REPORT ON BRIDGEPORT RESERVOIR BENEFICIAL USE IMPAIRMENT: LIMNOLOGY IN THE SUMMER-FALL 2000 AND COMPARISONS WITH 1989

Report prepared for the Lahontan Regional Water Quality Control Board,
South Lake Tahoe

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1.0 SUMMARY

Bridgeport Reservoir (volume = 40,500 af; area = 10.6 km^2; z_{max} = 13.3 m; z_{mean} = 4.6 m) is located at an altitude of about 6,460 feet on the eastern slope of the Sierra Nevada, northeast of the town of Bridgeport, Mono County, California. It drains a large basin (area = 930 km^2) relative to its size. Water quality in the reservoir is poor; high nutrient levels stimulate eutrophic conditions that are characterized by large blue-algae blooms, depletion of dissolved oxygen, fish kills, and malodors. These conditions have lead to the violation of at least ten basin plan standards set by the Lahontan Regional Water Quality Control Board (RWQCB) on an on-going basis. Thus, the reservoir is listed under section 303(d) of the federal Clean Water act.

A comprehensive study of the reservoir's limnology was carried out in summer-fall 2000 and compared with a smaller study conducted in 1989. The specific purpose of the 2000 study was to understand the physical, chemical and biological processes in the reservoir in relation to impairment of beneficial uses. The information will be used as part of the Total Maximum Daily Load (TMDL) process to determine numerical and/or narrative water quality standards to attain designated beneficial uses.

There were no changes in Bridgeport Reservoir's trophic state between 1989 and 2000 as determined by mean or maximum algal chlorophyll *a* levels, water clarity, or nutrients. The reservoir was very eutrophic in both years, but the physical and chemical lake mechanisms that typically induce such eutrophication differed. The findings suggests that eutrophication in the reservoir is promoted by robust mechanisms and will not easily be reversed. The only outlet from the reservoir's dam is at the bottom, which depletes deep, cold water, promoting lake mixing and consequent warmer temperatures on the sediments. Warmer sediments recycle nutrients more rapidly back to the algae on the lake surface. In 1989 the reservoir was dominated by very dense blue-green algae (cyanobacteria) blooms of *Aphanizomenon* (chlorophyll *a* up to 1,900 µg/L), low water clarity (<1 m Secchi depth, mean 1.2 m), and occasional reduced dissolved oxygen (3.8 mg/L). Maximum nutrients concentrations, especially ammonia (300-500 µg/L), total phosphorus (TP, 130-820 µg/L), and soluble phosphorus (>85 µg/L) were also very high. Maximum nutrients concentrations in 2000 were similar to the situation over a decade ago, including ammonia (>600 µg/L), TP (>500 µg/L), and soluble phosphorus (>85 µg/L). However, in 2000 dissolved oxygen (DO) depletion in the hypolimnion (bottom water) was more severe (<1 mg/L) and more persistent (5 weeks) than in 1989. In addition, in 2000 the ball-shaped blue-green alga *Gleotrichia* replaced the large flake-shaped blue-green alga *Aphanizomenon* as the dominant phytoplankton. Blue-green algae blooms (peak chl. *a* >800 µg/L) in 2000 were very dense, but about half that found in 1989, although the mean chlorophyll level was similar in both years (200-300 µg/L). Water clarity was slightly greater in 2000 (1.5 versus 1.2 m), but may only reflect the lower volume of runoff in 2000. For comparison, natural lakes in the region would be expected to contain clear water with few algae or nutrients (e.g. chl *a* <5 ug/L, Secchi depth ~ 8 m, ammonia <10 ug/L, TP <5 ug/L, and DO >8 mg/L).

Physical processes, especially the degree of vertical water mixing are a critical factor in the beneficial use impairment in Bridgeport Reservoir. In 2000 the water column showed a weak thermal stratification with lighter, warmer water (epilimnion) floating over a cooler denser layer (hypolimnion). This dimictic stratification in 2000 resulted in a period of hypolimnetic anoxia (zero dissolved oxygen, DO) in the 9-12 m bottom water layer. The anoxia lasted for about five weeks. In the dryer year of 1989 the reservoir was less deep and wind was able to stir shallower water for much of the summer with only short periods of thermal stratification followed by destratification, The polymictic circulation pattern in 1989 did not allow total depletion of DO in the temporary hypolimnion at 3-10 m. Hypolimnetic anoxia has two important water quality implications; fish kills and the release of undesirable or toxic forms of nutrients sequestered in the reservoir's sediments. Although stirring transfers needed oxygen to the sediments, it also mixes warm water along with oxygen. Warm sediments are undesirable since the increased temperature promotes nutrient recycling.

In the sunlit upper water of the reservoir's photic zone, high nutrient concentrations (i.e. ammonia and TP) in 2000 coincided with the period of thermal stratification and concomitant hypolimnetic anoxia. Laboratory measurements of the release of nutrients from isolated deep water sediment cores in the indicated negligible releases of soluble phosphate under oxygenated conditions, but an increase of over an order of magnitude to 1.4 $mg/m^2/d$ when anoxia was induced. Thus, field and laboratory results indicate that anoxia in the hypolimnion induced the regeneration of nutrients from the reservoir sediments during extended stratification. The field results show that nutrients released from the deep sediments were distributed vertically to the entire water column and thus were available to support the growth of phytoplankton in the photic zone. Thus, Bridgeport Reservoir has an unusual situation in that thermal stratification was sometimes strong enough to promote increased sediment release of nutrients, but weak or intermittent enough to allow nutrients to leak back into the upper waters. The net result is the very eutrophic state observed in Bridgeport Reservoir. Some support for an unstable hypolimnion was given by a diurnal study in 2000 that showed unexpected daily fluctuations in hypolimnion DO.

Blue-green algae dominated the plankton in Bridgeport reservoir. Four common large colonial planktonic forms (*Aphanizomenon, Anabaena, Microcystis, Gleotrichia*) were present. *Gleotrichia* was by far the most common form and a peak of 1,750 colonies/L was found in August compared with peaks of 350 colonies/L (*Anabaena*) and 196 colonies/L (*Aphanizomenon*). *Gleotrichia* was found for the first time in the reservoir, possibly indicating changes in nutrient loading since 1989, or changed conditions in the reservoir following the draining of the reservoir in the 1980s or the 1990's El Niño floods. As in 1989, a very large seasonal peak in blue-green algae abundance occurred in mid-summer. Phytoplankton were less abundant in the shallow, southern end of reservoir where there are stands of rooted submerged macrophytes. A recently developed sustainable lake restoration technique called biomanipulation encourages competition between blue-green algae and submerged plants. Such competition may be occurring naturally, if haphazardly, in Bridgeport Reservoir. As in 1989, heterocyst frequencies in the blue-green algae indicated low rates of nitrogen fixation and supported the chemical measurements of ample supplies of nitrogen to support the growth of even the very high density of phytoplankton. Buoyancy measurements showed that the common colonial blue-green algae possessed very rapid rising rates of 0.07-1.1 cm/s, sufficient to move from the sediment to the surface overnight. Thus all surface scums were produced by the algae floating to the surface, not by unusually high growth rates.

Synoptic measurements made simultaneously over the entire reservoir in 2000 indicated strong spatial variability in most biological and chemical parameters, but more monotonous distribution of physical variables, perhaps because the reservoir was open to strong wind action. Blue-green algae biomass as shown by surface chlorophyll distribution was also patchy with a major surface concentration of over 250 µg/L near the neck of the reservoir compared with an average of 76 µg/L for the entire reservoir on that day. *Gleotrichia* and *Anabaena* colonies closely mirrored the distribution of chlorophyll *a*. Surface ammonia was highest in the region over deep water indicating an efficient vertical transport mechanism. Surface temperatures ranging from 19.2 to 20.2 ^{o}C over most of the reservoir showed that temperature and surface drift did not correlate with the blue-green algal concentrations. Water transparency showed small variations (1.4 to 1.9 m) correlated with algal abundance. In contrast, surface dissolved oxygen varied considerably over the reservoir with lower values in the northern, deep end of the reservoir (5.9 to 7.7 mg/L) compared with the rest of the reservoir (8.1 to 8.8 mg/L).

Preliminary assessment of nutrient and pathogen loadings in tributaries to Bridgeport Reservoir in April-June 2000 showed substantial increases downstream of pasture in Bridgeport Valley. These loadings indicate that stock grazing could be responsible for external (i.e. originating in the watershed) nutrient inputs to Bridgeport Reservoir, similar to the conclusion drawn in 1989. Primarily the practice of flood-irrigation, but also allowing stock to wade in the tributary streams and reservoir edges, would be expected to exacerbate the transport of nutrients and fecal pathogens from cattle to reservoir. Estimated contributions to nutrient loading by bird populations (~5 tons) would be small compared to those due to the very much larger biomass of terrestrial livestock (~5,000 tons). The contribution of septic tanks in the region was apparently small since the large nutrient increases occurred only below irrigated pasture, not below septic tank systems, such as those at Twin Lakes. The contribution of the percolation basins of the town of Bridgeport cannot be determined from this study, but these flows do not contribute to the nutrients in the inflowing surface streams. Increases in concentrations of nitrogen species were small compared to those found for phosphorus species, based on the ratio of nitrogen to phosphorus typical of feces and urine. It is likely that denitrification is occurring in the flooded pastures, a conclusion supported by previous soil core analyses.

The second part of this study will analyze details needed for the solution of the Bridgeport Reservoir problem. They include GIS and nutrient measurements to allocate future nutrient loads and determine the size of passive and active restoration methods such as oxygenation, nutrient-removal wetlands, or an off-line reservoir.

2.0. RECOMMENDATIONS

Bridgeport Reservoir fails by a considerable margin to meet its designated beneficial uses. The cause appears to be the result of over 150 years of modification to its watershed to support livestock grazing, riparian zone removal, and consequent nutrient and pathogen loading to the reservoir. As population pressure increases, so does the pressure on land and water resources. Historically these pressures are countered by optimization of use, restrictions on operation, and use of mechanical solutions. There are several solutions to reduce excess eutrophication at Bridgeport Reservoir. Any solutions, however, must be considered in conjunction with the Total Maximum Daily Load (TMDL) process. Reductions in external nutrient loads from livestock grazing in the watershed, either through reducing the number of livestock or the implementation of Best Management Practices (BMPs), is the first obvious, theoretical solution to restoring beneficial uses at Bridgeport Reservoir. However, this approach presents several implementation challenges, and may not reduce pollutant loads sufficiently. Even if it is effective at reducing nutrient loads, it may not result in improved water quality for years or decades, due to the amount of internal nutrient regeneration from the reservoir sediments. Another solution to reducing external nutrient loads would be to install a nutrient-filtering wetlands between the pastures and the reservoir. If effective, this would probably allow at least some continued cattle ranching above the constructed wetland.

Another direct and long-term solution would be to relocate the reservoir off-stream, with water storage capacity similar to the current reservoir, but with spring filling and late summer releases only. Properly designed, such an off-line water storage system would be less eutrophic and easier to manage than the current system. This solution would prevent summer nutrient inputs (i.e. during the grazing season) from entering standing waters. A similar solution would be to convert the reservoir on the same site to an off-line system using a bypass.

A balance of external pollutant source control and active lake management is probably the only way to achieve the restoration of beneficial uses in both the short- and long-term, and in a sustainable fashion. For example, a hypolimnetic pure oxygen addition system installed in Bridgeport Reservoir would provide immediate and long-term improvements to water quality. It would ensure that the hypolimnion remains oxygenated throughout the summer, preventing fish kills and internal nutrient regeneration. In combination with measures to reduce external nutrient loads, over time the amount of artificial oxygenation required could be reduced, or even eliminated, such that this approach may provide immediate results and be sustainable over the long-term. Biomanipulation, where submerged aquatic vegetation and the zooplankter *Daphnia*, and the planktivore:piscivore fish ratio are managed to produce clear water holds promise for restoring shallow portions of the Bridgeport Reservoir. For the first few years, biomanipulation would require additional active management, such as oxygenation, to address internal nutrient regeneration in the deep portions of the reservoir.

Bridgeport Reservoir is small and is one of only two obstructions that now block the historical migration of Walker Lake cutthroat trout. Therefore, moving the reservoir off-line, combined with a constructed wetland to reduce cattle/horse waste effects, is the best and most sustainable long-term ecological solution to the current problems at Bridgeport Reservoir. With this optimal solution, cattle ranching could continue at a reasonable rate, river fish would not be threatened by anoxia and other health threats, and lake fishing opportunities would be preserved at the new off-line reservoir. Each part of this proposed optimal solution has been successfully carried out in California, but the parts have not been combined as suggested for Bridgeport Reservoir. These options are still possible at Bridgeport Reservoir. However, some options "outside the box" should also be considered. The entire range of conventional, new ecological engineering, and unconventional options are presented as recommendations below. It is hoped that these recommendations will be considered in terms of the long-term management of the entire watershed including the irrigated stock grazed lands, the open range, the wilder areas, the Walker River, and terminal Walker Lake, as well as Bridgeport Reservoir. The recommendations are based on the work in the years 1989 and 2000, and do not include information from the 2001 research which is expected to either reinforce or possibly modify some of the conclusion presented in this report. The year 2000 recommendations are:

A. With the current stock operations upstream and current reservoir location and hydraulic operation

- Construct a series of specifically designed nutrient/pathogen removal wetlands between the irrigated stock grazing fields and the reservoir. The TMDL targets for nutrients and pathogens would set the size of these wetlands which, if used alone, would have to meet specific nutrient and pathogen outflow targets (see next report).
- Re-designate the beneficial uses in the reservoir to less stringent target concentrations for nutrients, pathogens, algae, and dissolved oxygen. The new beneficial use targets would balance the current reservoir's benefit to the trophy trout population below the reservoir with the potential harm to fish caused by the lower water quality standards.
- Enact biomanipulation by increasing the piscivore:planktivore ratio and removing carp.
- Monitor to evaluate progress towards TMDL compliance

B. **With current stock operation upstream and reservoir location but modified reservoir operations**

- Install a hypolimnetic oxygenation system in the deepest part of the reservoir to meet dissolved oxygen standards and prevent nutrient and toxicant emissions from the anoxic sediments. This option would reduce ammonia, phosphorus and algae.
- Operate reservoir with a maximum depth of 5 m at the dam between June and August (may eliminate needed for oxygenation system). This option would substantially reduce water storage capacity and boating and fishing opportunities in the reservoir.
- Construct a more limited nutrient and pathogen removal wetland series between stock grazing and the reservoir (assumes oxygenation system is operated).
- Enact a biomanipulation as in #A.

C. With current stock operations and hydraulic operation but with new dam location

- Move reservoir to an off-line location with better water quality and quantity management possibilities.
- Regrade current reservoir and construct a bypass for most flow, especially in spring and summer.
- Construct nutrient and pathogen removal wetlands in the old upper reservoir bed.

D. **With current reservoir hydraulic operations and location but altered upstream stock operations**

- Remove or drastically reduce upstream irrigated stock grazing operations.
- Enact a biomanipulation as in #A.

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3.0. INTRODUCTION

3.1. Clean Water Act and Total Maximum Daily Loads (TMDLs).

The national plan to clean up lakes, streams, and estuaries is based on the Clean Water Act of 1972 and the various amendments made since it was first enacted. Initial implementation of the act focused on point sources of domestic and industrial wastes, the most obvious pollutant sources. As pollutant contributions from these source were reduced, the contribution of smaller, diffuse sources became proportionately more important. The culmination of the Clean Water Act is the regulation of all pollutant sources, such that beneficial uses of receiving water bodies can be restored and maintained. Regulation of all pollutant sources, including point, non-point, and natural background sources, is achieved through the development of a total maximum daily load (TMDL). The TMDL is an allowable pollutant load under which the beneficial uses of a water body can be maintained. The TMDL also includes a margin of safety to account for any uncertainty in the response of a particular water body to pollutant loading. The TMDL is formalized as follows:

TMDL = point sources + nonpoint sources + natural background + margin of safety

While point source discharges of wastewater can often be treated using end-of-pipe technologies, diffuse sources are mostly to be controlled by the implementation of Best Management Practices (BMPs). To date, the use of BMPs alone has generally not been effective to reduce non-point pollution. In part the disappointing performance of most BMPs can be attributed to a lack of regulatory authority to require their use and poor performance where they are used. However, there are many cases where there is no good BMP for the pollutant. The prime examples are storm flows in winter when temperature limits biological waste processing and volume overwhelms physical processes such as sedimentation. Removal of recalcitrant pollutants such as DDE or heavy metals also lacks reliable, proven BMPs, even in summer conditions. Thus there is a real need to improve the technical performance of current BMPs and to invent new ones.

When designated beneficial uses of a water body are impaired, the Clean Water Act requires that a water body be listed under Section 303(d) of the federal Clean Water Act as impaired. A TMDL must then be developed, including a quantification of the non-point source contribution of pollutants. The pollution contribution of each watershed use group is determined and reductions, if any, are then mandated such that the beneficial use will be restored. Thus the questions both of whether Bridgeport Reservoir has impaired beneficial uses as well as what is causing these impairments are of concern to regulatory authorities. In addition, to restore impaired beneficial uses and minimize uncertainty in the TMDL, it is vital to understand how the system functions.

Data gathered over the past decade indicate that the beneficial uses of Bridgeport Reservoir are impaired by poor water quality (Entrix Inc., 1990 and 1998; LRWQCB, 1999). Specifically the reservoir suffers from excess eutrophication (i.e. high primary productivity by algae), indicated by an over-abundance of blue-green algae (cyanobacteria), and low dissolved oxygen in deep water. Preliminary studies suggest the likely cause of the eutrophication is nutrient loading from the watershed, since the high Sierra source water should be nutrient-depleted. Possible sources of nutrients are human wastewater disposal, agriculture, atmospheric deposition, logging or clearing, and other land disturbance. In summer, nutrient recycling from the reservoir sediments (internal loading) enhances the degree of eutrophication. The following section was prepared by the Lahontan Regional Water Quality Control Board to describe the official position of the regulatory authority at that time.

3.2. DRAFT PROBLEM STATEMENT FOR BRIDGEPORT RESERVOIR TMDLS FOR NUTRIENTS.

(From a memo by T.Suk, Lahontan Regional Water Quality Control Board, 6/14/00)

Bridgeport Reservoir is located in the East Walker River Hydrologic Unit (HU # 630.300), on the East Walker River, immediately east of the town of Bridgeport, California (Mono County). The East Walker River watershed has been designated as a "Category 1" watershed by the California Unified Watershed Assessment. The reservoir is listed under Section 303(d) of the federal Clean Water Act as impaired due to nutrients.

Bridgeport Reservoir is highly eutrophic, with an enriched nutrient status that results in massive algae blooms and frequent adverse impacts to beneficial uses. Thick growths of algae interfere with recreational uses (e.g., swimming, boating, fishing), and the appearance of, and odors emitted by, decaying plant matter impair aesthetic uses of the water body. The taste of fish flesh can be compromised during algae blooms, and fish kills have occurred both within the reservoir and downstream. In addition to the documented nuisance conditions and fish kills, there exists a potential for the recurrent massive blooms of blue-green algae to produce compounds that can be toxic to other aquatic and terrestrial organisms, including food-chain organisms, wildlife, livestock, and humans.

The eutrophic conditions in the reservoir result in ongoing violations of at least ten water quality objectives contained in the Water Quality Control Plan for the Lahontan Region (Basin Plan), including: ammonia, biostimulatory substances, dissolved oxygen, pH, taste and odor, temperature, total dissolved solids, total nitrogen, total phosphorus, and turbidity.

The TMDLs will focus on achieving attainment of the applicable objectives for nutrients, as listed in Table 1. Any TMDL strategy that successfully controls the nutrient parameters listed in Table 1 should also result in improved conditions for the other six 'non-nutrient' objectives currently being violated.

It is important to note that site-specific objectives have never been developed for Bridgeport Reservoir. Therefore, the applicable objectives listed in Table 1 are a combination of: (1) the Basin Plan's region-wide objectives applicable to all surface waters of the Lahontan Region, and (2) site-specific objectives adopted for the East

Walker River, which feeds and drains the reservoir. Some of the currently applicable objectives may not be appropriate for the shallow, still water (lentic) conditions of Bridgeport Reservoir. Some of the applicable objectives may be too restrictive, while some may not be sufficiently protective. The effort to develop TMDLs for the Bridgeport Reservoir should include an evaluation of the adequacy of the current objectives, and a consideration of development of appropriate site-specific objectives for the reservoir and its inlets and outlet.

Table 1. Water quality standards for nutrients (Source, Lahontan Regional Water Quality Control Board, June, 2000).

Water quality parameter	**Water quality objective(s)** from: *Water Quality Control Plan for the Lahontan Region*, l995 ("Basin Plan")
Ammonia	Detailed equations for calculating allowable ammonia concentrations (based on temperature and pH) are located in the Basin Plan, beginning at page 3-3
Biostimulatory substances	Waters shall not contain biostimulatory substances in concentrations that promote aquatic growths to the extent that such growths cause nuisance or adversely affect the water for beneficial uses.
Total nitrogen (TN)	0.50 mg/L (annual average value) 0.80 mg/L (90th percentile value)
Total phosphorus (TP)	0.06 mg/L (annual average value) 0.10 mg/L (90th percentile value)

3.3. BRIDGEPORT RESERVOIR SETTING, OPERATION, BATHYMETRY, AND LIMNOLOGICAL SAMPLING STATIONS.

Setting. Bridgeport Reservoir is located on the East Fork of the Walker River about two miles northeast of the town of Bridgeport, California (Figs. 1 and 2). The East Walker River and its tributaries originate on the eastern slope of the Sierra Nevada. From the reservoir outlet the East Walker River flows about seven miles to the California-Nevada border and then eventually to Walker Lake in Nevada. The Bridgeport Reservoir watershed, with a drainage area of 930 km^2, ranges in elevation from 1950 – 3500 m amsl (Elkins, 2002). Terrain is diverse, including gently sloping pasture (Bridgeport Valley) just upstream of the reservoir, to forested and shrubby hills, to granite peaks at the top of the watershed. Bridgeport Valley is located on an alluvial fan composed of Tertiary and Quaternary volcanic rock, Pleistocene glacial deposits, and recent alluvium (gravelly sandy loam).

The Bridgeport Reservoir watershed has a semi-arid mountain climate. Summers are dry and warm, while winters are cold, with the bulk of annual precipitation falling as snow. Accordingly, peak stream flows occur during spring snowmelt runoff during May and early June. The mean annual precipitation near the crest of the watershed is 65 inches, while it is only 10 inches in Bridgeport Valley (Elkins, 2002).

The hydrology of East Walker River watershed is complex due to an extensive and complex network of diversions constructed to facilitate flood irrigation on pasture land in Bridgeport Valley. Five creeks drain the upper watershed, merging in Bridgeport Valley to form three creeks (Buckey Creek, Robinson Creek, East Walker River) that empty into Bridgeport Reservoir. Water is diverted via irrigation ditches from Buckeye Creek and Robinson Creek just upstream of the Valley towards East Walker River on the east side of the Valley. The diversions are not metered. The Valley is flood irrigated from spring through fall, resulting in saturated pockets of soil which remain for months at a time. Since water is diverted in an unknown manner between the various streams draining into the lake, it is very difficult to assign nutrient loadings to any one area.

The East Walker River watershed is a mixed-use basin. Livestock grazing, primarily by cattle, dominates watershed land uses from May through October. Photographs dating back to the 1870s record livestock in the Valley at present day numbers (approximately 10,000). There are some reports of even higher stock densities during the height of the mining period. Grazing now occurs in the hills above Bridgeport Valley, but livestock densities are greatest on the Valley's pasture, where irrigation has turned former sage-covered outwash benchlands to productive meadows (Robert Curry, pers. Comm., August 2000). Livestock in Bridgeport Valley are generally permitted to wade in the creeks and irrigation ditches, as well as the reservoir bed after the water level recedes during the irrigation season (Fig. 3).

Non-grazed portions of the watershed include forest, alpine tundra, and small urban centers. The town of Bridgeport, population 800, lies at the base of the watershed in Bridgeport Valley. A small residential subdivision is also located at Twin Lakes, along with a moderately large summer camping resort. The resort reported 21,000 visitor days for their 5-month season in 1979. Developments around Twin Lakes and in Bridgeport use septic systems and leach-fields for domestic waste disposal. In addition to these developed areas, public camping facilities exist in most of the watershed sub-basins. Trails throughout the watershed see a moderate amount of pack animal and foot traffic throughout the year.

Operation. The Walker River Irrigation District (WRID) completed construction of Bridgeport Reservoir in 1924 (Entrix, Inc. 1990). The reservoir dam is earth-filled with concrete cut-off walls and tile drains. It has a concrete, bottom-water outlet structure and an emergency surface spillway. The reservoir is used by WRID to store water for irrigating farmlands downstream along the East Walker River and in Mason Valley, Lyon County, Nevada. The reservoir is also used for recreation, with a productive sport trout fishery within the reservoir and on the East Walker River downstream of the reservoir.

WRID has water storage rights for Bridgeport Reservoir dating back to 8 August 1919. Water rights to Walker River water have also been decreed to users downstream of those served by WRID, including the Walker River Paiute Tribe at Schurz, Nevada. The State Water Resources Control Board (SWRCB), Division of Water Rights issued License 9407 to WRID for Diversion and Use of Water for Bridgeport Reservoir in 1970. The details of WRID's water rights and this license are summarized by SWRCB Order 90-18 (1990).

License 9407 confirms an appropriative water right to divert to storage a maximum of 39,700 af of water per-annum, and to make a maximum withdrawal of 36,000 af per-annum (SWRCB, 1990). Thus virtually the entire storage volume of the reservoir is used each year for irrigation.

Following a fish kill in 1988 during a maintenance draw-down of the reservoir, License 9407 was amended with a condition that the operation of Bridgeport Reservoir comply with Section 5937 of the California Fish and Game Code in order to protect the downstream fishery. To achieve compliance with this condition, the SWRCB (1990) mandated specific minimum reservoir pools and discharges as a function of annual snow pack conditions and forecasted daily minimum temperatures during winter, respectively.

