loading, but lowest during the winter when loading is high. The clas-sic explanation (Mortimer, 1941) is that phosphorus and iron form insoluble ferric phosphate under oxidizing conditions, but that soluble forms are released under anoxic conditions. Winter conditions in shallow, well mixed systems, especially the cool, oxygen-

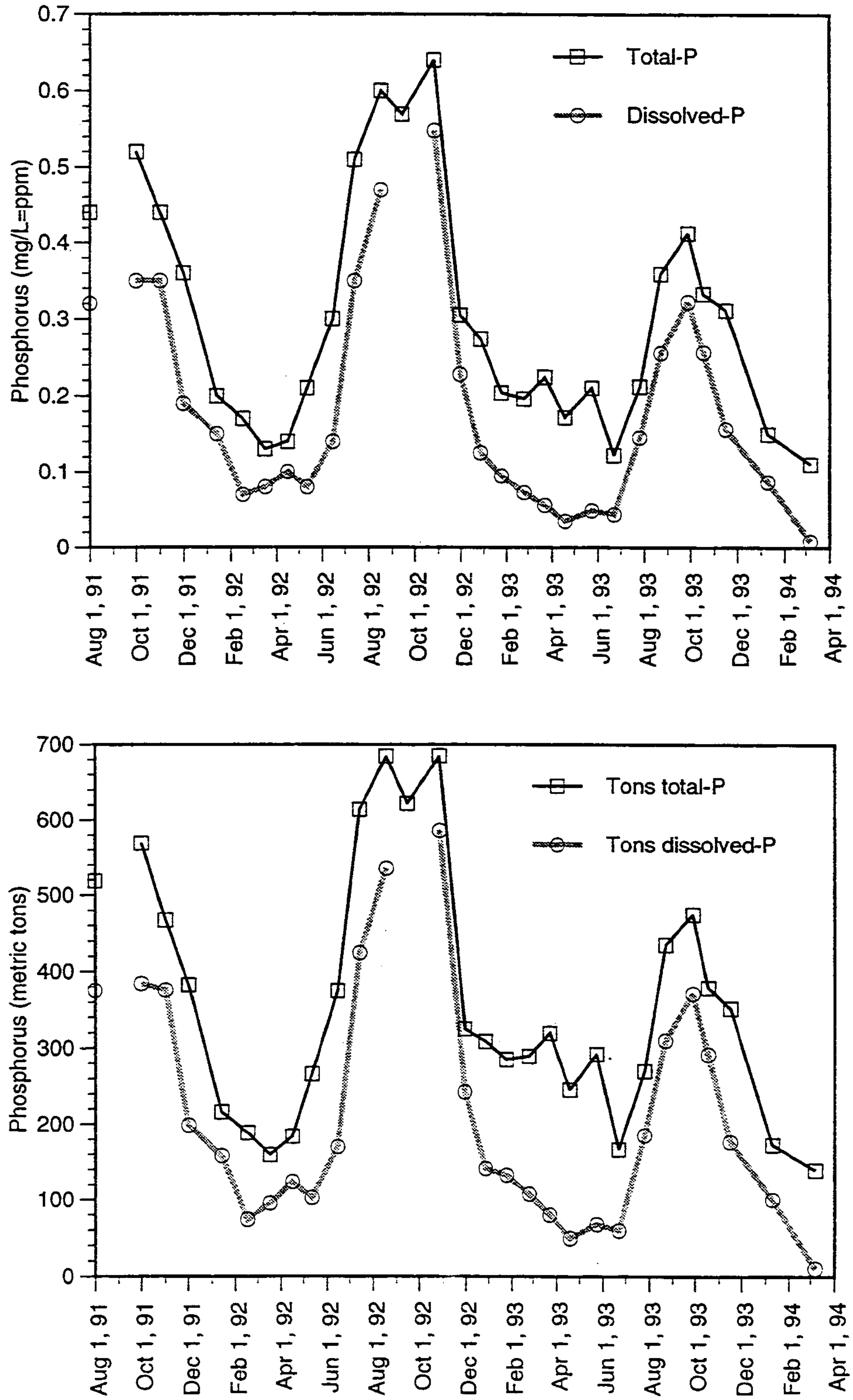


Figure 6.1. Concentrations (A) and tons (B) of total and dissolved phosphorus in Clear Lake water. Dashed lined indicate periods with more than a month between samples. Values before Sept. 1992 were taken from DWR data.

ated water, maintain an oxidized surface layer on the sediment that prevents phosphorus recycling. Additionally, clay and phytoplankton adsorb phosphorus and settle to the bottom at this time of year, depleting phosphorus from the water column. Deep lakes tend to recycle phosphorus at thermal overturn in fall and spring, but shallow, productive lakes tend to have late summer and early fall peaks of phosphorus emission from sediments. In Section 3.3.2 we discussed the experiments of Porcella *et at.* (1970) showing that Clear Lake sediments can recycle phosphorus to the water column even when the water is oxic. Under summer conditions, oxygen levels decline toward the bottom of the water column, the oxidized layer at the sediment surface becomes thinner (even absent for varying periods in deep water), and emission of phosphorus from the sediment surface begins. Many shallow lakes show the same seasonal pattern as Clear Lake (Sas, 1989).

Thus, the external load of nutrients is critical to lake water quality in the long run, while within any one year, the sediment emission processes of internal loading often controls the amounts of phosphorus (and probably iron) available to the algae for growth.

6.2 Methods

Sediment samples were collected each month from 9 routine sites around Clear Lake (coincident with water sampling). From west to east these were: off Rodman Slough delta, the middle of the Upper Arm, off Kelsey Creek delta, Soda Bay, Horseshoe Bend, the Narrows, off Rattlesnake Island in the Oaks Arm, the middle of the Lower Arm's western basin, and the Lower Arm eastern basin off Clearlake. Additional samples were taken periodically from two canals in the Keys, as well as off the Manning Creek delta, Lucerne, and Morrison Creek. Refer to the map in Figure 3.1 for Clear Lake geography. Sample collection initially consisted of scooping triplicate 250 cc (10 cm deep) grabs of surface mud using a 6" (15 cm) Ekman Dredge at each site. Dredge positioning and retrieval was carried out carefully so as to keep the surface sediment intact. Each grab was then bagged in one gallon Ziploc bags, stored on ice, and refrigerated for up to 36 hours. From October 21, 1991 to January 1993 the number of grabs per site dropped to two, which later reverted to three for each of the mid-arm deep sites (Upper, Oaks, and two Lower Arm basins). Results presented here are from samples taken on 32 dates over 32 months, ending in March 1994.

The dredge grabbed mud to about 15 an depth in the sediments, and the surface sample was then scooped from the top 10 an. Most active recycling of phosphorus should take place above this depth. Indeed the top 2-3 cm are generally found to be the most active (T. Hollibaugh, personal communication). A 10 cm sample will thus capture all of the active sediment, but might exaggerate the size of the inert pools due to the physical isolation of deeper sediments from exchange with the overlying water. In any case, as the data below show, a large fraction of the iron/aluminum bound phosphorus fraction recovered from these samples must have participated in the annual cycle to account for the phosphorus appearing in the overlying water.

Monthly water samples were collected from the same 9 sites. Concentrations of both dissolved and total phosphorus were measured using methods described above in **Chapter** 5 for data from September 1992 onwards. From August 1991 to August 1992 DWR data from one station in each arm were used (methods described in **Chapter** 4).

Total phosphorus in sediments was determined for each sample using perchloric acid digestion (Sherman, 1942). Sediment solids and extracting solutions were separated by centrifugation and decanted solutions were then filtered (Whatman #42) prior to phosphate determination (Murphy and Riley, 1962). Reagent blanks were carried through all steps and duplicate phosphorus determinations made on every eighth extractant. All standards were made in the extractants. Water content of sediment was calculated as the difference in mass between wet sediment and separate subsamples that were oven-dried for 24 hours at 110 C. Phosphorus concentrations in the sediments presented here were reported based upon dry weight in ug/g (ppm).

Fractionation analyses were performed upon the same samples used for total phosphorus determination. Samples were transported to the UC Hopland Research and Extension Center, where a phosphorus fractionation procedure was undertaken following that of Hieltjes and Lijklema (1980). Within 48 hours of being sampled, wet sediments containing the equivalent of 0.5g dry sediment were first extracted with 40 ml. of 1 M ammonium chloride adjusted to pH 7. These were shaken for two hours, decanted, and an additional 40 ml added for another 2 hours of shaking. This first step removed the loosely bound (available) phosphates. Secondly, samples were extracted by 40 ml. of 0.1 N sodium hydroxide and shaken for 17 hours to remove iron- and aluminum-bound phosphates. Finally 40 ml. of 0.5 N hydrochloric acid were added and the samples were shaken for 24 hours to remove the calcium-bound phosphates. Residual phosphorus consisted of the difference between total phosphorus and the sum of the measured phosphorus fractions. The residual fraction is interpreted by Hieltjes and Lijklema to be mainly organically bound phosphorus. Reviews of this and other methods for speciation of phosphorus are given in Pettersson et al. (1988) and Broberg and Persson (1988). We note that the interpretation of chemically defined fractions such as these are the subject of controversy, but that in many studies the base extractable (iron/aluminum) fraction is the most dynamic.

Monthly concentrations of water and sediment phosphorus were converted to mass units, in this case total metric tons (MT) in the lake water or sediments, to enable a direct mass-balance comparison of the relationship between sediment and water pools. Water concentrations for the whole lake were determined by averaging data from stations in each arm, multiplying each arm's average by that arm's proportion of total lake volume (Upper=0.63, Lower=0.27, Oaks=0.10), and adding these weighted averages for each arm. Multiplying weighted water concentrations by lake volume, derived from lake level readings on the local Rumsey Gauge and the area-capacity curve which relates lake level to water volume, provided the calculated masses in metric tons of both dissolved and total phosphorus in the water of Clear Lake.

Deeper, fine grained sediments contribute disproportionately to phosphorus storage and recycling, although there is no evidence that very deep sediments contribute disproportionately, as Home (1975) hypothesized. Since 80% of Clear Lake's lakebed surface is more than 17 feet (5 meters) below the high water mark (LCFCWCD Clear Lake area capacity curve), data from the 6 deeper water sites (Upper Arm, Horseshoe Bend, the Narrows, Oaks and 2 Lower Arm sites) were used to represent the lake. Averages of concentrations for each arm were weighted by that arm's proportion of the lake bed area deeper than -10 ft Rumsey (Upper=0.72, Lower=0.21, Oaks=0.07) to represent the lake as a whole.

To estimate total mass in the lake, the dry weight figures for concentration were converted to wet weight (averaging 87% water), assuming a density of 1.25 g/ml and a layer 10 cm thick to obtain an estimate of phosphorus per unit area, and then multiplied by effective lake bottom area. As we noted in Section 5.3.5, waves and currents will tend to prevent deposition of small particles near shore, so that fine sediment particles will be focused in deeper waters. The lake surface below -10 ft Rumsey (13,700 ha) corresponds roughly to the area of soft bottoms. Thus, with the assumption that only the soft sediments of deeper waters participate in the internal loading process, the 13,700 ha figure is used in the calculations that follow.

Points from September, 1992 and January 1993 (identified as outliers in Figures 6.2A and **6.2B)** exhibited concentrations of total phosphorus well outside the expected error about the samples' mean and do not conform to the expected seasonal pattern. All six stations reported unusual values at these outlier sampling dates, indicating that errors in sampling procedure did not cause the outlier values. The consistency in the trends of the iron/aluminum- and calcium-bound samples over the same period, which were taken during the same sampling run, suggests that the outlier values represent errors in analysis of total phosphorus. Further, the outlier data points are not reflected in changes in the overlying water. Thus, the only natural explanation of the outlier points would have to invoke complex vertical exchanges within the sediments, which are implausible (T. Hollibaugh, personal communication). We therefore conclude that the outlier points represent artifacts, and they are eliminated from the analyses that follow.

6.3 Results

6.3.1 Water column phosphorus concentrations

Total and dissolved phosphorus cycled seasonally (Figure 6.1), with concentrations increasing strongly during early summer, remaining high through the summer, then dropping in late fall to low winter and spring levels. Note that the curves in Figures 6.1A and 6.1B differ slightly in shape, due to differences in lake volume through the record. Total phosphorus ranged from 0.13 to 0.67 mg/l, and dissolved phosphorus from 0.01 to 0.57 mg/L. Dissolved phosphorus approached levels limiting to phytoplankton growth, which occurs at approximately 0.01 mg/L (Sas, 1989), during the spring of 1994. In total mass terms, about 500 MT of phosphorus cycled into and out of the water column in the 1992-3 year, and 300 MT during 1993-4. While the timing of the peaks in phosphorus in 1992-93 match those seen in the long-term data set (Figure 4.2), the amplitudes of the peaks in 1992 and 1993 were high, as in other drought years. Winter levels of both total and dissolved phosphorus were also very high in both the 1991-92 and 1992-93 winters relative to more typical years .

6.3.2 Sediment Phosphorus Dynamics

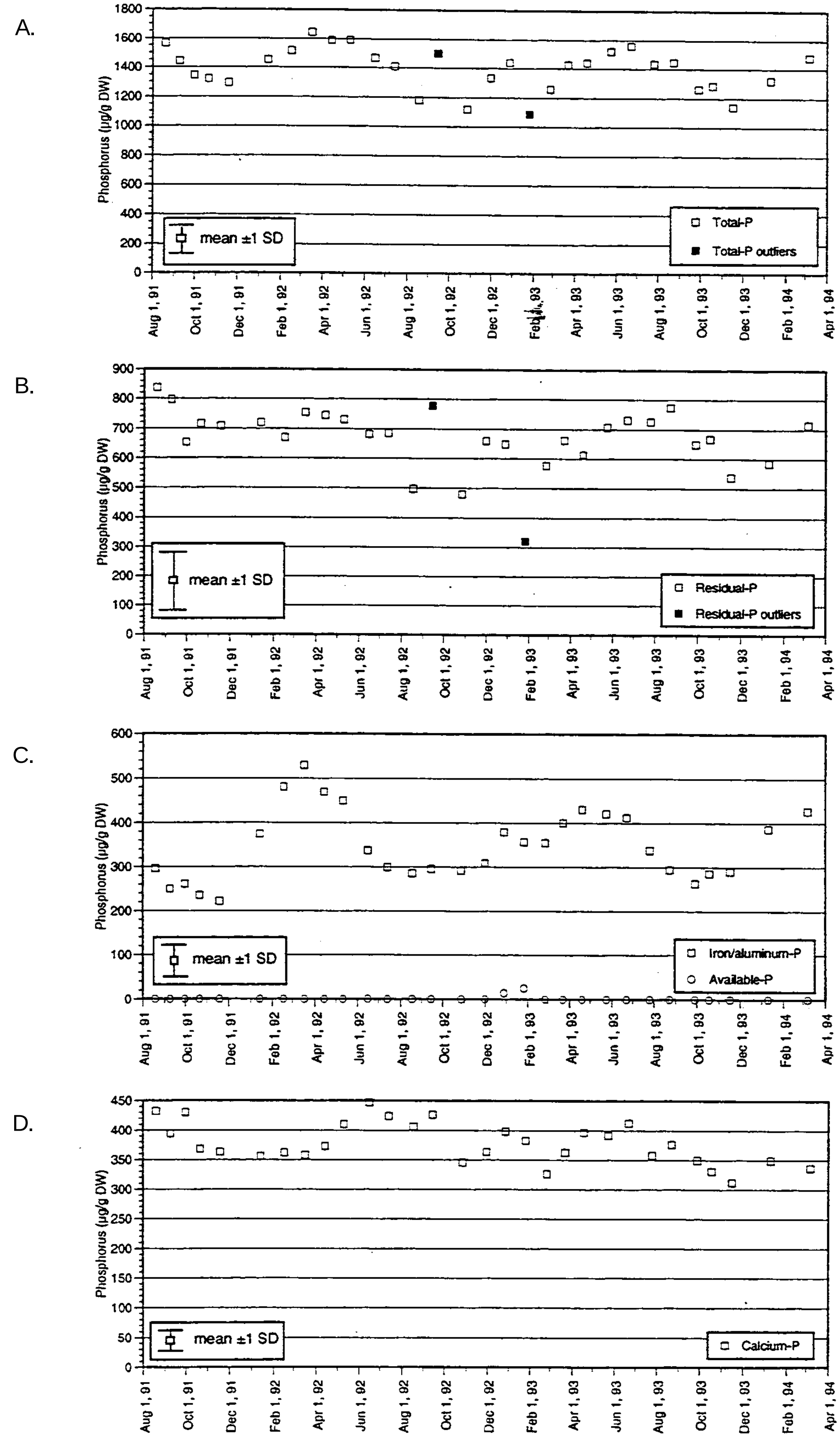


Figure 6.2. Concentrations of (A) sediment total phosphorus, (B) residual phosphorus, (C) iron/aluminum-bound and available phosphorus, and (D) calcium-bound phosphorus through time. Error bars show one standard deviation in each direction. Note differing scales of y-axis.

The phosphorus at the six deep water sites existed almost completely in one of the bound sediment-forms, with the single exception of the high creek discharge period of the 1993 winter when low levels of available phosphate appeared (shown in Figure 6.2C). The various bound fractions are much more important in the recycling process. Figure 6.2A-D combines results from the 6 deep water sites (Upper Arm, Horseshoe Bend, the Narrows, Oaks, and 2 Lower Arm sites) to plot the sample means (± 1 standard deviation) for total phosphorus and each phosphate fraction (n = 18 to 20).

6.3.2.1 Total Phosphorus

The sites exhibited clear differences in phosphate content, clustering in three groups. The lowest levels of total-phosphate were found at shallow sites with coarse-grained sediments off the Rodman Slough and Kelsey Creek deltas and the two locations in the Keys, having a seasonal and spatial range of 500 to 1,000 ug/g. Deeper water sites, representing the major portion of the lake, had levels of total phosphorus ranging between 1,000 and 2,000 ug/g. These sites included the Upper Arm, Horseshoe Bend, the Narrows, the Oaks Arm off Rattlesnake Island, and the two Lower Arm basins. The third sediment group was Soda Bay, which stood out from all other sites on the lake with total phosphorus levels rising to over 5,600 µg/ g on three occasions and seasonal shifts of 3,000µg/ g. Unfortunately, there are no measurements of sediment total phosphorus from more normal years to compare to our data. The unusually large amount of phosphorus cycling into the water column in the last few years suggests that sediment phosphorus levels are unusually high.

Total phosphorus in sediments, with the exception of the outliers (filled symbols in Figures 6.2A-B), exhibited a clear seasonal pattern during both years, increasing from fall to spring and dropping rapidly in the summer. The rise in total phosphorus in the sediments through the winter and spring must result primarily from losses of water column phosphorus due to precipitation and/ or algal uptake followed by settling, as concentrations of phosphorus in water over this period drop substantially (Figure 6.1A). External loading, interestingly, will have a slight diluting effect on lake sediment concentrations because the concentration of phosphorus in creek sediments was 1,000 µg/ g (Section 5.3.5), a little lower than the deep-water lake average of 1,440 ug/g. The low concentrations in sediments and high ones in water during the summer resulted from increasing water temperature and low dissolved oxygen, which both favor phosphorus solubility. In fact, the seasonal pat-tern for sediments were nearly the opposite of that for water throughout the year, which supports the hypothesis that phosphorus concentrations in water are due to release and resorption by the sediment.

With the outliers removed, the statistical association between total phosphorus gains in sediment and losses in the water column is significant ($p<0.01$), as shown in Figures 6.3a and 6.4a (n = 18). In terms of mass balance, it seems from Figure **6.3A** that from fall 1992 to spring 1993 the water column lost about 500 MT of total phosphorus, but the sediments gained about 800 MT. A year later, from fall 1993 to spring 1994, the water column lost 300 MT, and the sediments gained 500 MT. Given the difficulties we experienced with this determination, uncertainties in the assumption of the portion of sediments active in cycling, and the relatively few samples in time and space representing the muddy bottom, the agreement in sign and roughly in tonnage is encouraging.

6.3.2.2 Residual fraction

Of the bound fractions, the residual usually made up the largest percentage of total phosphorus (Figure 62B),averaging 680 ug/g. As a calculated value derived in part from total phosphorus, one would expect the residual to reflect many of the trends in total phosphorus, including the outlier points from September 1992 and January 1993, as it did. The shared outliers in total and residual phosphorus may have resulted from errors in the determination of total phosphorus or actual fluctuations in the residual fraction. If one interprets the residual as the organic fraction, then loss of phosphorus due to death and decay of microbes or some similar process might account for the fluctuations in the residual pool. However, the largest fluctuations in the residual fraction was due to the suspect outlier points in total phosphorus.

Neglecting the outlier points, the data suggest that this pool may have made a significant contribution to water column phosphorus in the late summer and fall of 1993, followed by uptake in the late fall **(Figure 63B) Figure 6.4B** shows a negative statistical relationship between the residual fraction and water column masses of phosphorus. This relationship is not significant ($P=0.09$). in the relatively small data set accumulated for the residual fraction so far.