Water is stored (i.e. inflows exceed outflows) during the winter (1 November to 1 March) and at any time during the irrigation season when the decreed rights downstream of WRID are satisfied (Entrix, Inc. 1990; SWRCB, 1990). The winter releases are a minimum of 20 to 30 cfs to comply with Order 90-18. Flood waters are also passed whenever required. During the irrigation season the natural inflow is passed through the Reservoir and delivered to the downstream users according to their rights. Delivery of stored water to WRID's members normally begins about 1 April. Each user may request delivery of water, pursuant to contract, at any time during the irrigation season. Daily mean discharge from the reservoir between 1 April and 30 September 2000 ranged from 85 to 300 cfs (USGS, 2001).

Bathymetry. The bathymetry of Bridgeport Reservoir was last surveyed in 1989 (Entrix, Inc., 1990). The maximum storage capacity of the reservoir at a spillway elevation of 6,460 ft amsl is 40,494 af (Table 2). The maximum depth of the reservoir, occurring near the dam, is 13.3 m, but fluctuates depending on the overall level of the water. In dry years the reservoir is held at lower elevations and in all years the water level is pulled down in the fall to supply water for irrigation downstream in Nevada.

The reservoir is shaped like a long arrowhead pointing almost due north with the point at the dam. By depth, it is divided into two regions, a narrow river canyon in which the dam is sited, and a shallow, wide region that was formerly a meadow (Fig. 2). Some of the upper meadow area is still used for pasture, as this region extends into the reservoir basin proper when the water level drops during the summer (Fig. 3).

When full to the spillway, about half of the reservoir is less than 3 m deep. Typically, the water elevation is less than the spillway elevation. For example, on June 6-7 1989 the elevation was 6447 ft, which eliminated about one third of the upper shallow region, but still left about half of the reservoir area less than 5 m deep. As the reservoir water level is lowered, its mean depth decreases less in proportion, which may have an impact on increasing its percentage of thermally stratified water since deep water stratifies more readily than shallow water. The depth-volume bathymetric curve is shown in Figure 4.

The hydraulic residence time (HRT) of Bridgeport Reservoir can vary considerably because of variations both in the stored water volume, and discharge flow. At maximum and typical water levels, the HRT is estimated to vary from about six weeks to 4.5 months, based on annual mean discharge from the reservoir (Table 2; see Appendix A-1 for calculation). This estimate seems reasonable, given that the volume of appropriated water annually by License 9407 is similar to the storage capacity of the reservoir.

Table 2. Physical characteristics of Bridgeport Reservoir. Note that the reservoir is operated for the benefit of irrigated agriculture downstream. In most years the reservoir is drawn down in the late summer and fall, greatly reducing reservoir area and maximum depth. Mean hydraulic residence time (HRT) for the reservoir is calculated using the annual mean discharge in East Walker River below the reservoir (USGS Site No. 10293000) for the available period of record, 1 October 1922 through 30 September 2000 (USGS, 2001). Range given after the mean HRT is based on the minimum and maximum annual means over that period. Elevations are given in units above mean sea level (amsl).

Parameter	**Water level at spillway**		**Water level at typical height**	
	US units	Metric	US units	Metric
Surface Elevation	6,460 ft	1,970 m	6,447 ft	1,965 m
Surface area, A	2,650 acres	1,070 ha	1,120 acres	454 ha
Mean depth, z	15 ft	4.6 m	11 ft	3.4 m
Maximum depth V_{max}	43 ft	13.3 m	30 ft	9 m
Storage volume V_s	40,494 af	49.9×10^6 m^3	12,539 af	15.5×10^6 m^3
Mean hydraulic residence time, HRT	138 days (46-544)	0.38 year (0.13-1.49)	43 days (14-169)	0.12 yr (0.04-0.46)

Sampling stations in 2000 and 1989. In 2000 five regular index stations were used (Fig. 4). They were located as follows: the dam index station near the dam at the north end of the reservoir, the mid-deep index station in the reservoir center, and a shallow-mid index station and two shallow index stations spread across the shallow, northern end of the reservoir. The dam index station was situated in the deepest part of the reservoir. The mid-deep index station was located upstream of the dam station in the thalweig or old river channel, and the shallow-mid index station further upstream along the long axis of the reservoir. The two other shallow stations were located to the east and west of the shallow-mid station.

In 1989, when the monitoring plan was focused on sediment transport, three stations were used all in deep water and with equal spacing along the dam. The samples were taken about 100 m from the dam face. The location of the middle 1989 station is similar to the dam index station used in 2000.

3.4. NOMENCLATURE.

Blue-green algae (cyanobacteria, cyanophyta, myxophyta). The indirect cause of the impairment to beneficial uses in Bridgeport Reservoir is high nutrient loading. The direct cause of impairment is extensive blooms of algae, specifically blue-green algae. Blue-green algae are also known by several other names (most recently cyanobacteria, but over the last 50 years have also been called cyanophyta or myxophyta). Algae are not a specific proper taxonomic group, being known more for what they lack (roots, intercellular transport) than what they have (Horne and Goldman, 1994). In general, algae are small autotrophic (i.e. energy derived directly from sunlight) organisms comprised of single cells, or a simple collection of single cells, that release oxygen during photosynthesis.

This functional definition would place the blue-green algae in the general group of algae. However, Stanier, using electron microscopy at UC Berkeley in the 1960s, determined that blue-green algae do not contain membrane-bound DNA and are prokaryotes, as are bacteria. Unfortunately using Stanier's definition complicates limnological discussion since most bacteria in lakes are neither photosynthetic nor autotrophic. Virtually all lake bacteria are heterotrophic, only consume oxygen, and depend on algae to release organic matter. A few rare lake bacteria use hydrogen sulfide rather than organic matter. Thus when discussing DO depletion and its impact on beneficial uses, it is ecologically more useful to think of the cyanobacteria as blue-green algae with similar functions as other algae in the lake's ecology.

3.5. PURPOSE OF THIS REPORT.

This report gives results from limnological studies made during the 2000 growth season (June-September) in Bridgeport Reservoir. A comparison is made with earlier data, especially the studies designed by Professor Horne that covered the same season in 1989. The 2000 study was designed to complement ongoing nutrient measurement studies in the watershed streams by the US Geological Survey and fishery use surveys by the Lahontan RWQCB.

4.0. METHODS

4.1. SEASONAL VARIATION OF PHYSICAL, CHEMICAL, AND BIOLOGICAL VARIABLES.

Regular measurements. Regular water quality monitoring at Bridgeport Reservoir was conducted at approximately 25 day intervals from June through September 2000. Measurements were made on seven occasions (the October survey was abandoned due to heavy snow on the mountain passes to the reservoir). No measurements were made during the winter ice cover due to the danger of such sampling without special equipment. Due to the elongated shape of Bridgeport Reservoir, five regular stations were used within the reservoir as described in Section 3.3. Two additional stations were set up on the inflow on the East Walker River and outflow from the reservoir during two monitoring events. Temperature and dissolved oxygen (DO) profiles were taken on each date along with samples for chlorophyll *a* and nutrients analyses. Temperature and DO were measured with a Hydrolab® surfer probe with a hand-held memory download. Records of the readouts were made in case of computer problems. Care was taken to properly calibrate the DO probe taking into account the reservoir's elevation. Water column samples were collected with a Van Dorn bottle approximately every 2 m at all in-reservoir sites while grab samples were collected at reservoir inflow and outflow stations.

Water samples were split into a number of sub-samples. Chlorophyll *a* was estimated with hot 90% methanol extraction and fluorometric analysis of samples filtered onto Whatman GF/C glass fiber filter within a few hours of collection following the methods devised for the International Biological Program (Golterman et al. 1978; Vollenwieder, 1971). Filters containing chlorophyll *a* were wrapped in aluminum foil and frozen for later analysis. The filtrate was then filtered again through Whatman 0.45-µm pore diameter membrane filters pre-washed with de-ionized water, then frozen for later analysis for dissolved nutrients by Berkeley staff.

Nutrients were measured using techniques from Standard Methods either at the Berkeley Laboratory or in certified commercial laboratories. Samples to characterize external nutrient inputs for the TMDL process were collected in the watershed by the USGS, and the data shown here are designed to interpret internal reservoir metabolic processes, such as internal cycling. Selected samples of unfiltered water were frozen for later total phosphorus (TP) analysis by Berkeley staff. Selected samples of unfiltered water were also sent to Sequoia Analytical, a State-certified professional laboratory located in Morgan Hill, California, for total and dissolved organic carbon and total Kjeldahl nitrogen analyses. Dissolved nutrient analyses included soluble reactive phosphorus (SRP) via the ascorbic acid method, nitrate via the cadmium reduction method, and ammonia via the phenate method (APHA, 1998). Detection limits were 30 µg/L for all dissolved compounds. Sequoia Analytical used EPA 415.1 and SM 5310C for total and dissolved organic carbon, and EPA 351.2 for total Kjeldahl nitrogen analyses.

Observations to determine the kind of algae dominating in the reservoir were also made. In eutrophic reservoirs, such as Bridgeport Reservoir, blue-green algae can fix atmospheric nitrogen and thus can be independent of external nitrogen sources. Nitrogen fixation only occurs when nitrate or ammonia fall to very low levels. Direct measurement of nitrogen fixation is quite complex, but an estimate of nitrogen fixation can be made by examining the number of heterocysts present in the blue-green algae colonies. Heterocysts, the site of nitrogen fixation in blue-green algae, are small, clear, thick-walled cells easily visible (with practice) under the light microscope. The numbers of heterocysts are correlated with the amount of nitrogen fixation, and the ratio of heterocysts to normal or vegetative cells gives an idea of the relative rate of fixation for each species.

4.2. SPECIAL STUDIES.

During the height of the summer bloom on Bridgeport Reservoir several special studies were made to determine aspects of the reservoir's eutrophication that were not revealed in the regular season-long study.

4.2.1. Special studies 1: Synoptic survey.

The spatial distribution of water quality variables in a eutrophic lake is often uneven. The peaks of abundance of algae, for example, may be missed by sampling at even the five regular index stations employed in the 2000 study. Spatial heterogeneity, or patchiness, is common during blooms of blue-green algae on lakes and accounts for some of their nuisance properties. A synoptic survey, where many samples are taken all over the reservoir at about the same time, overcomes any spatial heterogeneity, or at least determines its extent (Horne and Goldman, 1994). A synoptic survey is intended to give a "snap shot" of the entire reservoir at one time. It can be used to determine the location of the ideal index station or to calibrate existing index stations to the true reservoir mean for any parameter.

On 11 June 2000, the surface waters at 22 stations were sampled by surface dip using a large bucket. The sampling stations were located in a grid to cover the entire reservoir surface in a regular pattern. At some stations, where the water was deeper, both surface and 2.5 m deep samples were collected using a Van Dorn sampler. Selected samples were also monitored for algal species diversity, rate of algal rising, and productivity via the light/dark bottle DO production method (APHA, 1998). It is vital that synoptic sampling is rapid, so samples were taken using three boats to ensure that sampling was completed between 8:30 and 10:00 am. When sampling extends to longer periods there is a danger that the daily fluctuations in water quality will obscure spatial patterns.

4.2.2. Special studies 2: Diurnal study.

Eutrophic reservoirs exhibit large changes in some important water quality variables over a day. However, regular index station sampling only occurs at one time on each sampling date. Since several listed variables could change considerably with time of day (typically mid-morning), it is important to repeat much of the regular sampling over the course of an entire day. The available options were for a diel (24 hour) or a diurnal (dawn to dusk) sampling.

A Diel survey is the temporal analog of the spatial synoptic survey. It requires a full 24-hour study of reservoir parameters that vary with time and assists in determining the ideal time for sampling, once the index station has been selected. Diel variations in dissolved oxygen for example can kill or seriously affect fish and zooplankton but may only occur for a short period, often around dawn. Few regular measurements are made at these times so such conditions may often go undetected.

Preliminary surveys showed that the critical time for deep water DO was, unexpectedly, in the late afternoon, not the early morning. The shallowness of much of the reservoir combined with afternoon winds and the high altitude of the reservoir may be the cause of an afternoon low DO at depth. Together with the difficulty of launching boats in the low water of the reservoir in 2000, the low afternoon DO focused the study from a diel to a diurnal, or dawn to dusk, survey. Temperature and DO profiles with depth were taken as in the regular surveys at intervals down the depth column. Profiles were taken at five approximately equally spaced intervals from 8 am to 7 pm.

4.2.3. Special studies 3: Laboratory measurements of sediment nutrient release.

When the wind stirs the reservoir's sediments, more nutrients are released than if the reservoir were quiescent. If the sediments are anoxic, as occurs in deep water when the overlying waters lose oxygen in summer, the amount of nutrients released by wind mixing can increase. Anoxic sediments generally contain unbound phosphate and iron while sediments with oxygen usually have phosphate and iron combined in an insoluble precipitate.

Chamber incubations of deep sediment-water interface samples. Three samples of undisturbed sediment were collected at the dam index station in August 2000. Sediment was initially collected using a 15 by 15 cm Ekman dredge. The soft reservoir sediment plugs the jaws of the dredge, resulting in the collection of an undisturbed sediment surface with overlaying water. Once the dredge was brought to the surface, a sediment-water interface sample was sub-sampled from the dredge into a cylindrical Plexiglas chamber (Fig. 6). The chamber top was slowly pushed into the sediment and overlaying water captured in the dredge. A cap and gasket were then mated with the bottom of the chamber by hand while the chamber was still in the sediment. The capped chamber was pulled out of the dredge and bolted onto a round Plexiglas base. The sediment-water interface sample consisted of a sediment core 5-6 cm thick with a surface area of 71 cm^2, and 200-300 ml of overlaying water. After sediment collection, the chambers were gently flooded with bottom water. By keeping the chambers completely full of water, turbulence at the sediment-water interface was minimized during transport back to the laboratory.

Once in the laboratory, chambers were incubated in the dark at *in situ* reservoir sediment temperature (14°C). Sediment oxygen demand (SOD) was first measured. Chambers were filled completely with reservoir water and DO in chamber water overlaying sediment was measured with a Yellow Springs Instruments DO meter (model 55) and probe (model 5739). Oxygen demand (mg/L/d) for each chamber was calculated as the slope of the least squares best-fit line of DO versus time. This rate was converted to SOD (g/m/d) by multiplying by the volume of chamber water, and dividing by the chamber surface area. Oxygen uptake within chamber water was not measured, but was likely negligible. Numerous studies by Beutel (unpublished) show that chamber water DO consumption is less than 10-15% of the total oxygen demand measured in the chambers.

After the SOD test, chambers were incubated in four phases in the dark at *in situ* temperature (14°C): 1) high oxygen (aerated with air), 2) low oxygen (aerated with 10% oxygen), 3) anoxic (bubbled with nitrogen gas), and 4) re-aeration (aerated with air). The gas for phase 2 was obtained from Scott-Marin Inc., Riverside, California. Chambers were incubated with 1 L of reservoir water overlaying the sediment. Gas was injected into the chambers at a rate of approximately 5 ml/min, and this maintained a well-mixed water column but never re-suspended sediment. Water samples of 100 ml were removed every 2-3 days and analyzed for dissolved nutrients using the methods outlined above. De-ionized makeup water added to all non-aerated chambers was purged with the appropriate feed gas for 5 minutes prior to injection into the respective chamber to minimize introduction of atmospheric DO. De-ionized make-up water was used to avoid introducing nutrients into the chambers, which would have complicated calculation of nutrient accumulation rates and obscured interpretation of experimental results. Nutrient accumulation (mg/L/d) was calculated as the slope of the least squares best-fit line of ammonia mass versus time, taking into account the mass of nutrient removed during sampling. This rate was converted to a flux ($mg/m^2/d$) by multiplying by the volume of chamber water, and dividing by the chamber surface area.

Oxidation-reduction potential (ORP) of chamber water, a qualitative measure of the severity of anoxia at the sediment-water interface, was measured with permanently installed electrodes hand made after the method of Faulkner et al. (1989). To measure ORP, a Corning calomel reference electrode (epoxy body, No. 476406) was inserted through an access port into the top of the water column. The permanently installed electrode and the reference electrode were then connected to a Beckman Expandomatic SS-2 voltmeter, and a reading was recorded after 10 min. Standard ORP was also measured in ZoBell's solution each time it was measured in chambers. ORP values were then calculated relative to the standard hydrogen electrode using the equation presented in Standard Methods (APHA, 1998).

4.2.4. Special studies 4: Blue-green algae buoyancy and sinking study.

The active regulation of buoyancy of large colonial blue green algae is a key to their success (or becoming a nuisance in human terms). Little is known about the buoyancy or natural colonies of these algae since most studies have been made in the laboratory. The problem with laboratory studies of buoyancy is that the size of laboratory colonies is very much smaller than in nature. The speed of rising and sinking of the colonies is governed by Stokes Law and the shape of the sinking body. In lakes buoyancy depends on two main factors: effective cell diameter and cell density. Phytoplankton respire continually but DO depletion and danger to fish health only occurs late in the day, at night, or in deeper (darker) water when oxygen-producing photosynthesis no longer outpaces oxygen demand from respiration. Since blue-green algae can move up and down the water column quickly and with little energy loss, their movements radically change the oxygen patterns in the reservoir. Not all of the blue-green colonies in Bridgeport Reservoir sink to the lake sediments and reduce DO. Large amounts pile up and dry on the downwind shores by the dam where they exert little oxygen demand on the sediments. It is possible that this quirk is what has allowed the reservoir to have a salmonid fishery in the reservoir and below.

Rising and sinking studies were carried out using samples taken directly from the reservoir to the field laboratory. Large, visible Colonies of various blue-green algal species were selected using a pipette and placed very gently in the middle of a 1000 mL graduated cylinder filled with reservoir water. If the placement is carried out slowly, there is a minimal introduction of turbulence when the algae are added and when the pipette is withdrawn. The change in position of the visible colonies was measured over time and the speed of motion calculated.

4.2.5. Special studies 5: Hydrogen sulfide survey.

Hydrogen sulfide is a very toxic soluble gas that has killed salmonid fish in reservoirs and in downstream rivers and hatcheries receiving deep-water outflow from reservoirs. Hydrogen sulfide is produced under anoxic conditions in the sediments and is a typical result of eutrophication. Thus hydrogen sulfide is not desired in Bridgeport Reservoir. Deep-water samples were surveyed for the presence of hydrogen sulfide by odor (threshold for humans is very low at about 5 ppb).

4.3. Preliminary Review of the Initial Nutrients and Pathogens Samples Collected in the Watershed of Bridgeport Reservoir in 1989 and 2000.

The nutrient concentrations in the creeks flowing into Bridgeport Reservoir were collected and analyzed by the USGS using their own protocol. Plans were to collect data from April 2000 to April 2001. For this report, only the first spring data (April-June 2000) were examined. The full data set will be examined in terms of effects on the reservoir and for TMDL use in the next report in this series.

5.0 RESULTS

The results are shown in the conventional order of physical, chemical, and biological parameters. Comments and discussion are presented in the results section only if they are short. Additional discussion and opinions about the results and their meanings are presented in the discussion section. All raw temperature and DO profile data is collected in Appendix A-2. All analytical data is summarized in tabular form in Appendix A-3, and all raw laboratory data and reports are included in Appendix A-4.

5.1 REGULAR LIMNOLOGICAL MONITORING IN THE RESERVOIR IN 2000 AND COMPARISONS WITH 1989.

The water depth in these two years was very different and affected many of the lake variables such as the mixing regime and presence of toxicants such as ammonia. In 2000, a fairly normal water year, water depth was at a maximum of 13 m on 11 June, fell to about 9 m in late August and remained about 9 m until mid-September (Fig. 7a). The 1989 year was a critically dry year and consequently the reservoir was shallower since less water since less water was available for storage. The maximum depth was only 10 m in July, 8 m by mid-August and only 5 m by early September.

5.1.1. Seasonal isopleths of temperature and dissolved oxygen at the dam index station.

Temperature isotherms. The seasonal variation in temperature with depth for 2000 is shown in Figure 7a and can be compared with similar data collected in 1989 (Fig. 8). The data are also presented in tabular form in Appendix A-2.

Year 2000. Despite its sub-alpine location and relatively shallow depth, Bridgeport Reservoir in 2000 acted as a dimictic (two-mixings per year) system rather than a polymictic (many top-to-bottom mixings per year) system. Mixing occurred from mid-September until ice cover and again from ice out to mid-June. A summer period of moderately stable thermal stratification occurred in the deeper water and it is assumed that ice cover prevented wind mixing in winter. The expected polymictic characteristic of top-to-bottom mixing of the water column during windy conditions in summer did not occur.

Polymictic conditions occur in shallow lakes and are partially responsible for the more eutrophic conditions of such lakes since sediment nutrients can be recycled back up into the sunlit phonic zone during each mixing event. However, transport of deep-water nutrients from the sediments to the reservoir surface did occur in Bridgeport Reservoir in 2000, despite strong signs of stratification. Thus sometimes the weak stratification may have partially broken down. It is not fully clear if defining the reservoir in 2000 as dimictic is fully appropriate. Perhaps a new term such as weakly polymictic would be a better term. An alternate explanation is leakage of the hypolimnion water around the reservoir edges where wave action moves the thermocline up and down the submerged shore. In particular, the extensive shallow region at the upper end of the lake could be a site where nutrient-rich deep water finds its way to the upper waters.

For this project, sampling was limited to the summer season, but sufficient measurements were made to describe most of the year's temperature and DO cycles. In 2000 almost isothermal conditions were found on the first sampling event in mid-June, with morning temperatures of 18.2 ^{o}C at the surface and 16.7 ^{o}C at 12 m depth (Fig. 7a). The temperature difference of 1.5 ^{o}C was not sufficient to consider the water column to have stable thermal stratification since a normal wind should be able to overcome the corresponding density differences and mix the water column. That such mixing had occurred in the recent past is shown by the high bottom temperature. If no mixing had occurred the deeper water would remain at a low temperature, close to winter conditions. However, wind mixing was not able to remove some chemical stratification that had already begun to deplete DO in the deeper waters. The period of strongest thermal stratified occurred between 13 July and early September. During this time surface temperatures were 22-23 ^{o}C while the bottom water was 18-19 ^{o}C, a difference of about 3-4 ^{o}C (Fig. 7a). While this density difference was not enough to guarantee thermal stratification, mixing was at least reduced and insufficiently strong to satisfy deep-water oxygen demands.

In 1989 collections were begun slightly earlier than in 2000 and clear evidence of fully isothermal conditions of 14 ^{o}C were evident in mid-May and at 11 ^{o}C in early October (Fig. 8a). Weak stratification had set in by mid-June, as in 2000 (Fig. 8b) but was lost by mid-August. However, the bottom temperatures in 1989 were much warmer (21 ^{o}C) which would have produced a much less stable thermal stratification. In addition, 1989 was a critically dry year and the reservoir was shallower, especially later in the summer. The maximum depth in 1989 was only 10 m in July, 8 m by mid-August but fell considerably to only 5 m by early September. The 2000 data for these times were 13 m in June and 9 m from late August until mid-September. Thus water depths suitable for thermally stratified conditions in 1989 existed only between June and mid August. As with 2000, however, chemical stratification occurred in terms of moderate DO depletion in the deeper waters.

Dissolved oxygen. Oxygen isopleths (seasonal contour lines of equal concentrations of oxygen) reflect the interaction of the oxygen demand made by sunken algae and other organic matter, the amount of vertical mixing of surface oxygen, and inflows and outflows from the hypolimnion.

In 2000, the reservoir showed classical eutrophic oxygen isopleths with a period of hypoxia or low oxygen (< 1-2 mg/L) beginning about mid-July and lasting for about five weeks until approximately mid-August. The low DO extended from the sediments to 3 m from the surface (Fig. 7b). The very deepest water showed complete anoxia (zero oxygen), ensuring that the sediments would have the potential to release soluble phosphate, ammonia, and hydrogen sulfate. The presence of anoxic sediments alone does not necessarily produce nutrient or toxicant releases. The kind of sediments and the duration of the anoxia also plays a role.

In contrast to the low DO present in the deeper waters, the surface waters in 2000 were well supplied with oxygen reaching super-saturations of 145% (12.2 mg/L at 23.4 ^oC on 24 July; 13.1 mg/L at 20.2 ^oC, 11 September). The high DO values were due to favorable conditions for the blue-green algal photosynthesis that are abundant in the surface, photic zone of Bridgeport Reservoir. The favorable conditions were high light and abundant nutrients.

In 1989, no hypoxia (DO < 2 mg/L) occurred (Fig. 9a), in contrast with 2000 (Fig. 9b). Oxygen was depleted 2-3 m from the surface during the short period of weak stratification. However, the depletion was only to 4-5 mg/L and not to 0-2 mg/L as in 2000. At these moderate DO levels the releases of nutrients from the sediment is relatively small compared with releases under anoxic conditions. As in 2000, however, super-saturation with values similar to those in 2000 of about 140% saturation (13 mg/L at 19 ^oC) were observed in early September as the reservoir depth declined.

5.1.2. Seasonal and spatial variations in water clarity.

Water clarity is an important component of the beneficial uses of any reservoir. Water clarity or transparency was measured using a Secchi disc. High values of Secchi depth indicate clear water and a general picture of the trophic state of the reservoir can be assessed using the Secchi depth. Water with a Secchi depth greater than 8 m indicates oligotrophic conditions (for example Secchi in ultra-oligotrophic Lake Tahoe often ranges from 25-30 m). A Secchi depth of 2-8 m indicates mesotrophic conditions, a desirable range for most reservoirs, while less than 2 m indicates eutrophic conditions (Horne and Goldman, 1994, p. 465). However, the above trophic classification is based on the presence of algal chlorophyll alone. If other light absorbing mater is present in the water then the maximum Secchi depth achievable falls. In the case of Bridgeport Reservoir, the amount of colored water coming from upstream considerably reduces the Secchi depth. The origin of the colored water, likely due to the presence of humic and fulvic acids derived from vegetative decay, is not clear but may be associated with the flooded pastures created for livestock grazing over the last 150 years.

Seasonal variations in water clarity at the dam and mid-deep index stations. In 2000 the water in Bridgeport Reservoir at the dam index station showed low to moderate clarity. Secchi depths ranged from 0.75 to 2.75 m with a mean of 1.5 m (Fig. 10, Table 3). At the mid-deep index station, the corresponding values were similar (range 0.6 to > 3.6; mean = 1.5 m). For most of the time there was little difference between the mid-deep and dam index stations (Fig. 10).

Secchi depths of less than 2 m indicate eutrophic waters. By this definition Bridgeport Reservoir is eutrophic, but not hyper-eutrophic. However, the amount of algae in the reservoir is very large and the chlorophyll *a* levels indicate hyper-eutrophic conditions. This puzzle is resolved by considering the kind of algae present. All of the four common genera of blue-green algae in Bridgeport Reservoir form large colonies containing hundreds of thousands, sometimes millions, of individual cells. There are few other algae in the water since blue-green algae are effective competitors and exclude other algae by shading and perhaps other means (e.g. nutrient depletion, toxin production). The few non-blue-green algae that could grow in the reservoir are presumably kept at low levels by zooplankton grazing. An important phytoplankton grazing zooplankter *Daphnia* is abundant in the reservoir. However, *Daphnia* does not feed on mature blue-green algal colonies so plays little role in keeping the water clear in Bridgeport Reservoir as it is currently managed.

The net result is that the water between the blue-green colonies is relatively clear. Thus the water in the reservoir is more transparent than would be expected for such a large amount of algae. The chlorophyll in the center of the colonies takes no part in light absorption since it is already shaded by algae on the outside of the colony. The individual filaments of algae can glide in and out of the colony and presumably take turns being in the dark center and the illuminated exterior.

The spatial variation of water clarity around the reservoir is shown in Table 3. In the deeper, stratified part of the reservoir the Secchi depth averaged 1.5 m over the summer. The mean for the shallow water stations among the macrophyte beds was greater than 2.6 m. Considering that the September bloom was not included in the 1.5 m average and the Secchi depth exceeded water depth for some shallow water samples, water clarity in shallow water was probably double that in the deeper water. It is likely that the macrophyte beds were influential in reducing the shallow water algae. This conclusion has important bearing on the use of biomanipulation in the restoration of Bridgeport Reservoir (see discussion and recommendations).

Table 3. Water clarity (Secchi disc depth in meters) at the five index stations including the shallow stations near dense macrophyte colonies for Bridgeport Reservoir in 2000.

Date	**Dam**	**Mid-deep**	**Shallow**		
			West	Mid-shallow	East
11 June	2.75	3.6	>3.8	>1.5	3
21 June	0.95	1.0	1.65	>2	2.4
13 July	0.75	0.60	2.3	3.4	>2.5
30 July	1.8	2.3	>3	>2.5	>2.5
21 Aug	1.2	1.75	Ts	Ts	Ts
Mean	1.5	1.5	>2.7	>2.35	>2.6

">" = Secchi reached reservoir bed and was still visible. "Ts" = too shallow for boat access.

Water clarity in 1989 was similar to that found in 2000. Relatively deep values of greater than 2 m were uncommon and occurred in spring 1989 (e.g. June 14 = 4.8 m; July 10 = 2.2 m) while the average for the remainder of the season through October was ~ 0.5 m (n=6; range 0.25 to 0.9 m). The average for 1989 was 1.2 m (June-Oct.). Some of the low clarity water in 1989 may have been due to suspended sediment since a turbidity of 39 NTU was recorded at the same time as the lowest 0.25 m Secchi depth (water depth = 4.4 m at the dam at this date of 4 October 1989). However, for much of the rest of 1989 algal abundance, as shown by chlorophyll *a* values, was very high and even higher than in 2000.

5.1.3. Seasonal variations in nutrients with depth and location.

Total Phosphorus. Total phosphorus (TP) is a useful measure of the total potential for P-based algae crops in lakes. Usually TP in the growing season of spring to fall is dominated by the phosphorus in living material; detritus and soluble phosphate are present in fairly small amounts. Only soluble phosphate is available to algae, but approximately 80% of TP can easily be transformed into usable phosphate by a common enzyme, alkaline phosphatase, excreted by the algae when they sense cellular P-limitation.

In 2000 TP levels in the dam index station and the mid-deep index station were very high. At the surface TP ranged from about 160 to 600 µg/L between June and September (Fig. 11, Table 4). The mean value for the season was about 380 µg/L. For perspective, typical TP standards for lakes and reservoirs in temperate zones range from 5-40 µg/L, and 20 µg/L has been proposed by the Lahontan RWQCB for some lakes and reservoirs. Thus the mean TP in Bridgeport Reservoir currently exceeds typical standards by 19 times, and the maximum by 30 times.

Table 4. Seasonal variations in total phosphorus with depth for the five index stations in Bridgeport Reservoir in 2000. Concentrations labeled nd are less than the laboratory detection limit of 50 µg/L; means were calculated assuming that nd = half the detection limit.

Site	11 June	22 June	12 July	21 Aug	11 Sept	Seasonal Means (n)
Dam						
0 m	360	610	510	260	160	380 (5)
2 m			170			
4 m	190			150	130	157 (3)
6 m				80		
8 m	170	220	150		130	168 (4)
10 m						
12 m	80					
Mid-deep						
0 m	310	630	440	110	150	328 (5)
4 m	250	230	80	110	100	154 (5)
8 m		170	80			125 (2)
Shallow mid						
0 m	170	340	490		316	330 (4)
Shallow east						
0 m	210	290				250 (2)
2 m	280	310				295 (2)
Shallow west						
0 m	240	270	80		160	188 (4)
3 m	310	320				315 (2)
(sta 20, 22)				130		
Inflow	170	170	nd	nd	nd	83* (5)
Outflow	200	240	100	80	120	148 (5)

* 98 if nd ~ = detection limit of 50 ug/L.

TP at the dam index station showed the expected decline with depth, since TP includes algae, which in this reservoir are buoyant and tend to float near the surface (Table 4, Fig.

12a). Surface TP was always higher than deeper water samples. The 4m samples varied from 130-190 mg/L (mean = 114 mg/L), although only three dates were sampled (Table 4). The 8 m samples were also lower than those at the surface and ranged from 76-220 µg/L (mean = 150 µg/L for five dates).

At the mid-deep index station the situation was similar to that found for the dam index station (Table 4; Fig. 12b). TP at the surface ranged from 70-630 µg/L (mean = 330 µg/L). At 4m the mean TP was 151 µg/L (range 80-250). The 8 m TP levels were only measured in June and July 2000, and were 170 and 80 µg/L, again quite similar to those in the dam index station.

In the three shallow stations at the upstream end of the reservoir, samples could only be taken at the surface and at 2 or 3 m depth before the bottom was encountered. Thus only the surface samples can be compared with those in the deeper water. There were no large differences between the east, mid, and west shallow stations, although the shallow-mid station did have generally higher TP (Fig. 12c). Individual values are shown in Table 4. The average surface TP in the shallow part of the reservoir was about 240 µg/L considerably lower than found in the surface for the mid-deep and dam index stations (330 and 380 µg/L, respectively). However, TP was much higher in the shallow areas near the inflow creeks for some of the year when compared with ammonia. The contrast in distribution of the two nutrients suggests two inputs for TP (river and internal loading) but only one for ammonia (internal loading).

The TP data support the hypothesis that the presence of macrophytes and their attached green algae is unfavorable to the dominant planktonic blue-green algae. This finding supports the conclusion drawn from the water clarity data that biomanipulation may be a potential restoration technique for Bridgeport Reservoir.

Bridgeport Reservoir has a single, bottom water outlet and, as shown by the sediment discharged during its draining in the late 1980s, can pull from the deepest water layers. The mean reservoir outflow TP in summer-fall 2000 was 148 µg/L (range 80-240, Table 4). The mean TP concentration in the 8 m layer was 168 ug/L (range 130-220 µg/L), fairly similar to the outflow concentration. A single deep-water concentration (12 m) was 80 ug/L and the deeper water summer mean TP was probably a little lower than the 8 m average.

During the summer-fall 2000 the mean TP lost from the reservoir (148 ug/L) was about 50% higher than the inflow (83-98 ug/L, depending on the assumptions made for samples below detection, see Table 4). Internal loading of phosphate from the sediment to the deep water accounts for the higher outflow concentration in summer-fall.

In 1989 TP in surface waters showed similar patterns to those found in 2000. For 1989 TP concentrations of 70 µg/L in June increased to a sharp peak of 820 µg/L at the end of August, probably due to an algal bloom.

Soluble reactive phosphorus (SRP). While TP represents all living and dead organic phosphorus and particulate and soluble inorganic phosphorus, SRP is a more dynamic variable and shows the phosphorus directly available for algal growth. SRP was measured on only two occasions, 11 June and 11 September 2000 (Table 5).

Table 5. Seasonal and depth variations in bioavailable SRP (µg/L) for Bridgeport Reservoir in 2000. Concentrations labeled nd are less than the laboratory detection limit of 50 µg/L; means were calculated assuming that nd = half the detection limit and nd = the detection limit (in parentheses).

Dam index station	**11 June**	**11 Sept.**	**Mean**
0-8 m	nd	nd	25 (50)
10m	65	60	63
12m	60	-	60
Mid-deep index station			
0-4 m	nd	nd	25 (50)
6m	65	nd	45 (58)
8m	85	-	85
Reservoir outlet	nd	nd	25 (50)

As expected if SRP is in demand for living algae but released from anoxic sediments, SRP was high in deep water and below detection in shallow water (Table 5). SRP values of 60-85 µg/L in the deep waters indicate a very large potential for the sediments to supply bioavailable phosphorus in summer and fall. However, the SRP and TP in the sediments must originally have been derived from external sources such as watershed erosion, cattle grazing, sewage disposal, or wildlife.

Ammonia. Ammonia is a nutrient when providing nitrogen for algal growth, but a toxicant to fish and aquatic insect larvae at higher concentrations. Both ammonia and nitrate provide nitrogen for growth and provide ample growth stimulation at concentrations ranging from 50-500 µg/L (expressed as N). Beyond these concentrations, nitrate shows no toxicity for most species up to 10 mg/L. Ammonia can be toxic to aquatic life at concentrations greater than 100 µg/L, but the toxicity strongly depends on pH, temperature, water hardness, and DO concentration. Normally concentrations of 1-2 mg/L are harmful to salmonid fish, but no single number will be appropriate for all conditions.

At the dam index site ammonia showed very dramatic variations during the summer (Fig. 13). In late spring ammonia was below detection (30 µg/L) in the surface waters down to 8 m (Fig. 14, Table 6). In the bottom waters (10-12 m) moderate amounts of ammonia (mean = ~ 100 µg/L, range = 54-140) were present in early June. Ammonia below detection at 10-12 m depth on 22 June indicates that some mixing of deep and shallow waters may have occurred. Ammonia is a preferred nutrient for algae and the large biomass of blue-green algae present in spring would easily be able to remove 50-100 µg/L of ammonia in a few days following any mixing event. Larger amounts could exceed the uptake capacity of the algae bloom for ammonia since, unlike luxury consumption of phosphate, there are few non-toxic methods for algae to store nitrogen.

The situation of low ammonia in the majority of the water column with low to moderate values in deep water lasted from at least 11 June to early July when ammonia levels at all depths increased to a mean of approximately 490 µg/L (range 321-770, Figs. 13 and 14, Table 6). The high ammonia at all depths continued through July but began to revert to the spring condition by 21 August when the photic zone of 0-2 m had non-detectable ammonia, while the rest of the water column showed 58-130 µg/L (Fig. 14). However, samples taken earlier on the same day indicated up to 250 µg/L of ammonia in the general area of the dam station (see synoptic data). By 11 September, ammonia was non-detectable in 0-6 m and present at ~ 160 µg/L only in the deepest water layer (8 m since the reservoir was being drawn down for fall irrigation season). The onset of high ammonia in the reservoir coincided almost exactly with the onset of sediment anoxia in mid-July. The decline in high ammonia concentrations lagged behind the restoration of oxic sediment conditions by only a week or so, indicating rapid uptake by the fall algae bloom.

Table 6. Seasonal variations in ammonia with depth for the five index stations in Bridgeport Reservoir in 2000. Concentrations labeled nd are less than the laboratory detection limit of 30 µg/L; means were calculated assuming that nd = half of the detection limit.

Site	11 June	22 June	12 July	31 July	21 Aug	11 Sept	Mean
Dam							
0 m	nd	nd	681	232	nd	nd	162
2 m	nd	nd	770	263	nd	nd	182
4 m	nd	nd	414	460	131	nd	175
6 m	nd	nd	321	398	128	nd	149
8 m	nd	nd	348	368	58	164	161
10 m	54	nd	321	398			197
12 m	140	nd	553	607			329
Mid-deep							
0 m	nd	nd	360	136	nd	nd	93
2 m	nd	nd	116	nd	nd	nd	32
4 m	nd	nd	nd	81	nd	nd	26
6 m	54	nd	221	197	nd	231	122
8 m	57	nd	263	279			153
Shallow mid							
0 m	nd	48	nd	nd	nd		15
Shallow east							
0 m	nd	nd	nd	nd	nd		15
3 m	48	nd	nd	nd	nd		22
Shallow west							
0 m	nd	nd	nd	nd	nd		15
Outflow	45	nd	514	542	128	200	241
Inflow	nd	nd	nd	nd	nd	nd	15

Similar trends to those found in 2000 occurred in 1989 when surface ammonia rose to about 400 µg/L on 10 July 1989, fell to 100 µg/L in late July and early August, but rose again to 300 µg/L in late August, before falling again to 100-200 µg/L late September and October. The somewhat lower ammonia concentrations in 1989 (max 400 µg/L) relative to those in 2000 (max 680 µg/L) can be explained by the lower water elevation that restricted anoxic conditions in the deep-water sediments (Fig. 9a-b). An interesting set of values not measured in 2000, but measured in 1989 show that nitrate, the other nitrogen source available for algal growth, was a minor player in the nitrogen cycle. In 1989 values for nitrate in the surface waters near the dam ranged only from 30-100 µg/L between June and September (mean = 57 µg/L), compared with a mean for ammonia of 207 µg/L.

The surge of ammonia in the reservoir in 2000 was concurrent with the onset of stronger thermal stratification and low DO in the sediments (see Fig. 7a-b). Since ammonia is released under these conditions it is logical that the ammonia surge in the deeper water was a result of anoxic sediments. However, it is a peculiarity of the weak thermal stratification or polymixis of Bridgeport Reservoir that benthic ammonia is rapidly transported into the upper, photic zone waters. Once in the photic zone, ammonia would be available to supply the large fall blue-green algae bloom observed in 2000. Thus the anoxia in the bottom waters allows any nutrients accumulated over the last decade or more to be re-used at least twice, and amplifies eutrophication in the reservoir. Use of hypolimnetic oxygenation would substantially reduce or even remove this effect. Oxygenation would also remove any toxic threat to fish from ammonia.

Phosphorus as SRP is also released by anoxic sediments but is so dynamic in terms of active uptake that changes may not be very clear compared with the fluxes of ammonia. In 2000, the 31 July SRP samples were lost and SRP was only measured on 11 June and 11 September, which was insufficient to show changes. However, the clear mid-summer increases in ammonia are faintly reflected in the changes in TP where surface values at the dam index station rose from about 360 µg/L on 11 June to 500-600 µg/L in late June to mid-July and fell to 250 µg/L by 11 September. These changes were coincidental with those for ammonia, but TP reflects both living algae, dead algae and other detritus as well as inert soluble phosphorus and bioavailable TP and SRP.

5.1.4. Seasonal observations on the presence of hydrogen sulfide in deep water samples and other odors in surface waters in 2000.

Observations of the presence of hydrogen sulfide in bottom water samples showed no detectable sulfide (human nose detection limit is very low ~ 2-5 ppb). The observations were as follows:

- June 11 Dam station, 10 m – no hydrogen sulfide odor
- July 12 Surface at dam – strong fishy odor (characteristic of dense blue-green blooms)
- July 30. 8-11 m samples "spicy" odor but no hydrogen sulfide odor
- August 21, Deep water at dam, 9.7 m – no hydrogen sulfide odor
- August 22, Deep water sediment core, black surface but no hydrogen sulfide odor.

From the lack of sulfide it can be concluded that there was insufficient time of low DO in the sediments for hydrogen sulfide to form. Once anoxia occurs in the sediments, other electron acceptors such as nitrate or ferric iron must be used up first, before sulfate is reduced to sulfide. If the anoxic period had extended for a period on the order of eight weeks rather than the five weeks recorded in 2000, sulfide could form since there is an ample decaying mass of algae in the sediments in summer.

The formation of hydrogen sulfide requires the presence of sulfate, which is reduced under anoxic conditions. Water in the Walker River flowing from the Sierra Nevada mountains is soft, with little sulfate, so the production of hydrogen sulfide may also be muted due to lake of sulfate substrate for the sulfur-reducing bacteria.

Odor in fish flesh at Bridgeport Reservoir. The rental boat operator at Bridgeport Reservoir stated that he would not eat fish from the reservoir late in the season due to a "musty" odor. Musty odors are characteristic of some of the off-odors produced by both planktonic and benthic blue-green algae. In particular blue-green algae produce geosmin (earthy odor) and MIB (methyl-isoborneol, a musty odor). In commercial fisheries, for example for catfish, such odors are a problem and are dealt with by reducing the amount of blue-green algae in the water. Thus tainting of odors in fish flesh could be added to the impairment of beneficial uses of Bridgeport Reservoir that are attributable to blue-green algae and eutrophication. These flesh-tainting odors are often combated in commercial fisheries by using aeration or oxygenation.

5.1.5. Seasonal variations in algae as chlorophyll *a* with depth and location.

Chlorophyll *a* and algal biomass. Since blue-green algae (cyanobacteria) almost totally dominate the phytoplankton in Bridgeport Reservoir, their concentrations can be approximated by the measurement of chlorophyll *a*. Chlorophyll *a* is the green pigment that converts light energy into reductive power that supplies cell energy, and ranges from 0.5-2.0 % of the dry weight of an algal cell. It is thus a good surrogate for algal biomass and has the advantage that is much easier to measure than any other biomass method. Concentrations of chlorophyll *a* can range from very low values (< 0.1 µg/L) in unproductive lakes such as Lake Tahoe or the open ocean, to several hundred µg/L in extremely productive lakes or sewage oxidation ponds.

Chlorophyll *a* at the dam index station in 2000 ranged from high to extremely high. Beginning on 11 June, chlorophyll *a* in the surface water was 16 µg/L, a high value for a Sierra fed mountain lake. As the summer progressed the algae bloomed, reaching over 800 µg/L at the surface before rapidly falling back to 50-100 µg/L for the remainder of the summer and early autumn (Fig. 15, Table 7). The mean summer-fall surface concentration at the dam index station was 200 µg/L (n = 5).

In general the depth variation of chlorophyll and thus of the dominant blue-green algae followed the expected pattern with more algae at the surface or just below the surface in the upper two meters. For example, at the height of the July 2000 bloom almost all the algae in the dam index station water column were at the surface (806 µg/L), while only 250 µg/L were present at 2m, and less than 50 µg/L at 4, 6, and 8 m (Table 7). Blue-green algae owe much of their success in lakes to an ability to float to the surface using gas vacuoles. Thus their accumulation in surface waters is not surprising, and the distribution with depth is similar to that found in other lakes and even the open ocean.

Only when algal numbers were low, at the start of the bloom and in September for the dam station, was chlorophyll fairly evenly distributed with depth. In the 1989 study, particular attention was paid to the amount of deep-water chlorophyll since it also played a role in turbidity in the outflow. In terms of comparison with 2000, the 1989 deep-water data support the conclusion that most chlorophyll was at the surface. In both years the surface mean chlorophyll at the dam was 200-300 µg/L and the bottom values were almost always less than 50 µg/L and were frequently less than 30 µg/L.

Table 7. Seasonal and depth variations in chlorophyll *a* for Bridgeport Reservoir in 2000.

Site	10 June	22 June	12 July	31 July	11 Sept
Dam					
0 m	16	60	806	52	93
2 m	16	62	257	45	102
4 m	15	63	51	16	28
6 m	10	58	44	12	16
8 m	7	51	31	7	14
Mid-deep					
0 m	9	170	556	100	41
2 m	11	111	100	23	60
4 m	6	38	35	9	31
6 m	3	3	7	4	22
8 m		7			
Shallow mid					
0 m	1	3	11	76	37
Shallow east					
0 m	2	6	11	153	333
1.5 m	1.5	19			
Shallow west					
0 m	28	194	13	45	176
1.5 m	7	56			
Outflow	5	12	12	65	29
Inflow					

The value of 800 µg/L found in the surface water at the dam index station in 2000 was enormous, but even larger values were found for surface chlorophyll in 1989, when a peak of 1,920 µg/L was found at the dam index station. Blue-green algae blooms are extremely spatially heterogeneous (see synoptic section of this report) and a chlorophyll value of ~ 2,500 µg/L was found for a healthy surface bloom on 12 July 2000 in the NE corner of the reservoir not far from the 1989 index site. In addition, a decaying bloom in the shallow region of the reservoir showed a chlorophyll value of 1,100 µg/L on the same date.

A better comparison between 2000 and 1989 is the mean surface chlorophyll for the season (200 vs. 277 µg/L respectively). In both years sampling covered about the same time period, but the 1989 mean chlorophyll included an early low October value. Without the October value the mean for 1989 was about 295 µg/L. In addition, the 2000 study was based on five samples, while the 1989 data included 10 samples. In 1989 there were three dam stations and only the central one was used to determine the mean of 277 or 295 µg/L. The mean for the other two stations in 1989 was much lower due to lower peak values. Additional information can be gained by considering the results of the synoptic studies in 2000 that can be compared with the other regular dam stations in 1989 (see Section 5.2.1).

Chlorophyll *a* at the mid-deep index station. In terms of physical and chemical variables discussed above, the mid-deep station was similar to the dam index station. In terms of surface chlorophyll the two stations were also similar (Fig. 15, Tables 7 and 8). For example the mean surface chlorophyll was 205 µg/L (dam station) and 175 µg/L (mid-deep station). Both sites showed a large bloom in mid-July (806 and 555 µg/L for dam and mid-deep stations, respectively).

In terms of depth distribution, both dam and mid lake index station sites were also similar on most occasions with the majority of the algae and thus chlorophyll at the surface (Table 8). Only one time on 22 July did the dam station show a mixed, if low, population of algae while the mid-deep station continued the normal pattern of surface maximum.

Table 8. Seasonal variations in chlorophyll *a* in surface samples from the five index stations for Bridgeport Reservoir for 2000 (µg/L).

Date	**Dam**	**Mid-Deep**	**Shallow**		
			West	Mid shallow	East
11 June	16	9	28	1	2
21 June	60	170	195	3	6
13 July	806	555	13	11	11
30 July	52	100	45	77	153
11 Sept.	93	41	334	37	Ts
Mean	205	175	123	26	43

Chlorophyll *a* at the shallow index stations. Chlorophyll *a* in the surface waters of the three shallow water stations in 2000 were dramatically lower than those found in the deep stations (Fig. 16, Tables 7 and 8). The seasonal mean for the two deep stations was 190 µg/L while that for the three shallow stations was 64 µg/L. The shallow stations were generally less than 3 m deep so no comparative information on depth profiles with the 8-12 m stations could be gathered.

The low chlorophyll in the shallow stations could have several causes:

- Oxygenated or at least less anoxic sediments since shallow water mixes more easily. Oxygenated sediments release less nutrients during the summer.
- Dilution by inflowing creek water which has very low chlorophyll.
- Reduction of phytoplankton by competition with macrophytes and attached algae.
- Surface winds transporting phytoplankton toward the deeper portions of the reservoir away from the upwind shallows.
- Presence of macrophytes reducing turbulence and causing phytoplankton to sink.

Although some lowering of the shallow water chlorophyll must be due to the inflow, the volume of water entering the reservoir in summer was small. The question of spatial variation in nutrients and any thoughts regarding sediment anoxia are best addressed in the special synoptic survey that is reported in Section 5.2.1.

The chlorophyll *a* peak in the deep, northern part of the reservoir was not in phase with that found in the shallow water at the southern end. The highest chlorophyll in the shallow stations was towards the fall suggesting a source of nutrient stimulation that was confined to the shallow waters, or variation in the factors affecting nutrient and phytoplankton transport. Since the inflow to the reservoir enters in the shallow end, nutrients from the watershed could be a cause.

5.1.6. Observations of the phytoplankton: species composition and potential for nitrogen fixation (N_2-fixation).

Phytoplankton: species composition. In 2000 as in 1989, the phytoplankton in Bridgeport Reservoir was dominated by large, colonial blue-green algae. Under the microscope it was often difficult to find any other algal species, such as diatoms or small flagellates, that are normally found in lakes. In 2000 there were four common colonial forms of blue-green algae: *Aphanizomenon, Anabaena, Microcystis,* and *Gleotrichia*. The large colonies of these species can be distinguished by eye when on the reservoir. All these three genera are common in eutrophic lakes throughout the world.

Gleotrichia was found for the first time in the reservoir in 2000, yet it was by far the most common form found. Its invasion may be due to changes in nutrient loading since 1989, or changed conditions in the reservoir following the draining of the reservoir in the 1980s or the 1990's El Niño floods. *Gleotrichia* forms small balls of filaments extending out from a central core and resembles a pincushion or hedgehog. It has characteristic heterocysts only at the inmost end of the filament, in the core region. It was present in the surface plankton on every occasion the reservoir was sampled in 2000. *Anabaena* was also common and present on each occasion. *Aphanizomenon* and *Microcystis* were not always observed, but may have been present in smaller colonies.

The blue-green surface plankton were not distributed evenly around the reservoir in terms of biomass (see synoptic survey, Section 5.2.1), but patchiness is common for blue-green algae in lakes. The algae were also not distributed evenly in terms of species (see also synoptic survey). Uneven distribution of blue-green species has also been found in lakes, but is less commonly discussed in the literature. In 2000 the most obvious spatial differences seen in the regular season sampling was the dominance of *Gleotrichia* in the deeper water, from the dam upstream to the center of the reservoir.

Although systematic examination of the phytoplankton was outside the scope of this project, the algae were examined closely by eye, hand lens, and under the compound microscope. The color of these four species is characteristic of some aspects of their health and metabolism. For example, sometimes the large colonies that are easily visible by eye are a pale yellow or yellow-green and at other times they present a vibrant dark green color. The pale and yellow cast can indicate a lack of nitrogen in the cells since chlorophyll contains nitrogen and is often depleted or denatured when environmental nitrogen is scarce. On all occasions examined in 2000 the cells that were common in the phytoplankton appeared healthy and green. This finding is in agreement with the large amount of ammonia measured in the summer in the reservoir.