6.3.2.3 Iron- and aluminum-bound fraction

The base extractable iron/aluminum-bound fraction accounted for about a third of the total phosphorus in the sediment. There was a very clear seasonal pattern of uptake of Iron/aluminum-bound phosphorus by sediments in winter and spring and loss dur-

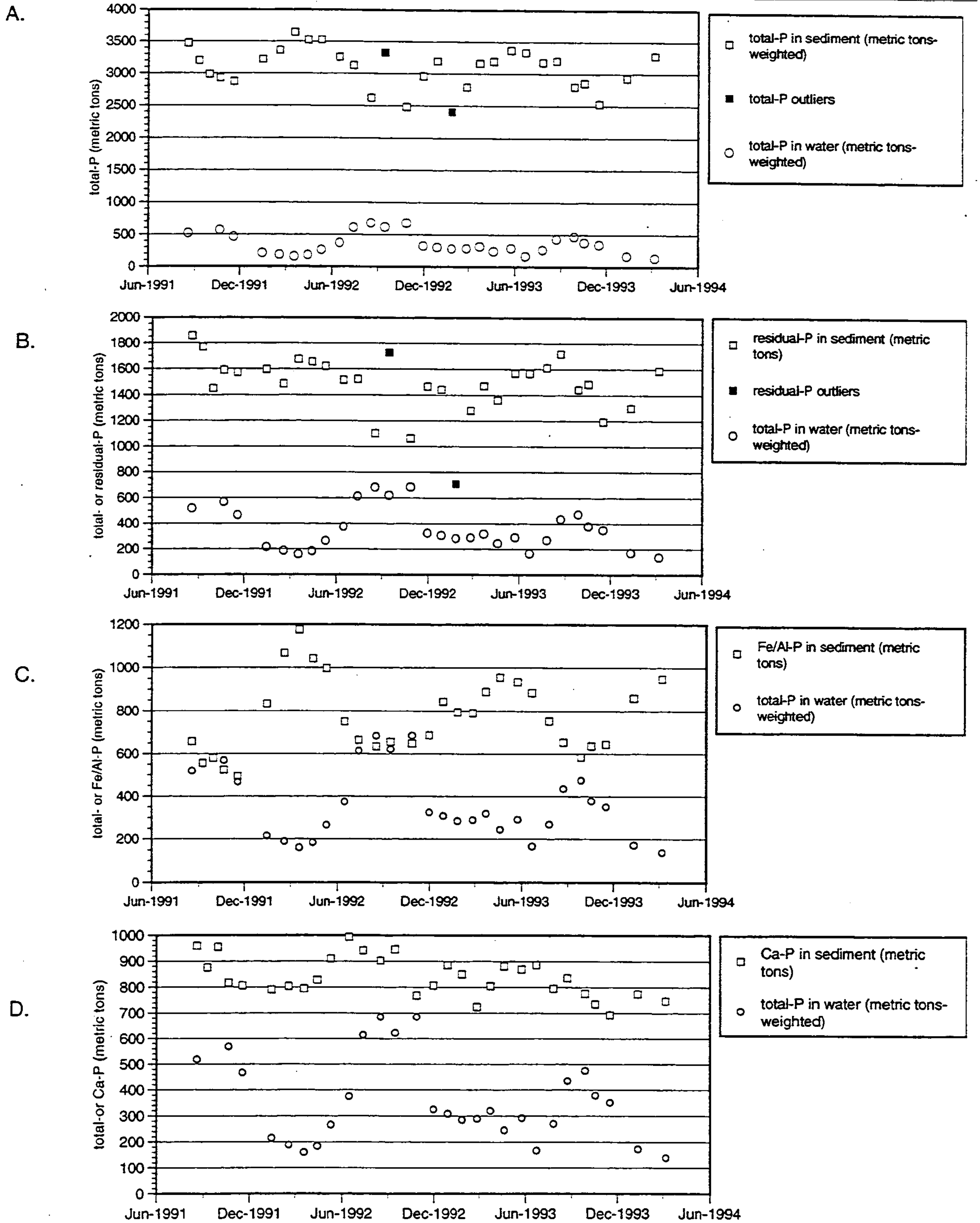


Figure 6.3. Concentrations of (A) total phosphorus in sediments and water, and (B) residual phosphorus in sediments and total phosphorus in water, (C) iron/aluminum-bound phosphorus in sediments and total phosphorus in water, and (D) calcium-bound phosphorus in sediments and total phosphorus in water over time.

ing summer and fall (Figures 6.2C and 6.3C). In the activity and increase the oxygen content of water. The winter period of maximum storage, nearly 40% of summer season produces algal crops that deplete total phosphorus was in the iron/aluminum-bound oxygen when they decay on the bottom. fraction, whereas this figure fell to closer to 25% in

summer and fall. This seasonal pattern closely fol- The relationship between decreasing levels in the lows the classic idea of sediment phosphorus cycling. sediment iron/ aluminum-fraction and increasing Iron-bound phosphorus should be mobilized from levels of total phosphorus in the water column is also sediments when the potential for anoxia at or near close one. Figure **6.4C** shows the highly significant the sediment surface is highest; conversely, sediment statistical association ($P<0.001$, $R^2=0.666$) between the phosphorus release should be inhibited when the two. The slope of the relationship is -0.765 ± 0.128, in-surface layer of the sediments is oxidized due either dictating that mass balance (a slope of -1) between to higher oxygen content of the overlying water or the iron/aluminum fraction and total phosphorus in lesser oxygen depleting microbial activity in the sedi- water is within two standard errors. A slope of -0.76 ments. Low temperatures in winter reduce microbial interpreted literally would mean that the. sediment

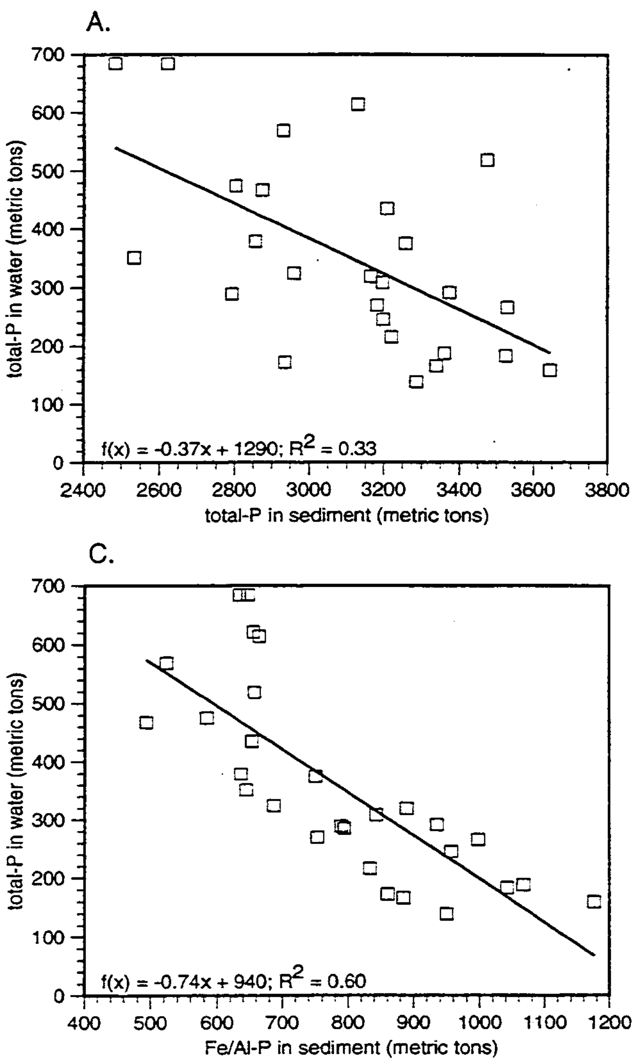


Figure 6.4. Concentrations of (A) total phosphorus, (B) residual phosphorus, (C) iron/aluminum-bound phosphorus, and (D) calcium-bound phosphorus in sediment plotted against total phosphorus in water from Aug. 1991 to Apr. 1993. Total and residual plots include outlier points. Water values before Sept. 1992 were taken from DWR stations CL1, CL3, and CL4.

iron/aluminum fraction accounts for 76% of the phosphorus entering and leaving the water column, leaving 24% to be accounted for by other pools, such as the residual, "organic" fraction. Given the uncertainties involved in the estimates of both sediment and water tonnages and the limited data available, the results strongly suggest that the iron-and aluminum-bound pool dominates the phosphorus recycling dynamics in Clear Lake.

6.3.2.4 Calcium-bound fraction

In contrast to the residual and iron/aluminum-fractions, concentrations in the acid extractable calcium fraction were relatively constant at the six deep water sites, with slight losses in the fall and winter of 19914 and erratic gains in spring and summer (Figure 6.2D, 6.3D). This pool constitutes about a third of the total phosphorus in sediments. Because this fraction varied relatively little, it is unlikely that a significant portion of sediment/water exchange of phosphorus occurs in the calcium-bound fraction. Calcium-bound phosphorus is generally considered strongly fixed and not likely to participate significandy in the biological cycle of phosphorus. Statistical analysis shows no significant relationship between sediment calcium-bound phosphorus with water column phosphorus (Figure 6.4D). Unlike any of the other fractions, the calcium-bound fraction showed increases in summer. Higher pH conditions due to evaporative concentration and photosynthesis may have the effect of increasing the deposition of apatite (calcium phosphate) during summer.

6.3.3 Results of Multiple Regression

The multiple regression of tons of each of the three bound fractions with tons of total phosphorus in the water produces a highly significant model (Table 6.1). This model accounts for more of the variation in water total phosphorus than any fraction alone. The iron/aluminum fraction was the major reason for significance of the multiple regression model of the combined sediment fractions, as expected from the analysis of the individual fractions (Figure 6.4). The most interesting feature of the multiple model is that its R^2 (0.746) is higher than that of the iron/aluminum fraction alone (R^2=0.666), indicating that the other fractions also play a role in cycling of phosphorus between sediment and water. Indeed, the contribution of the residual fraction is significant in the multiple regression model when the base extractable iron/aluminum fraction is controlled ($P < .05$). Even the weak pattern of summer increase of calcium bound phosphorus approaches significance in the multiple modeL The results of multiple regression, coupled with the loss of residual phosphorus in late summer and fall of 1991-93 evident in **Figure 6.3B,** provide evidence that the organic residual pool participates actively in recycling, though it is considerably less important than the iron/aluminum fraction.

6.4 Historical Variations In Internal Load

It is noteworthy that the 1991-94 period is very high in water column phosphorus, particularly during winter, compared to earlier years. **Figure 6.5 show** the historical internal loading estimated by subtracting the wintertime minimum phosphorus tonnage from the following summer/fall maximum, using a volume weighted average of the DWR data from the three arms. For years in which winter data were missing, the lowest preceding winter or springtime value was used. The mass of phosphorus cycling during drought conditions was unusually large. For the 1969-73 high phosphorus years the internal load was 200-400 MT/yr, about half the extremes reached in the recent drought of 1988-1992. The amounts cycling in the 1973-87 period were even lower, ranging between 90 and 210 MT/yr. Figure 6.5A shows the increasing flux in total phosphorus in water beginning in 1987 to a peak in 1990 of 650 MT and the continuing elevated flux in 1991-92. The DWR record shows that

	Tons total-P in water		
	R^2	/-value	Slope
whole model	0.784	<0.001	1.0±0.5
FRACTION:			
iron/aluminum		<0.001	-0.64±0.12
residual		0.039	-0.28±0.12
calcium		0.070	0.619±0.316

Table 6.1. Results of multiple regression analysis of sediment fractions (MT) against water total phosphorus (MT) from 1991-1994.

the 1991-94 period for which sediment data exist was historically odd, with much greater annual fluctuations than in the previous twenty years. The oddity of the past few years must be kept in mind when-ever one compares recent data with historical aver-ages and likely future conditions. Most importantly, the more normal years cycled much less phosphorus to the water column, meaning that reducing loading would have greater effect in non-drought years than the 1990-94 period would indicate.

6.5 Discussion

The very active seasonal cycle of sediment absorption of phosphorus during winter and spring and recycling, or internal loading, during summer and fall is well illustrated by the 1992-93 and 1993-94 years, although winter minima and summer-fall maxima are both usually considerably lower over the period of the DWR record (1969-92) (Figure **6.5B).** The summer maximum mass of total phosphorus in the water column in the 1991-93 ranged from 500-680 MT, which fell in winter 1992-94 to about 200 MT. The estimated loss down Cache Creek during that period was 12 MT in 1991-92 and 83 MT in 1992-93, leaving internal loading (recycling) from the sediments to account for a gain of an estimated of about 500 MT in 1991-92 and about 230 MT in 1992-93. About 200 MT of phosphorus remained in the water column during this period compared to 50 MT in the normal years from 1974- 1986.

The iron/aluminum-bound pool certainly accounts for most of the phosphorus reaching the water. The residual fraction appears to play a secondary role, and the calcium bound pool is nearly inert. The regular variation in the iron/aluminum-bound pool between 220µg/g (450 MT) and 530p.g/g (1,200 MT) (Figure 6.2C, 6.3C) and its negative regression with total phosphorus in water (Figure 6.4C) strongly indicate that it is the major fraction involved in phosphorus cycling, at least under the unusual conditions of the past drought years and the more typical 1993 winter. The nearly significant correlation between water column phosphorus and the residual pool and the significant influence of the residual fraction in the multiple regression model suggest that an organic fraction is also participating in the active phosphorus cycle. The calcium-bound pool exhibits little seasonal variation, with perhaps only a weak tendency to precipitate calcium during summer.

The total phosphorus variation in sediment tends to overestimate water changes, although the statistical correlation relating the two variables is significant. While variation in the iron/aluminum-bound fraction slightly exceeds water column changes, variation in sediment total phosphorus exceeds it by a greater margin. From the total phosphorus data in Figure 6.3A we estimate that as much as 1,000 MT of phosphorus might have been cycling from the sediments in 1992-94. Water column data from **Figure** 6.1B, by contrast, suggest that only about half that amount is cycling. The estimated outflow loss of 12 MT and 83 MT in the two years is insufficient to account for this gap. Given current data, we must accept that the internal loading process has uncertainties on this scale. However, the data from sediments are notoriously variable in space, and the presence of outlier points suggest difficulties with the total phosphorus analysis for sediments. Given the reasonably close match between the water column estimates and the well-behaved iron/aluminum fraction fluctuation, we are inclined to accept that approximately 250-500 MT of phosphorus cycled actively during the period of this study.

6.6 Conclusion

From the management perspective, it is clear that the sediments of Clear Lake have a large pool of actively recycling phosphorus, as much as 500-700 MT in drought years. It is not, however, an inexhaustible pool. With only half to a third that much phosphorus cycling in the water column during more typical years, average gains by external loading, and losses to sediments and outflow at about 150 MT/yr (estimate from Section 5.3.9), perhaps 1 to 3 years would be necessary to turn over the normal stock of cycling phosphorus in the water. Even in drought conditions, the pool cycling would turn over in 3 to 5 years. The largest uncertainty in these estimates of phosphorus load in water is the behavior of the sediment loss term under conditions of reduced loading, as shown in the model in Section 5.3.10. If the losses from water to the sediments remain at the typical levels of 130 MT, response of the lake to improvements in loading will be rapid. If lower loading results in a smaller loss to the sediments response will be slow. During the drought years of low loading, the sediments acted as a large source of phosphorus. In the most pessimistic scenario, something like the 500 MT of phosphorus that cycled at the height of the drought years would have to be lost via the relatively small discharge down Cache Creek (historical average 25 MT/ yr).

The significance of the turnover time figure is that it provides a theoretical estimate of the time it will take the lake to get halfway to a new equilibrium if (1) loading were cut, and (2) the existing stock of actively cycling phosphorus behaves as if it were contained in a simple reservoir being diluted by cleaner streams. It suggests a relatively quick response to external load reductions if the sediment acts as a reasonably large sink but a rather slow improvement if it acts as a large source of phosphorus (as during the drought). The level of total phosphorus in Clear Lake sediments is also not particularly high relative to other lakes studied, as discussed in more detail in Section 9.2, lending some support to the more optimistic estimates, making reducing external load a viable strategy. If the pessimistic estimate turns out to be more realistic, some form of sediment inactivation may be a desirable supplementary strategy.

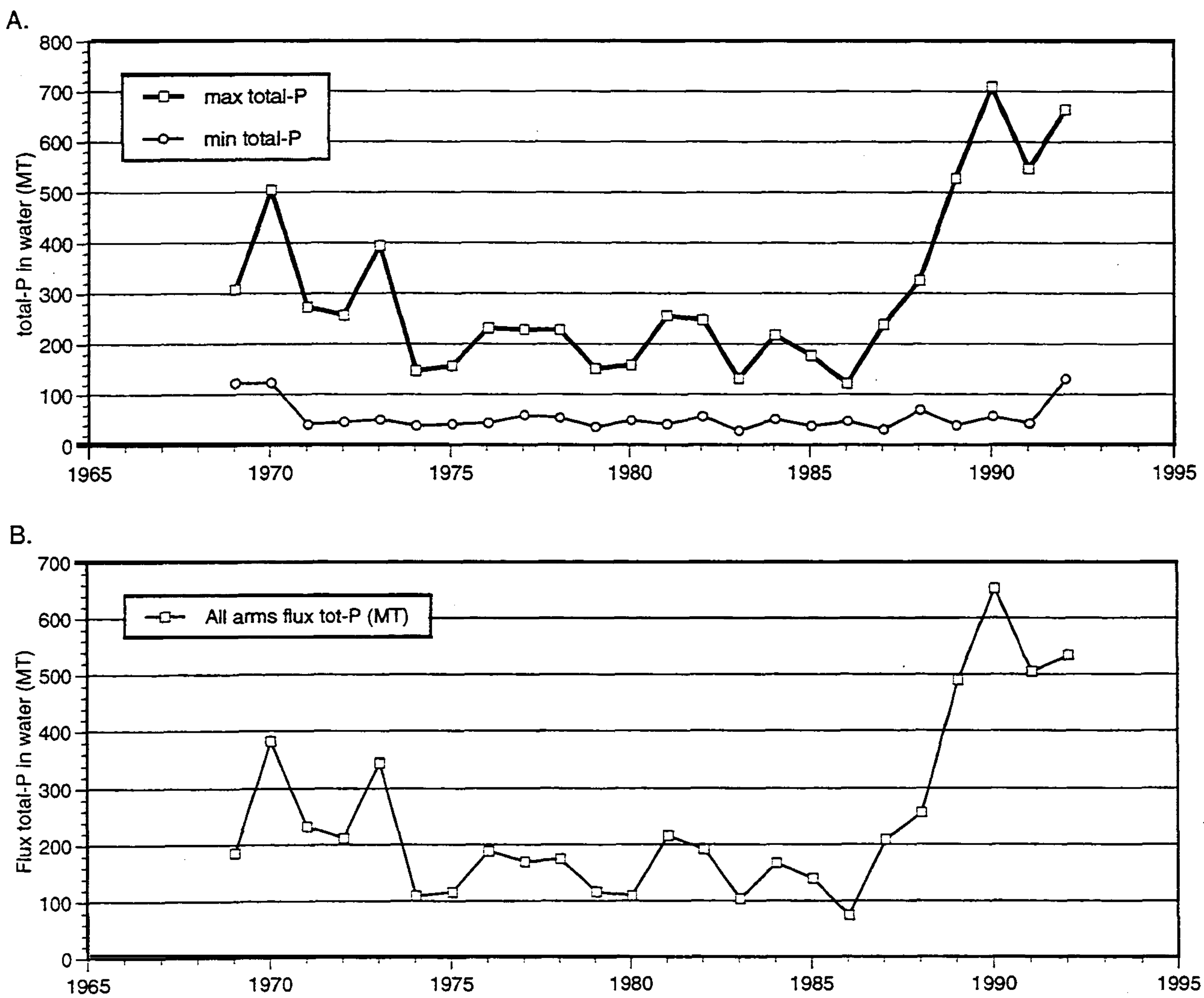


Figure 6.5. Annual maxima and minima (A) and flux (B) for total phosphorus in water through the DWR record.

7 Nutrient Bioassays

Xiaoping Li

7.0 Abstract

To verify the Home hypothesis that Clear Lake algal growth is nitrogen and iron limited, and that nitrogen fixation is iron limited, we conducted similar bioassays under the much different conditions of 1992-93. Also tested where the toxic effects of added metals on the health of algal cells.

Algal growth was significantly enhanced in every experiment bye combination of chelated iron and nitrate. In a few experiments nitrate alone was stimulating. Nitrogen fixation responded to chelated iron, though the effect was statistically significant only during bloom conditions in June and August 1993. Addition of phosphorus to lake water had little effect on algae, and phosphorus levels were already high in the lake water that was collected for the tests.

Additions of the metals copper and compounded zinc+manganese were generally inhibitory to algal growth, as has been observed in other studies.

The study broadly confirms the historical pattern where algae are limited in their growth by a shortage of both available iron and nitrogen when phosphorus is abundant.

7.1 Introduction

Nutrient bioassays are essentially fertilizer trials. Samples of plankton are collected from the lake and subsamples are fertilized with various combinations of nutrients. Nutrient bioassays are used to determine which nutrients limit phytoplankton growth. In typical eutrophic lakes, loading from erosion or sewage, which is high in phosphorus and low in nitrogen relative to plankton needs, made nitrogen limiting. Nitrogen limitation in turn favors scum forming blue-green algae because many genera fix nitrogen from the atmosphere (many terrestrial plants, especially those in the pea family, have symbiotic bacteria in root nodules that do the same). The objective of remediation is to force the system back into the phosphorus limited regime, so that non-scum forming types of algae are favored by competition for phosphorus rather than nitrogen. In some systems, the picture is complicated by limitation by other nutrients. The bioassay work of Wurtsbaugh and Home (1983) indicated that iron limited nitrogen fixation in Clear Lake, thus causing nitrogen to be limiting to overall phytoplankton productivity. The experiments described here are designed to check these results under conditions that have changed dramatically because of drought.

7.2 Methodology

7.2.1 Experimental Design

A series of nutrient bioassays was performed in Soda Bay, Clear Lake. Field bioassays of 5-8 days were conducted from Sept. 1992 to June 1993. Bioassays were designed to be nutrient enrichment experiments (Goldman 1978). In such experiments, samples of lake water containing phytoplankton are subdivided into experimental treatments which receive varying amounts of chemical nutrients and are then incubated for a period of time. The experimental treatments for this study were Control (unenriched), added NO_3, added PO_4, added Fe; added EDTA(Na_2), added EDTA+Fe, and added EDTA+Fe+NO_3, and focused on the contribution of nitrate, phosphate and iron and its chelated forms to algal productivity.

Several additional experiments were designed for metal treatments (Cu and Mn+Zn) to estimate the effects of metal toxicity on algae growth and nitrogen fixation.

Nutrient additions to the cultures were usually made at daily intervals but in two cases (Experiment 4 and Experiment Cu) at the start of the experiments. Nutrient levels are shown in Table 7.1.