Potential for nitrogen fixation as assessed by heterocyst frequencies. The ability to fix gaseous nitrogen into cellular nitrogen is an ability possessed by few organisms. Prior to human intervention, the world's nitrogen cycle was mostly maintained by a balance between nitrogen fixation in wild legumes or non-legumes and denitrification (nitrate to nitrogen gas in estuaries and the coastal ocean). In all cases the active agents were bacteria growing in a kind of symbiosis with the plants. In the case of the aquatic ecosystem, some, but by no means all, blue-green algae can fix nitrogen. The blue-green algae show a common root with bacteria in having this ability. Three of the four common species found in Bridgeport Reservoir (*Gleotrichia*, *Anabaena*, and *Aphanizomenon*) are capable of fixing nitrogen gas in the oxygenated surface waters, but will not do so unless other sources of nitrogen, such as nitrate or ammonia, have been used up. Under conditions of low nitrate and ammonia nitrogen fixation will occur so long as other nutrients such as available phosphorus, iron and molybdenum are adequate. It is generally thought that nitrogen fixation confers an advantage over other algae in N-limiting conditions, but other factors such as buoyancy are also important in determining the dominant algae in lakes and reservoirs.

The potential for nitrogen fixation in blue-green algae in lakes is shown by the presence of specially adapted cells called heterocysts. These thick-walled, empty-looking cells form from normal vegetative cells in response to a shortage of nitrogen. The absolute number of heterocysts or the ratio of heterocysts to vegetative cells (H:C) can be used to predict the amount of nitrogen fixation (Horne and Goldman, 1972; Horne and Galat, 1985). The H:C ratios vary between species. Some species, such as *Anabaena*, show maximum fixation at ratios of about 1:5 or 1:10, while other genera, such as *Aphanizomenon* or *Nodularia*, only reach ratios of 1:40. The ratios observed in 2000 are shown in Table 9.

Table 9. Heterocyst to cell ratios (H:C) for potentially nitrogen-fixing blue-green algae in Bridgeport Reservoir for summer 2000. Samples are from surface waters.

Genera	H:C ratio (date)	Site	Comments
Anabaena	1:35 (7/13)	Shallow	Low N_2-fix
	1:27 (7/31)	Shallow	Low N_2-fix
	1:43 (7/31)	Deep, dam	Low N_2-fix
Gleotrichia	1:37 (6/11)	Deep	?*
	1:500 (7/15)	Shallow	Low N_2-fix
	1:45 (7/31)	Shallow	?*

*The relationship of low N_2-fixation and H:C for *Gleotrichia* is not known. *Aphanizomenon* was not regularly assayed for heterocyst frequency since only a few were observed in flake colonies containing thousands of filaments.

From the H:C data it appears that nitrogen fixation was at low levels for most of the 2000 season. The amount of nitrogen fixation would be suppressed if nitrate and ammonia concentrations exceeded 100 μg/L in the reservoir (Horne and Commins, 1987).

Ammonia alone was present at concentrations greater than 100 µg/L for much of the reservoir by the end of June (Table 6).

5.2. SPECIAL LIMNOLOGICAL MONITORING IN 2000.

5.2.1. Synoptic survey.

A synoptic survey was carried out in the morning of 21 August 2000. A total of 23 stations were set up covering all of the reservoir. Measurements were made at the surface and 2.5 m (where deep enough). Temperature, DO, Secchi depth, ammonia, total phosphorus, chlorophyll *a*, and microscopic observations of plankton were made at each station. In addition, primary production was determined from changes in DO over time in sealed bottles. The layout of the 23 synoptic stations is shown in Figure 17. Using three boats, the volunteer expert assistance of Dr. James Roth, and nine graduate students, the synoptic was completed in less than two hours beginning at 8:30 am. The weather was calm and clear with virtually no ripples on the reservoir. The conditions thus represented a typical late summer morning in the eastern Sierra Nevada, and should be representative of most summer days.

Temperature synoptic. The distribution of temperature at the surface and depth of 2.5 m 21 August 2000 is shown in Figure 18a-b. There was little temperature heterogeneity in surface waters, with temperatures ranging from 18.2 to 20.4 ^{o}C across the reservoir, but only by about 1 ^{o}C for portions of the reservoir away from the cool inflows of its tributaries. The warmest water was pushed towards the dam at the north end of the reservoir, reflecting the prevailing winds down the E. Walker River drainage. At a depth of 2.5 m, which approximates the bottom in the extensive shallow regions of the reservoir, a strong band of cooler water was found (18-19 ^{o}C, Fig. 18b). The cooler water could have resulted from an interflow formed the tributary inflows, which have a greater density than the warmer surface waters of the reservoir. In the deeper parts of the reservoir the upper water column was more vertically homogeneous.

The surface measurements from 2000 can be compared with a smaller synoptic survey made in 1989 (Fig. 18c). A similar situation to that found on 21 August 2000 prevailed on 21 September 1989 (Fig. 18c). Surface temperatures away from the E. Walker River varied by only 0.8 ^{o}C, similar to 2000, although the water was slightly warmer that morning (19.9 vs. 20.7 ^{o}C). Once again, the warmer surface water appeared to have been blown to the north end of the reservoir, towards the dam. The inflowing water from the E. Walker River was cooler in 1989 (16.4 ^{o}C) than in 2000 (18.2 ^{o}C), probably due to the difference in sample dates.

Dissolved oxygen synoptic. The distribution of DO in surface samples is shown in Figure 19a. There was a gradient of DO from the shallow, southern section of the reservoir, which varied from 8.8 to 10.4 mg/L, down to a low of 5.9 mg/L at the dam. Low DO is often, but not always, found in the mornings in the surface waters of eutrophic lakes. The low DO near the dam was probably due to overnight respiration by a surface bloom blown there by the afternoon winds. Conversely it is likely that the higher DO, closer to saturation, was due to the relatively low phytoplankton concentrations at the shallow end of the reservoir.

Water clarity synoptic. The distribution of water clarity as determined by the Secchi disc depth synoptic survey is shown in Figure 19b. The entire reservoir showed Secchi depths of less than 2 m, a strong indication that the ecosystem is eutrophic. The poorest water quality was shown in the central section of the reservoir, where chlorophyll and algae were most concentrated. The lowest value found was 1.4 m, and the highest value was 1.8 m.

The Secchi depths measured in the synoptic survey show that the low values measured during bloom periods on Bridgeport Reservoir using only the deep index stations were probably representative of the entire reservoir. Even though the algae blooms were very patchy, a simple measurement such as water clarity showed overall and widespread degradation in water quality. Future improvements in water clarity can reliably be measured on a routine basis using only the deep index stations.

Ammonia synoptic. The spatial variation of ammonia for both surface and 2.5 m layers was significant (Fig. 20a-b). Apart from the first sample located near the dam index site, there was little difference with depth. However, there were large spatial differences across the reservoir (Fig. 20a). In the first two stations surface ammonia ranged from 150-200 µg/L. Moving upstream to the next three stations ammonia decreased to 120 µg/L and then to 60-80 µg/L. All of these first five stations were located in the old canyon where the reservoir is deepest and most sheltered. Ammonia in the remainder of the reservoir was below the laboratory detection limit (< 50 µg/L), with the exception of the surface water at station 11 (60 µg/L). A similar distribution of ammonia to that in the surface was shown by the 2.5 m synoptic samples from the deeper section of the reservoir (Fig. 20b). A concentration of 250 µg/L was found at 2.5 m near the dam, and ammonia gradually declined upstream at this depth.

The seasonal studies indicated that ammonia had been generated in the deep water sediments in the weeks prior to the synoptic survey. The multiple simultaneous sampling brings out the exquisite details of the journey of this sediment ammonia up into the upper water column and its spread through the reservoir. The loss of ammonia as it spreads upstream is most likely due to dilution, but some uptake into algae could have occurred.

Chlorophyll *a* synoptic. The distribution of algae, as shown by the concentrations of chlorophyll *a*, was very different from that shown by ammonia. Instead of the maximum over deep water near the dam, as for ammonia, surface chlorophyll showed a peak in the central part of the reservoir where the canyon broadens out into the former meadowlands (Fig. 21a). Three stations (5, 8, and 11) showed chlorophyll values of 150-210 µg/L, compared with a reservoir-wide average for that day of 76 µg/L. Stations 6, 7, and 10, adjacent to the three high chlorophyll sites, contained 60-90 µg/L, again higher than the overall mean value. Only station 18 (90 µg/L), in the shallow central part of the reservoir, did not conform to the pattern in rest of the reservoir. Most of the remaining sites had chlorophyll values around 40 µg/L, with a low of 25 µg/L (Sta. 19).

The most obvious explanation for the observed chlorophyll pattern was a bloom of blue-green algae in the center of the reservoir. This may indicate that the center of the reservoir is a principal location of vertical nutrient transport from the hypolimnion to the photic zone. Such mixing at this location may result from both convective circulation or upwelling.

Blue-green surface blooms are typically spatially heterogeneous. Nonetheless, surface blooms of blue-green algae, because they are so buoyant, can be easily moved around by the wind. In Clear Lake, California surface blue-green blooms have been tracked moving at 5-15 cm/sec or over a km in an afternoon, under conditions of light winds (Wrigely and Horne, 1974; Wrigely and Horne, 1975).

Algal species synoptic. Counts of algae colonies were made by direct observation. In Bridgeport reservoir these colonies represent the vast majority of the algal species present. The results of the synoptic survey of algal colonies are shown in Figures 21b, 22a-c, and Table 10.

Table 10. Algal colonies present relative to chlorophyll *a* and primary production in synoptic samples from Bridgeport Reservoir on 21 August 2000. Net production was measured in mg oxygen produced/L/h.

Station	***Gleotrichia*** (Colonies/L)	***Anabaena*** (Colonies/L)	***Aphanizo-menon*** (Colonies/L)	**Total algae** (colonies/L)	**Chl *a* (µg/L)**	**Net produc-tion**	**Photos index** (Net/chla)
Surface							
1	210	0	0	249	39	0.87	2.23
4 b	896	36	0	932	47	0.96	2.04
6	350	16	16**	382	89	0.89	1.00
7	980	36	28	1,044	75	10.9	1.45
8	1,400	350	4	1,754	217	1.18	0.54
8 repeat	1,400	80	16	1,496	217	1.18	0.54
12	350	16	52	418	53	0.87	1.64
13	154	11	11	176	64	0.59	0.92
15	140	32	40	312	75	0.88	1.17
16	196	0	60	256	67	1.14	1.70
18	70	8	72	150	93	1.0	1.08
21	70	28	36	134	42	0.99	2.36
22	42	0	0	42	68	0.98	1.44
0 m mean					**76**	**0.94**	**1.24**
2.5 m samples							
1	72	4	4*	80	116	0.36	0.31
4a	20	0	0	20	56	0.51	0.91
8	182	12	16	210	51	0.78	1.53
15	20	4	4	28	89	0.93	1.045
2.5 m mean					**78**	**0.65**	**0.83**

* Very large flake (1 to 1.3 cm long). ** 0.5-1.0 cm long flakes.

The relationship between algae colonies and chlorophyll *a* was moderate, with the total blue-green algae visible colony numbers (Table 10) accounting for 52% of the variation in chlorophyll *a* (Fig. 23a). For the dominant algae, *Gleotrichia*, the relationship was a little weaker (R^2 = 0.41, Fig. 23b). A stronger relationship between chlorophyll *a* and colonies was not expected since the colonies vary in size.

The algal colonies were counted in the same bottles that were used for measurement of photosynthesis (i.e. primary production synoptic, see below). Thus the two assays can be directly compared. No relationship was found between colony number and photosynthesis (primary production, R^2 = 0.12) or photosynthetic index (PI, R^2 = 0.12). The slight negative relationship with the PI was most likely due to a few large colonies that would make inefficient use of their chlorophyll, since much of the pigment in the center is shaded by the pigment in the outer layer of filaments.

Primary Production synoptic. The primary production of phytoplankton is the basis of the production of oxygen in reservoirs. Primary production was measured by the conventional light and dark bottle technique using the changes in dissolved oxygen over time to find both net photosynthesis (light bottle DO increase) and respiration (dark bottle DO changes). Gross photosynthesis, or that available for growth, is found by adding the light and dark bottle changes. The amount of depletion of DO in the water column can be estimated from this technique and used to size any restoration systems, such as hypolimnetic oxygenation.

Of the 23 synoptic stations, 13 were assayed for primary production, with several stations tested at both surface and 2.5 m depth. The results are shown in Table 11. The mean net production for surface waters was 0.94 mg oxygen/L/h (range 0.59 to 1.18). Values of 0.5 to 1 mg oxygen/L/h are very high levels of productivity, indicating that there were ample nutrients and sunlight to support healthy algae in Bridgeport Reservoir.

Dark respiration showed a mean of -0.09 mg oxygen/L/h (range -0.08 to -0.16) or about -10% of the net production. Such a low respiration contribution indicates that the blue-green phytoplankton in Bridgeport Reservoir at that time were healthy and probably not nutrient limited, at least over the few hours of the incubation.

There was little relationship between chlorophyll and primary production in the surface waters of the reservoir (Fig. 24a). Spatial variations in chlorophyll explained only about a quarter of the spatial variations in net photosynthesis (R^2 only 0.23). A strong relationship would be expected if all the chlorophyll was similarly active (e.g. Horne et al., 1979), but in this case there were different species of blue-green algae in different parts of the reservoir and some may have been more isolated or more nutrient-rich than others. Chlorophyll *a* is a major site for the storage of nitrogen in algae and ammonia was not evenly distributed at this time (Fig. 20a-b). Chlorophyll distribution did apparently explain about 55% of the spatial variation of respiration in the reservoir. However, if the single outlying station 8 (217 µg/L chl a) was removed the R^2 fell to less than 1%.

There was a good relationship between chlorophyll *a* and respiration as measured by the dark bottle short-term oxygen consumption. About half of the changes in oxygen were accounted for by respiration (r^2 = 0.55, Fig. 24b). However, if the example with the largest concentration of chlorophyll is removed from the regression, the relationship between oxygen and respiration is poor and the regression coefficient falls to almost zero. Thus only for the denser blooms can the more easily measured chlorophyll be used to determine the approximate amount of oxygen demand that would be imposed by nocturnal or deep (dark) aphotic zone respiration of the blue-green algae in Bridgeport Reservoir.

Table 11. Primary production in surface water synoptic samples from Bridgeport Reservoir on 21 August 2000**.** All samples were incubated in surface light to indicate potential productivity for the 2.5 m samples. Units are in mg DO/L/hr for DO and µg/L for chlorophyll *a*.

Station	**Chl a (µg/L)**	**Net photosynthesis (light bottle)**	**Respiration (dark bottle)**	**Gross Photosynthesis (light + dark)**	**Photos index (Net/chl a)**
Surface samples					
1	39	0.87	-0.09	0.96	2.23
3	56	0.71	-0.08	0.79	1.27
4 b	47	0.96	-0.10	1.05	2.04
6	89	0.89	-0.09	0.98	1.00
7	75	1.09	-0.10	1.19	1.45
8	217	1.18	-0.16	1.34	0.54
12	53	0.87	-0.08	0.96	1.64
13	64	0.59	-0.09	0.68	0.92
15	75	0.88	-0.096	0.97	1.17
16	67	1.14	-0.10	1.26	1.70
18	93	1.0	-0.09	1.09	1.08
21	42	0.99	-0.11	1.10	2.36
22	68	0.98	-0.13	1.11	1.44
Surface mean	**76**	**0.94**	**-0.09**	**1.03**	**1.24**
2.5 m samples					
1	116	0.36	-0.15	0.51	0.31
4a	56	0.51	-0.10	0.61	0.91
8	89	0.78	-0.09	0.87	1.53
15	78	0.93	-0.16	1.09	1.05
2.5 m mean		**0.65**	**-0.13**	**0.78**	**0.83**

Although there were identical light conditions during the incubation of the surface and 2.5 m samples, there was a substantial difference in the net productivity and respiration. The samples from deeper water were about one-third less productive and respired at a greater rate (-0.13 mg/L/h relative to –0.09 mg/L/h), despite the similar biomass (chlorophyll) levels present (Table 11). The difference between the surface and 2.5 m samples is probably due to light limitation since nutrients were abundant and over half of the surface light has been absorbed at 2.5 m.

Photosynthetic Index (PI) or photosynthetic efficiency. The ratio of net primary production to chlorophyll a is a measure of the health of the algae. The PI is affected by damaged or inefficient chlorophyll a, low available nutrients and previous light conditions (the experiments were run with near optimal light). In Bridgeport Reservoir the surface samples had a much higher PI (mean = 1.24) than samples collected from 2.5 m (mean = 0.83) indicating that the algae from the deeper water had spent some time in gloomy deeper water and were less "fit" than those collected from the surface (Table 11). Implicit in this explanation is that there was not a large amount of vertical interchange between surface and deeper water by wind mixing or active buoyancy regulation by the algae. The spatial distribution of the PI indicates that the densest accumulations of algae were not photosynthetically efficient. The low efficiency (low PI) is not surprising since much of the chlorophyll in dense algal colonies or aggregates of colonies in dense scums is self shielded from external light sources.

5.2.2. Diurnal study.

The diurnal (daylight hours only) study was carried out on 11 September 2000 from near dawn to dusk (8 am to 7 pm). The results are shown in detail in Appendix A-2 and Figures 25a (temperature) and 25b (DO).

Diurnal temperature variations. The onset of temporary thermal stratification and consequent isolation of the deep water and sediments from top-to-bottom mixing (holomixis) was clearly demonstrated in Bridgeport Reservoir (Fig. 25a). In the early morning the surface water temperature had fallen to a low of 17.6^{o}C due to evaporative cooling of the warm water by the cool night air in this high elevation lake. Bottom water temperatures in the early morning were 15.4^{o}C, only 2.2 degrees below the surface temperature and would have been easily mixed with surface waters if there had been a morning wind that day. However, the calm conditions, clear air, and cloudless skies characteristic of the eastern Sierra Nevada Mountains rapidly heated the lake so that by late afternoon the surface water temperature had risen a full 4 ^{o}C and reached 21.8 ^{o}C (Fig. 25a). At this time the surface water was 6.4 ^{o}C higher than the bottom water, which remained unchanged through the day due to the insolating effect of the upper water and the high turbidity of the water that absorbed almost all solar light and heat. A strong diurnal thermocline became established at 2-3 m depth. The upper and lower waters were thus quite stable and would resist wind mixing until the upper layer had cooled.

Diurnal variation in dissolved oxygen. The variation in DO over the day (Fig. 25a) followed the trends established for temperature (Fig. 25a). An oxycline or sharp change in DO with depth became established at 2-3 m and paralleled the thermocline. The surface DO of 10 mg/L at 10 am rose to 12.6 mg/L by late afternoon, despite the water being warmest at that time and thus with the lowest solubility for oxygen. In contrast, the DO in the deepest water above the sediments fell from 3.6 mg/L in the early morning to 1.9 mg/L in mid morning and remained below 3 mg/L for the rest of the day. DO values below 2 mg/L are termed hypoxic and fish are unable to survive. In addition, low DO values near the sediments usually indicate the potential for the release of nutrients and toxic substances such as ammonia, hydrogen sulfide and soluble phosphate. However, it takes some time (hours) before the low DO at the sediment-water interface actually shows releases of formerly sediment-bound chemicals. Thus short term dips in DO are serious for fish but may not affect sediment chemistry to any degree.

An unexpected occurrence was the high early morning DO of 11-16 mg/L in the upper 2 meters of water (Fig. 25b). The high DO value was not coincident with a temperature stratification (Fig. 25a) but disappeared as the sun rose further and surface DO fell from 16 to 10 mg/L by 10 am. Dawn in the region at this date was about 6 am and sunrise over the eastern hills occurred at 6:45. Thus about one hours photosynthesis could have occurred prior to the first measurements of the day at 8 am, and may have accounted for the high early DO since there was no wind. The high DO occurred in a thick layer of water about 3 m deep and so eliminates the effects of a dense surface scum. Measurement errors are also possible but the probes used did not appear to give any other unexpected results.

Thus the most likely explanation for the high early morning DO is provided by consideration of the values found at the opposite end of the day. Just before sunset at 19:00 hours the DO in the reservoir had risen to 13.1 mg./L (Fig. 25b). Given the large amount of algae in the lake it is possible that the DO in the evenings in Bridgeport Reservoir could reach 15-20 mg/L and most of this could carry over to the next morning. In most lakes, DO is consumed overnight by the algae that produced it. However, blue-green algae sink and rise rapidly (see Section 5.2.4) and normally sink in the afternoon and evening to gain access to nutrients in the aphotic zone. The dark DO consumption by blue-green algae was measured in August as low as -0.08 mg/L/h or about 1 mg/L for a 12 hour night. Therefore algal dark respiration would have only reduce the daily DO consumption by about 6%. Such a respiration rate would lower DO from 17 to 16 mg/L overnight. The much greater loss of 6 mg/L in DO over the mid-morning can be ascribed to the light winds that efficiently strip oxygen from the surface waters.

5.2.3. Internal Loading: Sediment disturbance and nutrient release study.

Algal blooms have two sources of nutrients. One is from external loading from streams and atmospheric deposition. The other is internal loading from the sediments. Deep water sediments are the normal source of nutrients in summer in shallow lakes. The deeper water loses oxygen causing chemical changes in the sediments. Anoxic sediments can release ammonia, phosphate, iron, manganese, and hydrogen sulfide. Anoxia in the sediments occurs in Bridgeport Reservoir, but not all summer, and perhaps not every year. The effect of anoxia on phosphate releases is summarized in Fig. 26a-c and the details indicated in Fig. 27.

As can be seen from Figures 26a-c and 27, there was no release of phosphate from the deep samples for 19 days of incubation under conditions of 20% (normal air) or reduced oxygen supply of 10% oxygen (via controlled gas additions). However, once the sediments were artificially made anoxic, phosphate releases occurred almost at once in all three replicated chambers (Figs. 26a-c, 27). The release rate of phosphorus under the experimental conditions used are shown in Table 12. The rate of phosphate release under aerated conditions was very small (0.11 $mg/m^2/d$) but rose almost 15 fold when the sediments became anoxic. Once oxygen was added back to the system, phosphate releases ceased and the released phosphate disappeared from the chambers (Fig. 26,

Table 12). The release of phosphate under anoxic conditions and its re-precipitation back to the sediments as ferric phosphate is a well-known process, but does not always occur. In Bridgeport Reservoir the sediment iron and phosphorus content is apparently ideal for this classical release and sequestering process. This evidence that Bridgeport Reservoir sediment release phosphate under anoxic conditions and sequester phosphate in the presence of oxygen are further evidence that internal recycling of nutrients is occurring in Bridgeport Reservoir. It also indicates that deep water oxygenation is a potentially effective restoration method for the reservoir.

Table 12. Release rate of soluble, bioavailable phosphate from sediments under various conditions in laboratory for Bridgeport Reservoir in summer 2000. Values are means from three chambers treated identically.

Day	**Conditions**	**Concentration (µg/L)**	**Release rate ($mg/m^2/d$)**
19	Aerated	11	0.11
22	Anoxic	345	1.6
24	Anoxic	568	1.2

5.2.4. Blue-green algae buoyancy and sinking study.

Rising rates. The dominance of large colonial blue-green algae in lakes and reservoirs is often due to active regulation of buoyancy. The speed of rising of colonies from the plankton of Bridgeport Reservoir in summer 2000 is shown in Table 13. The mean rates of rising through the water column as determined by the studies using wild algae ranged from 0.07 cm/s for the balls of *Gleotrichia* to 0.17 cm/s for the clumps of *Microcystis* and 0.13 cm/s for the large nail-clipping shaped Aphanizomenon (Table 13). A coiled clump of *Anabaena* showed the fastest speed of 1.1 cm/s.

Table 13. Blue-green algal buoyancy measured in Bridgeport Reservoir in 2000. Natural rate of rising and rate of sinking when gas vacuoles were crushed artificially by additional pressure.

Algal taxon	**Rising rate (cm/s)**	**Sinking rate (cm/s)**
Gleotrichia	0.03, 0.12, 0.07	0.39, 0.32, 031
Mean	**0.07**	**0.34**
Microcystis	0.22, 0.25, 0.07, 0.12, 0.13, 0.20, 0.18	
Mean	**0.17**	
*Aphanizomenon**	0.18, 0.07	
Mean	**0.13**	
Anabaena	1.1	

*Not all colonies rose, some appeared to have neutral buoyancy at the time.

The observed speeds of rising are very rapid compared with other algae or laboratory tests of cultured blue-green algae, indicate that in Bridgeport Reservoir the phytoplankton could rise between 2.5 m/h (*Gleotrichia*) and 40 m/h (*Anabaena*). Thus the blue-green algal colonies in the reservoir could easily move from the deeper, nutrient-rich water to the surface within a few hours, assuming only small amounts of wind induced turbulence existed.

Sinking rates were not observed on this occasion since all of the algae showed either neutral or positive buoyancy. However, the maximum sinking rate can be estimated for some of the blue-green algae since it is the same as the maximum rising rate. Thus is was theoretically possible that most of the lake nuisance algae could sink to the lake bed within a day if is was not very windy.

5.2.5. Fate of algal carbon

Fate of floating scums. Dense floating scums of blue-green algae are an unsightly feature of most eutrophic lakes in summer and fall. Generally, the most dense accumulations occur on calm days at the downwind end of the lake. If winds increase or change direction the floating masses will either be mixing into the water column or drift away from the shore. In Bridgeport Reservoir the main site of accumulation is near the dam. If the wind is onshore some algae will be washed up and stranded on the shore, but they soon die and decompose.

An unusual feature of Bridgeport Reservoir is the accumulation of dried blue-green algae in piles several feet thick on the shoreline. The piles last through the winter, probably due to the region's cold and very dry climate. Presumably, the algae that dry out and partially decompose on the shore do not use any of the oxygen in the lake. Thus they do not contribute to the summer anoxia which is the root of many of the water quality problems in the reservoir.

It was hoped to compare the estimated amount of algae sinking to the sediments with those washed up on shore. However, as explained above in 5.2.4 only positive buoyancy was found in the algae in Bridgeport Reservoir in 2000. In addition, no accumulation on the shore occurred, as was seen in some previous winters. Conditions were not apparently suitable for this accumulation to be blown further on the dry area of the shore where large piles of algae had been found in some past years.

The lack of summer accumulation of algal carbon on the shore shows that, at least in 2000, the main fate of blue-green algal carbon is to sink to the lake bed and form sediments, be decomposed, or to be washed out of the bottom water outlet of the reservoir. From a management viewpoint this location is important since any algae sinking at this spot will be sucked out of the reservoir via the bottom release gate. The large amount of dissolved organic carbon passing out of the reservoir (Table 14) indicates some loss of carbon to downstream.

Dissolved and total organic carbon. Dissolved organic carbon (DOC) is the main component of total organic carbon (TOC) in natural waters. In most studies, TOC and DOC refer only to the smaller size fractions but often include algae but exclude floating leaves or grasses. DOC can contribute to the depletion of oxygen in lakes and some mathematical models add as much as 20% of the DO depletion to external, but unspecified sources. The values for TOC, DOC and also the related total organic nitrogen (TKN, total Kejeldahl-N = organic N + ammonia) are shown in Table 14. Since ammonia was usually less than 1 mg/L, TKN is a good approximation of total N.

Table 14. Total and dissolved organic carbon and dissolved organic nitrogen in Bridgeport Reservoir and drainage streams. Units in mg/L. The June samples are surface collections, the remainder of the data in the reservoir are composites (0-8 m, dam site; 0-2 m, shallow sites). Conservative, hardness related chemistry taken in June as the Walker River as entered the reservoir was (total alkalinity = 58 mg/L; TDS = 82 mg/L) and in the reservoir at the Dam (total alkalinity = 84 mg/L, TDS = 122 mg/L).

Component	**Reservoir**				**E. Walker River**		
	Dam	**Mid-deep**	**Shallow-east**	**Shallow-mid**	**E. Walker River @ Highway 395**	**E. Walker River, reservoir inflow**	**E. Walker River, reservoir outflow**
6/11/00							
TOC	4.3	-	4.2	1[1]	1.3	7.2	
DOC	5.2	-	3.9	3.7	1.1	6.2	
TKN	0.2			<0.2			
7/20/00							
TOC	6.5					5.1	
DOC	5.6					5.1	
TKN	0.9		0.8	1.1		0.3	0.8
7/31/00							
TKN	0.7	11.6	2.8	1.1		0.3	1.0
9/11/00							
TOC	5.9		8.8	6.1		2.9	11.8

[1] Probably an analytical error since it is an outlier and is much less than DOC.

The Walker River picks up a large amount of DOC in the short distance from the highway 395 bridge (1.1 mg/L) and the reservoir inflow (6.2, Table 14). The increase could be due to livestock excretion but could also be due to leaching of yellow colored humic substances from the wetlands near the lake. The increase is quite large for natural waters but not outside the range of values of DOC recorded elsewhere (Hessen & Tranvik, 1998).

In the reservoir the DOC and TOC were about the expected value for a eutrophic reservoir (3.7-8.8 mg/L, mean = 5.4 mg/L). Since the TOC/DOC mean for inflow was similar (5.3 mg/L), the lake would be in balance. However, it is unlikely that such a eutrophic reservoir would not generate at least 2-4 mg/L DOC during the summer. Thus much of the inflowing DOC must be labile (e.g. animal waste) and refractory (e.g. humic substances).

The organic nitrogen (TKN) in the reservoir provides a check on the DOC/TOC data since TKN is part of the TOC pool. Values of 0.2 mg/L or less for TKN in the early summer correlate generally with other biomass indicators and show a peak during the summer bloom in late June and July 2000 (Table 8).

5.2.6. Hydrogen sulfide survey.

Hydrogen sulfide was not detected by odor in any of the deep samples taken in Bridgeport Reservoir. Even when DO in the deep water was low, not sulfide odor was present. These findings indicate that the redox potential in the deep sediments was above -200 mv. The likely explanation is that there was not sufficient time for the redox to drop to very low levels before the reservoir water column turned over bringing at least some oxygen to the sediments. Low concentrations of sulfate may have also restricted sulfide production.

5.3. Preliminary Review of the Initial Nutrients and Pathogens Samples Collected in the Watershed of Bridgeport Reservoir in 2000.

Extensive collections of water quality data, including nutrient and fecal pathogen indicators for tributaries to Bridgeport Reservoir were made in 2000. The analysis of these data together with GIS mapping will be presented in the next report in this series. However, some initial data were collected in Spring and early summer 2000 and are analyzed here, since there is concern that the spring melt conditions may be one way in which excess nutrients reach the reservoir from the watershed. The data are summarized in Table 14.

5.3.1. Nutrients and pathogens above and below the stock pastures. Dividing the data into two groups, above the main Bridgeport Valley and below, allows characterization of water quality upstream and downstream of livestock grazing. Some of the upper stations were downstream of minor amounts of grazing activity in Sierra foothills, but grazing was much less intense than in Bridgeport Valley. In addition, the water distribution in the valley is complex, with water moving in a network of gravity fed irrigation ditches that cross Bridgeport Valley. The water interchange has the effect of moving any signal from valley processes around spatially and temporally as the water is diverted here and there.

Several preliminary conclusions can be reached from the data shown in Table 15. First, there was an increase in total phosphorus and soluble inorganic phosphorus as the water passes through the valley. For example the mean TP doubled from 19 µg/L to 39 µg/L between the upper sites and Bridgeport Reservoir (Table 15). Phosphate almost quadrupled from 4.3 to 16 µg/L over the same distance. The maximum recorded values of TP (116 and 107 µg/L) were almost double the Basin Plan standard of 60 µg/L.

The cause of the increase was most likely due to excreta and urine from cattle and horses grazing in the valley. There were no other likely sources of an increase in TP and phosphate of the magnitude measured. Confirmation of the source of the nutrients as being from livestock is indicated by the large increase in fecal coliform bacteria found for the same sites above and below the valley. The mean spring fecal coliform bacteria concentrations found above the valley were 2.6 colonies/100 mL, while the mean spring concentration below was 28 times greater (74 colonies/100 mL; Table 15). It is almost inconceivable that anything other than livestock could be the source of the measured amount of pathogens below the Bridgeport Valley pasture. There are almost no dwellings in the area between the upper sites and Highway 395, and coliform bacteria have such a short half life that this source is unlikely to be even a small contributor for most stretches of the creeks.

Table 15. Nutrient and fecal coliform concentrations above and below areas of stock grazing in the Bridgeport Reservoir drainage in spring-early summer 2000. Data from USGS and Lahontan RWQCB. Units are μg/L as N or P for nutrients and colonies/100 mL water for fecal coliform bacteria.

Site	**Date**	**Nitrate**	**Ammonia**	**TP**	**PO_4-P**	**Fecal Coliforms**	**Flow (cfs)**
Above Valley							
Virginia Creek nr. Bridgeport	4/12	7	4	33	13	2	20
	5/10	10	4	25	14	1	23
	6/5	15	<2	45	6	11	30
Robinson Creek @ twin lakes outlet	4/12	<5	<2	E4	<1	<1	7
	5/10	<5	<2	<8	<1	<1	8
	6/5	7	<2	<8	<1	-	12.5
Buckeye nr Bridgeport	4/12	<5	<2	14	2	2	65
	5/10	5	<2	10	2	2	137
	6/5	22	<2	18	<1	-	213
Arithmetic Mean		**8.3**	**1.7**	**19**	**4.3**	**2.6**	**57**
Below stock grazing areas							
Buckeye @ 395	4/12	<5	4	116	10	15	12
	5/10	<5	10	11	2	73	46
	6/5	13	2	32	<1	180	150
Suager @ 395	4/12	37	3	107	45	6	20
	5/10	52	4	44	38	2	13
	6/5	83	<2	59	35	59	5.2
Robinson @ 395	4/12	7	5	7	23	7	11
	5/10	<5	<2	<5	12	7	39
	6/5	8	3	8	16	200	124
Inlet to reservoir							
E. Walker River @ reservoir	4/12	<5	<2	51	23	3	34
	5/10	<5	12	42	19	82	71
	6/5	8-10	5-11	44	7	270	143
Robinson Creek @ reservoir	4/12	<5	<2	14	6	2	5
	5/10	<5	4	18	2	88	36
	6/5	10	8	25	<1	120	123
Arithmetic Mean		**16**	**4.4**	**39**	**16**	**74**	**55**

A clearer concept of the effect of the stock excretion is shown by simplifying the data by selecting the mean values for a single creek, Buckeye Creek, as it moves from the presumably more pristine upper reaches to the reservoir (Table 16). Here TP increased almost fourfold from 14 to 53 µg/L and phosphate doubled from 2 to 4.3 µg/L, while fecal coliform bacteria increased over 40 fold.

Table 16. Arithmetic mean nutrient and fecal coliform concentrations in Buckeye Creek above and below areas of stock grazing in the Bridgeport Reservoir drainage in spring-early summer 2000. Data from USGS and Lahontan RWQCB. Units are µg/L as N or P for nutrients and colonies/100 mL water for fecal coliform bacteria.

Site	Nitrate	Ammonia	TP	PO_4-P	Fecal Coliform	Flow (cfs)
Above Valley						
Buckeye nr Bridgeport	**10**	**<2**	**14**	**2**	**2**	**138**
Below stock grazing areas						
Buckeye @ 395	**6**	**5.3**	**53**	**4.3**	**89**	**69**

Similar, but more detailed conclusions can be drawn from Robinson Creek, where three stations were available (Table 17). In this creek TP increased from 4 µg/L in the upper stream to 7 µg/L at Highway 395, and to 19 µg/L at the inflow to the reservoir. Phosphate showed an increase from undetectable (< 1 µg/L) at the upper site, to 17 µg/L at the highway, then fell to 3 µg/L in the final short section to the reservoir. Fecal coliform bacteria rose from less than 1 µg/L (upper) to 71 µg/L (highway 395), and to 70 µg/L at the reservoir. Although the phosphate may have been expected to rise or remain high for the final section between the highway and the reservoir, soluble phosphate is a very dynamic substance and can be absorbed or released easily by soils.

Table 17. Arithmetic mean nutrient and pathogen concentrations in Robinson Creek above and below areas of stock grazing in the Bridgeport Reservoir drainage in spring-early summer 2000. Data from USGS and Lahontan RWQCB. Units are µg/L as N or P for nutrients and colonies/100 mL water for fecal coliform bacteria.

Site	Nitrate	Ammonia	TP	PO_4-P	Fecal Coliform	Flow (cfs)
Above Valley						
Robinson Creek @ twin lakes outlet	**4**	**1**	**4**	**<1**	**<1**	**9.1**
Below stock grazing areas						
Robinson @ 395	**6**	**3**	**7**	**17**	**71**	**58**
Inlet to reservoir						
Robinson Creek @ reservoir	**5.3**	**4.3**	**19**	**3**	**70**	**55**

The overall, but preliminary, conclusion to be reached from the evidence shown in Tables 15-17 is that the stock grazing in the valley was responsible for all or most of the increase in TP, phosphate and pathogens as the water flowed through the valley. This conclusion is based on the magnitude and direction of the trends in pollutant increases along the creeks, which would be hard to explain in any other way. The conclusion is preliminary because the tables only examine data from April to June, 2000 and do not include the other nine months of collection. However, most of the inflow of water occurs in the spring and early summer so the data examined cover a very critical period. The data are also preliminary because the means shown have not been flow-weighted to account for the greater flows in June during the snow melt. However, the fact that high concentrations of TP and coliform bacteria occurred during high flow would tend to increase the effect of grazing, not decrease it. Finally, nutrients, chemical constituents, and pathogens other than phosphorus and fecal coliform bacteria were measured by USGS. The entire year's data will be examined in the second main report (2001 report) in this series of reports.

5.3.2. Nitrogen.

Unlike phosphorus and pathogens, increases in nitrate and ammonia concentrations downstream of the Bridgeport Valley pasture were modest. Mean nitrate in the inflowing stream doubled from 8 to 16 μg/L following passage over the irrigated lands (Table 15). Ammonia showed a similar doubling from 1.7 to 4.4 μg/L. While a doubling of TP was considered large, the increases in nitrate should be much larger, given that the ratio of the two elements in fecal matter and urine is about 10:1 N:P by weight. Thus an increase in TP of 8 μg/L should be accompanied by an increase about 80 μg/L nitrate. Urea is quickly converted to ammonia in the soil and then usually converted to nitrate since bacteria can derive energy from these oxidation steps. The best estimate of bioavailable nitrogen for algae growth is total inorganic nitrogen (TIN), which is the sum of nitrate, nitrite, and ammonia. The measured and predicted TIN values above and below the stock grazing areas are shown in Table 18.

Table 18. Arithmetic mean nitrogen concentrations above and below areas of stock grazing in the Bridgeport Reservoir drainage in spring-early summer 2000. Data from USGS and Lahontan RWQCB. Units are μg/L as N or P for nutrients. TIN = nitrate + nitrite (if present) + ammonia, expressed as N. TIN gives the best estimate of bioavailable nitrogen for algae growth. Predicted TIN is based on measured TP using a N:P ratio of 10:1 by weight.

Site	**Nitrate**	**Ammonia**	**TIN**	
			Measured	Predicted
Above Valley	8.3	1.7	10	100
Below stock grazing areas	16	4.4	20.4	390

Measured TIN concentrations were lower below the stock grazing areas than would be predicted based on measured TP concentrations and the N:P ratio for urine and fecal matter. Since stock excrete both N and P there must be some loss or retention of TIN in the soil. Although ammonia can be retained by soils in the humus, nitrate is highly soluble and is easily lost from soils to groundwater (Canter, 1997). The Bridgeport Valley pastures have been used for about 150 years at approximately the same livestock densities. Thus, TIN should be in equilibrium. The most likely explanation is loss of nitrate by denitrification in saturated soil of the pasture. This explanation is supported by findings from excavations and soil cores in the Bridgeport Valley pastures, showing that their soils have been wet and alternately reducing and oxidizing seasonally for a long time (Bob Curry, pers. communication, August 2000).

5.4. BIOMASS OF LIVESTOCK IN THE BRIDGEPORT RESERVOIR DRAINAGE AND WATERBIRDS ON THE RESERVOIR SURFACE.

Eutrophication results from input of inorganic nutrients, regardless of source. Since Bridgeport Reservoir often hosts large flocks of waterbirds, the contribution of their excreta should be considered in relation to other sources. On a seasonal basis, the concentrations of the major nutrients in the reservoir are obviously controlled by internal loading in summer (Figs 8-9, 11-15). Thus the presence of birds would add slightly to this main nutrients flux but might be important on an annual basis.

A good estimate of the annual loading of bird excreta requires a more detailed estimate of bird presence, species composition, day and night behavior, and feeding sites than was possible with this project. However, a comparison can be made in terms of biomass of livestock in the drainage and birds on the reservoir. An exact number of livestock in the drainage was not available, but an approximate count of the number of livestock indicated about 10,000 head. Assuming an average weight of mature and young cows or horses as 1,000 pounds over the summer season, the approximate biomass of livestock is 5,000 tons. Counts of birds on the reservoir in the early morning in late summer were made. Since the birds tend to move during the count, an estimate of 1,000 to 4,000 birds was made by counting small areas and extrapolating to the whole reservoir. Erring on the highest side, as many as 4,000 birds might be present. Using an average duck weight of 2.5 pounds (mallard) the biomass of waterbirds was 5 tons.

6.0. DISCUSSION

6.1. THERMAL STRATIFICATION, SEDIMENT ANOXIA, AND THE AMPLIFICATION OF EUTROPHICATION.

The bottom-release design of the reservoir contributes to warm hypolimnion temperatures in Bridgeport Reservoir. Cold water that would remain year-round in the lake bottom is released downstream. In a lake, outflow is from the warmer, surface water leaving, the cooler layer intact. In dry years, Bridgeport Reservoir is shallow enough that the water mixes top-to-bottom in summer so that the sediments are heated by surface water. The warm water temperatures have a mixed impact on DO and eutrophication in the reservoir.

The water temperature in the hypolimnion (bottom water) during the short, thermally-stratified season was high (16-19 ^{o}C). A deep lake at the same altitude and sub-alpine climate would have hypolimnion temperatures in the range 0-6 ^{o}C. Given that surface temperatures ranged from 20-22 ^{o}C, a difference of only 3 to 6 ^{o}C from the hypolimnion, it is unlikely that stable stratification would have occurred for long. Warm hypolimnion water has two counterbalancing effects on DO and related impairments of beneficial uses. The warm bottom water and sediments recycle nutrients and toxicants much faster than in lakes with cold sediments, but a shorter period of stratification and accompanying sediment anoxia limits the period of nutrient recycling. In some sheltered reservoirs at lower elevations thermal stratification can last for up to nine months (i.e. from March to December for Upper San Leandro Reservoir, EMBUD, Oakland; volume = 40,000 af).

The sediment temperature measured in Bridgeport Reservoir (19 ^{o}C), is about 15 ^{o}C warmer than a similar conventionally-stratified lake where winter temperatures fall as low as they do in Bridgeport. A calculation of the increased nutrient cycling can be made based on Arrenius's Law. This generally applicable relationship for biochemical reactions states that an increase in temperature of 10 ^{o}C would cause a doubling of the rate of metabolic reactions. For Bridgeport Reservoir the law predicts bacterial recycling of nutrients and toxicants to be about three times more rapid than for a water body with cold hypolimnion sediments. Similar effects occurred with the much larger Camanche Reservoir (V = 425,000 af) during the 1986 drought and release of water for irrigation (Horne, 1989). In general, warm sediments are not good for water quality.

While the rate of nutrient recycling is enhanced due to the reservoir's bottom-release design, the top-to-bottom mixing also limits the period over which hypolimnetic anoxia can accelerate nutrient recycling. However, even the relatively short period of hypolimnetic anoxia is partially responsible for the extreme eutrophication in Bridgeport Reservoir. There is sufficient time for the anoxia to occur and for the release of bioavailable forms of nitrogen and phosphorus from the sediments to the water column. There may be some opportunity for balancing these two effects by managing the operational depth of the reservoir and the outflow rate between mid-June and mid-September. But, since both of these variables depend on both the amount of precipitation during the previous winter and the needs of downstream water users, there are considerable constraints if the reservoir is to be optimized for beneficial uses through operation of the reservoir. The most feasible alternative for reducing the internal recycling of nutrients within Bridgeport Reservoir would be to install a hypolimnetic oxygenation system. These systems have been shown to be a relatively low-cost method of substantially reducing nutrient recycling without upsetting temperature stratification in small and large reservoirs, including Camanche Reservoir (EBMUD, Mokelumne River).

6.2. BLUE-GREEN ALGAL BUOYANCY AND AMPLIFICATION OF EUTROPHICATION IN BRIDGEPORT RESERVOIR.

Blue-green algae float, a trait which has enhanced their success at Bridgeport Reservoir because of its bottom-release design. Water released from the bottom of the reservoir usually has lowest concentrations of blue-green algae. Bottom releases effectively concentrate buoyant living algae in the upper waters in the reservoir's photic zone. Increases in the oxygen demand in the sediments will occur when the algae die and sink to the reservoir bed but these algae will be sucked out of the discharge only if they sink close to the outflow intake. In most of the deep water, the outflow current is inadequate to suspend sediments unless the reservoir is almost dry (Entrix, Inc. 1990).

6.3. WERE THERE DIFFERENCES IN THE DEGREE OF EUTROPHICATION IN 1989 AND 2000 IN BRIDGEPORT RESERVOIR?

There were no important differences in the mean or maximum chlorophyll *a* levels in the reservoir in 1989 and 2000. In addition, water clarity and nutrients were similar and high in both years. Nonetheless, and important difference in the dominant nuisance algae occurred over this period.

Aphanizomenon, the common blue-green algae characteristic of very eutrophic waters, dominated the reservoir phytoplankton in 1989. A brief survey of the reservoir in summer in 1992 showed that *Aphanizomenon* still dominated the system and was present in very large amounts. However, by 2000 the reservoir was dominated by a new blue-green alga, *Gleotrichia*, another large colonial species that, like *Aphanizomenon*, controls its position in the water column by buoyancy regulation and can fix atmospheric nitrogen. *Aphanizomenon* was still present, even common in 2000, but did not dominate the algal biomass as in 1989 and 1992. In the 1990s *Gleotrichia* invaded shallow Clear Lake, California which is situated in the Coast Range about 150 miles north of the San Francisco Bay Area. Clear Lake had been dominated by *Aphanizomenon* since at least 1970 and is still common. Not as much is known about *Gleotrichia* as the other common species of blue-green nuisance algae. It is not certain if the presence of *Gleotrichia* in Bridgeport Reservoir indicates an improvement or a degradation.

6.4. Support for biomanipulation as a viable method for improvement of water quality in Bridgeport Reservoir.

Biomanipulation is one of several common lake and reservoir management methods. It is one of the most sustainable restoration techniques for shallow lakes and relies on natural methods to reduce the characteristics of eutrophication. The essence of the method is to facilitate the establishment of rooted aquatic plants in shallow water areas. These plants stabilize the sediments, induce the growth of attached algae that may out-compete nuisance phytoplankton for nutrients, and provide shelter for zooplankton that eat phytoplankton.

Typically, the establishment of rooted macrophytes is established by removing benthivorous fish such as carp that disturb the sediment and prevent macrophyte recruitment. In Bridgeport Reservoir, rooted macrophyte populations appear to have increased during the 1990s, with coincident improvements in water quality at the shallow, southern end of the reservoir. The increase in macrophytes may have occurred with the import of sediments in one of the 1990s El Nino storm events, according to some local anglers.

A test of beneficial use restoration using biomanipulation is to estimate the amount of nuisance blue-green algae and water clarity in the open water and compare that with the same parameters measured near the macrophytes. As shown below in Table 19 there is a distinct improvement in the macrophyte region.

Table 19. Feasibility of biomanipulation for Bridgeport Reservoir. Spatial variation of limnological variables in Bridgeport Reservoir in 2000 in terms of support or not for biomanipulation as a sustainable option. TP is shown in µg/L.

Parameter	TP in deep water (ug/L)	TP in shallow water (ug/L)	Supports biomanipulation
Secchi depth (m)	1.5	>2.6	Yes
TP (µg/L)	380, 326	240	Yes

Thus, biomanipulation may be a viable active management option for improving water quality within the shallow, littoral portions of Bridgeport Reservoir. However, provision will need to be made for good boat access if further macrophyte stands are encouraged. Weed harvesting using simple mechanical cutters attached to small boats is a normal method of lake management and could be incorporated into biomanipulation at Bridgeport Reservoir.

7.0 GLOSSARY

Anoxic. The absence of dissolved oxygen (DO) in water. Often called anaerobic. In lakes and reservoirs DO lower than 2 mg/L gives functional anoxia since few organisms can live at such low oxygen concentrations. Anoxic conditions in the deep water produce dramatic changes in lake water and sediment chemistry.

Autotrophic. Organic matter produced using only sunlight and inorganic chemicals such as carbon dioxide, phosphate and nitrate. Most algae and higher plants are autotrophic. Opposite of heterotrophic where bacteria, fungi and animals use pre-formed organic matter created by plants and algae.

Benthivorous. Feeding on benthic (sediment-dwelling) food. Usually applied to fish such as catfish or carp.

Biomanipulation. A sustainable management technique for eutrophic shallow lakes. The essence of the method is to enhance several natural processes that lead to clear lake water. The main components of biomanipulation are the enhancement of protection of the large zooplankter *Daphnia* from fish predation, the restoration of submerged weed beds, the reduction of large bottom-grubbing fish (usually carp), and the maintenance of a higher ratio of large predatory fish to small zooplankton-eating fish.

Blue-green algae (Cyanobacteria). The group of algae that dominates eutrophic lakes. The main reason for their success in the phytoplankton of eutrophic lakes is the ability to regulate their buoyancy in the warm well-stratified conditions of summer. Blue-green algae do well in less productive conditions too but if nutrients are high, they become very abundant. Cyanobacteria differ greatly from other bacteria in their ecological role since they photosynthesize using carbon dioxide and release oxygen just like higher plants. Other bacteria do not produce oxygen. Cyanobacteria share the simple cell structure of bacteria but since algae are not a scientific classification like bacteria, either name Cyanobacteria or blue-green algae can be used.

Dimictic. Refers to the number (two) of top to bottom mixing seasons experienced by the lake. One mixing season is in the fall after turnover but before ice cover. The second mixing season is in early spring when the ice has melted but the lake has not warmed enough for summer thermal stratification. Most deeper lakes (>10m) away from the coast and in the temperate regions are dimictic.

Epilimnion. The warmer, less dense layer of water that floats over the deeper cooler and denser water layers in a lake during the summer thermal stratification period. In winter the cold water of the lake usually mixes easily by wind action unless ice cover is present.

Eutrophic. A condition in a lake characterized by an abundance of nutrients. Eutrophic lakes are distinguished by low dissolved oxygen in deep water, green water color, large crops of phytoplankton, especially blue-green algae, low water clarity and large crops of some fish (e. g. carp, bluegill).

Humic acids/ humic substances. Humic acids are naturally occurring, complex and heterogeneous refractory organic compounds that usually impart a tea color to natural waters. Humic substances are the remnants of the decomposition of leaves and twigs and usually have a high molecular weight. In nature they play a role in chelating possibly toxic metal ions and in shading shallow water organisms from dangerous UVB radiation, especially at high elevations.

Hypolimnion. The deep cold and dense layer that underlies the epilimnion.

Thermocline. The region of sharpest temperature gradient with depth. The thermocline region, which is called the metalimnion, separates epi- and hyp-limnion and prevents nutrients from passing from the deeper to the shallower lake waters.

Limnology. The study of enclosed bodies of water. Primarily lakes, reservoirs, rivers, streams, wetlands and estuaries.

Macrophytes. The community of large plants that occur in and around lakes and wetlands. Most macrophytes are rooted (e. g. pondweeds and cattails) but some float (e. g. duckweed). Macrophytes can be submerged, floating, emergent or sometimes show mixtures of these locations.

The mixed layer. The upper water layer of a lake that is regularly stirred by the wind. The mixed layer is the epilimnion in stratified lakes.

Oligotrophic. The opposite of eutrophic. A condition in a lake characterized by a paucity of nutrients. Oligotrophic lakes are distinguished by blue water color, high dissolved oxygen in deep water, small crops of phytoplankton, high water clarity and small crops of fish.

Phytoplankton. The freely suspended algal population. Most phytoplankton are microscopic but can sometimes be seen when they form colonies of thousands or even millions of individuals (e. g. the large colonial blue-green algae in Bridgeport Reservoir are sometimes the size of peas and are easily seen with the naked eye). Most phytoplankton cannot swim or move independently of the water currents. However, some larger ones such as blue-green algae and dinoflagellates can move actively with various special mechanisms.