Chemical and bioassay measurements were conducted at the same time so as to obtain both bioassay and lake chemical data. During the period of time when these experiments were conducted the phytoplankton in Clear Lake were dominated by blue-green algae for all but a short period in mid-winter. One of reasons for this dominance is the nitrogen fixation capacity of some blue-green algae (Home, 1975).

Treatments	NO_3 (.M-)	PO_4 (ug/L)	Fe (u /L)	EDT	Cu (u4/L)	Mn (µg/L)	Zn (µg/L)
Control	—	—	—	—	—	—	—
N	50	—	—	—	—	—	
P	—	**5**	—	—	—	—	—
Fe	—	—	20	—	—	—	—
EDTA	—	—	—	**5 lmo!.**	—	—	—
EDTA+Fe	—	—	20	**5** µrnol	—	—	—
EDTA+Fe+N	50	—	20	**5** µcool	—	—	—
Cu	—	—	—	—	10	—	—
Mn+Zn	—	—	—	—	—	10	10

Experiment 4 (added at start of the experiment

Treatments	NO_3 (µg/L)	PO_4 (µg/L)	Fe (µg/L)	EDTA	Cu (µg/L)	Mn (µg/L)	Zn (µg/L)
Control	—	—	—	—			
N	200	—	—	—			
P	—	50	—	—			
EDTA	—	—	—	10 µmol			
Fe	—	—	25	—			
N+P	200	50	—	—			
EDTA+Fe	—	—	25	10 µmol			
EDTA+Fe+N	200	—	25	10 µmol			
EDTA+Fe+P	—	50	25	10 µmol			
EDTA+Fe+N+P	200	50	25	10 µmol			

Experiment IV 'dalYaddition)

Treatments	NO_3 (u9/L)	PO_4 (ug/L)	Fe (ugfL)	EDTA	Cu (µq/l-)	Mn (u /L)	Zn (i g/L)
Control			0	5 µcool			
Fe(10)			10	5 'mot			
Fe(20)			20	5 µcool			
Fe(30)			30	5 !awl			
Fe(40)			40	5 µmol			

Experiment Cu (added at start of the experiment)

Treatments	NO_3 (µg/L)	PO_4 (µg/L)	Fe (µg/L)	EDTA	Cu (µg/L)	Mn (µg/L)	Zn (µg/L)
Control					0		
Cu (5)					5		
Cu (10)					10		
Cu (20)					20		
Cu (50)					50		

Table 7.1. Experimental design.

Chlorophyll-a and nitrogen fixation rate were chosen as parameters of algal growth.

7.22 Field Collection and Incubation

Lake water was collected with a Van Dom Sampler at depth Om, lm and 2m from Soda Bay, Clear Lake. A series of 1 gal. translucent, flexible, polyethylene containers (cubatainers) were filled with 3 liters of raw lake water each and incubated at a depth of lm after nutrient addition to each treatment. Subsamples from the experimental containers were collected and analyzed every two days before nutrient additions (except for Experiment 1 and Experiment Cu, where subsamples were collected each day). All treatments were performed in duplicate, and the Control in triplicate.

7.2.3. Measurement and Data Analysis

Chemical measurements of nitrate, ammonia, total phosphorus (TP), soluble reactive phosphorus (SRP), total Fe, filterable (soluble) Fe and other chemical and physical parameters (DO, Temperature, pH) were run during the period of each experiment (Table 7.2) Nitrate was measured by the hydrazine method, ammonia by low level method, SRP by colorimetric method and TP after acid hydrolysis. Iron was measured by the modified ferrozine method as total Fe in raw water samples and filterable (soluble) iron in filtered samples through O.45µ. membrane filter °anic and Byron, 1989).

To measure the chlorophyll-a level, 100m1 subsamples collected from the experimental cubatainer was filtered through glass fiber filters (Whatman GF/C type) and extracted at 4 °C for at least 24h with 100% methanol. Extracts were analyzed fluorometrically with a Turner Model 111 calibrated against lettuce chlorophyll-a.

Nitrogen fixation rates were estimated by the acetylene reduction method as described by Home and Goldman (1972). 50m1 of water withdrawn from cubatainers was placed into a 60m1 serum bottle and then injected with 5m1 of freshly generated acetylene followed by vigorous agitation for 15s. After incubation *in situ* for 2h at a water depth of lm in Soda Bay, the samples were retrieved then shaken vigorously and the gas in the serum bottle was withdraw with a 10ml vacutainer for subsequent gas-chromatographic analysis.

Except as noted, statistical differences between treatments were analyzed by one-way analyses of variance followed by a Dunnett's test (Montgomery 1991) to compare each treatment against the controL

7.3 Results And Discussion

7.3.1 Bioassays Results

Tables 7.3 and 7.4 show respectively the changes in chlorophyll-a levels and rates of nitrogen fixation observed during the several experiments, which are discussed below. Chlorophyll-a was significantly stimulated by additions of EDTA+Fe+NO_3 and NO_3 treatments, whereas nitrogen fixation was not (**Figure 7.1AB).** For all eight experiments EDTA+Fe+NO_3 treatments had the most significant difference from controls: two experiments where $P<0.01$ and six where $P<0.05$. In only three of the eight bioassays did nitrate additions stimulate algae growth significantly ($P<0.01$ or $P<0.05$), though the effects of NO_3 treatments on chlorophyll concentration were positive for the other experiments. The primary productivity of Clear Lake is limited by nitrogen during the period when the lake has a deficit of nitrogen species (Goldman and Wetzel 1963; Home 1975). The stimulation of algae metabolism was enhanced by combined chelated iron and nitrate in most experiments. This suggests that iron and nitrate are co-limiting factors to algae growth of Clear Lake.

Fe treatments, EDTA treatments and even Fe+EDTA treatments occasionally stimulated chlorophyll-a level, but more frequently had little or slight negative effects on algal growth (Figure 7.1A). Phosphate additions had no significant effects on algal growth from the dry year (1992) or the wet year (1993). Dissolved phosphorus (soluble reactive phosphorus) reached fairly low levels in the spring of 1993, but only on June 20 was the expected threshhold of limitation (10 gg/l) reached.

Impacts on nitrogen fixation rates (Figure 7.1B) show that there was a significant difference between treatments and controls in the summer bloom (Experiment 8, June 16-24, 1993) but not in the other seasons. This might have been partially due to the high variance between replicates that obscured real treatment effects.

Iron plays a key role in both photosynthesis and nitrogen metabolism, but it is unknown which forms of iron are actually used by the phytoplankton *in situ* (Rueter and Petersen, 1987). According to our results, the algal growth response to adding iron as a chemically stable chelate (such as the EDTA+Fe+NO_3 treatment to chlorophyll-a level and the EDTA+Fe treatment to nitrogen fixation) was much greater than the response of adding iron or EDTA alone, with the exception of experiment 9 when EDTA alone greatly stimulated nitrogen fixation. It seems certain that the complex forms of iron, rather than other iron spe-

Date	pH	DO (mg/L)	Temp. (°C)	NH_4-N (µg/L)	NO_3-N (µg/L)	Total Fe (u9/L)	Soluble Fe (1^{1}9/^{L})	SRP (1^{1}9/^{L})	TP (^{1}9/^{L})	Chl-a (1^{1}9/L)	N_2-
9/5/92	7.92	8.25	23.00	174	0.9	_	29	**296**	499	40	
9/10/92	7.20	8.20	22.80	131	1.5	651	4.8	257	511		
9/23/92	7.26	8.10	22.10	125	1.3	434	6.6	334	543	29	12
9/28/92	7.86	8.10	21.20	98	1.3	516	5.0	**356**	556		
10/20/92	7.48	8.20	18.00	260	4.4	**355**	8.1	**298**	528	37	42
1219/92		10.80	p.80	18	8.1	327	6.1	125	139	21	2~9
12/16/92	6.99	11.00	8.00	43	13	290	7.3	78	115		,
2/6/93		10.90	7.80	131	105	428	27	**52**	76	8.8	49
2/13/93	6.20	10.90	7.80	6.9	88	216	12	41	49		
3/9/93	5.58	10.20	11.60	123	95	269	11	36	63	12	60
3/17/93				23	60	**275**	52	27	60		
4/29/93				170	5.2	68	13	17	**53**	.5	78
6/17/93	8.10	8.30	20.80	80	7.6	99	16	30	59	'1	118
6/20/93				412	6.0	119	**25**	10	73	41	**375**
8/5/93	7.10	7.60	**25.50**	402	3.8	169	26	101	197	76	769
8/11/93	7.00	7.80	24.00			**227**	20	166	235	54	**262**

SRP Soluble reactive phosphorus
TP Total phosphorus
Chl-a t Chlorophyll-a
Nitrogen fixation as C_2H_4 nmol/l.h

Table 7.2. Chemical analyses of lakewater, Soda Bay, Clear Lake, CA. From Sept. 5, 1992 to June 20, 1993 inclusive.

Chlorophyll a Concentration (ug/l)

Exper. Number	Date	Field Values	Control Treatment	NO_3-N Treatment	p	PO_4-P Treatment	p	Fe Treatment	p	EDTA Treatment	p	EDTA+Fe Treatment	p	EDTA+Fe+N Treatment	p	Cu Treatment	p	Mn+Zn Treatment	p
1	9/5/92	40.4	41.4	45.0	ns	42.7	ns	44.2	ns	40.9	ns	50.0	ns	50.1	*	18.3	**	-	-
2	9/23/92	29.3	28.2	33.9	ns	-	ns	28.6	ns	27.0	ns	26.9	ns	38.0		18.9	**	17.1	**
3	10/20/92	37.3	28.3	32.4	ns	-	ns	28.0	ns	27.5	ns	27.6	ns	37.1	*	19.4	*	26.4	ns
4	10/20/92	37.3	15.6	21.7	**	16.5	ns	17.3	ns	17.3	ns	14.9	ns	18.8	**	-	-	-	-
5	12/16/92	20.6	45.4	55.2	**	43.6	ns	41.2	ns	42.5	ns	47.1	ns	53.7	*	31.8		41.0	ns
6	2/6/93	8.8	13.0	14.3	ns	12.6	ns	13.3	ns	12.7	ns	13.0	ns	16.3	*	-	-	-	-
7	3/9/93	11.4	13.4	16.1	*	13.9	ns	12.6	ns	14.2	ns	13.0	ns	17.6	**	-	-	-	-
8	6/17/93	11.0	14.5	15.6	ns	14.8	ns	16.6	ns	15.0	ns	14.2	ns	20.4	**	-	-	-	-
9	8/5/93	75.6	117.7	118.8	ns	150.0	ns	128.6	ns	133.8	ns	177.7	ns	247.7	ns	-	-	-	-

Table 7.3. Effects of nutrient additions on Chlorophyll a levels (ug/l) in Clear lake algal bioassays. Values indicate means of initial and subsequent treatments. Results which differ significantly from Controls are shown with (*) for p<0.05 and (**) for p<0.01. "ns" means that the results were not significant. (p>0.05)

Nitrogen Fixation Rates (C_2H_4 nmol/i.h)

Exper. Number	Date	Field Values	Control Treatment	NO_3-N Treatment	p	PO_4-P Treatment	p	Fe Treatment	p	EDTA Treatment	p	EDTA+Fe Treatment	p	EDTA+Fe+N Treatment	p	Cu Treatment	p	Mn+Zn Treatment	p
1	**9/5/92**	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
2	9/23/92	12.1	27.3	**27.2**	ns	-	-	23.8	ns	20.0	ns	**28.5**	ns	**29.7**	ns	**5.2**	*	5.0	*
3	10/20/92	41.9	118.6	136.4	ns	-	-	132.8	ns	83.6	ns	127.0	ns	111.2	ns	**45.5**	*	76.8	ns
4	10/20/92	41.9	107.8	99.6	ns	90.3	ns	106.9	ns	98.4	ns	120.5	ns	117.2	ns	-	-	-	-
5	12/16/92	28.7	39.9	44.6	ns	**42.7**	ns	42.7	ns	**42.2**	ns	44.4	ns	47.1	ns	**38.2**	ns	41.6	ns
6	**2/6/93**	48.9	49.5	51.8	ns	**49.9**	ns	48.0	ns	62.1	ns	63.6	ns	50.9	ns	-	-	-	-
7	3/9/93	60.4	92.1	81.0	ns	86.6	ns	79.3	ns	94.6	ns	98.6	ns	69.4	ns	-	-	-	-
8	6/17/93	118.5	**315.3**	300.5	ns	350.0	ns	**323.4**	ns	324.0	ns	**372.9**	ns	**295.6**	ns	-	-	-	-
9	8/5/93	769.4	653.8	664.3	ns	884.7	ns	714.1		1226.1	ns	930.1		1040.7	ns	-	-	-	

Table 7.4. Effects of nutrient additions on nitrogen fixation rates (nmol/l) in Clear lake algal bioassays. Values indicate means of initial and subsequent treatments. Results which differ significantly from Controls are shown with (*) for p<0.05 and (**) for p<0.01. "ns" means that the results were not significant. (p>0.05)

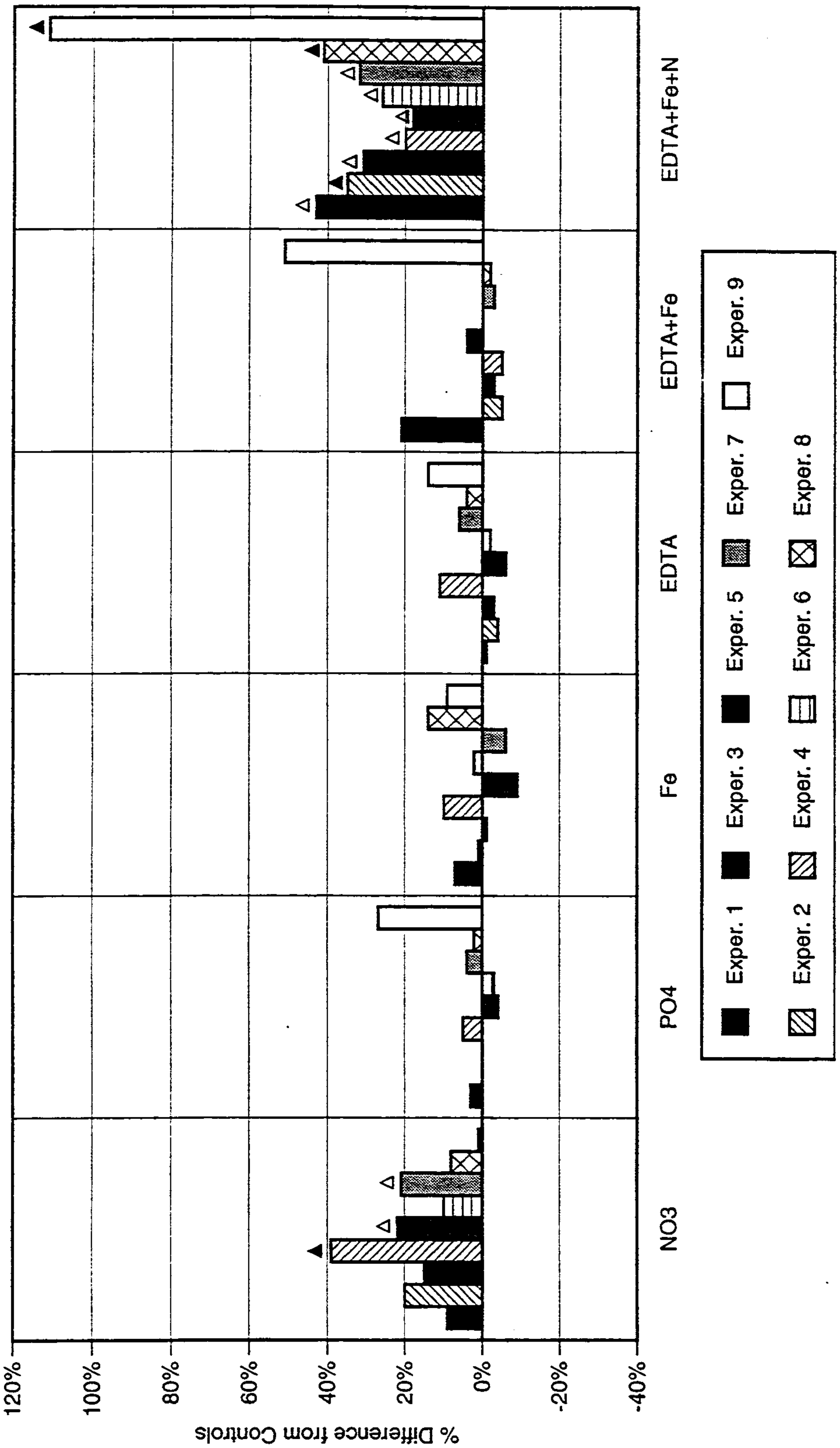


Figure7.1A. Effects of nutrient additions (as percentage deviations from the Controls) on Chlorophyll a levels (µg/L) in Clear Lake algal bioassays. Results which differ significantly from Controls are shown with (Δ) for $p<0.05$ and (▲) for $p<0.01$.

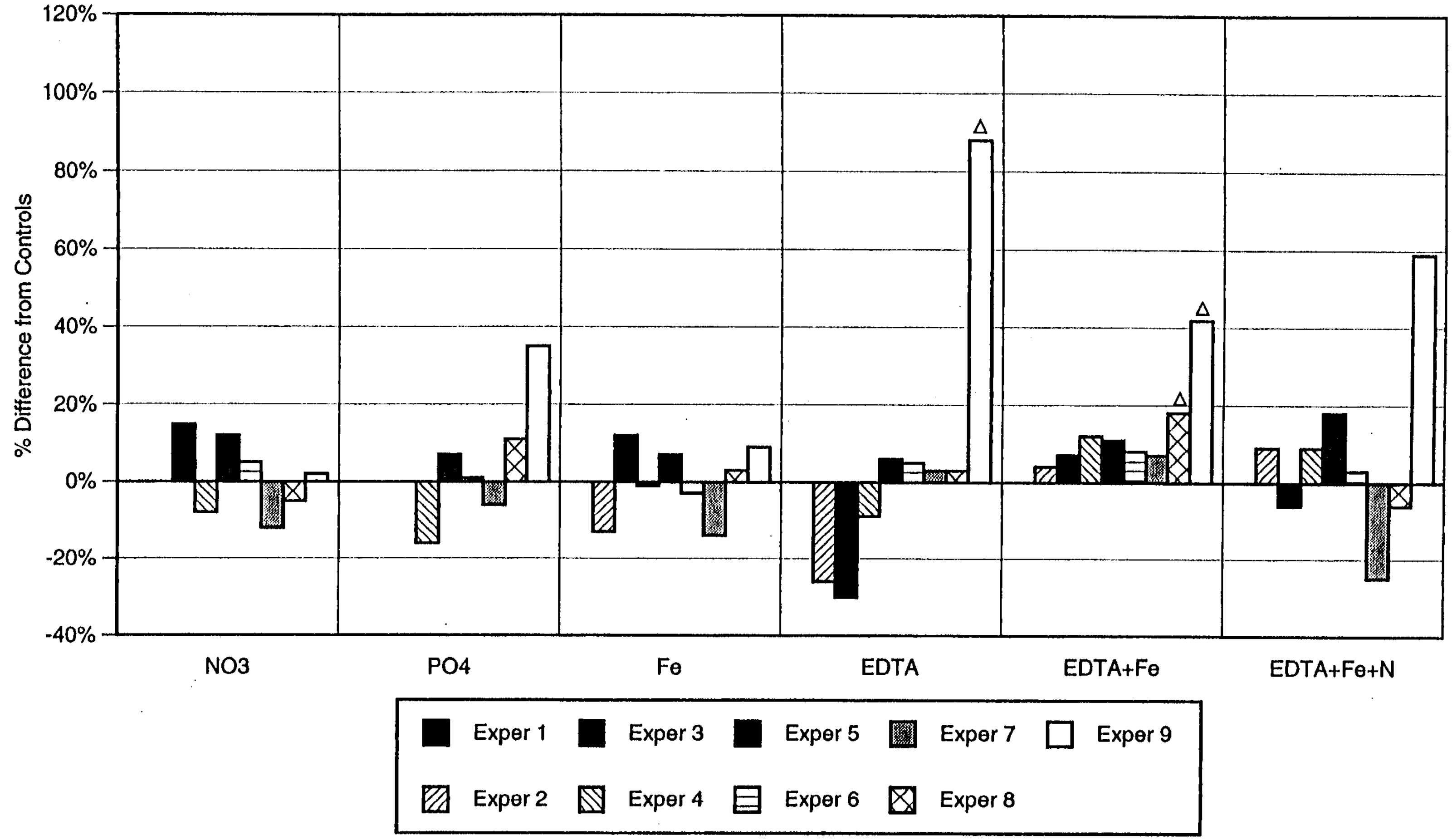


Figure7.1B. Effects of nutrient additions (as percentage deviations from the Controls) on nitrogen fixation rates (as C_2H_4 nmol/L) in Clear Lake algal bioassays. Results which differ significantly from Controls are shown with (Δ) for p<0.05 and (▲) for p<0.01.

des, were main limiting forms to algal growth. So the concentration of total iron alone may not be a good indicator of iron availability to have information on iron speciation and complexation (Morel, 1983). The strong effect of EDTA alone in experiment 9 is thus not surprising. EDTA may have made some limiting metal ion available or complexed a toxic one.