Piscivore:planktivore ratio. The ratio of fish-eating fish (piscivores) to zooplankton-eating fish (planktivores). If the ratio is too high then zooplankton are reduced and cloudy algae-filled water can occur. Maintaining the correct ratio is a key to the biomanipulation lake management strategy.

Polymictic. Mixing of the lake top-to-bottom on more than two occasions per year. Shallow lakes (~5m) are usually polymictic since the wind can stir the water to this depth. Polymictic lakes are usually more eutrophic than monomictic or dimictic lake with the same nutrient loadings.

Thermal stratification. The term that describes the separation of a lake in summer into a warm upper zone (epilimnion) and a cooler, denser deep layer (hypolimnion). Warm water is less dense that cool water so the two layers are stable and the wind cannot mix them together. Thermal stratification occurs in all kinds of water bodies but is only stable over the summer in lakes that are deeper than about 10m.

Zooplankton. The freely suspended small animal population that eats phytoplankton and is food for small fish. The smaller zooplankton are microscopic and have limiting swimming ability. The mid-sized and larger zooplankton are just visible to the human eye and migrate vertically in the lake each day to feed in the epilimnion at night but move to deeper darker waters to avoid fish predation each day. If the deep water is toxic (no oxygen) then these zooplankton must hide in submerged macrophyte beds or perish.

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FIGURES

Figure 1. Location map: Bridgeport Reservoir watershed. Source: Entrix, Inc. 1990.

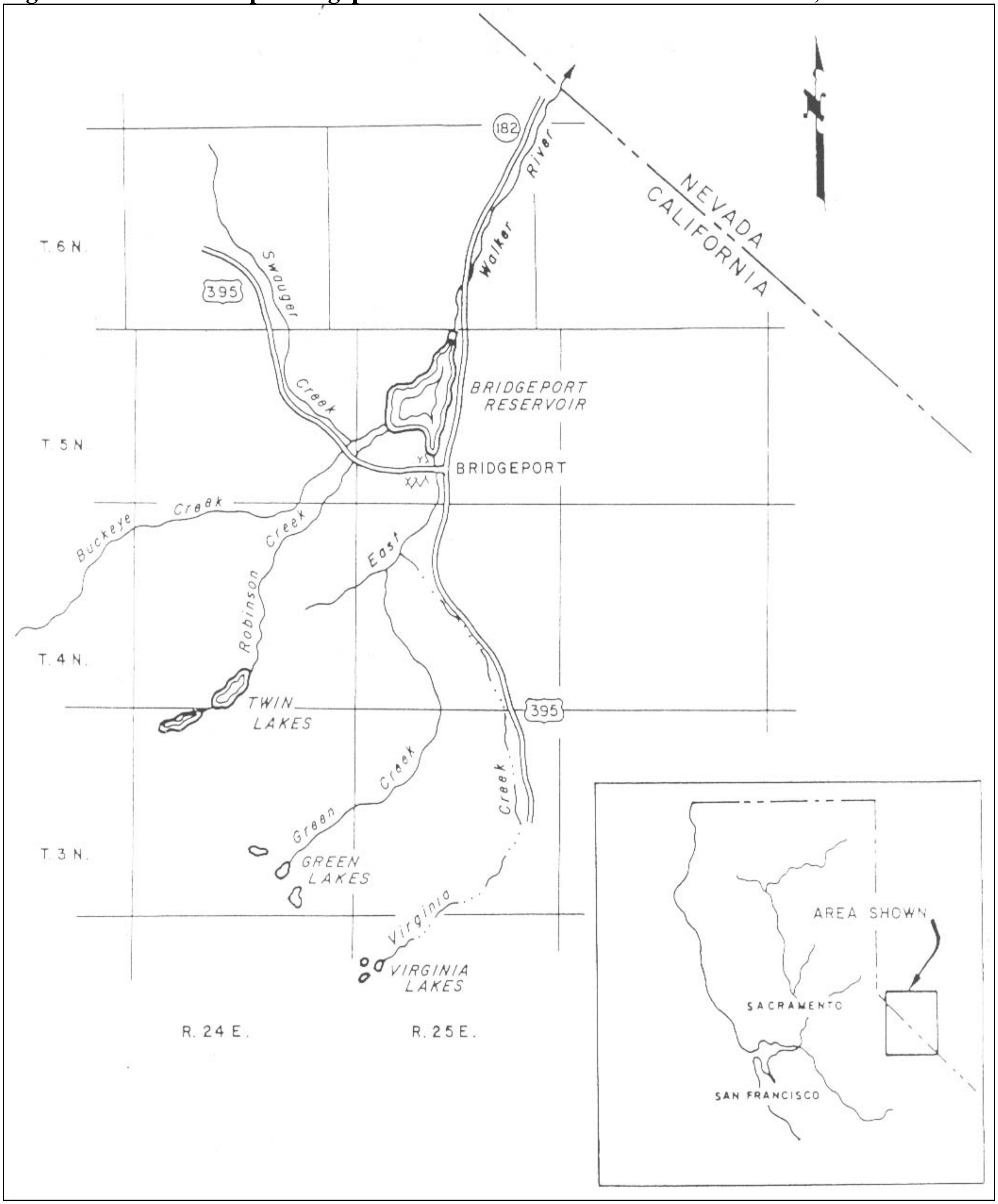


Figure 2. Bridgeport Reservoir depth contours from bathymetric survey completed in June 1989. Source: Entrix, Inc. 1990.

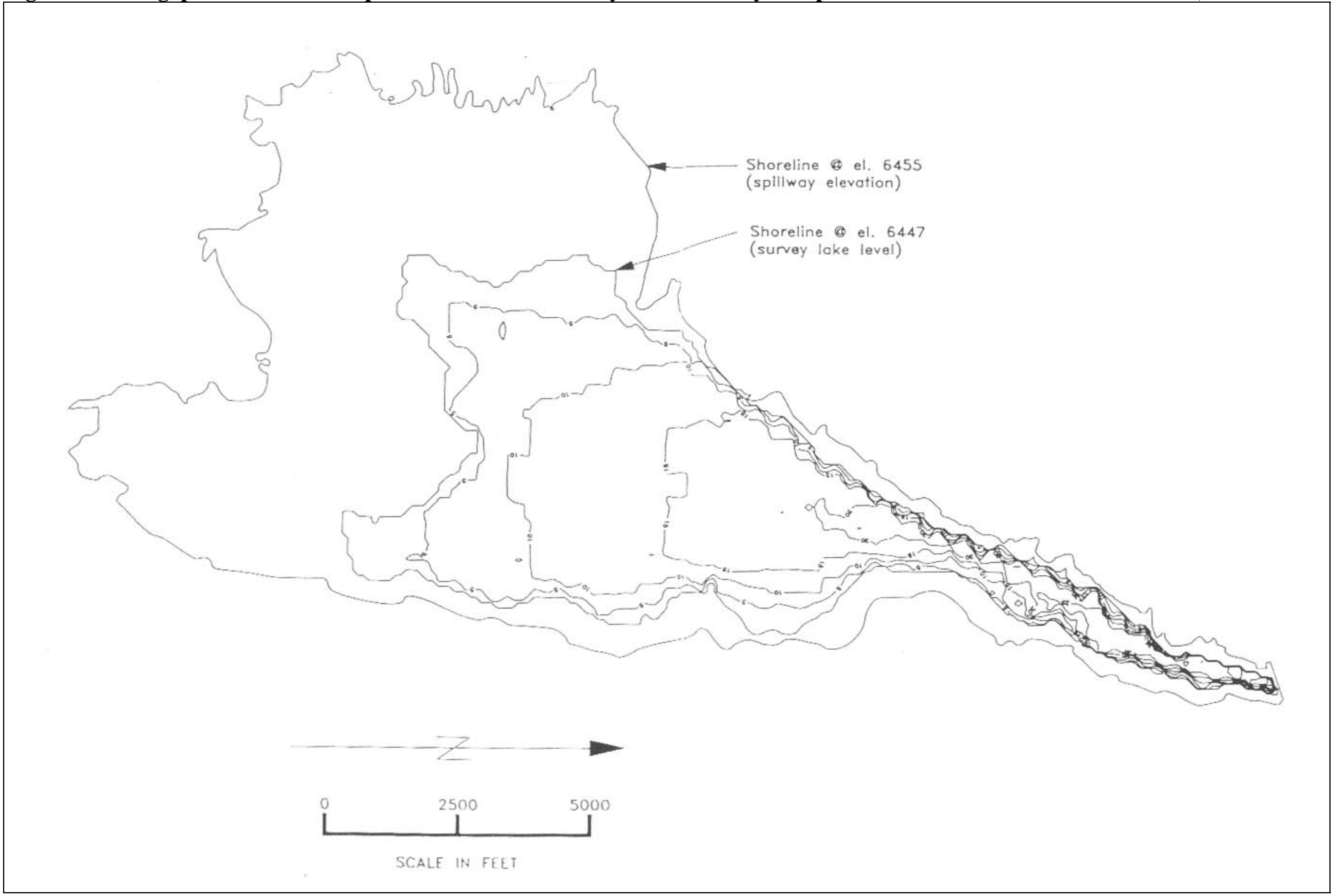


Figure 3. Water line map of Bridgeport Reservoir. Note that the south end of the reservoir is used as pasture when the reservoir is not full. Source: Entrix, Inc. 1990.

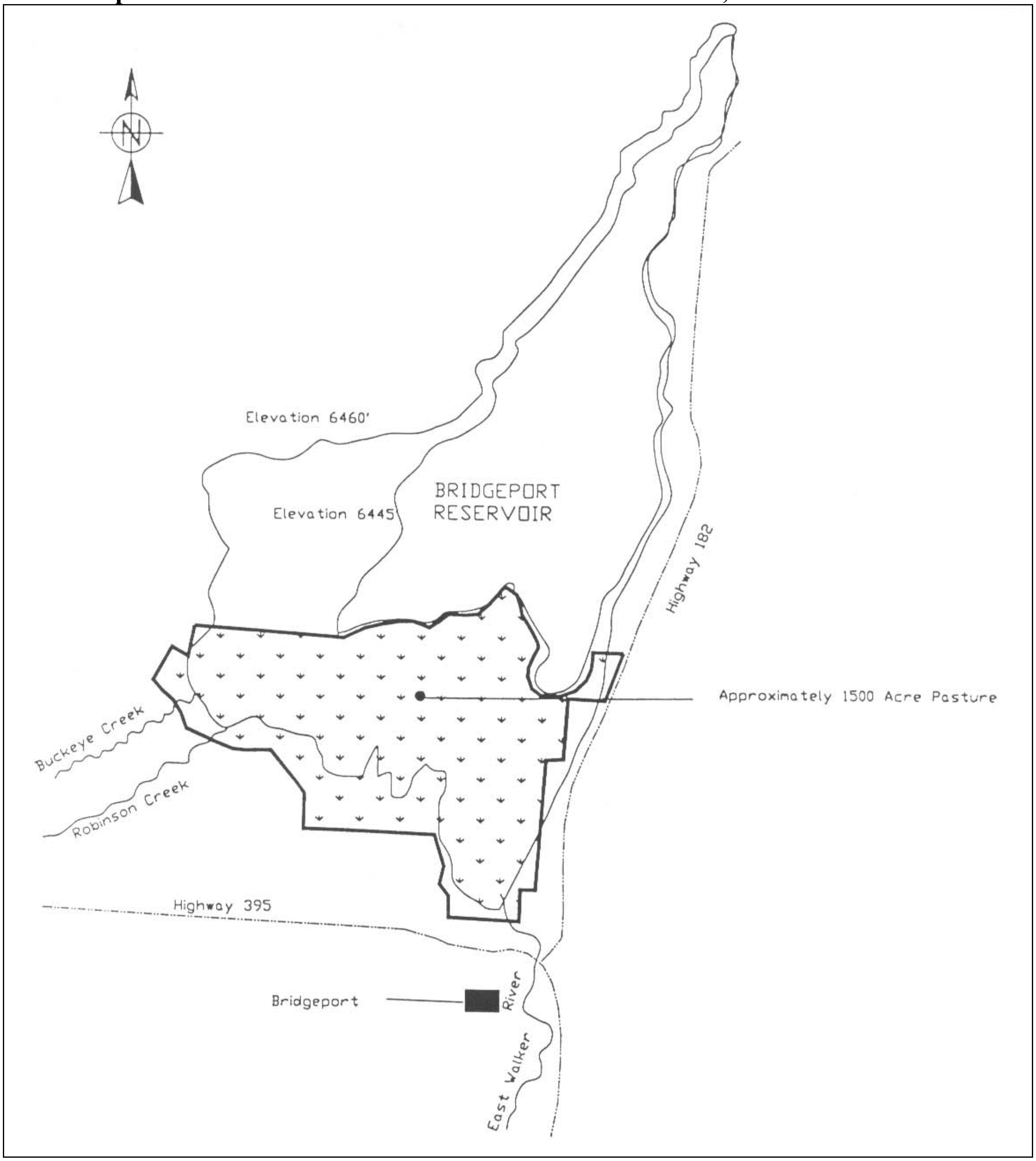


Figure 4. Location of the five limnological monitoring index stations at Bridgeport Reservoir for the 2000 study.

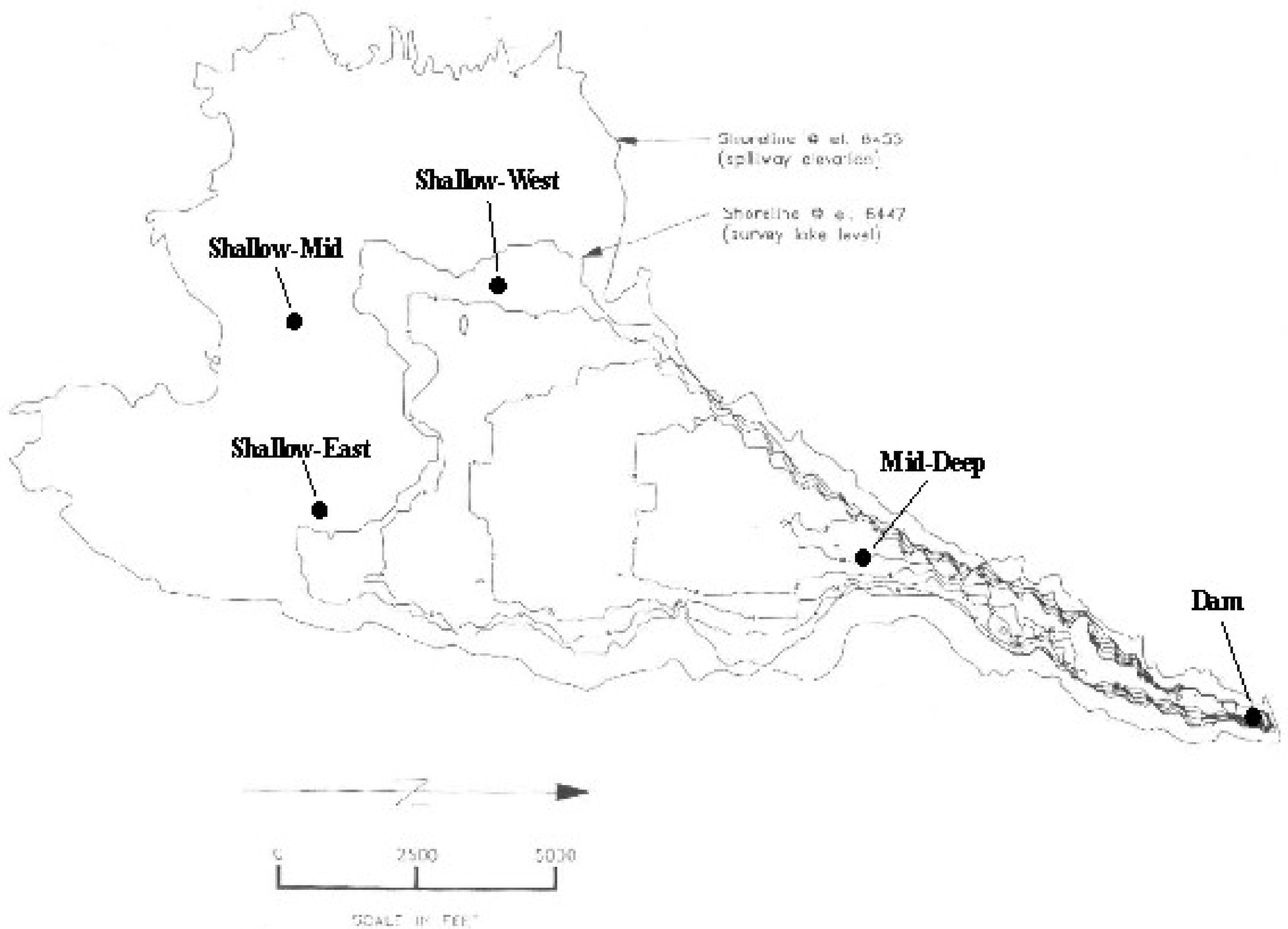


Figure 5. Volume-depth bathymetric curve for Bridgeport Reservoir. Based on data from the 1989 bathymetric survey (Entrix, Inc. 1990).

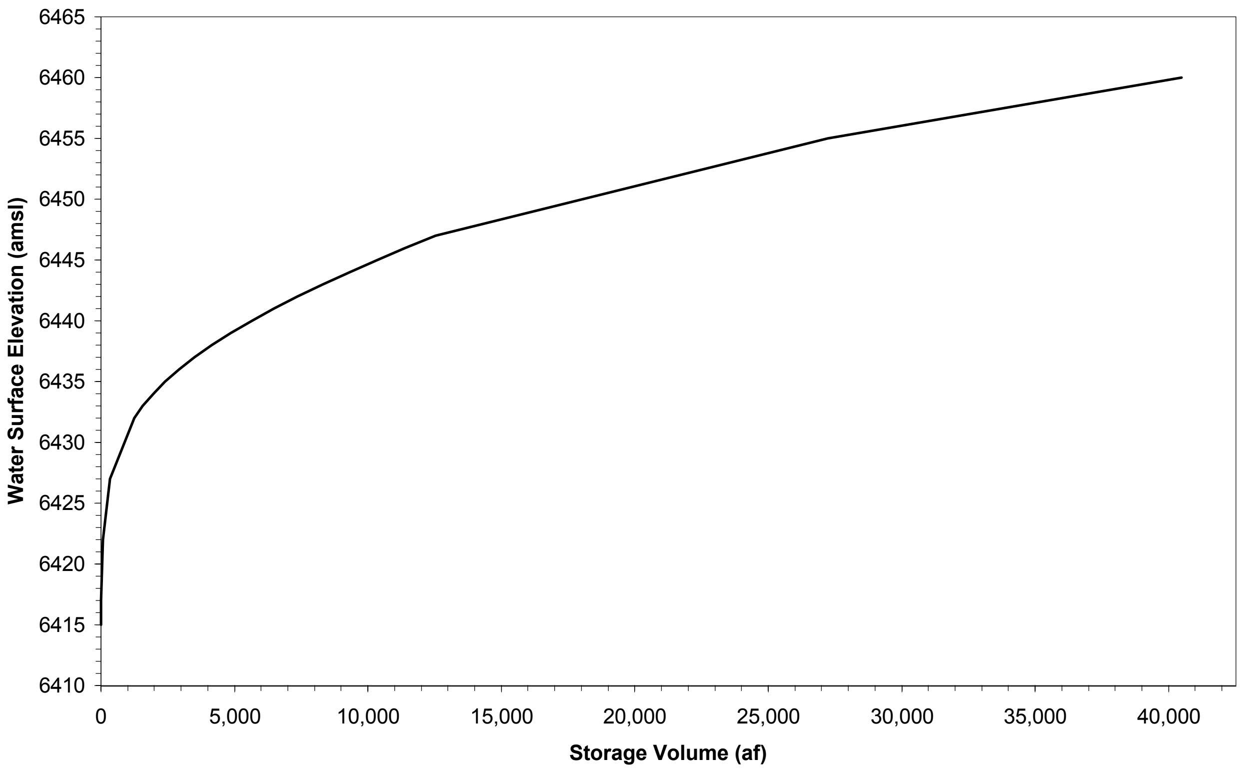


Figure 6. Experimental apparatus used for Bridgeport Reservoir sediment nutrient release experiments in 2000.

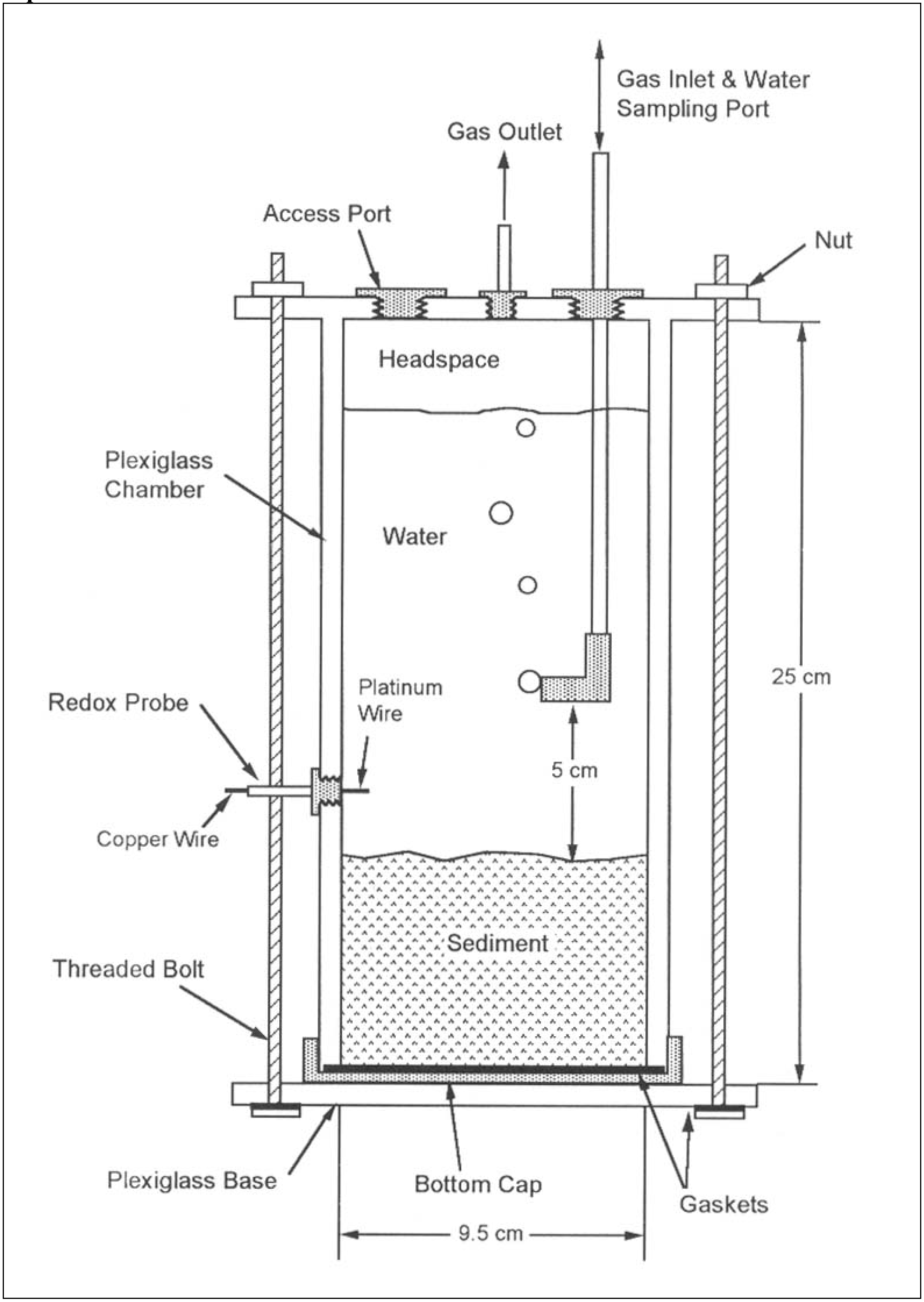
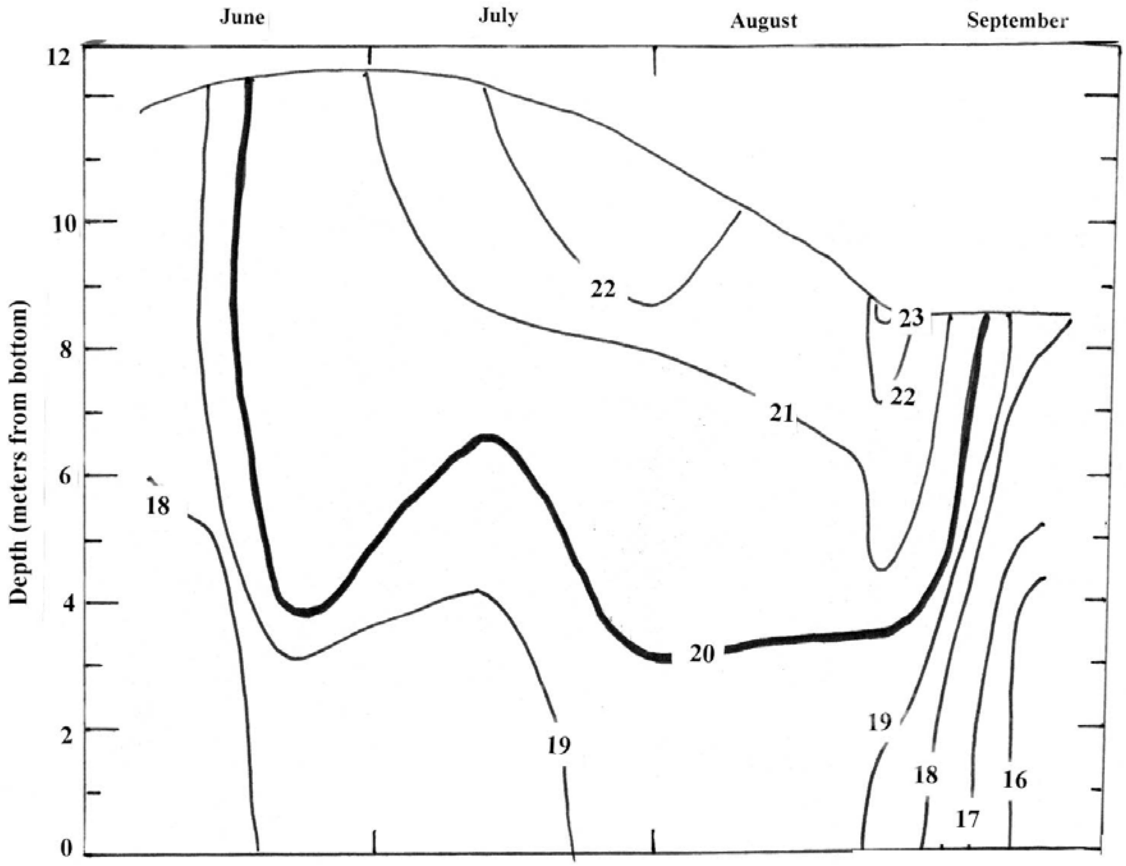


Figure 7 a-b. (a) Temperature isotherms (lines every ^{o}C), and



and (b) dissolved oxygen isopleths (lines every mg/L) for Bridgeport Reservoir in 2000.