It has been suggested that nitrogen fixation by blue-green algae in highly eutrophic Clear Lake was severely inhibited by trace amounts of copper (Home and Goldman, 1974, Home, 1975). Our field bioassays show that copper and Zn+Mn treatments had markedly toxic effects not only on nitrogen fixation but also on chlorophyll-a level of blue-green algae. **Figure** 7.2A shows that for most of the chlorophyll measurements taken, Cu and Zn+Mn treatments had significantly negative differences relative to controls. Figure 7.2B shows similar impacts on nitrogen fixation from addition of varied amounts of Cu. Perhaps

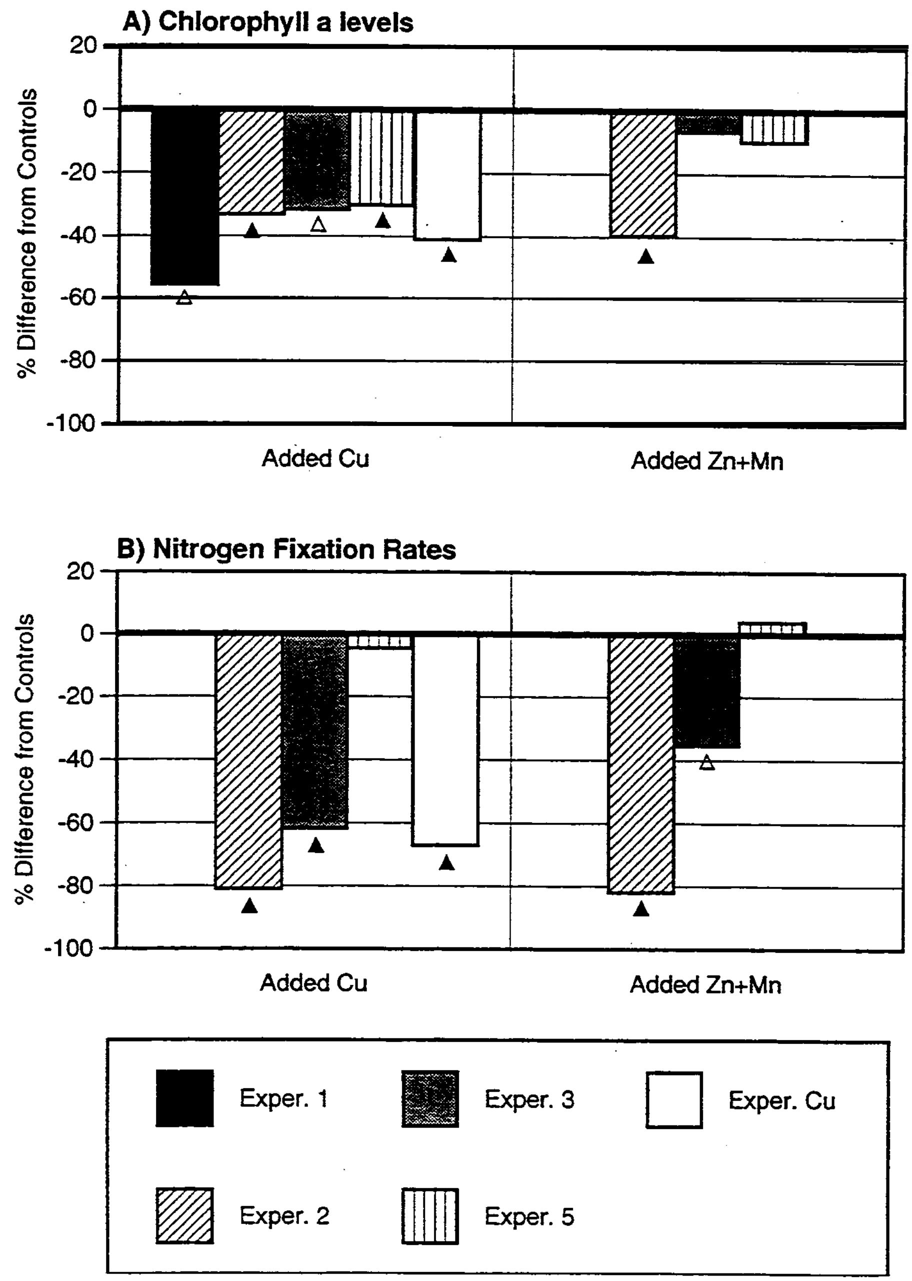


Figure7.2. Effects of copper (Cu) additions (as percentage deviations from the Controls) on (A) Chlorophyll *a* levels (pg/L) and (B) *nitrogen fixation rates (as C_2H_4* nmoVl) *in* Clear lake algal *bioassays.* Results *which differ significantly from* Controls *are shown with (A) for p<0.05* and *(A) for* p<.01.

because of chelation of Cu by organic material in the lake water, lower concentrations (5 µg/L) had no toxic and perhaps a small stimulatory effect on both chlorophyll-a level and nitrogen fixation in the first 1-2 days (Figure 7.3). However, higher concentrations of Cu caused a dramatic and permanent drop in photosynthesis and nitrogen fixation of blue-green algae. These results are similar to those reported by Wurtsbaugh (1983). In this case, the major blue-green algae was *Gleotrichia* rather than *Aphanizomenon;* the former genus first noted in Clear Lake as a dominant form in 1992.

7.4 Summary

Broadly speaking, this study confirms the historic pattern of nitrogen and iron limitation of phytoplankton photosynthesis and nitrogen fixation in Clear Lake. Most frequently, a combination of chelated iron and nitrate were required to stimulate algal growth as measured by chlorophyll-a levels. During the experiment conducted during cyanobacterial bloom conditions, chelated iron stimulated atmospheric nitrogen fixation. Copper and zinc plus manganese treatments we generally inhibitory. The experiments were conducted under a regime of high phosphorus levels, and no stimulation by this element was detected. Unfortunately, there is as yet little bioassay data from periods of potentially limiting phosphorus levels, such as were routine from 1975-1985, to test the hypothesis that it can be an important limiting element in Clear Lake.

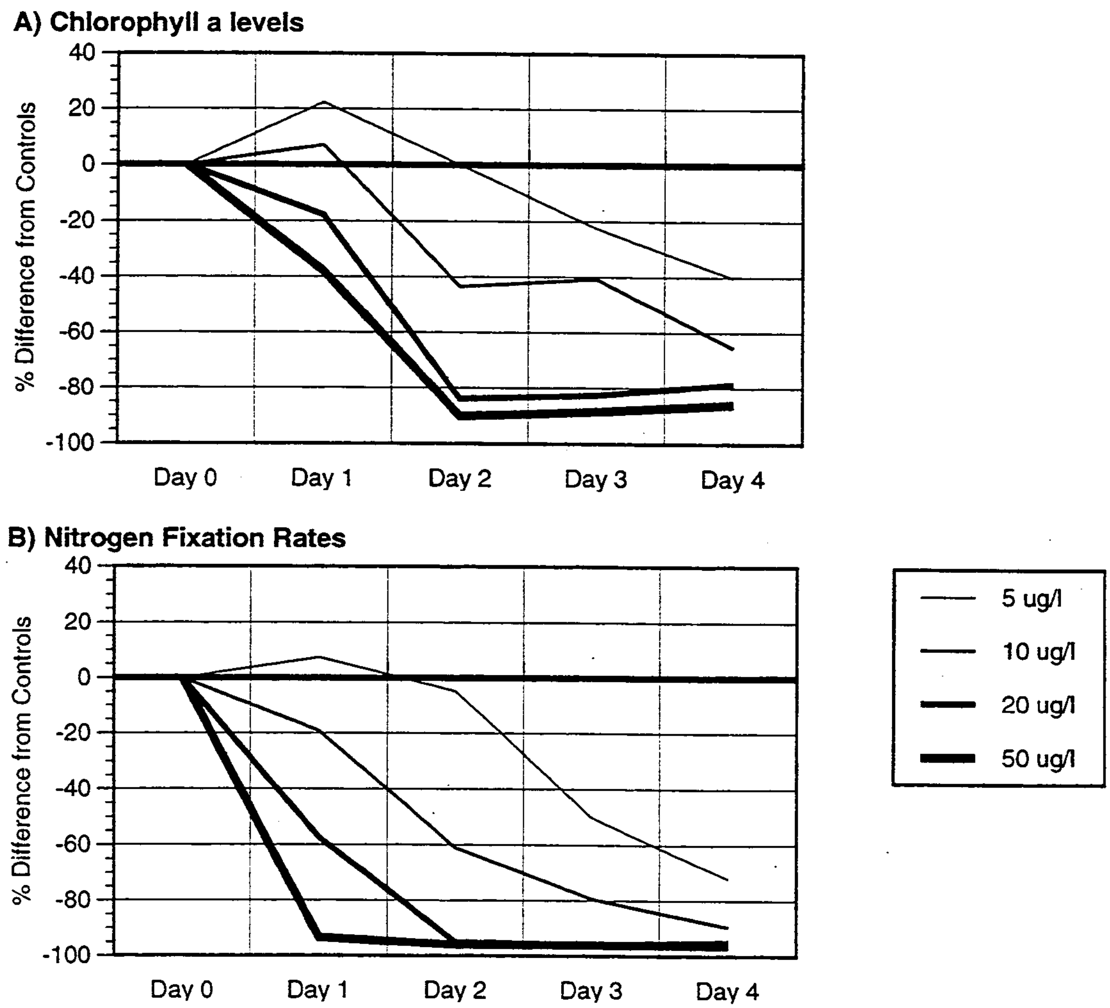


Figure7.3. Effects of copper (Cu) concentrations (recorded as percentage deviations from the Controls) on (A) Chlorophyll a levels (µg/L) and (B) nitrogen fixation rates (as C_2H_4 nmol/l) in Clear lake algal bioassays conducted during early summer blooms in Soda Bay, Clear lake.

8 Wetlands and Riparian Vegetation

B. Follansbee and S. Why

8.0 Abstract

Tributary and marginal wetlands have a strong influence on water quality for Clear Lake. Marshes trap sediments during winter creek flow, tules absorb nutrients through their roots and retain them in above-water plant parts, and streams are stabilized by riparian vegetation. Wetlands can also serve as important habitat for wildlife.

Disturbance of riparian corridors in the Clear Lake watershed has led to stream destabilization and subsequent erosion of stream banks. Gravel mining in creek beds is the major source of disturbance in most cases, although some kinds of agricultural activity, off-road vehicle use, and development can contribute to or aggravate this problem.

The reclamation of Robinson and Tule Lakes represent the most significant losses of wetlands for the Clear Lake system. Tale Lake still captures about 70% of the nutrients in Scotts Creek during winter runoff periods. Restoration of wetlands to Robinson Lake might yield similar results for nutrient loads in Middle and Clover Creeks. A significant reduction in lake phosphorus inputs is likely if Robinson lake were managed as a seasonal wetland.

A better understanding of wetlands extent and function should provide useful tools for lake managers. Ongoing research efforts devoted to the Clear Lake wetlands system should be an important part of a long-term management strategy.

8.1 Introduction

At the time of European settlement, the Clear Lake Basin possessed large acreages of a variety of wet-land types. Although much still exists, impacts of modem activities have been extensive. The primary types are: riparian forest and scrub along tributary streams (Scott Creek, Middle Creek, etc.); emergent tole marshes (Tule Lake, Rodman Slough to Bloody Island, and Anderson Marsh); fringe tole marshes along the majority of the lake margin; and macrophyte beds to some depth offshore.

Each of these communities could in principle have important effects on water quality, aesthetics, recreation, and fisheries. Riparian forest and scrub slow flood waters causing the precipitation of coarse and medium grained suspended solids, and stabilize a defined channel thus minimizing erosion of alluvial soils in the floodplain. Forested reaches of streams store sediment and nutrients on the floodplain and retard their flow into the lake. The effects of emergent marshes include slowing flood waters causing the retention of fine grained suspended solids in wetlands, stabilization of delta sediments at tributary mouths in the lake, and uptake of inorganic dissolved nutrients from the water passing through the marshes. Emergent marshes later release many of the previously absorbed dissolved nutrients in organic forms during fall senescence of the vegetation. Emergent marshes may also cause permanent deposition of nutrients like phosphorus and iron, and may be significant sites for the reconversion of nitrogen to the gaseous form. Fringe marshes stabilize shore-side fine sediments against wave erosion and uptake nutrients from the water column in competition with algae. Submerged macrophyte (waterweed) beds have some of the same effects as emergent marshes.

All types of wetlands tend to be important for fish and wildlife habitat (unfortunately including some-times noxious wildlife such as mosquitoes). The value of fringing marsh as fish habitat at Clear Lake is well documented by Week (1982). Open habitats established after clearing toles will generally support 64-90 fish/km of shore, whereas shoreline with dense toles supports nearly 700. Heavy wildlife use of Anderson Marsh, the least impacted large marsh on the lake, is some indication of what Tule and Robin-son lakes were like as wildlife habitat. Macrophyte beds are thought to have the similar value (R. Macedo, DFG, personal communication). Tule marshes and submerged weed beds compete with or interfere with human uses such as farming, navigation, and recreational beaches.

We conducted surveys of stream channels to determine the general extent of recent damage to riparian systems. Data on marsh wetlands and fringing tole marsh is reviewed. The stream water quality studies give preliminary information about the operation of wetlands for nutrient storage, as described in Chapter 5. We also give a basic description of the macrophyte communities that were important during the unusual (but perhaps natural) Clearwater 1991-1993 years.

8.2 Preliminary Estimate Of The Role Of Wetlands In The Clear Lake Ecosystem

8.2.1. Methods Of Observation

It is beyond the scope of this study to do a comprehensive analysis of wetlands. We have developed preliminary information using simple methods.

(1) The historical analyses reported in Section **3.3.1** indicate the general outlines of wetland loss as recorded in documents. As regards tule marshes fringing Clear Lake and once filling the major wetlands of Tule and Robinson Lakes, DFG biologist L.E. Week (1982) conducted a detailed survey of losses using the airphoto record back to the 1940s plus old maps and descriptions to determine losses. It is unlikely that significant improvement in Week's estimates are possible due to the limitations of the airphoto and other records. In any case, the general patterns are quite clean

(2) We have made comprehensive investigations of stream channels by foot and off-road motorbike in order to study recent changes geomorphic and vegetation changes. Cat Woodmansee and Stephen Why used off-road motorcycles to investigate conditions on the main stems of Kelsey, Scott and Middle Creeks as high as practical. As mentioned above in connection with the discussion of nutrient budgets, surveys indicate considerable damage to Kelsey Creek, Scotts Creek and Middle Creek.

(3) Nutrient budget estimates reported in Chapter 5 document the nutrient retention process in the existing Tule Lake and are useful to make preliminary estimates of the function of reclaimed/restored wetlands to retain phosphorus.

(4) In midsummer of 1991 and 1992 semi-quantitative surveys were made of the submerged macrophyte community by areal survey and sampling from boats.

8.2.2 Riparian Disturbance And Restoration

Large acreages of riparian communities have been degraded mainly through gravel mining of the streambeds of Clear Lake tributaries. The destruction of riparian vegetation and extraction of gravel from the streambed causes destabilization upstream of the mining area, within the mining area, and downstream of the mining area through changes in the stream morphology and energy content of the flowing water. Lowered groundwater tables may cause declines in riparian communties, and especially make their re-establishment difficult, due to a lack of surface or near-surface water in summer and fall.

This stream destabilization has resulted in impacts on additional riparian areas, including the erosion of large quantities of coarse materials from the floodplain and streambed, and the erosion of large quantities of rich alluvial soils from streamside terraces. An example of these impacts is the section of Scotts Creek mined between 1969 and 1971 for gravel to build the roadbed of Highway 29. The riparian vegetation was removed between milepoints 16.43 and 18.75 from the floodplain during the course of the gravel extraction operation. This section of the floodplain is still largely devoid of riparian trees and the creek flow is not stabilized in a defined channel.

Immediately following the gravel extraction the active floodplain was eroded approximately three feet on the Benneck property (milepoint 16.30) just down-stream of the gravel mine. Following the erosion of the floodplain the realigned channel began eroding the adjacent stream terrace and exporting the alluvial soil offsite. In the winter of 1992 the channel shifted 20 feet laterally on the Benneck property and removed a portion of a gravel bar and the early successional vegetation that had colonized the ban These types of erosion were observed throughout the impacted reach of Scotts Creek and may be indicative of the effects of instream gravel mining on floodplain and terrace stability in other stream systems within the Clear Lake basin.

Another type of impact is agricultural encroachment into the riparian corridor. The threat of alluvial soil erosion from the stream terraces by flood flows is drastically increased by this narrowing of the riparian corridor. Clear examples of this encroachment on Scott Creek are between milepoints 8.65 to 9.80 and milepoints 12.92 to 15.67. In the latter two, gravel extraction has disturbed long sections of channel by increasing flow velocities above and below actual locations of extraction, causing downcutting and bank loss as the streams attempted to re-equilibrate. A similar destabilization and downcutting of Kelsey Creek accompanied gravel mining and/or the cutting of the bar and marsh at its mouth to provide a boat harbor for Clear Lake State Park in 1958-59. Substantial disturbances, such as a down cutting of about

6 feet on Adobe Creek, have occurred throughout the Basin.

Visual surveys during off-road motorcycle tours and other travel suggest other causes of increased erosion rates. Clean cultivation of walnut orchards on steep slopes, steep road cuts along the lake shore, dirt roads in the uplands, and similar land practices contribute to the problem. With current information, quantitative estimation of the sources of erosion products by land use type is unfortunately not possible. A general calculation in 1972 by USDA Soil Conservation Service (SCS) (Brown and Caldwell, 1982) estimated that 54% of erosion was from streambanks compared to 41% from land and gully erosion, so sediments from stream channels are perhaps as important as all other sources of sediment combined. Goldstein and Tolsdorf (1994) reached a similar conclusion based on standard SCS methods, but with very limited field work. Studies described at the end of this chapter are underway to improve understanding of sources of erosion products more quantitatively.

8.2.3 Evaluation of Wetland Effects on Nutrient Loading

Wetland loss is a potentially important contributor to accelerated nutrient flows into Clear Lake, due to the loss of nutrient storage in wetlands. Wetland restoration is potentially an important tool to improve water quality. The major losses occurred due to the reclamation of the Rodman Slough and Tule Lake areas for agriculture in the early part of the Century. According to the study of Week (1982), the original area of wetland was about 3,642 ha. (9,000 acres) of which about 84% of the total (3,038 ha; 7,519 acres) has been heavily impacted or lost entirely. Half the loss was complete by 1952, and virtually all was complete by 1968. Almost half the total loss estimated by Week was the result of the reclamation of Tule and Robinson Lakes. Other major losses include the filling of a broad fringing marsh in front of Lakeport in the early 1900s, and the excavation of the 81 ha. (200 acre) Clearlake Keys subdivision from the marsh at the mouth of Schindler Creek in the 1960s. Week (1982) shows the basic time history of loss.

As far as water quality is concerned, these impacts do not necessarily remove all of the nutrient retention functions of wetlands. Tule Lake still floods in the winter and acts as a retention basin for Scotts Creek until pumped out in the spring. Thus, the evaluation of the present system and comparison with the pre-reclamation effects of the wetlands on water quality presents complex questions. The stream water quality data collected during the past two winters suggests that on the order of 70% of Scotts Creek phosphorus load is removed by Tule Lake (Chapter 5). Unfortunately, most of the wetland loss is absolute, more like that of Robinson Lake than Tule Lake.

If the Robinson Lake wetland between the town of Upper Lake and Clear Lake functioned as well as Tule Lake to remove phosphorus from Middle and Clover Creeks, a material increase in nutrient loading probably occurred in the late 1920s when this system was diked and drained. Levees now confine even flood flows to the relatively narrow Rodman Slough and immediately adjacent fringing marshes. The delta off Rodman Slough has apparently been aggrading rapidly.

The loss of retention and settling capacity in the historic Robinson Lake can be quantified as follows: The 1992-93 phosphorus load from Middle and Clover Creeks was about 80 MT (**Table** 5.10), so if pre-reclamation Robinson Lake was as efficient at retaining phosphorus as Tule Lake is now (70% retention, Section 5.3.8), approximately 56 MT would have been retained, which is about 18% of our estimated total 1992-3 load based upon the extrapolation to unguaged streams by the area method, or 26% based on the average flow method. Scotts Creek below Tule Lake, carrying 31MT of phosphorus in 1992-3, would also flow through Robinson Lake, so an additional 22MT of phosphorus might be removed by further filtering in Robinson Lake, for a grand total of 25% of the budget by the area method and 35% by the flow method. This calculation would be an overestimate if Best Management Practices were also implemented on stream reaches above the wetlands. Using Goldstein and Tolsdorf's (1994) estimate of a 37% reduction in load due to BMP, the total load reaching Robinson Lake would be only 70 instead of 111MT, and the amount deposited in Robinson Lake at 70% deposition efficiency 49MT. This figure is equivalent to 16% or 22% of the 1992-3 load, depending upon extrapolation method. If we exclude the Scotts Creek portion of the load from the calculation on the grounds that passage through Tule Lake would already have removed all of the phosphorus that a wetland could remove, the post-BMP storage in a restored Robinson Lake would be 35MT, or 11% or 16% of the total budget, depending upon extrapolation method. These figures indicate that the wetlands were historically very important in regulating the nutrient inflow into Clear Lake.

Fringing marshes have been heavily impacted by development. Much of this loss is due to small-scale clearing of beaches for second-home sites and lakeshore businesses. Much of the clearing was apparently aesthetic rather than functional; people preferred clear, unobstructed beaches instead of "swamps" in front of their properties. Aesthetic preferences seem to be changing as people come to understand the value of wetlands. The regulation of wetland filling and disturbance is now quite stringent, with permits required (usually denied) by the Army Corps of Engineers for filling and dredging and by the Department of Fish and Game for any disturbance that might affect fish habitat. The Lake County Planning Department now interprets these rules as requiring even new home builders to refrain from any construction practice that damages marshland. Any damage that is necessary must be offset by a 3-for-1 mitigation practice. The Lake County Lakebed Management Department provides advice on Tule planting, which currently occurs on a small scale. A certain amount of local natural recovery of fringing marsh is also visible, but neither planting nor natural recovery are yet sufficient to materially compensate for past losses. We believe that fringing marsh area is (and was) too small to have important impacts on nutrient budgets; its value as habitat is much more important.

8.2.4. The Submerged Macrophyte Community

Several methods were used to document the unusual macrophyte growth that occurred in the summers of 1991 and 1992. Mr. Lee Beery flew Peter Richerson on surveys on July 19, 1991 and July 11, 1992. The aircraft flew at about 80 knots, 300' above the lake, making a complete survey of the shore in one hour. Richerson made notes on a chart of the lake as the plane flew along the shore. At this elevation, clumps of macrophytes as small as about 1 m2 were visible if they reached the surface. The rough notes from the flight were converted to a prose description within 24 hours of the flight. On July 15,1991 Richerson and Stephen Why visited the Rodman Slough and Manning Creek mouth areas to collect samples for identification. On July 26, 1991 Richerson and Why revisited the Rodman Slough area to make a semi-quantitative estimate of macrophyte biomass. A transect was established off the mouth of Rodman Slough from 1 to 2 meters depth in an area of generally heavy weed growth. Along the transect, an iron frame 1 m^2 was thrown haphazardly 12 times, and the macrophytes within the frame pulled up (using snorkeling gear) and bagged in plastic garbage bags for subsequent biomass estimates. Wet weight was estimated from samples gently shaken dry. Dry weight was estimated from samples left to air dry for 48 hours.