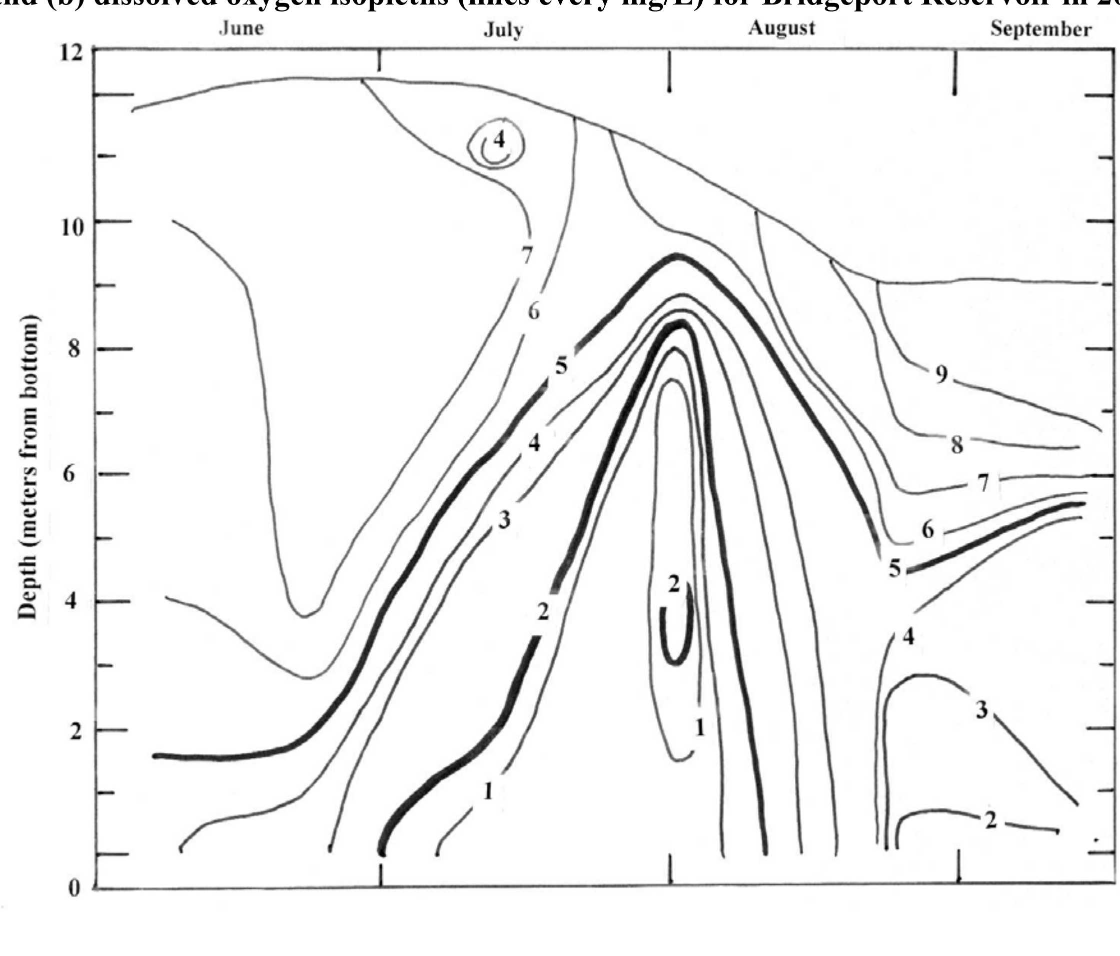
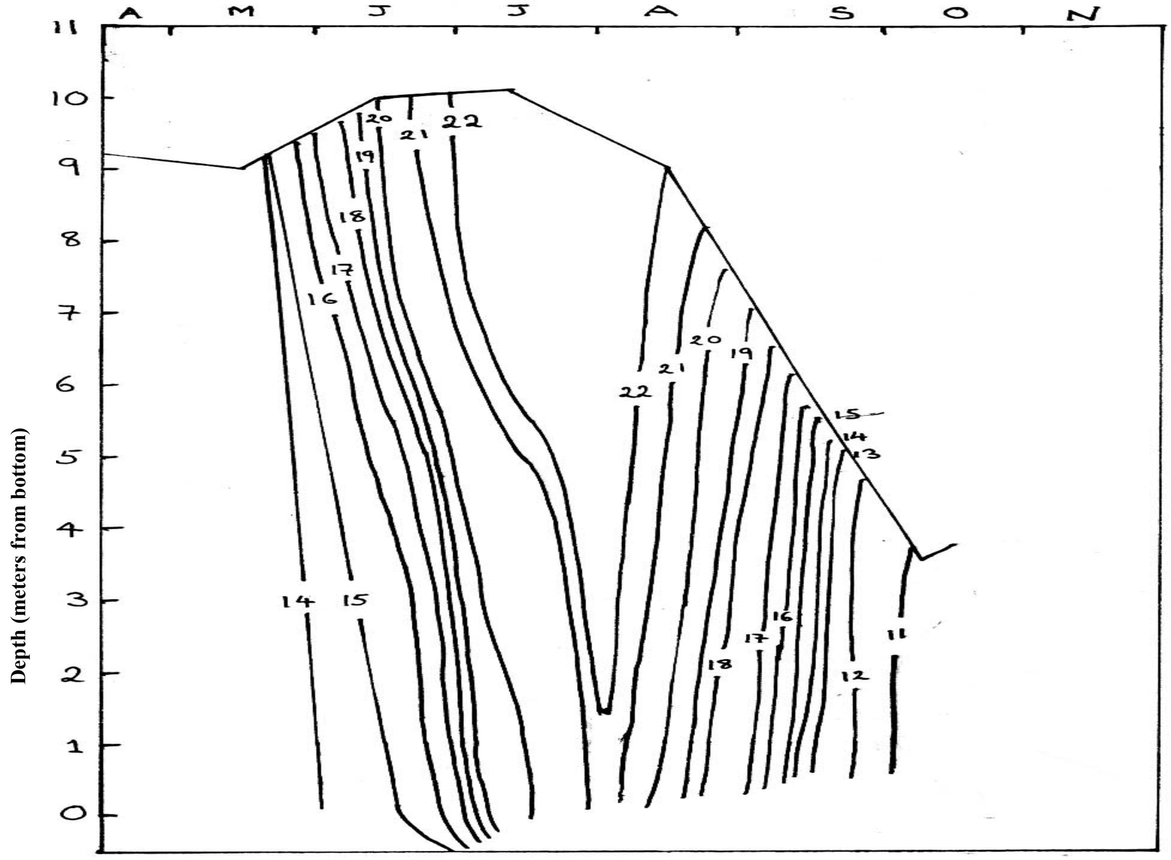


Figure 8 a-b. Comparison of thermal stratification in (a) 1989 (polymictic condition), and



(b) 2000 (weak dimictic condition) for Bridgeport Reservoir.

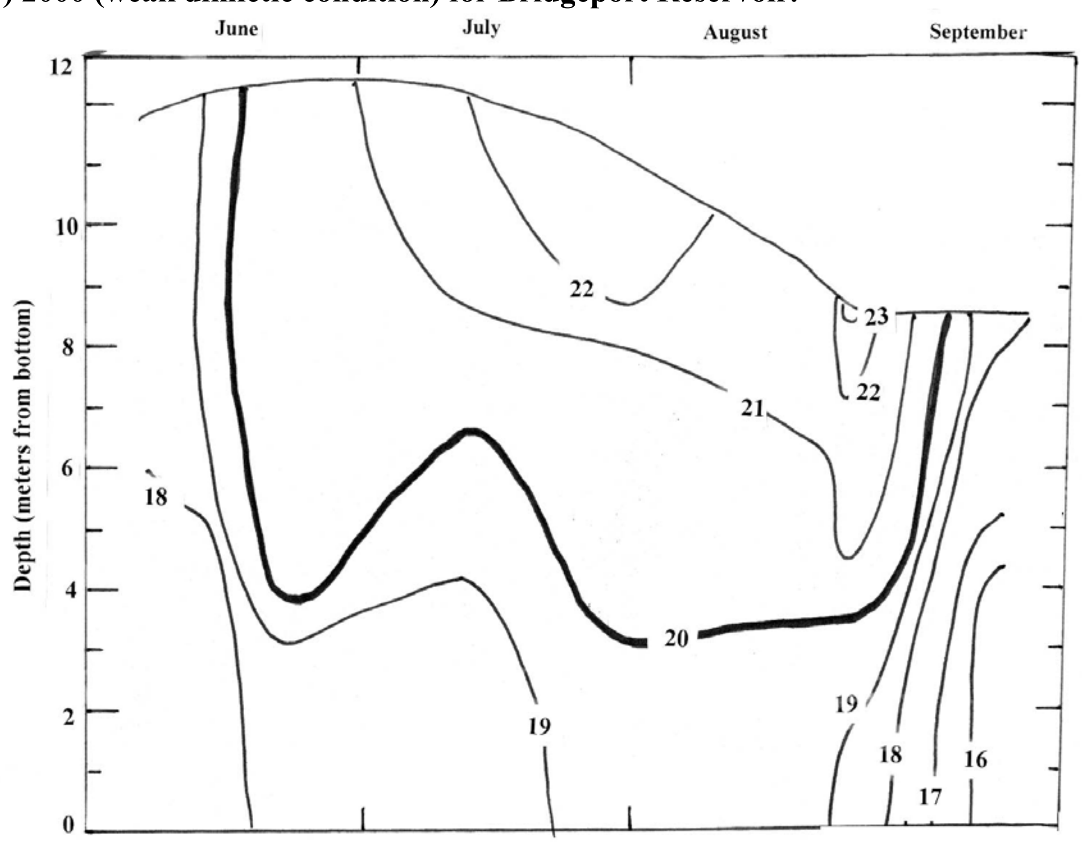
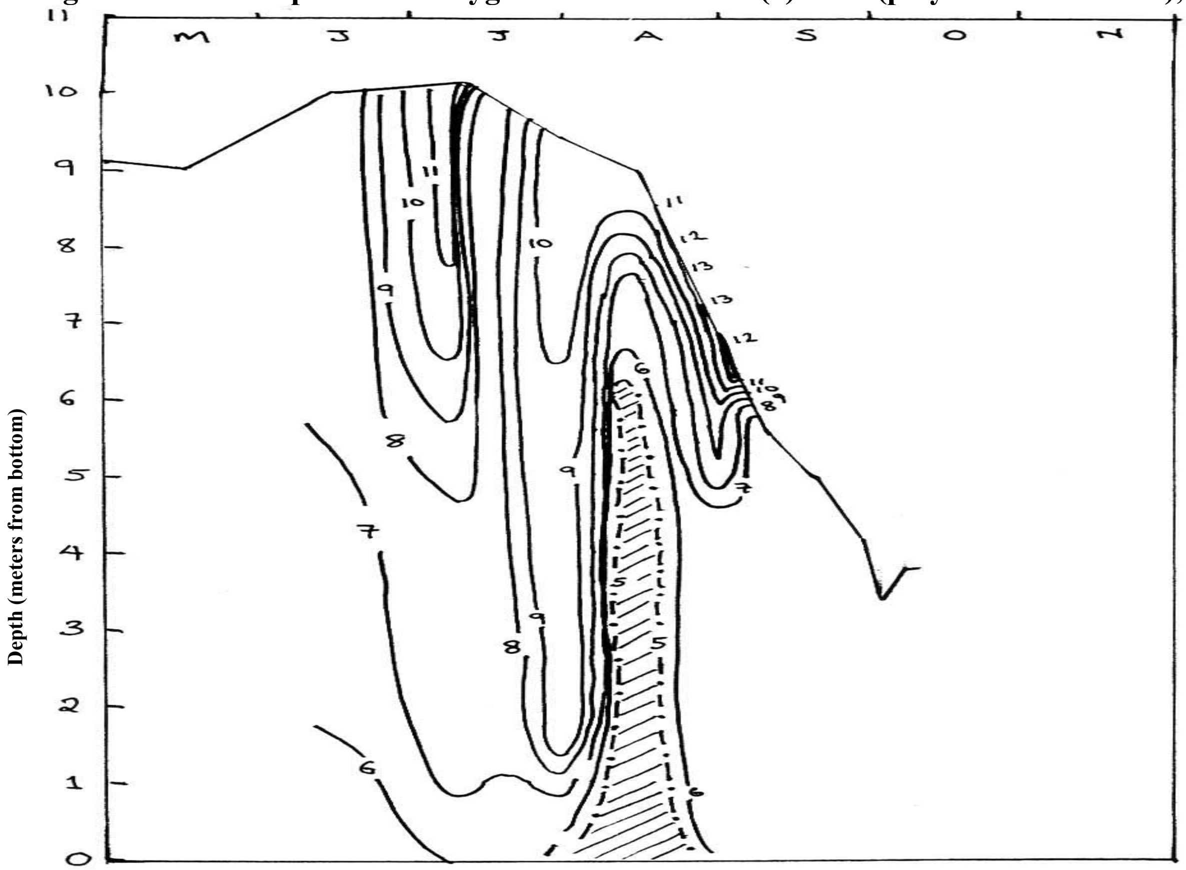


Figure 9 a-b. Comparison of oxygen stratification in (a) 1989 (polymictic condition), and



(b) 2000 (weak dimictic condition) for Bridgeport Reservoir.

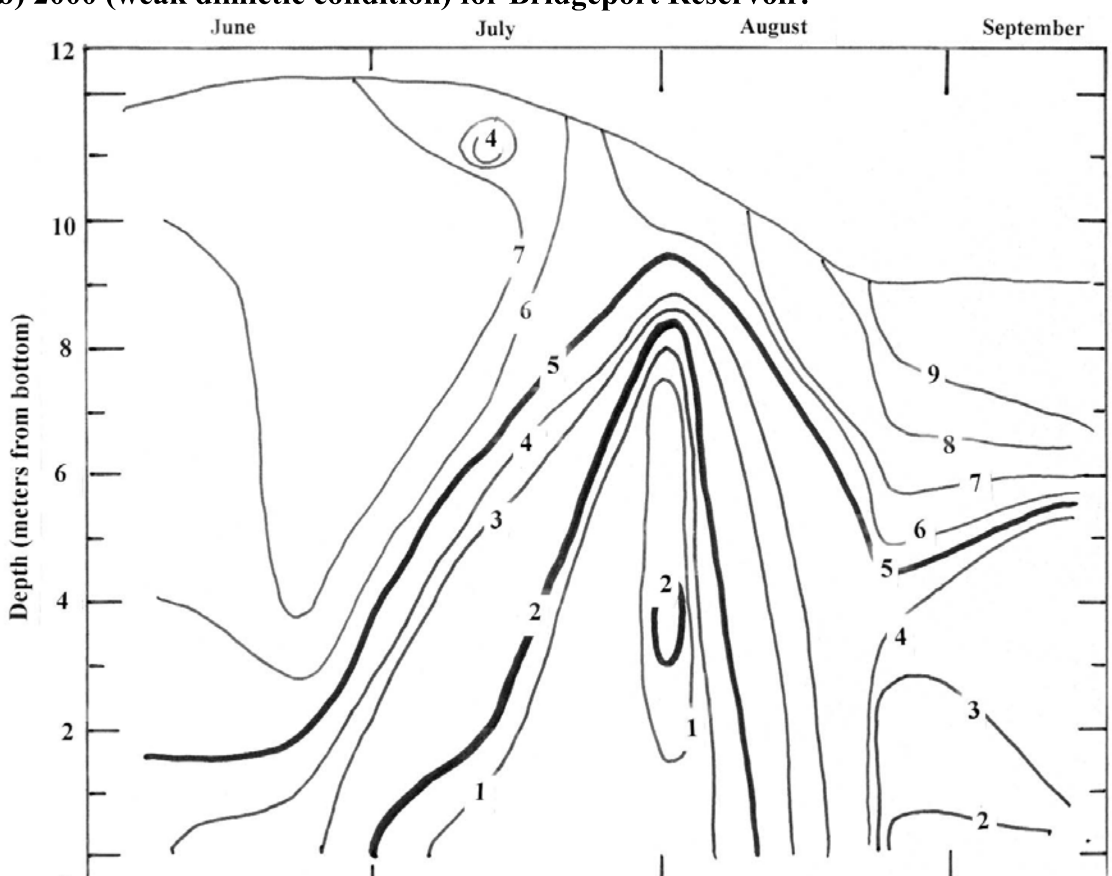


Figure 10. Seasonal variation in water clarity (Secchi disc depth) at Bridgeport Reservoir in 2000.

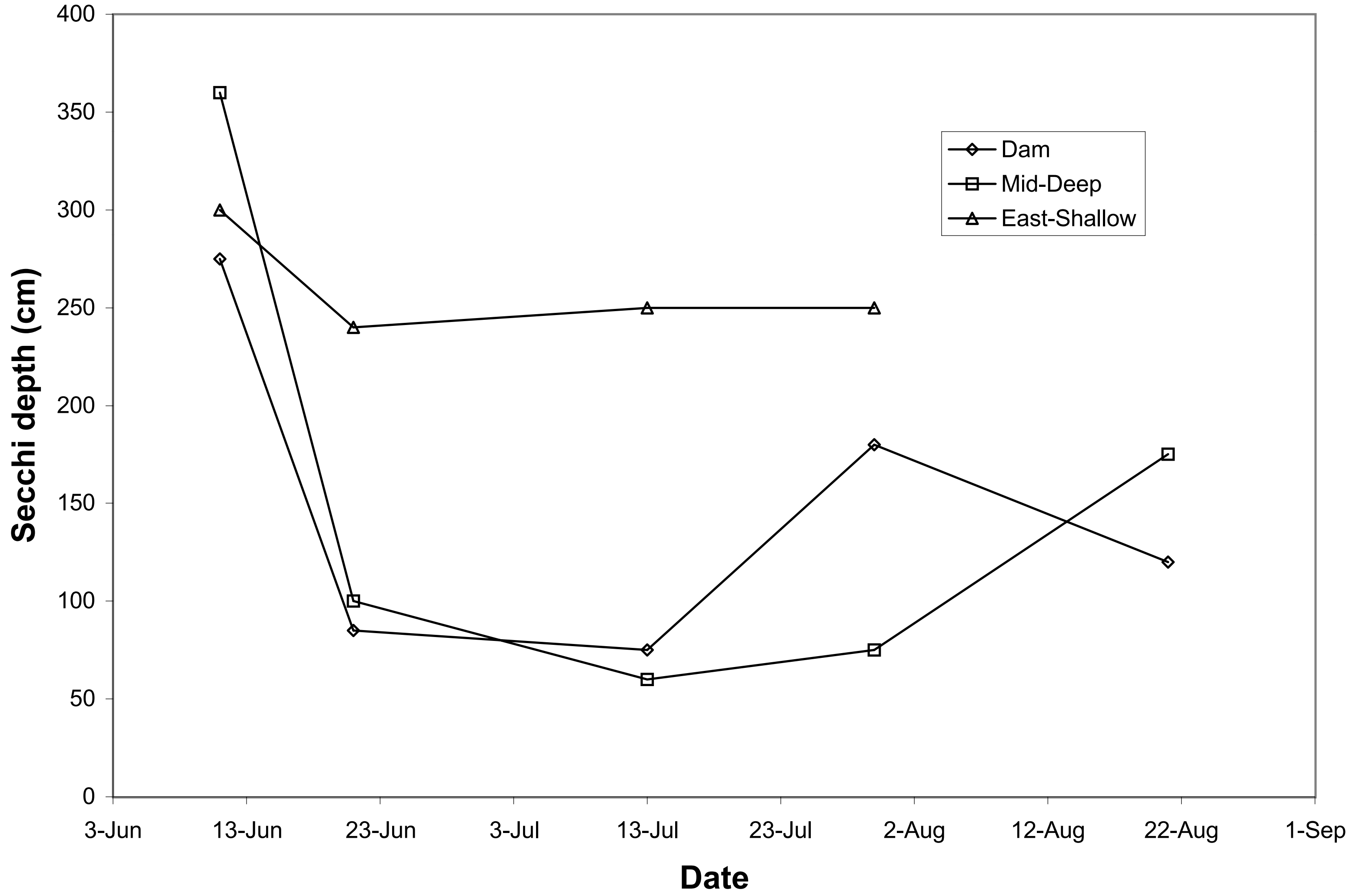


Figure 11. Seasonal variation in total phosphorous (TP) in surface water at three index stations at Bridgeport Reservoir in 2000. Both deep and shallow waters are high in TP, unlike ammonia, suggesting input from the creeks as well as internal loading.

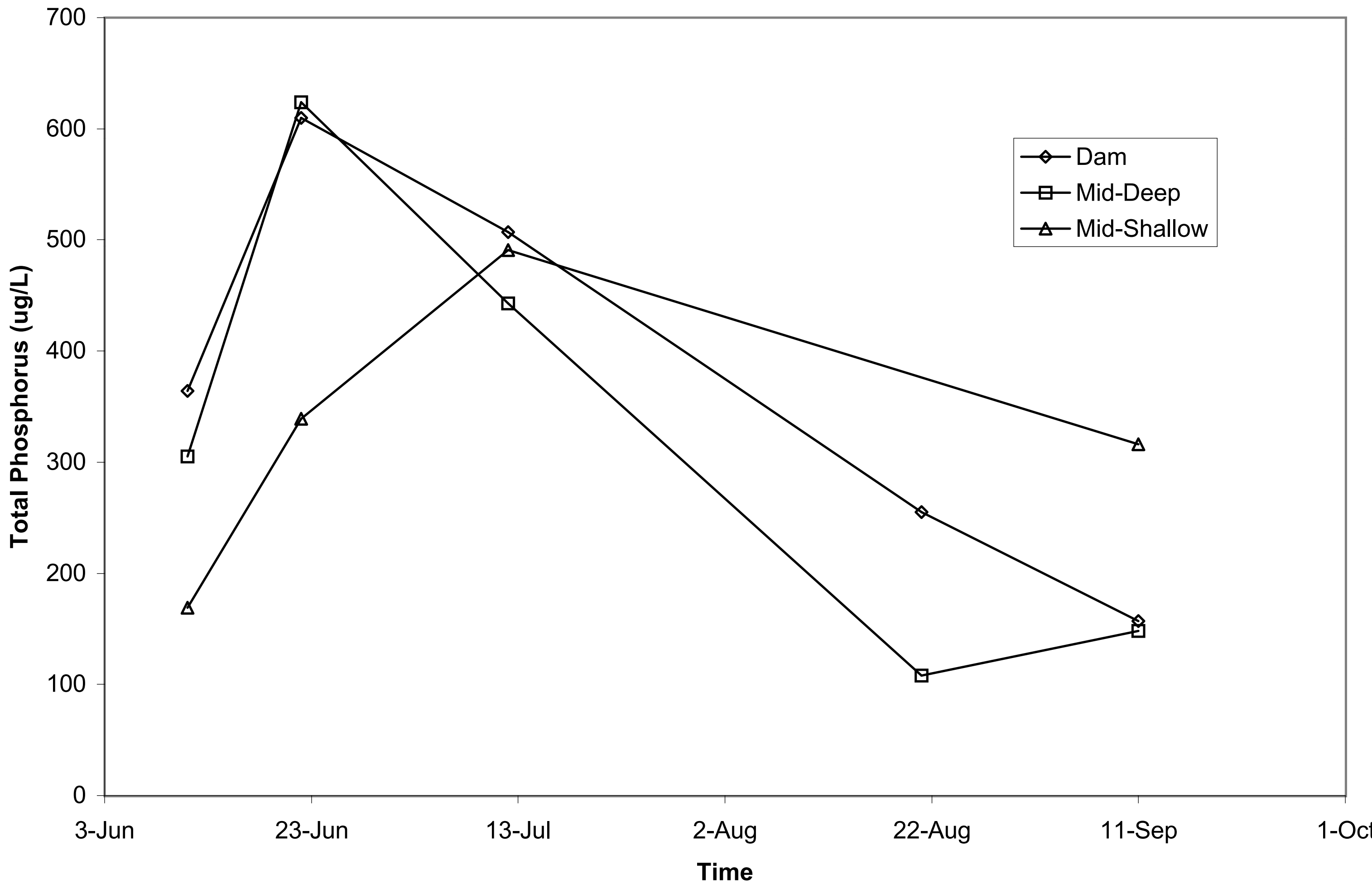


Figure 12 a-c. Seasonal and depth variation in total phosphorus at five index stations at Bridgeport Reservoir in 2000.