Submerged macrophytes (waterweeds) established fairly dense stands over much of the lake's shallows during the clear-water 1991 and 1992 summers. Lesser but still significant growths occurred in the 1993 and 1994 summers. Although relatively clear years in the 1975-85 period gave rise to isolated growths of these forms, and various species can always be found in favorable locations, growths in 1991 and 1992 appear to exceed anything in living memory. Week (1982) did not find waterweed communities of sufficient note to even mention in his report. The only documentation we have been able to discover of significant communities of this type are for the pre-1925 period **(see Section 3.3.1)**

Three species of the common waterweed genus *Potamogeton, P. pectinatus* (sago pondweed), P. *amplifolius* (bigleaf pondweed) and P. *crispus* (curlyleaf pondweed) were responsible for most of the biomass. In 1991, when our observations were more detailed, sago and curlyleaf pondweeds were co-dominant in most stands, and bigleaf pondweed a distinct subdominant, though common. *Ceratophyllum dermersum* (coontail) and *Ludwigia peploides* (water primrose) were also dominants in limited areas, such as the shallower areas of Soda Bay. Growths were heavy enough to restrict navigation into and out of Rodman Slough and many areas with rather lighter infestations. Swimming was impeded on many beaches. Fishing was complicated by the presence of submerged weed, although Fish and Game biologists consider the growths favorable as fish habitat (R. Macedo, pers comm.

Aircraft surveys showed quite variable growths of the weed community. In general, areas with soft bottoms had heavier growth than areas with sand to cobble bottoms. In both years, the area around Rodman Slough had approximately 75% cover of weed on the surface out to a depth of approximately 2 meters. The area around the mouth of Manning Creek in the "comer" south of Lakeport was also rather densely covered (50% cover) to similar depths. In contrast, in both years, growths in the Oaks Arm and much of the Lower Arm were restricted to isolated plants. Local variation, with some quite dense patches in harbors, characterized the northwest shore of the Upper Arm. Some areas varied strikingly between the years; the east shore of Buckingham was rather thickly covered in the 1991 survey, but noted as dean in 1992.

The biomass sample from the area of dense weed growth at Rodman Slough showed high variability, with a mean dry weight of 36.7 g/m^2 (SD = 66). This biomass is quite comparable of bloom densities of phytoplankton, as reported by Home (1975). Given that less than 10% of the lake's surface area was covered by weedbeds, it is unlikely that macrophytes are qualitatively important in the lake's nutrient bud-get. As a problem for navigation and shallow-water contact recreation, and as a potential for fish and insect habitat, a large macrophyte community has important implications if 1991-92 dear-water conditions become routine.

8.3 Preliminary Results of a Mesocosm Study of Native Emergent Marsh Communities for Nutrient Removal and Sediment Retention at Clear Lake

8.3.1 Introduction

Restored, managed marshes on tributary streams at Clear Lake could serve as settling basins and temporary nutrient storage/transformation sites. The study is designed to quantify some aspects of several plant communities that could be used in restored marshes. The results of the research should indicate which community is optimal for use in restored, managed marshes on tributaries to the lake.

8.3.2 Experimental Approach

A mesocosm and microcosm study of five local macrophytes for nutrient (nitrogen and phosphorus) uptake and partitioning, productivity, invasibility, and ease of establishment was initiated. Five native marsh communities were established in replicated mesocosms in June 1993 and in replicated microcosms in March 1994. The communities consisted of one emergent species [tuber rush *(Scirpus tuberosus)*, water primrose *(Ludwigia peploides)*, common reed *(Phragmites communis)*, tule *(Scirpus acutus)*, or cattail *(Typha latifolia)]* and floating water fern *(Azolla filliculoides* and *A. mexicana)*. Water fern harbors symbiotic nitrogen fixing blue-green algae, serving as a nitrogen source (after decomposition) and takes up phosphorus. The plants are being grown on native marsh soil using wellwater with no outflows or added nutrients.

The Lake County Mosquito Abatement District is cooperating through sampling larval invertebrate production in the five communities and eventually will test alternative control methods (hormonal growth inhibitor, fungus, *Bacillus thuringensis)* in the mesocosms.

Water samples (both water column and soil pore water) were collected at three week intervals during the growing season and five week intervals in the dormant season. At the same time, redox potential and growth were measured. Biomass and soil samples were collected at the end of the growing season. Water, plant and soil samples were analyzed for nitrogen (ammonia, nitrate and dissolved organic nitrogen), and dissolved and total phosphorus.

8.3.3 Preliminary Results and Discussion

After one growing season the tule, tuber rush and cattail established and fully occupied their mesocosms. Water primrose and common reed initially established, but were later heavily damaged by aphids and subsequently outcompeted by invaders, and water ferns failed to establish well due to the late introduction date.

First year results for the study indicate that tule has the highest biomass production **(Table 8.1) followed** by cattail and tuber rush. Tuber rush had the highest nutrient content followed by tule and cattail. The higher nutrient content of tuber rush is primarily due to the less fibrous nature of this species' stems and foliage as compared to tule and cattail.

To determine the total nutrient uptake for a species, the nutrient content is multiplied by the total biomass production. Using this calculation, tule has the highest total nutrient uptake followed by cattail and

	Nutrient Content (a)				
	Nitrogen		Phosphorus		**Primary Productivity**
Species	above gnd	below gnd	above gnd	below gnd	**(g/sq. m/day)**
Scirpus acutus	0.7	0.4	0.1	0.1	7.47
Scirpus tuberosus	1.7	0.8	02	0.3	1.55
Typha latifolia	0.7	0.5	0.1	0.1	4.95

(a) figures are dry weight percentages for aboveground/belowground biomass nutrient content.

Table 8.1. Biomass production rates and nutrient content for macrophyte species grown in experimental mesocosms.

tuber rush. Other factors to consider in selecting a community for use in managed marshes include: tule was the primary species occupying the marshes that have been lost in the basin, tule has higher habitat value for wildlife than cattail or tuber rush, abundant rhizomes of both tule and cattail are available locally, and tule rhizomes can better withstand deep, continuous flooding following harvest than cattail.

8.3.4 Future Research

The meso- and microcosms are ongoing and will be monitored through early 1995. The second year of data for the mesocosms and the one year of data for the microcosms will provide additional information for the choice of a marsh community for use in managed, restored marshes.

8.4 Work in Progress

We have collaborated with Lake County to submit a Non-Point Source Demonstration Grant to the Water Resources Control Board, which has been approved. This project will demonstrate Best Management Practices in the Scotts Creek Watershed. We plan to collaborate with Lake County, the Soil Conservation Service, Resource Conservation Districts, and Goddard and Goddard Engineering to develop other quantitative studies of the erosion problem.

8.5 Conclusions

Wetlands, floodplains, and riparian corridors are important parts of the Clear Lake ecosystem. Before massive disturbance of these communities beginning in the 1920s, they retained considerable nutrient loads in depositional environments outside the lake proper. Damage to them is the main reason for an approximate doubling of phosphorus loads to the lake, and their rehabilitation is the main means by which water quality in the lake can be improved.

9 Alternative Methods for the Control of Blue-Green Blooms

9.0 Abstract

Scum-forming blue-green blooms are typically controlled either by removing sources of phosphorus from inflowing waters and/or by chemical treatment of lake waters to limit recycling of this nutrient. An estimated 50% of the phosphorus currently entering the lake is a product of human disturbance. Phosphorus (and probably iron) fertilization of algal blooms can be substantially reduced by management practices designed to reduce non-point sources of sediment from damaged creek channels, roads, and similar sources. Rehabilitation of wetlands and floodplains to settle nutrient-laden sediments would have important benefits. Control of point source inputs, such as wastewater from sewage treatment plants, is currently reasonably effective, and only marginal benefits will be obtained by improving the plants. In-lake treatments might be effective, but would be expensive and have potential hazards to the ecosystem. Direct management of blue-green populations by harvest, poisoning, or biomanipulation are not practical in Clear Lake, although skimming and similar practices are useful on a small scale for relief of local problems.

9.1 Introduction

The purpose of this chapter is to review the methods which might be used to control blue-green blooms in Clear Lake. Scum-forming blooms caused by floating cyanobacteria are a common problem in lakes around the world, and much attention has been de-voted to studying this problem over the last 30 years (NAS, 1969; Likens, 1972, Schindler, 1974, Vincent, 1987, Vollenweider, 1987; NRC, 1992). Since several pioneering attempts to control eutrophication (excessive nutrient supply) were successful, many well-documented projects have been conducted aimed at reducing blooms, and a large body of experience has been obtained. Recent reviews of these practical projects and the reasons for their success or failure include Moore and Thornton, (1988), Sas *et al.* (1989) and Cooke *et al.* (1993). This chapter follows the comprehensive outline of Cooke *et al.* (1993) in order to cover at least briefly all currently conceivable methods by which scum forming blue-green algae might be controlled in Clear Lake.

The findings of this report (**Chapters** 3 to 8) confirm the long-standing hypothesis that the blue-green al-gal blooms of Clear Lake are a product of high in-puts of iron and phosphorus. The situation in Clear Lake is similar to many other eutrophic lakes that suffer from bloom problems. The main sources of phosphorus are erosion products carried by streams into the lake with winter flood flows. Our historical analysis (Chapter 3) indicates that scum-forming blooms were uncommon in the 19th and early 20th Centuries. The DWR data series indicates consider-able reduction in phosphorus loading in the 1970s after gravel mining in streambeds and similar disturbances were curtailed **(Chapters 4, 6).** The preliminary geological assessment by USDA Soil Conservation Service (Goldstein and Tolsdorf, 1994) indicates that approximately 50% of current sediment load derives from non-natural sources. There is rough agreement between our limited measurements of sediment (and phosphorus) load (**Chapter** 5) from streams and the theoretical calculations of the SCS study. Our historical survey (**Chapter** 3), observations of stream channels, and estimates of the effects of wetland loss (**Chapter** 8) are consistent with a large increase in sediment load after 1925. The sediments of Clear Lake store substantial quantities of phosphorus and iron (**Chapter** 6), but responses to the (hypothetical) reduced loading after 1970 and to natural stress of the drought from 1986-92 were fairly rapid (**Chapter** 6).

Several features of Clear Lake must be considered when evaluating the practicality of treatment alternatives, compared to the general experience with algal eutrophication. First, Clear Lake is a relatively large lake. Intensive, expensive treatments that might be practical for a small lake may not be so in this system. Second, Clear Lake is a multi-use system. It must be managed for fish production, wildlife protection, recreation, water supply and nuisance insect control as well as for algal blooms, precluding aggressive single-purpose management strategies that might risk causing other problems. Third, the lake is unusual in having bloom formers limited most directly by iron. Ideally, management strategies would seek to enhance the iron limitation, which would be very effective in reducing bloom formers. However, few data exist for iron in Clear Lake, and iron chemistry in lakes is more complex and less well under-stood generally than phosphorus (see **Section** 5.3.11). The only practical strategy available would seem to be to take advantage of the fact that iron and phosphorus tend to cycle together and the fact that phosphorus was at or near limiting levels from 1974-86.

Thus, the general strategy we assume in this chapter and the next is that measures to control phosphorus input will also limit iron input. Even if this assumption is wrong, if phosphorus can be sufficiently reduced, it will control the abundance of scum formers regardless of iron limitation. Under this strategy, action to improve water quality can be taken, whereas any other assumption merely leads to recommendations for more studies, which have no prospects of even medium term payoffs, because of the difficulties of understanding iron.

Much like any practical management decision, the strategy is not risk-free. Clear Lake is a large, complex, ever-changing ecosystem. There are significant gaps in the existing data, and hence in our understanding. Measures recommended may not work, and may have unintended side effects. On the other hand, risks and uncertainties are not so large as to preclude action. A stepwise, adaptive management approach with monitoring of results does limit risks to quite modest levels relative to potential benefits. Monitoring and further studies are the environmental management equivalent of bookkeeping. Initial investments should be carefully assessed, and expanded, reduced, or modified as results indicate.

Two important factors need to be considered for a more detailed appreciation of likely responses by Clear Lake to phosphorus control compared to other lakes.

9.1.1 Phosphorus Stored in Sediments

First, phosphorus stored in sediments will provide an internal load of phosphorus that will take some time to reach equilibrium with a reduced supply of inflowing phosphorus. Modeling of the nutrient budget under low-loading assumptions demonstrates that water quality improvements are very sensitive to assumptions about sediment behavior (Section 5.3.10). General experience (Cooke *et al.*, 1993) suggests that levels of total phosphorus in sediments of above 1,000 µg/g dry weight are associated with slow recovery and those with lesser amounts with fairly prompt recovery. With the exception of Soda Bay (3,000-4,000 ug/g), Clear Lake sediments are at or near 1,000 µg/g. The average of 6 deep-water stations used for mass balance calculations in Chapter 6 was 1,440 µg/g. However, according to Sas *et al.* (1989:84) none of the four lakes in the study with levels less than 2,500 µg/g exhibited net phosphorus release from the sediments in the first two years after load reduction. Thus, sediment pool of stored phosphorus in Clear Lake sediments is considerable, but not high enough to delay a response to source controls for many years. Response will be rapid if sediment storage rates remain like historic averages, but slower if remediation slows the rate of sediment storage. Direct chemical treatments of sediments with alum are a method of coping with high internal loading if source reduction methods are inadequate.

It is also important to note that our data on sediment phosphorus come from the late drought years in which internal loading was extremely high due to a lack of outflow, reduced sediment storage and/or drought-induced chemical changes affecting recycling rates. The combination of natural load reduction during the drought combined with low water outflows did result in considerable net outflow of phosphorus from the sediments. On the other hand, high flow input years like 1983 and 1986 resulted in high rates of storage in sediments. (Table 5.15, Figure 5.12).

In the DWR record, we observe the fairly fast drop in phosphorus concentration after 1973, presumably due to the sediment internal load reaching equilibrium with a lower external load due to an end of some of the most disruptive periods of stream channel disturbance. Phosphorus concentrations in the water column during the 1975-85 period are consistent with the lake having reached a rough equilibrium with a lower external load of phosphorus and a lower surface sediment concentration for internal loading during the summer season, compared to the 1992-3 period for which we have data. Unfortunately, the lack of close monitoring of stream loading and sediment concentrations until the last two years do not permit a critical assessment of these issues. It is notable that the two high flow years in the 1980s (1983 and 1986) did not result in high phosphorus the following summer, an indication of improved loading conditions relative to 1970 and 1973 especially, when relatively moderate water flows were associated with high levels of phosphorus the following summer. The situation in Clear Lake justifies cautious optimism. Sediment stores of phosphorus are not so grossly excessive as to indicate a very slow response to reductions in loading; rather, responses to changes in loading seem to be rapid.

9.1.2 Are Species Shifts Away from Scum-Formers Likely?

The second fundamental idea to understand is that the most dramatic improvement in water quality in Clear Lake will occur if the occurrence of scum forming blue-greens (cyanobacteria) is reduced. The classic rule-of-thumb is that cyanobacteria are limited by dissolved phosphorus concentrations of less than 10 ug/l or ratios of nitrogen to phosphorus of more than 8:1 by weight.

How effective might nutrient load reduction be in improving water quality in Clear Lake? There are two ways to answer this question. First, given the findings in this report, is there any hope of reducing phosphorus levels in Clear Lake to levels that might limit scum-forming biomass? The historical data from the DWR monitoring program suggest grounds for optimism_ As described in Sections 43 and 4.4, declines in phosphorus levels after 1972 did confine the blooms to a shorter season for many years, compared to the apparently hopeless situation studied by Home and Lallatin in the late 1960s and early 1970s. If phosphorus loading were reduced still further against this baseline, substantial benefits to water quality are probable. Further, the drought period dear-water years (1991-2) show how iron might limit scum formers, even if phosphorus is non-limiting.

The lake was apparently fairly free of major blue-green scums until after 1925, when earthmoving projects in the basin and wetland reclamation accelerated. Thus, a program to reduce sediment loads to the lake on the order of 50%, in part by source control, and in part by deposition in rehabilitated wetlands, is a practical goal. The historical evidence from Clear Lake suggests that, if the excess load of nutrients due to human activities were reduced to predisturbance levels, it would be effective in limiting blue-greens. Substantial reductions in lake phosphorus concentrations below those in the 1973-86 predrought years have excellent promise of maintaining lake conditions that favor more desirable algal species, such as diatoms and green algae, over scum-forming blue-greens.

9.2 Nutrient Load Reduction

Controlling sources of nutrients is an obvious and common strategy to improve water quality due to excessive algal growth. Sources of excessive nutrients can be controlled by eliminating sources, diverting high-nutrient waters out of the basin, or by removing nutrients from inflowing waters by advanced waste-water treatment or by settlement in wetlands. Approaches differ depending upon the sources of excess nutrients and the nutrients to be controlled. "Point" sources, such as sewage effluent treatment, require especially different strategies than "non-point" sources, such as erosion.

Clear Lake can also be compared to experiences in managing other lakes around the world by nutrient load reduction. The technique is often but not always successful (Cullen and Forsberg, 1988; Cooke *et al.*, 1993). Sas *et al.* (1989) summarize the experience in Europe based upon a sample of 9 deep, stratified lakes and 9 shallow systems like Clear Lake. In shallow lakes, inlake phosphorus declines with reduction in external load, but proportionately less than the reduction in phosphorus load. Shallow lakes tend to recycle phosphorus fairly efficiently, so halving phosphorus inflow can be expected to reduce lake phosphorus concentrations to 64 ± 6% (95% confidence interval) of pre-reduction levels. On the other hand, on average algal biomass (chlorophyll) responded to nutrient reductions rather well in shallow lakes, so that a 64% reduction in in-lake phosphorus had the effect of reducing algal biomass to 54±7% of pre-treatment concentrations among the 4 lakes sampling. Thus the statistical experience in phosphorus load reduction in other lakes lends weight to the hypothesis that human inputs account for the eutrophication of Clear Lake. If human-caused erosion did double phosphorus loads, that should have had a dramatic impact on water quality. As a result, restoring the natural phosphorus supply should produce marked reductions in algal biomass.

9.2.1 Point Source Reduction

In Clear Lake, point sources are a minor source of phosphorus load. Even before the present spray field sewage treatment systems were adopted, Home (1975) estimated that only 20% of the phosphorus supply was from sewage waste. Current treatment plants are theoretically zero discharge land disposal systems that store treated water during the winter, and sprinkle irrigate spray fields during the summer. The limited data from Lyons Creek **(Section 5.3.6)** suggest that some phosphorus is discharged from spray fields, and considerably more by occasional high-water overflows. However, the current loading by waste treatment plants is today much less than 20% of the total external load, and probably on the order 1-3%, mostly from overflows. Uncertainties in this estimate are very large, because no routine monitoring of sewage treatment plants has been conducted. And our own data from one sample from one overflow event cannot be extrapolated with much confidence. The Central Valley Regional Water Quality Board has issued a Cease and Desist Order mandating that overflows stop by the year 2001. Nevertheless, there is no indication that major improvements in algal growth can be achieved by further improvements in the sewage treatment plant system, such as diversion or advanced treatment.

Leakage from sewage collection pipes is a potential explanation for very high populations of scum formers in some enclosed locales, such as the Clearlake Keys canal system at the Northeast end of Oaks Arm. Fecal coliform bacterial counts are often elevated in this location (Martin Winston, Personal Communication). In such situations with poor exchange systems with the open lake, limited dilution can amplify the effects of even relatively small leaks. Such leakage might also make some contribution to the whole-lake nutrient budget The lack of data on these sources makes quantitative estimates impossible, but they are presumably small on a whole-lake basis.

No other significant point sources of phosphorus or iron inflow into Clear Lake are currently known. It is conceivable that undiscovered sources exist. For example, we have no explanation for the very high levels of phosphorus in Soda Bay sediments compared to other locations. Only relatively minor monitoring and exploratory efforts on point source controls of nutrients are justified by current knowledge. Localized natural sources of iron from subsurface springs could conceivably be an important source of loading by this element, but no positive evidence for large flows has been discovered. Natural springs might be difficult to control as point sources, depending upon their number and structure.

9.2.2. Non-point Source Reduction

Given that the overwhelming mass of phosphorus flows into Clear Lake originate from erosion, and that approximately half of this flow is a result of human disturbances (Goldstein and Tolsdorf, 1994: Table 1; see discussions in Chapters 5 and 8), non-point source control is potentially the most important and practical tool for nutrient load reduction. Unfortunately, compared to point source control, an effective strategy is a matter of examining the details of the watershed, stream reach by stream reach. In designing a detailed program of non-point source control, contributions from a large number of individually small sources on each influent stream must be reduced. The data on which to base detailed, reach-by-reach recommendations are expensive to acquire. Our own data, reported **in Chapter 5,** are limited to two years (only 1992-3 with significant flows), and are not detailed. They do suggest that Middle Creek is an especially large source of erosion products. Goldstein and Tolsdorf (1994: Table 4) estimate contributions from specific standard SCS methods, albeit extrapolated from limited field work. They estimate that the largest sources of erosion products are channel erosion (34%) and that road cuts are an important secondary source (13%). They estimate that a 37% reduction of erosion is possible by the use of best management practices. The upcoming Scotts Creek Watershed Non-Point Source Demonstration Project is an example of the combination of action and monitoring that is suited to prudent actions designed to make real improvements in loading while learning to employ best management practices in the complex setting of actual creek conditions.

Non-point source contributions from wildfire, construction, urban storm runoff, and septic leachfields are all estimated to be much smaller sources than channel erosion and road cuts. However, few data exist to make a critical evaluation of possible contributions from these sources. In the case of septic leachfields, it is difficult to imagine them making a quantitatively significant contribution to the overall lake budget, even though many substandard installations exist.

9.3 Lake Protection

If best management practices to control point and non-point sources of nutrients are insufficient, measures may still be taken to protect the lake from the impact of these loads. Below we identify possible options.

9.3.1 Source Diversion

Heavily loaded waters may be diverted out-of-basin. The plan to use sewage waste from the treatment plant at the City of Clear Lake as recharge water for the Geysers Steam Field is an example. However, the scope for expansion of this strategy to a solution for Clear Lake's loading problems is limited by the fact that more than 95% of phosphorus load is erosion products in the main tributaries, not point source wastewater outfalls. Even if it were hydraulically feasible to divert large amounts of nutrient-rich water from the Middle Creek system, for example, the problem of water rights would be difficult to solve. Natural runoff contaminated by non-point sources must be managed quite differently from point sources both from an engineering and legal standpoint.

9.3.2 Nutrient retention in reservoirs and wetlands.

The apparent effectiveness of the Tule Lake system in storing phosphorus-laden sediments suggest that the Robinson Lake wetland (See Figure 3.1) once played a similar role for Middle and Clover Creeks (Scotts Creek also flowed into the Robinson Lake wetland after leaving Tule Lake). These three creeks contribute about half of the total flow into Clear Lake (Table 5.3). Tule marshes at the mouth of Kelsey Creek may have also stored significant quantities of sediment moving downstream in that system. The current Reclamation system that prevents the Scotts, Middle and Clover Creek systems from flooding their historic wetlands allows this large, sediment-laden flow to discharge directly into the lake. Even with successful best management practices implemented on Middle Creek, providing opportunities for flooding and retention of nutrients on the historic flood plain, as at Tule Lake, is an important strategy to consider. Similar measures to permit the maximum possible floodplain deposition of nutrients carried by other major creeks (such as Kelsey, Manning, and Adobe Creeks) offer considerable potential to reduce phosphorus loading from these streams, though the size and high sediment load of Middle Creek suggest that projects at Robinson Lake have the greatest potential. Following Goldstein and Tolsdorf's (1994) estimate that 37% of the load of the creeks could be controlled by best management practices, if floodplain/wetland storage removed 70% of the remainder, an estimated additional 11%-22% of the total load to the lake could be eliminated if Robinson Lake still existed, depending upon assumptions (Section 8.2.3). Wetlands and floodplains along the lower courses of streams, such as Tule Lake, are also insurance against failures of best management practices to be effective on the upstream reaches of creeks. In the absence of successful management of erosion near its sources, a restored Robinson Lake might store as much as 25%-35% of the current phosphorus load. Of course, attempting to rely entirely on wetlands to retain nutrients is a relatively short-term strategy, since high sediment loads will fill wetland basins.

The practicality of restoring the Robinson Lake Reclamation area is favored by the poor condition of the dike system. Under current conditions, an unplanned conversion of the Robinson Lake area to wetland will occur when a sufficiently high-runoff winter occurs. A project that diverts flow through all or part of Robinson Lake would result in significant lowering of the nutrient load from Middle and Clover Creeks. A multiple use project that flooded all or part of Robinson Lake in the winter followed by pumping it dry in spring, much like the current operation of Tule Lake, would result in significant lowering of nutrient load while reducing stress on levee, although the costs and practicality of such a system require investigation.

Wetland restoration has many other benefits besides nutrient load reduction (e.g. increasing waterfowl production and fish habitat), which may increase its fundability. On the other hand, some studies indicate that permanently flooded systems retain less phosphorus than seasonal wetlands, probably due to the efficient recycling of phosphorus in anaerobic sediments. Depending upon how the system is operated, flooding may limit or preclude farming activities and residential development. Increases in wetland area may also increase production of mosquitoes and other noxious insects.

Other design considerations include the necessity to size retention systems to obtain sufficient retention periods for effective sediment removal during high-flow periods. The highest flows contribute the most **nutrients (Chapter 5),** and the most effective system will have to store these flows. Major events on the Scotts-Middle-Clover creek system will deliver 10-20 million m^3 (10,000-20,000 acre feet) of water over a period of a few days after a storm. For efficient retention of nutrients, the system must be capable of temporarily storing such volumes. The historic Robinson Lake stored approximately 10 million m^3 (10,000 acre feet), leading to retention times similar to those for Tule Lake's storage of Scotts Creek flood waters.

Upstream reservoirs also store some nutrients. The effects of proposed systems such as Lakeport Lake and Pomo Dam are difficult to estimate without the results from detailed studies of the development of sediment loads along the length of streams. If the largest source of sediments is channel erosion on the relatively lower reaches of streams, the effectiveness of upstream reservoirs is not likely to be great. Also, low sediment load discharge from reservoirs will tend to erode downstream reaches as erosion and deposition approach equilibria again. High costs and environmental impact concerns have reduced interest in reservoir construction in any case. These projects have been rejected by Lake County voters because of their expense, and the US Army Corps of Engineers flood control study draft documents (ACE, 1992) indicate that upstream reservoirs are not cost-effective for flood control.

9.4 Dilution and Flushing

Introducing large volumes of low nutrient water into a lake to reduce the residence time (i.e. flushing the system) is a commonly used treatment procedure. The problem is that dilution with low-nutrient water on a scale commensurate with the size of Clear Lake would require huge volumes of water. In the late 1960s, the proposal to use Clear Lake as part of the conveyance for an interbasin transfer of Eel River water into the Sacramento system was estimated to have some effect of reducing nutrients in Clear Lake (Kaiser Engineers, 1968; Dept. Fish and Game, 1972; Dept. Parks and Recreation, 1972), but only at rather high diversion rates. However, the great expense and large environmental impacts of this project caused it to be abandoned. The Eel River is currently protected under the Wild and Scenic Rivers Act. In the current climate, such projects face insurmountable policy problems. No active planning for this project has occurred for many years, and it is unlikely ever to be built. There appears to be no prospect of obtaining dilution water for Clear Lake from the Eel River or any other source.

9.5 Alteration of Operations of Cache Creek Dam

Since Clear Lake is operated as a water storage reservoir by Yolo County Flood Control and Water Conservation District, there may be some opportunity to operate the storage system so as to accelerate nutrient outflow from the lake. The current operation of the system has this effect as an unintended consequence. The Cache Creek outflow presumably has phosphorus levels similar to those of the DWR Lower Arm station, and total phosphorus concentrations approximately double during the summer irrigation season compared to the winter storage season (Figure 4.2B). Storing winter flood outflows until internal loading reintroduces phosphorus (and presumably iron) into the water column, and then releasing summer flows during the irrigation season approximately doubles phosphorus outflow relative to a no-dam situation in which flood flows would have left the lake in the winter and spring. In a year in which Clear Lake fills to 7.56 Rumsey, 397 million m^3 (322,000 acre feet) of water are stored above 0 Rumsey for later irrigation release. This is nearly 30% of the total volume of the lake at 7.56 Rumsey. Without the dam, this water would exit the lake by late spring when phosphorus levels in the water are still low. Peak summer phosphorus values are sometimes quite high but only for a few weeks. In principle, water releases could be timed to coincide with maxi-mum phosphorus levels by operating the Clear Lake and Indian Valley reservoir systems jointly to maximize nutrient "harvest" from Clear Lake. The complicating factor is that Cache Creek outflow is a relatively small loss term compared to sediment storage (Table 5.15). It is not certain that marginal adjustments in irrigation releases would be sufficient to have an appreciable effect on phosphorus export from the system, but it is important to be sure that the timing of Yolo County Flood Control and Water Conservation District is maintained in a way that favors phosphorus export.

flows are stored, and the lake will drop to its lowest level more slowly as controlled releases for irrigation occur. In late spring and early summer, the lake will be deeper than under natural conditions, and hence will warm more slowly. The fringing tule marsh systems may be affected by the altered draw-down regime. We have not been able to discover any mechanism by which these relatively small effects could have a major impact on the growth of blue-green scum formers. LCFCWCD has studied the effects of the operation of the dam on late summer-fall lake level minima, and only very small differences can be detected in levels compared to before the operation of the dam (T. Smythe, personal communication).

Many people have expressed the opinion that the operation of Cache Creek Dam has a major detrimental effect on lake water quality. The opinions of Judge Jones quoted in Section 3.3.1 are typical. Public sentiment in Lake County notwithstanding, we have not been able to discover a mechanism that would generate major deleterious effects.

9.6 Phosphorus Inactivation

In cases where external nutrient load reduction designed to produce desired changes in trophic state (i.e. eliminate scum-formers) is inadequate or impractical, chemical inactivation of phosphorus is often attempted. The idea is to prevent phosphorus from recycling into the water column by keeping it in insoluble iron, aluminum, and calcium compounds in the sediments. The precipitating chemicals can be applied directly, or oxygen may be introduced into deep water to keep the various phosphorus minerals (especially iron minerals) in the sediments in the insoluble state. In Clear Lake, considerable benefit could be achieved by suppressing the internal loading of phosphorus (and/or iron) during the peak summer growth months. Practically, such treatments are costly, potentially hazardous to fish and wildlife, and uncertain in effect.

9.6.1 Inactivation by Aeration

Rusk (1983) reports on the relatively modest effects of attempted aeration in Clear Lake, which was intended to reduce nutrient recycling **(Section 3.3.3).** Sas *et al.* (1989:xiv) note that the oxygen content of overlying water seems to have relatively little effect on phosphorus release rates. Our data on internal

The use of Cache Creek Dam for water storage un- loading support the idea that all soft, phosphorus-
doubtedly has many other subtle effects on the lake. rich sediments in the lake lose phosphorus to the
Compared to the natural situation, water levels will overlying water during the summer. A simple scheme
tend to be higher in the winter and spring as flood to oxygenate small, deep holes to reduce phospho-

rus loading would depend upon internal loading sources being localized, which is not the case. Aeration is not a practical treatment for Clear Lake.

9.6.2 Inactivation by Chemical Treatment

Large-scale treatment of lake sediments by phosphorus inactivation is a conceivable strategy for Clear Lake. Cooke *et al.* (1993) give a good review of the principles and experience with various strategies. Compounds of iron, calcium and aluminum that form insoluble phosphates can be applied to remove phosphorus from the water column and precipitate it on the bottom. Ideally, the application remains effective at reducing phosphorus recycling from the sediments for several years.

Treatment with iron is undesirable because of the risk that it might raise iron concentrations, eliminating iron limitation in Clear Lake. As Section 6.3.2 indicates, iron fractions fail to sequester phosphorus under summer conditions in Clear Lake, though they are important in sequestering phosphorus during the winter. That iron is incapable of complete control of phosphorus during the critical warm-season period is quite possible. Also, the pH of Clear Lake is above that where the best results from iron additions are expected.

Calcium inactivation is most effective at pH >9.5, considerably higher than occurs in Clear Lake sediments. According to the fractionation studies re**ported in Section 6.3.2,** substantial amounts of sediment phosphorus are calcium-bound, but there is little evidence of active participation in the release cycle. It has been suggested that in relatively hard-water (high natural calcium availability) lakes, fluorine might be added to the lake to precipitate highly insoluble fluorapatite, the cavity-fighting fluorine mineral of teeth (Ramon Margalef, personal communication). However, this form of treatment has never been attempted.

The use of aluminum salts (aluminum sulfate, alum) to control phosphorus is a very common treatment when external load control fails or shows slow response. Phosphorus is strongly adsorbed to the aluminum hydroxide that is formed when aluminum is added to the water. A floc forms which settles onto the sediments, and prevents phosphorus recycling for some years, at least under ideal conditions. The aluminum complex is much more inert than iron and calcium phosphate minerals in the face of low oxygen concentration. Hence, phosphorus levels tend to be controlled during the critical summer bloom sea-son. The pH and alkalinity levels of Clear Lake are sufficiently high that an effective dose of aluminum could be administered without driving pH down to dangerously low levels (below 6.0). Much experience with dosage calculations and application techniques has been obtained as alum application is a relatively routine treatment method (Kennedy and Cooke ,1982; Cooke *et al.,* 1993).

Cooke *et al.* (1993) claim that nearly every carefully conducted aluminum treatment has been successful in slowing phosphorus release, and that changes in trophic state is common. The effects of treatment have been documented up to 12 years after application. Previous literature suggested that shallow lakes may not benefit from alum treatment, but the project by Welch *et al.* (1988) in Washington State and other projects reviewed by Cooke *et al.* (1993) have showed a high percentage of successes, especially when treatment was coupled with external loading control. Aluminum treatment may be especially helpful in shallow lakes where internal loading during the summer season dominates over external loading in the growing season. Clear Lake exemplifies this to the extreme, having virtually no external loading during the summer dry season. The phytoplankton depend entirely upon recycled phosphorus during the critical bloom season.

9.6.3 Environmental Hazards of Alum Use

Cooke *et al.* (1993) review a number of studies showing adverse effects of aluminum applications to fish and benthic biota. They consider alum application an environmentally hazardous process in soft-water lakes, but not in well-buffered high pH systems like Clear Lake. California Department of Fish and Game has concerns about alum application (R. Macedo, personal communication). Undoubtedly, a thorough Environmental Impact Report would have to be made, and slight risks to the environment documented, before any chemical treatment would be approved.

9.6.4 Logistics and Cost

Alum has to be applied so as to form a relatively uniform layer of aluminum hydroxide floc (precipitate) on the sediment surface. Systems for accomplishing this are well worked out for small, calm lakes, but in a large, windswept system such as Clear Lake, considerable development and testing would be required to design apparatus to lay an effective floc.

According to Cooke *et al.'s* (1993) review, the costs of aluminum application have fallen in recent years to around $600/ha. Since all data come from much smaller systems than Clear Lake the actual application might be somewhat more economical, exclusive of design and development costs. If we assume $500/ ha, and assume that 2/3 of Clear Lake's surface area is underlain by soft sediments that recycle phosphorus (11,700 ha), a lakewide treatment with alum would cost about $6 million. Since one treatment should be effective for several years, and since losses to the tourist industry due to algal scums exceed $7 million per year, alum treatment has potential for a good benefit/cost ratio. It is also obvious that such a project would face considerable environmental, developmental, financial, and institutional hurdles.

9.7 Sediment Removal

In small lakes, it is sometimes feasible to dredge and remove nutrient-rich sediments to achieve control of scum-forming algae. There are two major reasons why such a treatment is impractical in Clear Lake. First, the size of the lake makes this a very expensive option. Supposing that 50 cm of mud would have to be removed from 11,700 ha of lake bottom, 58 million m^3 of mud would have to be dredged. It is important to note that lake sediments are 10s to 100s of meters thick, and dredging would be completely impractical unless deep sediments contain much smaller pool of iron/aluminum phosphorus than surface sediments. Cooke *et al.* (1993:Table 17-7) compare the costs per hectare of alum and dredging projects. The <u>minimum</u> costs they cite for sediment removal is $\$1/m^3$, leading to costs on the order of at least a few thousand dollars per hectare, rather than a few hundred dollars per hectare for alum treatment. Most projects reported costs of several dollars per m^3. Thus, dredging projects are usually used only when it is desired to increase the depth of the lake as well as remove nutrients.

Second, sediment removal from Clear Lake also faces major questions regarding environmental effects. Dredging is destructive of benthic organisms on which many fish feed. The disposal of large volumes of dredge spoil is difficult to accomplish. Large areas of Clear Lake sediments contain considerable quantities of mercury, arsenic and chlorinated hydrocarbon pesticide residues. The more highly contaminated sediments would be hazardous to disturb and expensive to dispose of.

Supposing that some form of in-lake treatment of sediments ultimately proves necessary, alum treatment promises to be clearly more cost effective and environmentally benign than dredging.

9.8 Biomanipulation

Water clarity and algal biomass can be manipulated in many lakes by increasing the grazing rate of zooplankton upon algae. This can be done by either removing the smaller zooplankton-eating fish by poisoning or introducing predatory fish that consume the zooplankton feeders. Higher zooplankton populations in turn lower phytoplankton populations by increased feeding rates.

Three problems limit the potential for biomanipulation in Clear Lake. First, the experiments of Elser and Goldman (1991) described in Chapter 3 indicate that increased zooplankton grazing has little effect upon blue-green algal biomass in Clear Lake. The scum-forming blue-greens are highly resistant to grazing because of their large size and poor nutritional quality, so these results were expected. Second, Clear Lake already has large populations of picivorous largemouth bass. It is likely that these populations are already in balance with zooplankton-eating forage fishes. It is unclear what strategy a fisheries manager could use to manipulate fish populations to increase grazing on phytoplankton. Third, The Clear Lake gnat is a zooplankton-consuming noxious insect, which seems to be held in check by the predation or competition of the Inland Silversides. Although this species has not been a serious problem for many years, populations still exist, and the potential for a population explosion exists (Art Colwell, personal communication). Any biomanipulation measure that increased zooplankton populations would run the risk of increasing gnat populations.

A few fish are known to graze directly upon blue-green algae, especially certain filter-feeding species of Tilapia. Cooke *et al.* (1993: 385-387) give a concise review of experiences, mostly in Florida, where algae-eating species have escaped. The prospects for utilizing algae feeding fish in Clear Lake are not positive. Successful algal control is not often achieved in Florida, and fisheries managers consider Tilapia a serious pest that interferes with game fish. These tropical species would probably not survive winters in Clear Lake, and a large-scale hatchery would have to be operated to supply new populations every spring.

Cooke *et al.* (1993) summarize their discussion of biomanipulation with the observation that the strategy is still quite experimental and most likely to be effective in relatively low-nutrient lakes. There is no currently known biomanipulation strategy that offers a promising solution to Clear Lake's problem with scum-forming blue-green algae.

9.9 Algal Harvest

Direct harvest of scum-forming populations with skimming machinery is frequently suggested as a solution to Clear Lake's algal scum problems (Ed Hedrick, personal communication). Since these algae float on the surface, the same basic technology as used for oil skimming is applicable to removal of scums. No large-scale skimming systems for the removal of algal biomass have been demonstrated on any lake, but engineers who have examined the scums are of the opinion that no insuperable mechanical problems exist. A pilot-scale project by Cindy Dreps, under the supervision of Professor Roger Garrett (Agricultural Engineering, UCD) is currently under way.

Two quite different types of harvest strategies need to be considered:

9.9.1 Harvest as a Lakewide Control

Might it be possible to actually remove enough algae from the lake during the bloom season to directly reduce population sizes and nutrient concentrations? (During blooms, most of the limiting nutrient will exist in the algal biomass, not dissolved in the water.) One problem with this strategy is that under most conditions, only a small portion of the potential scum-forming biomass is actually at the surface in scums. Most remains suspended in the water, so that only a small proportion of the population is susceptible to skimming, at least on any given day. There may be exceptions to this generalization. The massive accumulations of very buoyant *Microcystis* at Horseshoe Bend, the Keys area, and other downwind locations in September-October 1990 may have concentrated a substantial fraction of the population in highly localized areas. No precise measurements were made, and in any case this episode was apparently unique in the known history of Clear Lake.

The second problem is the mass of material that would have to be collected and disposed of. Large blooms produce a biomass of 100-500 ml/m^2 of lake surface area. For a lake as large as Clear Lake, the total bloom crop will amount to 20-100 thousand metric tons of algae (wet weight). Scums contain much water between colonies of algae, so that the total volumes to handle would be roughly twice this figure. No figures exist in the literature for comparative purposes, but the problem of skimming and disposing of wet algae is somewhat similar to dredging and disposing of sediment. If costs of skimming plus disposal of a few dollars per metric ton could be achieved, and if a sufficient quantity of algae are exposed to skimming, it is conceivable that skimming could be an effective means of removing algae and nutrients from the system, comparable in cost and effectiveness to alum treatment.

It is often suggested that algal harvesting could support itself through sales of the algae. Indeed, one genus of blue-green algae, *Spirulina,* is grown in ponds as a food supplement and for specialty chemicals. However, these systems depend upon the steady production over a long season of a rather controlled product. The episodic nature of the blooms in natural systems such as Clear Lake, the variability in species composition and presumably chemical constituents, and the fact that the genera present in Clear Lake may contain toxic substances (Dept of Health Services, 1991) all suggest that it will be difficult to develop a valuable product (Algae Subcommittee Finding 91-8). On the other hand, tests made for Lake County in previous years suggest that harvested material could be composted on agricultural fields or disposed of in similar ways without hazard. Harvest and disposal operations on the scale required for lakewide effects will face major questions of environmental impacts at transfer points and disposal sites.

The most obvious problem with the harvest alternative is simply that it is unproved. With no experience from other systems for guidance, development costs are likely to be substantial and risks of failure high.

9.9.2 Harvest for Local Amelioration of Scum Problems

The use of spraying, sprinkling, and airboat mixing for the short-term improvement of conditions on specific beaches and in confined channels are proven, if limited, strategies for algal scum control in Clear Lake. Small- to medium-scale harvesting technologies might well be much more effective, since current techniques of mixing the biomass into the water column are only effective until the scums rise again to the surface, generally after 24 hours or so. However, repeated harvest will frequently be required as new scums are blown ashore.

Small-scale harvest has much the same spectrum of promise and problems as large-scale harvest. The main difference is that small-scale harvest schemes offer immediate payoffs and pilot scale of harvesting will not require the same magnitude of investment as an immediate attempt at harvest on a large scale.

9.10 Copper Application

Copper sulfate application to control blue-green scums is a classic control technique. Home and Goldman (1974) proposed, on the basis of bioassay studies, that trace amounts of copper might suppress nitrogen fixation and blue-green growth, although subsequent bioassay experiments by Wurtsbaugh (1983) did not replicate the extreme sensitivity of blue-greens to copper additions. Lakewide copper treatments at concentrations normally used are fairly expensive (AWWARF, 1987), on the order of $100/ha, or about $2 million for one lakewide treatment Treatments are only effective for short periods of time. Both short-term toxicity effects on many organisms and long term deterioration from repeated applications are known from copper treatment (Hanson and Stefan, 1984). Copper treatment is neither cost effective nor environmentally sound as a means of controlling scums in Clear Lake.

9.11 Waterweed Control

As evidenced by the 1991 and 1992 summers, low blue-green biomass and relatively transparent water in Clear Lake leads to growth of rooted aquatic plants out to depths of about 2 meters (Chapter 8). Growths heavy enough to inhibit navigation and the enjoyment of beaches and <u>swimming</u> docks were widespread. Heavy growths of algae prevent waterweed growth by shading, but reduction of algal biomass is likely to make weed growth more common. (This result is not inevitable. Species shifts away from scum formers toward smaller suspended algae can reduce subsurface light penetration because smaller particles are more efficient light absorbers than larger ones for equal volumes.) Since the purpose of this report is to suggest alternatives for algal scum control, a detailed discussion of waterweed control is beyond its scope. However, since waterweed growth is a possible consequence of control of algae, a sketch of the potential for controlling it is important complementary information.

Waterweed growth in recreational lakes is an extremely common problem, and many alternative chemical, biological, and mechanical treatment methods are employed for control. Cook *et al.* (1993) give a multi-chapter review. The most common methods of treatment are herbicides, harvesting and the introduction of grass carp. Development costs for registration of herbicides in California are likely to be high in view of potential environmental impacts, and there appears to be little cost advantage over mechanical removal. Many cost reports suggest that harvest costs are on the order of $300-2,000/ha. Since mass clearance is usually not desirable for environmental reasons or necessary for satisfactory usage, the total area cleared is usually limited to a strip in front of beaches and docks, and paths for navigation. Costs on the order of $3-5 per frontage meter of property may be a reasonable estimate. Home (1975) gives a shoreline length for Clear Lake of 114 km. If we suppose that half of this length might require clearance twice per year in a year with serious waterweed infestation, weed removal costs would reach about $500,000.

Grass carp are <u>specialized</u> consumers of waterweeds (such as the Potamogetons) which have dominated the biomass in Clear Lake. Grass carp introductions are an inexpensive but controversial method of waterweed control. Sterile triploid fish are stocked in sufficient quantity to consume the biomass of waterweed. Costs on a per hectare basis are about 20% of harvester costs.

Controversy arises because of general fears attending these introductions. There is worry that triploid stocks will contain viable diploids or otherwise manage to reproduce and spread. The impact of grass carp on other species of fish and ecological variables more generally is sometimes large and is not predictable.

Problems have arisen in attempting to control the amount of waterweed grazed. Clearly, grass carp cannot be expected to tailor their grazing to human desires, and all-or-nothing control is the frequent result. This reduces the cost advantage relative to harvesting unless complete eradication of waterweeds is desired. The current position of CDF&G that waterweed growth is desirable habitat is not favorable to the eradication strategy.

There is high uncertainty regarding problems that waterweed growth will present as water clarity increases due to nutrient control. Selective harvest is. a simple, proven, and environmentally benign approach to this problem. At least until much more serious problems than occurred in 1991 and 1992 appear, it will be a sufficient strategy to mitigate waterweed problems.

The cost of selective harvest is small relative to estimated losses due to water quality deterioration from algal scums. It is our impression that the problems associated with waterweed growth in 1991 and 1992 indeed caused less problems for aesthetics and recreation than algal blooms. Even with only limited hand methods available for waterweed harvest, most residents and recreators appear to prefer clear water to algal bloom conditions. Fishermen may be an exception (Richard Macedo, personal communication). Waterweed beds are considered excellent fish habitat, but do have the effect of allowing fish to spread out. Angling is thus more difficult. We conclude that, on balance, the waterweed problems that may result from clear water conditions are a tolerable cost for the benefit of reduced blooms of scum-forming blue-greens.

9.12 Conclusions

Non-point source control of the sediment load to Clear Lake, combined with the rehabilitation of floodplains and wetlands, promises to reduce the external loading of nutrients to Clear Lake to near-natural levels. The rehabilitation of eroding stream channels, control of erosion from roads, and re-establishment of sediment-trapping wetlands in the Rodman Slough area will result in the largest reductions. The information currently available on internal loading suggests that the response of the lake to a major reduction in external load will result in prompt, substantial improvements in water quality, including substantial declines in the occurrence of scum-forming blue-green algae.

If the response to non-point source control and wetland rehabilitation proves slow or limited, alum application to control phosphorus emission from sediments is a potentially effective strategy.

On the basis of current information, no other strategy for reducing the magnitude of scum formers shows high promise of making a qualitative difference, although there are some high-risk ideas worthy of further investigation.

Strategies for direct abatement of nuisance conditions in local areas are a separate consideration from lake-wide control strategies. Continued development of skimming and weed harvesting technology for these limited purposes is worth pursuing.

10 Recommended Strategies

10.0 Abstract

Our findings suggest that non-point source sediment reduction is the most important method of reducing algal blooms in Clear Lake. We recommend an aggressive strategy of adaptive management of non-point source loading. Stream reaches should be selected for application of Best Management Practices on as large a scale as State, Federal, County, private, and grant funds will permit. Monitoring of the effectiveness of these alternatives should be as thorough as funding permits. Cost-effective management practices should be expanded to cover all applicable sources, and unsuccessful ones dropped or modified.

These findings also suggest that wetlands on the lower reaches of streams are very effective at reducing sediment and nutrient inflow into the lake. The current Tule Lake system is quite effective, even Though intensively farmed. Restoring nutrient retention functions to the historic Robinson Lake has high promise of considerably reducing the load from the highly erodable Middle Creek drainage, and should likewise be pursued aggressively.

Monitoring of the lake itself should be expanded and modernized. The current monitoring of the lake by the Department of Water Resources, and streams by DWR and Lake County, is inadequate for rational management. The stream inflow record is short, limited to a modest fraction of the watershed, and is inadequate to evaluate the contributions of specific non-point sources. In an ever-changing system that is being managed intensively, monitoring is an essential part of evaluating management strategies and responding to ongoing natural and unintended human-caused changes.

Clear Lake is still an incompletely understood system. All of the recommended nutrient controls recommended here carry an element of risk. Some promising strategies cannot be recommended for action without further investigation. An ongoing program of research and development ought to be conducted at Clear lake to reduce uncertainties and develop promising new strategies.

Clear Lake is the natural environmental analog of a major business investment. To imagine that the lake can be managed effectively to support the tourist industry, aesthetic benefits of the lake to residents, and other environmental benefits without monitoring and research expenditures is like imagining that one can manage a business without ongoing bookkeeping and financial analysis. The lake will continue to change because of natural and human changes in the basin. Human adaptation to and management of these changes requires knowledge that can be obtained only through monitoring and research.

10.1 Recommendations for Action

10.1.1 Channel Protection and Rehabilitation

The most important action that should be taken to reduce nutrient loads to Clear Lake is to apply Best Management Practices to all downcutting and actively eroding stream channels. Our surveys, The Lake County Aggregate Resources Element of its General Plan and the direct estimation approach of Goldstein and Tolsdorf (1994) show that all major tributaries carry high unnatural sediment burdens due to channel damage. These sediments are the main vehicle of phosphorus loading to Clear Lake.

The pending Scotts Creek Watershed Non-point Source Demonstration Project, developed based on the early results of this study, exemplifies the action we recommend. The Scotts Valley watershed will be divided into four sectors from headwaters to Tule Lake, and the kinds of damage to each sector evaluated. Then, examples of Best Management Practices will be applied to the problems of each sector, and monitored for effectiveness. Best Management Practices include restoration of streamside riparian corridors, and construction of channel stabilization and grade control structures designed to reduce water velocity and encourage sediment deposition.

The most effective strategies based upon demonstration projects should be expanded as rapidly as funding can be obtained to all high sediment yield stream reaches, with priority to Middle and Kelsey Creeks (and secondarily to Scotts Creek) because of their size and importance to the nutrient load. The Option 9 proposal to expand the scope of application of Best Management Practices to a considerably larger number of stream reaches than possible with the NPS demonstration project is an example of this strategy.

Because of present State and National policy concerns with nonpoint source pollution, erosion, and wetland habitat rehabilitation, implementation of this recommendation should receive substantial assistance from grants. Lake County agencies and districts, under the leadership of the Flood Control and Water Conservation District and Lakebed Management, should pursue aid from such agencies as the US Environmental Protection Agency (Nonpoint Source programs, Clean Lakes Phase II and Phase III; managed by the State Water Resources Control Board), USDA Soil Conservation Service (which assists private landowners to implement best management practices through Resource Conservation Districts), California Department of Transportation (which funds environmental mitigation projects of highway construction), California Department of Fish and Game (which acquires land and funds projects for habitat improvement and conservation), and the US Biological Survey (which promotes habitat conservation). Since the US Forest Service and Bureau of Land Management have substantial land ownership in the basin, the county should seek assistance from these agencies, especially for projects on their lands.

10.1.2 Wetland Protection and Rehabilitation

10.1.2.1 Restoration of Robinson Lake

We recommend that Lake County seek funds to reestablish a functional wetland system in the former Robinson Lake. As large an area as practical (at a minimum an area the size of Tule Lake) should be allowed to flood during high water events, so that the nutrient rich sediment transported by Middle and Clover Creeks is deposited in the Robinson Lake basin rather than the main body of Clear Lake.

Such a system, whether farmed as Tule Lake or devoted to wetland habitat or both would store substantial quantities of nutrients that now enter the lake from Middle and Clover Creeks via Rodman Slough. (Historically Scotts Creek below Tule Lake also flowed into the main lake via Robinson Lake.) The technical water quality objective should be to store flood flows on the old lake bed for as long as possible to remove phosphorus and iron by settling. (On-going monitoring of Tule Lake should be used to calibrate the desirable retention time more precisely, see below.) Other objectives, such as retaining farm operations versus habitat development for fish and wildlife will affect the details of the restoration of the wetland system. The potential for obtaining grant funds for wildlife habitat restoration argue *for* including this component, but water quality objectives may be met quite well with a multiple use system that retains a large component of agricultural use in the summer.

The dike system used to reclaim this land from flooding is substandard and will eventually be lost, much as large areas of the Mississippi Valley diked floodplain was breached and then abandoned after the 1993 flood. It does not seem to be financial feasible to upgrade the levee system to current standards (L Smythe, personal communication).

Rather than await the uncontrolled failure of the reclamation system at an uncertain point in the future, we recommend a planned approach. Landowners should benefit from a planned transition to a system in which full flood protection is not provided because the planned realization of water quality and habitat benefits can be used as a basis for seeking funds for the purchase of land or acquisition of easements to compensate landowners. Uncontrolled loss of the dike system is unpredictable, and may not result in the most effective restoration of Robinson Lake.

The County should (1) develop a plan for the future utilization of Robinson Lake and (2) seek the necessary funds to implement the plan. The plan should develop engineering design alternatives and evaluate their benefits and costs. The plan or plans that are acceptable to Lake County citizens should be the basis of grant proposals seeking aid in restoration of Robinson Lake.

10.1.2.2 *Small Wetland and Flood Plain Protection*

No other single wetland project has the same quantitative possibilities for water quality improvement as the Robinson Lake project, because of the sizes of the creeks and basin involved. However, many small-scale wetland and flood plain rehabilitation projects are possible along the courses of many streams and at their mouths. Fringing tule marsh in the lake itself also undoubtedly has the effect of tying up some nutrients that would otherwise turn up as algal biomass. In aggregate they may also have some positive effect on water quality, though they are probably more important as fish and wildlife habitat. We do recommend that a 3-for-1 mitigation policy for wetlands be adopted, and that other county policies that favor protection and rehabilitation of all wetlands be maintained and strengthened. A 3-for-1 policy seems a reasonable choice given the possible failure of mitigation projects, and a desire to see a net increase in functional wetland habitat Mitigation measures will frequently not result in habitat as desirable as that removed. Experience may suggest that a larger or smaller ratio better meets County objectives.

10.1.3 Road Erosion Reduction

Goldstein and Tolsdorf (1994) identify erosion caused by road cuts, fills, steep ditches, and poor culverts as an important secondary source of erosion (13% of total sediment yield). Local, State, and Federal agencies responsible for roads in the Clear Lake drainage basin should undertake a comprehensive program of road erosion reduction. Improvements to privately maintained roads also needs to be addressed.

Lake County should seek the cooperation of responsible parties, including its Public Works Department, CALTRANS, US Forest Service and Bureau of Land Management, and private entities to survey the road system and identify sections with potential erosion problems. A policy for new and reconstructed facilities should be developed to ensure Best Management Practices are included in all project designs and construction. These entities should be encouraged to budget funds to improve cuts, fills, and drainage facilities, and to develop sedimentation structures to capture erosion products. A reasonable goal is to reduce sediment yields from roads by 50% over the next ten years. The Planning Department should examine new road construction in private developments to ensure that design and construction practices minimize erosion potential.

10.1.4 Construction, Excavation, and Filling

Goldstein and Tolsdorf (1994) estimate that sediment contributions due to these practices are relatively modest (4% of total sediment yield). Nevertheless, earth moving along the shore of the lake or in the vicinity of streams where it might reach the lake is a cause for concern. In steep areas, soil disturbance some distance from active streams may of course reach the lake. We recommend that the present County grading ordinance and environmental review procedures be used as tools to evaluate and regulate projects which may contribute generate significant sediment loads.

10.1.5 Erosion Protection After Wildfire

Goldstein and Tolsdorf (1994) also consider this to be a minor source on average (4% of sediment yield). However, wildfires vary greatly in size (see Tune Line in Figure 3.7), and subsequent erosion depends substantially on weather and after-fire practices. A sufficiently large fire might well have measurable impacts in the immediately following years. In the event of very large fires, or fires in terrain that is likely to cause large-scale erosion, we recommend that County officials request that California Department of Forestry and the Federal land management agencies take effective measures to control erosion.

10.1.6 Operations of the Cache Creek Dam

The Cache Creek Dam, operated by Yolo County Flood Control and Water Conservation District, stores winter runoff for irrigation in the summer. The current drawdown regime accelerates the loss of phosphorus from the lake because irrigation releases tend to occur in summer when phosphorus levels in the lake are high. Since the Cache Creek Dam is operated in conjunction with Indian Valley Reservoir, there is some flexibility in the drawdown schedule. In negotiations with Yolo County FCWCD, Lake County should ensure that the favorable pattern of mid- to late summer drawdown of Clear Lake continues.

10.1.7 Localized Issues and Controls

The aforementioned recommendations deal primarily with lake-wide scale issues. It was not the goal of this study to deal with each specific localized area along the shoreline of Clear Lake. However, there are other more localized issues that may need to be addressed for specific shoreline problems.

During the recent dear water years waterweed infestations have interfered with recreational use of shoreline and shallow water areas. Small-scale private efforts to dear weeds around docks and so forth are generally quite successful. Contra Costa Landscaping demonstrated the potential of harvester machinery to dear larger areas. The County should encourage such demonstration projects by private concerns as long as waterweeds are a problem.

The Clearlake Keys subdivision's canals, similar enclosed waterbodies, and lee shores such as Horseshoe Bend have repeated problems of scum-forming algae build-up during summer and/or fall seasons, depending on the year. The causes of these blooms may be related to more localized problems associated (1) with sewage system leakage, (2) nutrient loading from lawn fertilization and disposal of nutrient-rich material such as yard debris, (3) the fact that this region is at the end of a long wind fetch which may contribute to the build up of algae and other floating debris, (4) runoff from Highway 20 and other roads, (5) shoreline topography that promotes accumulation and stranding, and (6) other unique aspects of more enclosed waterways (such as in-creased temperatures and poor circulation). Further, the potential remedial recommendations that apply to lake-wide solutions (above) will almost certainly not apply to these localized situations. Skimming and disposal of scums is much more likely to be useful on a small scale in confined channels than as a technique for managing the problem populations themselves on a lakewide basis.

A first step in resolving issues of localized scum-forming blooms will be to implement some of the more sophisticated monitoring plans addressed later within this chapter. If there are found to be elevated levels of other nutrients (e.g. from leaky sewage systems), and if there is a desire to control these localized problems, then appropriate private and/or county control measures should be implemented. The enclosed nature of the channels at sites such as the Clearlake Keys subdivision naturally raises temperatures in these areas and this, together with any elevated nutrients, would tend to exacerbate the scum-forming bloom problem. One obvious solution in this instance would be to reduce the level of exogenous nutrient input (i.e. reduce or eliminate any sewage input; use basins to trap sediment input) to these channels. If wind were found to be a si
cant factor in driving floating debris and specifically algae into the Keys, then an air fence or boom-type device could be installed at the entrance to the Keys to prevent the majority of floating debris from entering this waterway. Short term solutions to localized problems, such as algae and weed harvesting, may be effective solutions to localized problems even if they are not applicable on a lakewide basis.

The county should carefully review any future plans for residential developments built on confined channels, given their history of scum buildup problems. Plans should include an element ensuring good circulation or other practical mitigation measures, and inspection, monitoring, and enforcement provisions should be clear to developers.

Skimming technology is primarily applicable to localized problems. Homeowners associations and similar grass roots organizations should take the primary role in developing solutions to these problems, with the assistance of private entrepreneurial efforts. The Clearlake Keys Property Owners Association has had considerable success controlling the worst effects of blooms in their canals with their spray-boat and weed harvester machinery. Oil skimming technology is highly developed, and might well provide an economical, effective means of collecting and disposing of scums (Steve Ricks, Clean Bay Cooperative, personal communication).

10.2 Establish A Lake Management Agency to Implement Recommendations

We recommend that Lake County establish lead authority for lake management in one County Agency. Lake County has been delegated responsibility for managing the lake's waters by the State of California. It has a legal responsibility for, as well as a huge economic and environmental quality-of-life stake in, the lake environment being as high-quality as possible. From the economic point of view alone, Clear Lake and the tourist industry it generates makes it one of the two most important natural resources in the County, together with agriculture. Since this is a technical study not a management system report, our recommendations focus on performance goals, not organizational details.

We have noted repeatedly in this report how the problems that affect the lake interact. Algal bloom control strategies require attention to the ecology of the whole drainage basin. Algae cannot be managed in isolation from the fishery, wildlife, noxious insects or waterweeds. Nor can water quality issues be divorced from water supply or flood control problems.

The people of Lake County would be best served by a County agency structure in which one agency has clear authority to deal with all major elements of the lake's interacting environmental quality problems. The three agencies that currently have significant management responsibilities related to the lake include the Flood Control and Water Conservation District, Lakebed Management, and the Mosquito Abatement District. Each has by history and by legislation a narrow mandate. Each has a technical staff with expertise relevant to managing the lake. None has clear responsibility for algae control. The lack of clear action authority was noted by Lallatin (1966), and its persistence is an important part of the reason why progress on the problem has been so modest over the last 30 years. The recent reorganization of the Flood Control and Water Conservation District and Lakebed Management Department so that they report to a common administrative head is a step in the right direction. The inclusion of both agencies within the Public Works Department also promises to make better use of the County's technical expertise.

While such an agency needs to take a strong leadership role, it must coordinate its activities with other county agencies, most notably Building and Safety, Planning, Public Health and Air Quality. Numerous State and Federal regulatory and land/water management agencies also have important roles. The Clear Lake Basin Integrated Resource Management Committee currently provides overall coordination of Basin management efforts. This function needs to be served, whatever the exact form the structure takes.

10.3 Adequate Financial Resources

A natural resource like Clear Lake is the analog of a business investment. Using tourism as an example, it is the basic "production facility" of an industry producing tens of millions of dollars of revenue each year. For lack of a solution to the algae-caused water quality problems, it is estimated that at least $7 million per year in tourist business is lost (Goldstein and Tolsdorf, 1994).

Like many natural resources, the scope for private management is limited. The lake is a public resource that must be managed by a public agencies. The current financial situation places severe constraints on local government agencies, especially on support of even the most worthy projects from general funds. The recommendations of this report imply significantly greater investments for management on the argument that such investments will have a high payoff in improved lake quality. No matter how high the expected payoffs, the actual financing of the management effort will be a major challenge. The public debate on actions to improve water quality at Clear Lake need to include a discussion of resources to implement improvements.

A comprehensive set of recommendations on financing is also beyond the scope of this report. On the basis of experience on this project we do recommend attention to the following funding mechanisms:

<u>10.3.1. Grants</u>

The County has had success with grant applications, and the prospects for future success with them is excellent. Much of the specific project work on nonpoint source control appears to fundable from Federal sources like EPA and SCS. The problem will be raising local cost share contributions for an accelerated program. We explored one project with the US Geological Survey on iron geochemistry, but a 50% local cost share match made this project impractical. Since grants have uncertain prospects, narrow purposes, and fixed durations, it is impossible to fund permanent, routine functions entirely with grants.

<u>10.3.2 State Agency In-kind Contributions</u>

Currently, the Department of Water Resources monitoring program is a key element in the management of Clear Lake, although it needs expansion (see below). California Department of Fish and Game undertakes some work in Clear Lake from time to time. The University of California contributions to this project via UCD and Hopland Research and Extension Center are another example. Maintaining and expanding State agency in-kind and possibly cash contributions to environmental management should be a County goal. The difficulty with such funding includes the difficulties of interagency cooperation. Diverging policies and priorities are often difficult to reconcile, even with the best of intentions.

<u>10.3.3 User Fees</u>

Tourists, fishermen, and rimland residents are the most direct beneficiaries of lake environmental quality. Support for Lakebed Management already comes from annual encroachment and permit fees for docks and pilings. There may be additional fees that would be perceived as a fair method of raising funds to support better management of water quality, fisheries, or other environmental quality values.

<u>10.3.4 Special Assessment Districts</u>

For services like algae or waterweed removal, or possibly to support more comprehensive programs, district boundaries might be established on the model of flood protection and sewer districts, and fees assessed in proportion to benefits received. Prior to establishing fees and/or assessments, estimates of the value of environmental quality improvements to the Lake County economy must be refined and mechanisms for funding the basic lake management actions, monitoring and investigations must be developed.

<u>10.3.5 Cooperation with Private Non-profit Groups</u>

Organizations like the Lake County Land Trust, Nature Conservancy, and Ducks Unlimited have programs of land acquisition, purchases of conservation easements, and similar programs aimed at habitat protection. The goals of these groups is generally compatible with the riparian and wetland enhancement projects recommended here. Lake County should discuss common projects with them.

10.4 Recommended Monitoring Program

The current monitoring system, based on Lallatin's plan of 25 years ago, needs to be substantially upgraded to support effective management. The frequency of sampling needs to be increased, reporting needs to be prompt, parameters need to be added, and the stream water quality sampling program begun with this project needs to be continued and enhanced.

10.4.1 Inlake Water Quality Monitoring

The 3 station DWR monitoring program is based on monthly sampling since 1968, as described in detail in **Chapter** 4. The program is excellent in many respects, and has greatly increased our understanding of Clear Lake. It should be improved in the following *ways:*

10.4.1.1 Additional Parameters

Chlorophyll has never been routinely sampled in Clear Lake. This is unfortunate, since this is the commonest measure of algal biomass used. For comparative purposes, it should be added to the routine sampling program. The measurement is simple, requiring much less time and effort than phytoplankton enumerations.

Since iron is frequently the geochemically limiting element, it should be collected at every sampling date, rather that twice per year as currently done.

10.4.1.2 Frequency

A monthly sampling interval is very sparse. Major events can easily occur between sampling dates, as the spiky nature of the data presented in Chapter 4 show. Weekly or biweekly sampling, at least during the periods of rapid change and major blooms (May-October) is necessary to provide a clear understanding of each year's events.

10.4.1.3 Timely Reporting

For some management purposes, such as planning the county dispatch of air and spray boats, timely reporting of current conditions in the lake would be highly desirable. Some data, such as temperature, oxygen, and chlorophyll (using fluorometery) can be derived from direct field measurements. Chemical samples can be stored, and their production is dependent upon laboratory schedules. Phytoplankton samples are easily stored, and there is a temptation to delay examination under the pressure of other scheduling constraints. Reduction and analysis of data is more time consuming. Field-collected data should be in the hands of county staff the day of collection, and copies of chemistry determinations should be sent as soon as they are completed. Phytoplankton enumerations should be simplified by mechanical averaging (pooling of samples from different depths) so that a meaningful picture of populations can be maintained on an almost real-time basis. An important aspect of timely reporting is that it generates a lively interest in ongoing events and stimulates the creative imagination of managers and researchers in a way that long-delayed data cannot.

Lake County should negotiate and collaborate with DWR on modifications of the current monitoring program to meet the objectives listed above.

An annual report should be prepared that summarizes all data on the lake and interprets the results for management purposes. As with data collection, the County agencies (including the Mosquito Abatement District, which collects important data on fish and invertebrates) need to develop a rational system for accomplishing this task, in collaboration with DWR The agencies should sponsor an annual workshop to present and discuss rough drafts of annual report contributions.

10.4.2 Stream Water Quality Monitoring

With the assistance of Hopland Research and Extension Center, the County and UCD initiated a basic program of stream nutrient loading, **as** described in Chapter 5. However, the record which comprises only 2 years (one very dry), is restricted to the larger gauged streams, and covers only phosphorus and suspended solids in any detail. Due to the short record and incomplete coverage of drainages, there is considerable uncertainty especially regarding historical loading. It is important to continue current monitoring to develop basic baseline information against which to judge improvements expected from source control. More specific monitoring programs will be required to evaluate control projects for effectiveness.

10.4.2.1 *Continued Monitoring*

The monitoring program on the three major tributaries as outlined in Chapter 5 should be continued. Although iron measurements can be considered only experimental, they should be continued. Nitrogen measurements (nitrate, ammonia, total nitrogen) should be added to phosphorus measurements to verify Home's (1975) conclusions about the relative importance of stream inflow and in-lake nitrogen fixation on the total nitrogen budget of the system. Inadvertent control of nitrogen loading in an effort to control phosphorus and iron might maintain low ratios of nitrogen to phosphorus, maintaining conditions that favor scum formers.

Since the availability of iron and phosphorus to algae is difficult to estimate from chemical measures, we recommend the additional use of bioassay methodology, as in **Chapter** 7, to evaluate the growth-stimulating potential of stream waters. Samples of winter flood flows should be stored frozen until summer high populations of algae are available for experimentation. The ability of flood flow waters to stimulate algal growth and nitrogen fixation in the subsequent summer period can then be compared to expectations based on chemical analysis. If results of direct bioassays are very different from estimated loads, different priorities for control may be required. For phosphorus, the high rates of loss due to sediment burial are consistent with much of the measured total phosphorus load not actively participating in the biological phosphorus cycle. Some drainages way contribute quite disproportionately to the fraction that does actively cycle.

10.4.2.2 Additional Monitoring

As **noted in Chapter 5,** a major uncertainty in nutrient loading estimates emerges from the fact that the larger creeks tend to drain high-elevation watersheds, with high runoff per unit area. Smaller, low elevation drainages are a disproportionate share of the unguaged area contributing to the lake. We recommend installing gauges on representative smaller watersheds. A significant run of data from these creeks will substantially reduce the uncertainty of loading estimates.

A major uncertainty in the estimation of the effectiveness of the Tule Lake system in removing nutrient load is the difficulty of gauging the hydraulic outflow from the system. We recommend a hydraulic modeling approach to solving the effect of lake stage on discharge out the channel below Tule Lake. Tule Lake also needs to be monitored during low flows and when agricultural drain water is flowing into the lake. These flows are likely to be high in nutrients. Although flows maybe modest, in aggregate they may make a contribution to the nutrient bud-get.

10.4.3 Monitoring of Effectiveness of Action Projects

Remediation projects inevitably have a trial-and-error component. We recommend that a monitoring plan be developed for each project, designed to estimate effectiveness at preventing erosion, storing nutrients, etc. The Soil Conservation Service has developed an array of methods for estimating sediment erosion rates due to various processes and types of watersheds. Monitoring staff should be trained in the use of these methods. Sediment and nutrient concentrations should be measured using sampling designs that demonstrate the success or failure of practices implemented. Control sites should be monitored for comparison with treated sites.

10.4.4 Wastewater Treatment Plant and Storm Drainage Monitoring

Basic water quality monitoring should be conducted on streams draining sewage treatment plant spray fields and on effluent lost during flood events. Storm drainage from urban areas may also be very high in available nutrients and should be monitored. Even though current data support the idea that these are small sources on a lakewide basis, the large volumes of concentrated nutrients handled by wastewater and storm drainage systems makes them a potential hazard, and one the first that occurs to the public as a source of nutrient problem. Good management practice dictates sufficient monitoring to ally public fears about the load generated from these sources, even if the technical uncertainty about their effects were minimal. The monitoring need not be extensive, and could be quite minimal if the expected result that little nutrient loading results from treatment plants and storm drainage is verified. Uncertainties in the present data are very high, given that wastewater stream and urban runoff phosphorus is relatively easy to control if it does prove significant

10.4.5 Nutrient Outflow Monitoring

More frequent nutrient samples should be taken at Cache Creek Dam to estimate accurately the volume of phosphorus leaving the system via the outflow (DWR collects data at the outflow every second month currently). This is necessary to compute accurately not only the total annual outflow rate but also its timing. The current dam operation is hypothesized to favor outflow of nutrients from the lake, and more accurate information on this effect would help to accurately quantify this effect

10.5 Recommended Research and Development Program

Lake County should conduct a program of research and development aimed at promising (but unproved) strategies to control scum-forming algae in Clear Lake.

It is important to recognize that Clear Lake is a unique and still poorly understood system. It is in the county's interest to stimulate as much research and development on the lake as these limiting principles (and budget realities) permit.

10.5.1 Improved Sediment Yield Estimates

The Goldstein and Tolsdorf (1994) estimates of sources of sediment used well-proven field methods. However, the amount of field data that were collected were too small to give a precise estimate of sediment sources. A better estimate of the total lakewide averages is important for overall assessment of the problem. Action projects must be targeted based on reasonably precise estimates of sediment yield at the level of creek sub-watersheds to be efficient. Where, precisely, do the worst problems lie? More extensive field work involving a significant expenditure of effort by a trained hydrogeologist, would be required to provide a reasonably comprehensive picture of erosion patterns in the basin. The best approach would be to use the SCS field methods, together with the recommended monitoring associated with project activities like the Scotts Creek Project, to calibrate the USDA-SCS Agricultural Non-point Source (AGNPS) sediment model. This model, adequately calibrated, should give rather precise estimates of sources of sediment in quite some geographical detail. Calibrating and running the model would require additional time of properly trained staff. The County, the Resource Conservation Districts and SCS should collaborate on a project to implement the AGNPS approach. BLM, the Forest Service, the County Planning Department, and the UCD Center for Ecological Health Research should be requested to provide assistance with modeling and Geographical Information Systems efforts related to this project.

10.5.2 Phosphorus Inactivation By Chemical Treatment

The use of alum to control phosphorus release from sediments is a potentially valuable eutrophication control technique for Clear Lake. It should be investigated as a potential backup strategy for load reduction in case Clear Lake reacts slowly or inadequately to erosion control measures. The expense, complexity, and possible environmental impacts of such a treatment demand a step-by-step approach. Investigation of the possible effectiveness of alum application should begin with simple laboratory microcosm experiments. If these experiments suggest promising results, they should be scaled up using meter-square plastic tubes anchored in confined channels as experimental systems. Ultimately, whole confined channels and then the semi-isolated Lower Arm could serve as steps in the scale-up to a whole-lake treatment. Experiments to establish the effects of alum treatment on bottom organisms and fish would have to accompany the evaluation of effectiveness in preventing phosphorus release. Equipment and applications procedures suitable for establishing an even distribution of aluminum floc on the bottom of a large, turbulent system like Clear Lake is a significant challenge.

Other possible chemical treatments, such as calcium or fluorine treatment, should also be tried in laboratory-scale experiments, and developed further if preliminary results are promising.

Implementation of this recommendation may be to develop microcosm systems as part of the Superfund supported program evaluating the hazards posed by mercury contaminated sediments, as the same technology could be used for both types of experiments.

10.5.3 Iron Geochemistry

Investigations of iron geochemistry must start at a rather basic level. There is no immediate prospect of practical control strategies emerging from such research. Still, the poor understanding of how iron enters and cycles in Clear Lake, and how it regulates the nitrogen and phosphorus cycles, is a major background uncertainty. The County should offer such assistance as is possible to University and Geological Survey researchers, who have expressed interest in these questions. Local offers of in-kind assistance and endorsement of the importance of the problem to funding agencies will be helpful in attracting a program on iron.

10.5.4 Phosphorus Availability

The largest loss term in the phosphorus budget is burial in sediments, and the largest gain is particulate phosphorus in streams. It is important for some control strategies to better understand if there are any biological inert fractions of phosphorus that are directly buried in sediments. For example, we note in **Chapter 9** that modifications of the operation of Cache Creek Dam might be used to accelerate losses of phosphorus during the summer. However, since the outlet loss term is a relatively small part of the total loss as we currently understand it, a strong case for a moderate increase in the outflow loss term cannot be made. Similarly, erosion control measures would ideally be directed at the sources of sediment yielding the most available phosphorus fractions.

The simplest approach is to apply the same phosphorus fractionation methods described in **Chapter 6**. If different watersheds are delivering substantially different amounts of base extractable iron/aluminum phosphorus, it would be possible to refine the targeting of stream reaches for remedial actions. Differences in the soils and geology of particular watersheds may affect the portion of the total phosphorus the is in this highly variable pool.

Other factors may affect remediation strategies. For example, phosphorus availability is undoubtedly a function of grain size, with coarser, less weathered grains having less available forms of phosphorus. Grain size affects the mobility and settling velocities of particles. Thus erosion control structures and sediment trapping facilities will tend to trap particles of some sizes more efficiently than others. Initial experiments should focus on size-fractionating high-flow suspended sediments, and subjecting size fractions to chemical fractionation analysis and comparing base extractable to total phosphorus. Size fractionated sediments should also be investigated using bioassay responses using methods suggested for monitoring in **Sectio**n 10.4.2.1.

If initial results suggest a large unavailable component in the particulate fraction, a more extensive program of research in phosphorus recycling should be conducted. Microcosm experiments to verify the resistance of certain fractions of the load to mobilization under more realistic conditions than simple extraction would be useful. Analysis of sediment cores for phosphorus availability as a function of depth might clarify whether or not active recycling pool is confined to a relatively small fraction of phosphorus near the sediment surface. More sophisticated experiments might be required, depending upon the corn--plexity of processes in the sediment column. UCD researchers are currently planning experiments to study iron and mercury biogeochemistry in Clear Lake sediments, and phosphorus cycling investigations could be added easily.

10.5.5 Study of Sewage and Septic Leachfield Leakage

Studies should be designed to estimate the effects sewage leakage and septic leachfield as lake-wide phosphorus and iron sources. The initial investigations should be low cost pilot programs designed to set an upper bound on these contributions. If upper bounds are as low or lower than those assumed in this report, intensive studies are not warranted.

Special Districts and other providers of sewage services should conduct or commission a sufficiently detailed study to estimate phosphorus loading from sewage leakage into confined channels to compliment the elementary Questa studies of nitrogen emission. The Clearlake Oaks Water Company, especially, should conduct such an investigation at the Clearlake Keys subdivision. This study should be paired with another to estimate the extent to which scum-forming algae are driven into the Keys by wind. Such evaluations are necessary to estimate the extent to which confined channels will benefit from overall lake quality improvements, and the extent to which special measures are required for them (see above under recommendations). In our study, it was not possible to devote attention to the complex problems of confined channels.

10.5.6 Algae Skimming

Skimming technology is primarily applicable to localmed problems and, therefore, should be mainly developed with entrepreneurial private investment. Skimming technology for collecting spilled oil is a well developed technology. The County expedite demonstration projects aimed at establishing the feasibility of skimming for local purposes insofar as this is practical given very limited financial resources.

We consider the hypothesis that skimming is a practical tool for lakewide control of blooms or nutrients highly speculative. The first step in evaluating the hypothesis should, however, be taken. Using the recently acquired UCD field fluorometer, it is a relatively simple matter to estimate what fraction of the blue-green population is concentrated in surface scums that are susceptible to harvest. Using this data, a first-cut estimate of the practicality of lake-wide skimming is possible. The skeptical opinion of most technical observers regarding lake-wide skimming will probably be confirmed by this experiment, but the level of public controversy generated by large scale skimming proposals over the years will only be resolved by a fair demonstration of its feasibility or infeasibility.

10.5.6 Economic Assessments

Goldstein and Tolsdorf (1994), using survey data from Hinton (1972), could calculate only a very conservative lower bound on the estimate of economic losses due to impaired water quality at Clear Lake. A reasonably accurate estimate of the benefits to improved water quality and the costs of treatment measures is important background information for all other activities which require expenditures of public funds. For example, grant proposals are likely to have bet-ter success if accurate benefit-cost estimates are included. Benefits should exceed costs, but in the absence of an accurate estimate of benefits from improved water quality it is difficult to estimate whether quite expensive projects should be contemplated (the conservative estimate of a $7 million per year certainly justifies expenditures of a relatively modest scope). UCD and USDA Soil Conservation Service hope to collaborate on an extension of the Goldstein analysis. County agencies should assist with this study and ensure that it is designed to answer the questions that will be most useful for policy and project decisions.

10.6 Summary

Lake County in cooperation with individuals, voluntary organizations, local, state and federal agencies can implement practical actions and policies, primarily for erosion control, that will substantially improve lake water quality. Continued monitoring and research will ensure success and may develop other cost effective strategies.

The heavy load of sediment carried by tributary streams can be cut significantly by erosion manage-

tions are working as planned and that unplanned natural and human changes are detected and dealt with.

Some of the most basic processes in Clear Lake, especially the dynamics of phosphorus and iron, are poorly understood. A better knowledge of these processes would reduce the risks associated with remedial actions. Some remedial actions that might prove very effective, such as chemical treatment of sediments, require considerable development work before they can be recommended as safe and effective treatment strategies. Therefore we recommend that the County support a program of research on the most important processes and promising treatment alternatives.

The preceding recommendations are based on the idea of adaptive management–learning *by doing*. They derive from our best current understanding of how the lake works, but we never understand perfectly. All real-world decisions involve risk and uncertainty. We can't do nothing just because knowledge is not

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A

Responses from Public Hearing on 6/14/94 on the Draft Final Report entitled "The Causes And Control of Algal Blooms In Clear Lake: Clean Lakes Diagnostic/Feasibility Study for Clear Lake, California"

The Public Hearing commenced approximately 7:00pm. Sue Arterburn (Director: Flood Control) provided introductions for Dr. Peter Richerson (Project Director), Dr. Tom Suchanek (Project Co-Director), Mr. Stephen Why (Field Scientist) and Mr. Tom Smythe (Water Resource Engineer). Dr. Richerson then provided a summary of the study, emphasizing the Alternative Methods available for the control of blue-green blooms and a set of Recommended Strategies to deal with this problem. Questions were then received from the audience. The following dialog represents the question/response session that followed Dr. Richerson's presentation.

Participants who signed the attendance sheet

Bob & Jane Butler	Brad Lamphere	Cat-En Woodmansee
Donna Lowdermilk	Dorothy Riggle	Frances B. Huffman
George Riggle	Glenn Dishman	Jerry McQueen
Karan Mackey	Lon Sonner (sp?)	Louise Talley
Mary Ann McQueen	Mort Joel (sp?)	Roger Phelps
Rose & Al Coldenbroff (sp?)		

Q = Question, R = Response

Q: What is the source of phosphorus in the lake that causes the blue-green blooms? This question was stimulated by the use of the word 'fertilization" during the presentation to refer to the process by which phosphorus acts as a fertilizer to stimulate blue-green growth.

R We did not mean to give the impression that "fertilizers" (as applied to agricultural crops) are the primary source of the phosphorus causing the algal blooms, nor a significant problem in this regard. We should modify the Executive Summary to be sure the term "fertilization" is used more properly and dearly.

Q: A question was raised in reference to one potential treatment option, "sediment removal". The question focussed on the concept that sediment removal might create more movement of mercury (bound to these sediments) around the lake and that this would be an undesirable effect.

R It was agreed that movement of massive quantities of sediment, especially if performed near the Sulphur Bank Mercury Mine, would have this effect Sediment removal, however, is not a likely alternative because of the extreme logistical difficulties associated with it.

Q: If the lake's water clarity is improved, wouldn't there be an increase in "water weeds", which would then reduce the 'flow" of the lake, or maybe increase sedimentation which would then "choke" the lake and reduce flow?

R While it is true that an improvement in the water clarity of Clear Lake would likely increase the abundance of shallow water weeds, there is no expectation that this would significantly reduce the natural flow pattern of the lake. Some alteration of the flora of the lake would be expected, but for most people, the switch from slimy green algae over most of the lake surface to dearer water with some additional weeds along the shoreline is probably the more desirable alternative.

Q: Does the Aggregate Gravel Management Plan comply with the stream rehabilitation recommendations? R Yes.

Q: Is there any opportunity for algae to be used as a food source for animals or humans?

R: Yes, but...for two main reasons, this is likely not very feasible. (1) Current technologies are not sophisticated enough to handle the logistics involved in the collection and transport of the algae, and (2) at present it is likely not to be cost effective. In addition, there is still some question about the potential influence of natural toxins that might be associated with thick, concentrated algae.

Q: Are there any costing comparisons between alum treatments and other methods?

R: Currently there has not been a detailed economic analysis of these alternatives, but that is an excellent

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idea that should be followed up on. Rough calculations are provided in Chapter 9.

Q: What is the toxicity of blue greens?
R: The toxicity of Clear Lake bluegreens was tested by California Department of Health Services. They are somewhat toxic.

Q: Are faulty sewage systems a problem in creating blue-green blooms?
R Analysis of the DWR 24 year database, plus our own studies indicate that on a lake-wide basis, sewage systems do not represent any significant problem. Rather, our analyses indicate that phosphorus, and likely iron, from erosion products are the most significant contributors to lake-wide blue-green blooms. However, in enclosed regions, such as the Keys, leaky sewage systems may contribute significantly to localized algal blooms.

Q: If alum were used as a treatment alternative, would it be removed in water treatment plants before people drink the water from those sources?
R From a human health perspective, alum is very safe. It is even purposely added to some drinking supplies for clarification purposes.

Q: When U.S.G.S. took their deep cores, what did they find with respect to historical and pre-historical levels of phosphorus?
R: Uncertain whether phosphorus was measured on these cores but even if it was, it is probably not interpretable in terms of the types of phosphorus that is important for the growth of blue-greens.

Q: A request was made to explain more thoroughly the concept of phosphorus "cycling", and whether this meant it was being dissolved into the water column or taken up biologically and incorporated into biota.
R Mostly it dissolves and becomes incorporated into phytoplankton.

Q: What about involving students in a monitoring project?
R A good idea, but mostly, the level of sophistication needed to conduct the types of routine, fairly technical measurements would not warrant the time it would take to train and calibrate students' performances sufficiently well and would not make it cost effective.

Q: If we had improved water quality, might we expect to see the appearance of Hydrilla, a water weed that can grow to 17 feet deep?
R It is possible that *Hydrilla* might grow in Clear Lake as it is a state-wide threat.

Q: Please clarify the concept of competition between blue-greens and other algae in the lake.
R: The analysis of the DWR database indicated that normal stream flow years likely contributed significant sedimentary material rich in phosphorus (and likely iron) which contributed to heavy blue-green growth at the expense of other types of algae (like diatoms and green algae). The recent drought years (1986-1992) provided little input of these elements and it is likely that because scum-forming blue-greens were not able to obtain the necessary iron they needed, they declined in abundance which allowed other non-scum-forming types of algae to predominate; hence, the increased water clarity.

***Comment:** The solution must start by controlling erosion.*
R: Yes, we believe that this is one of the strongest recommendations we can make to reduce the nutrient input to the lake. With such erosion/nutrient source reductions, the blue-green blooms should be much less serious, as before 1925.

Comment: A comment was made questioning the validity of the $7.0M loss of revenue because of the blue-green algal bloom problem.
R: This is a minimal estimate based on how much extra time current recreators would spend at a cleaner Clear Lake.

Q: What is the brown colored algae that has been present in the past few years?
R: There have been several: last year *Gleotrichia* and this year *Anabena* and *Gleotrichia.*