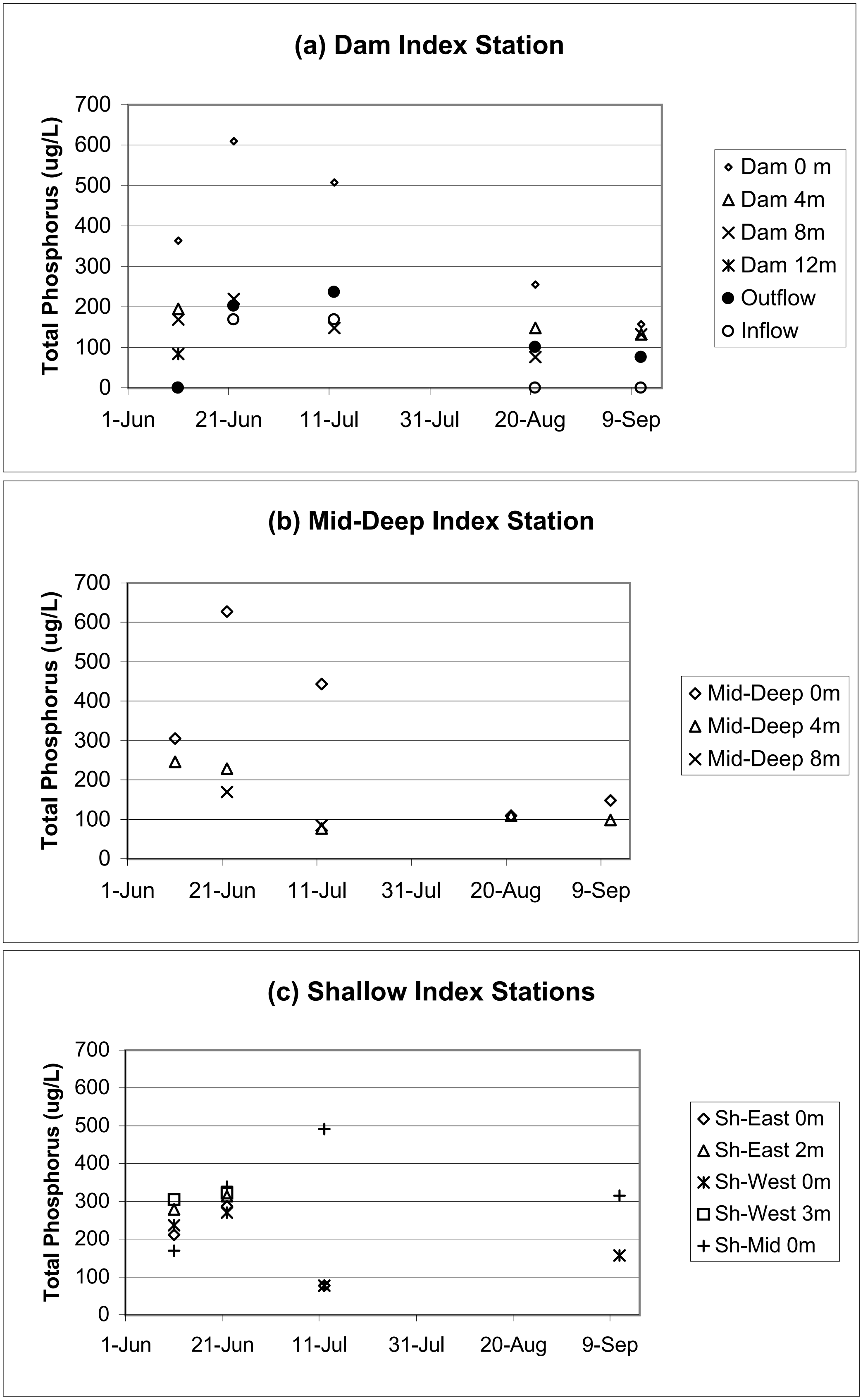


Figure 13. Seasonal variation in ammonia in surface water at three index stations at Bridgeport Reservoir in 2000. The flux of ammonia is coincident with the period of benthic anoxia in the deep water.

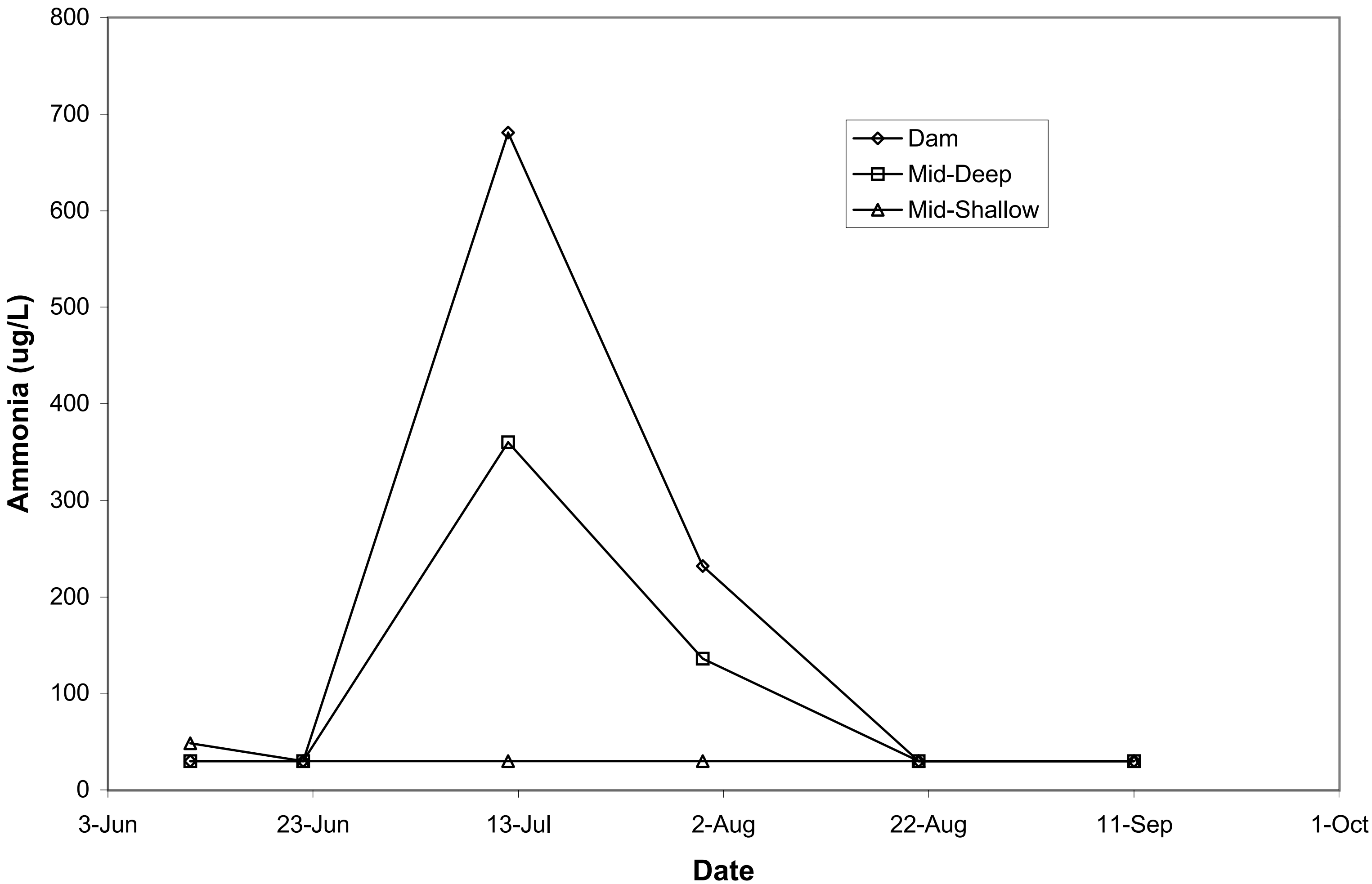


Figure 14. Seasonal and depth variation in ammonia at the dam index station in Bridgeport Reservoir in 2000.

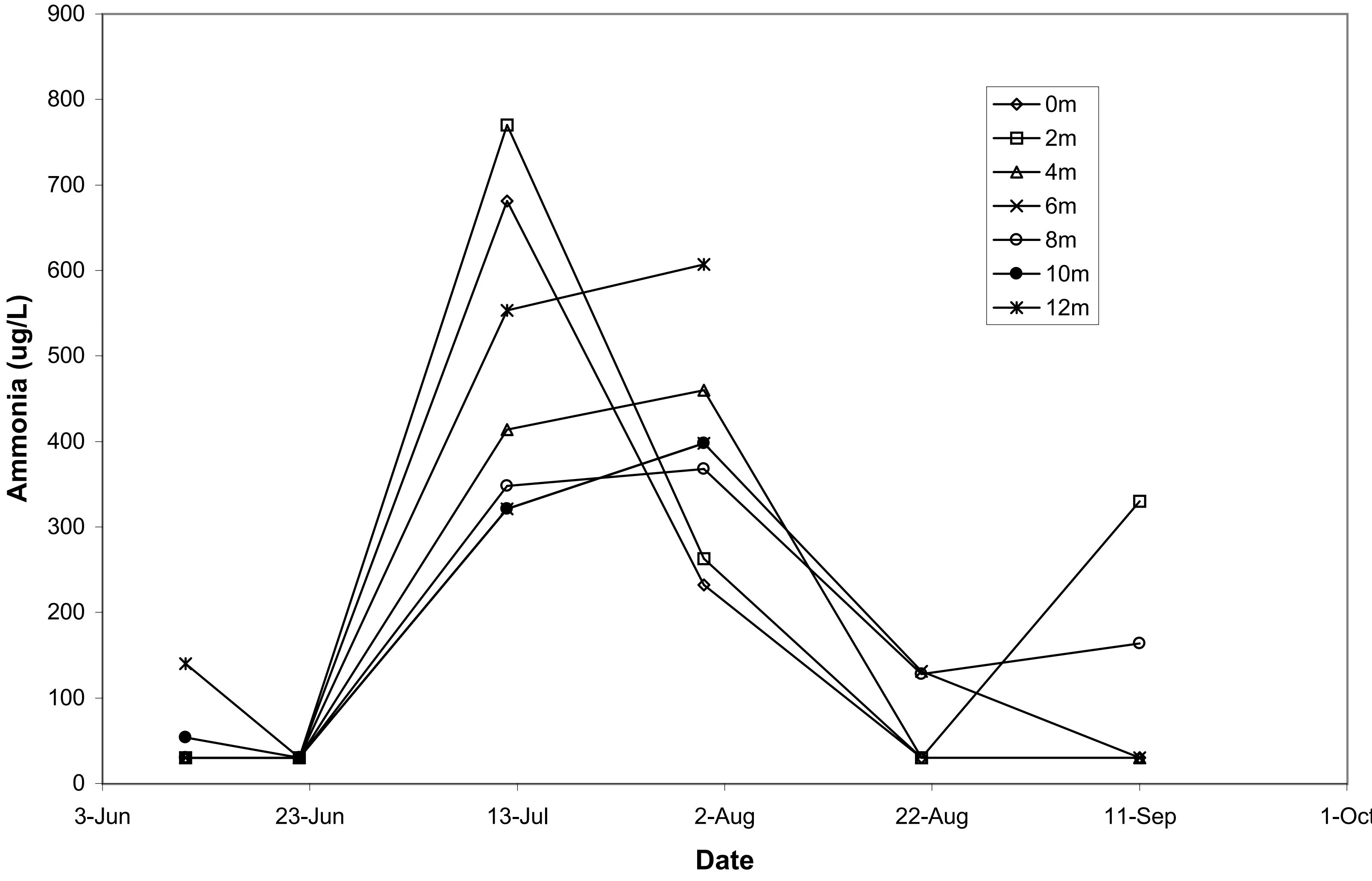


Figure 15. Seasonal variation in chlorophyll *a* in surface water at three index stations at Bridgeport Reservoir in 2000.

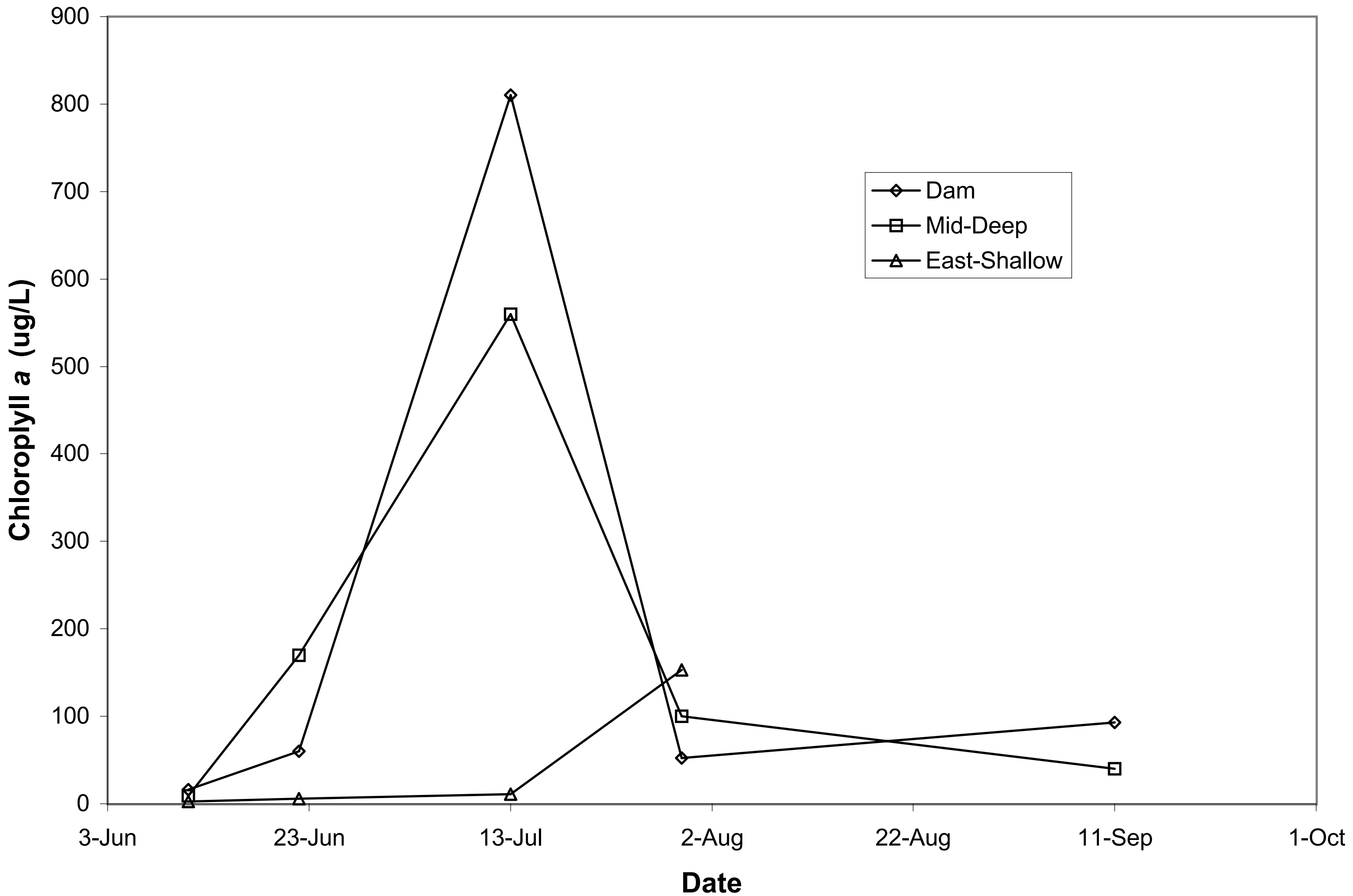


Figure 16. Seasonal variation in chlorophyll *a* in surface water at the dam and three shallow index stations at Bridgeport Reservoir in 2000.

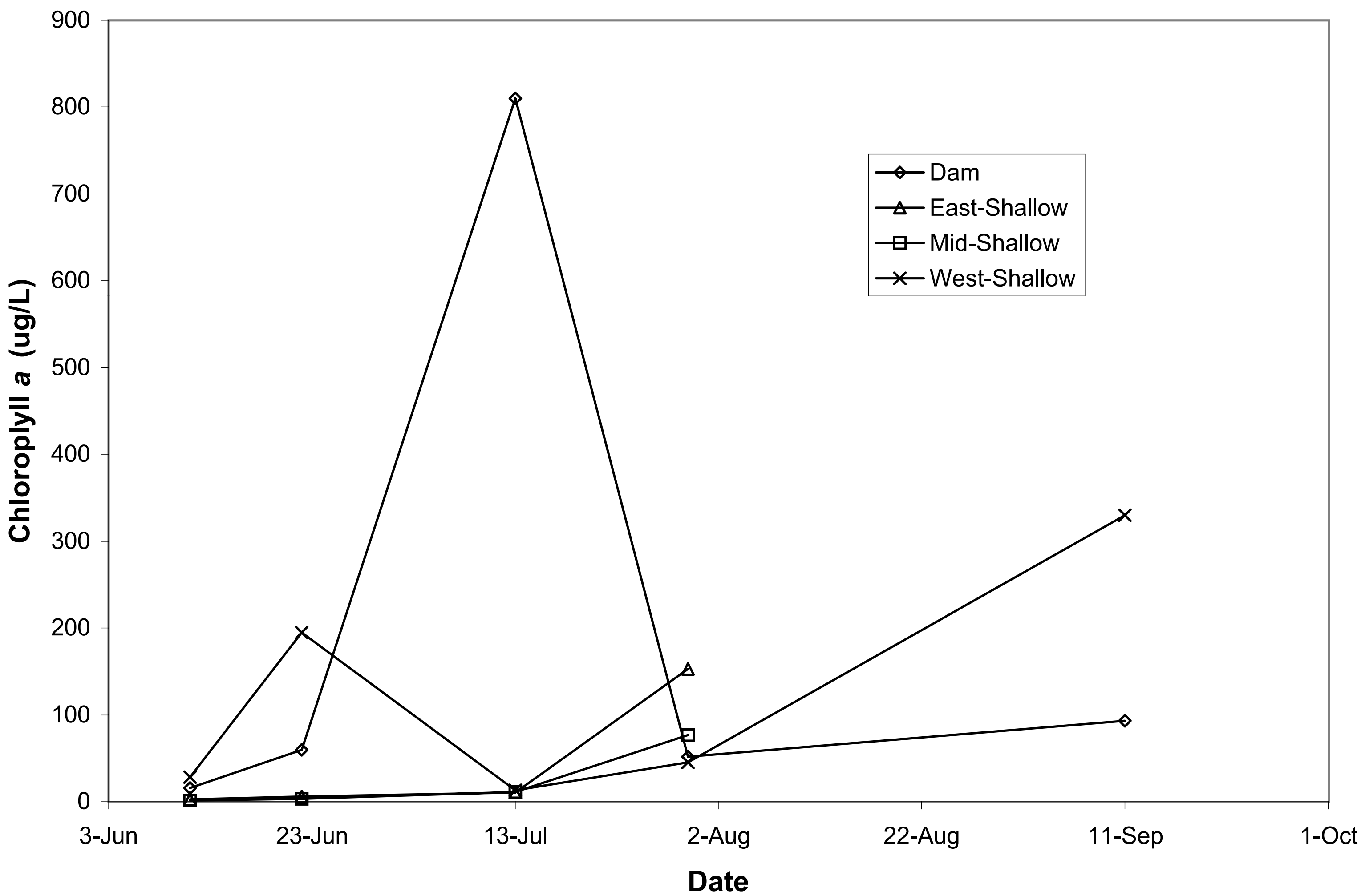


Figure 17. Map of stations used for the synoptic survey on 21 August 2000 at Bridgeport Reservoir.

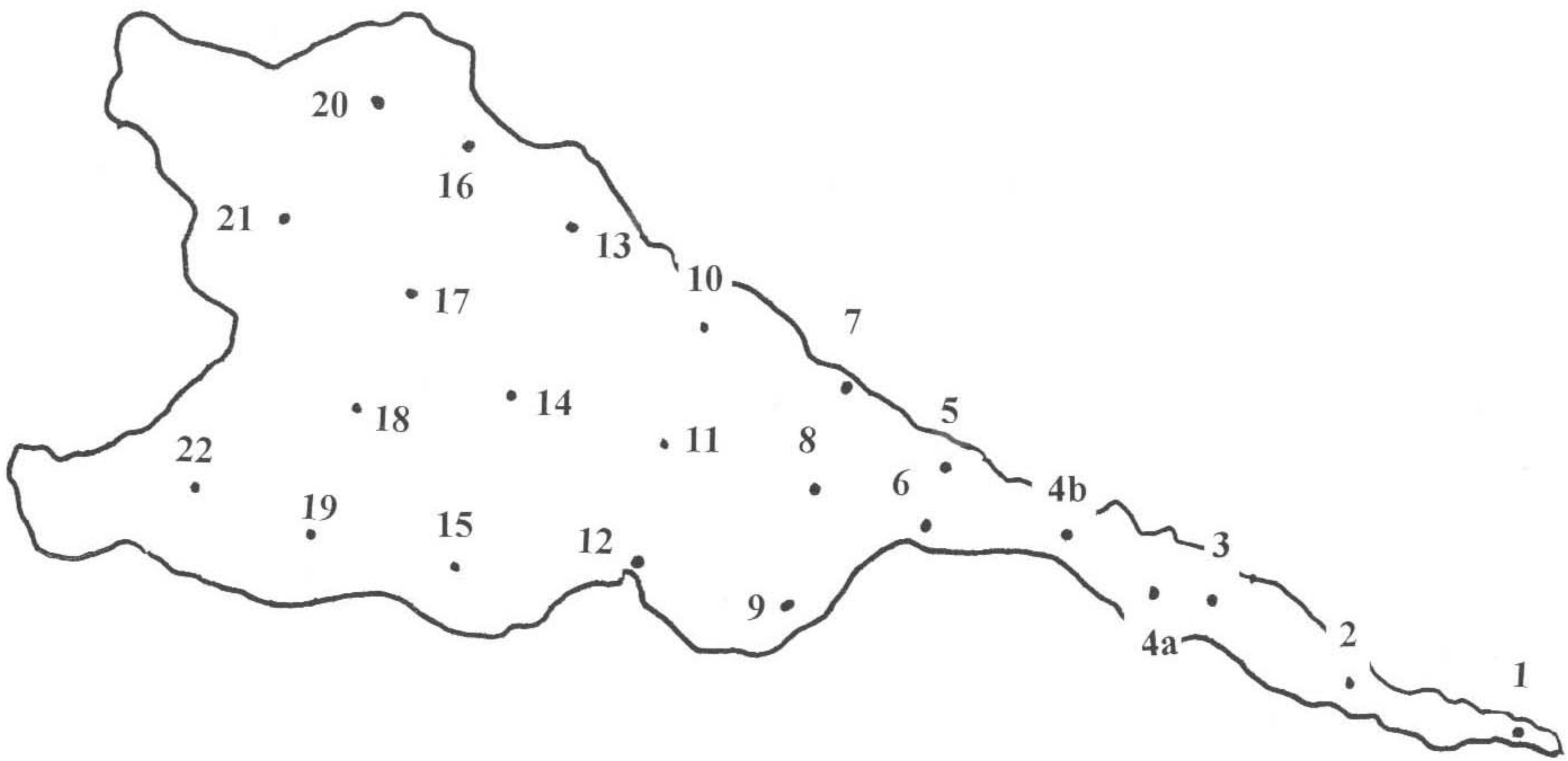


Figure 18 a-c. Water temperatures (^oC) for Bridgeport Reservoir synoptic studies (a) at the surface, and (b) at a depth of 2.5 m on 21 August 2000, and (c) at the surface on 21 September 1989 (Entrix, Inc. 1990).

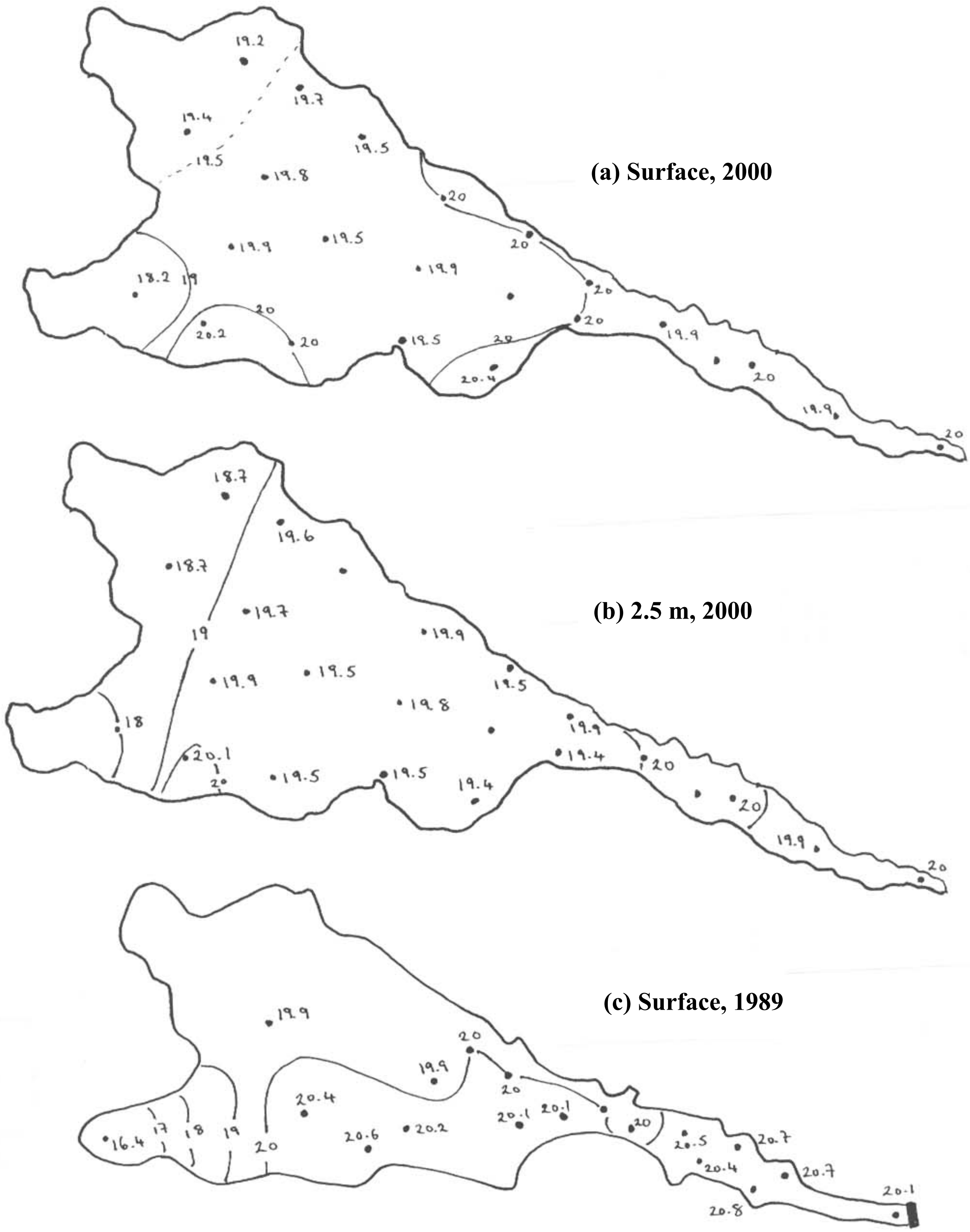
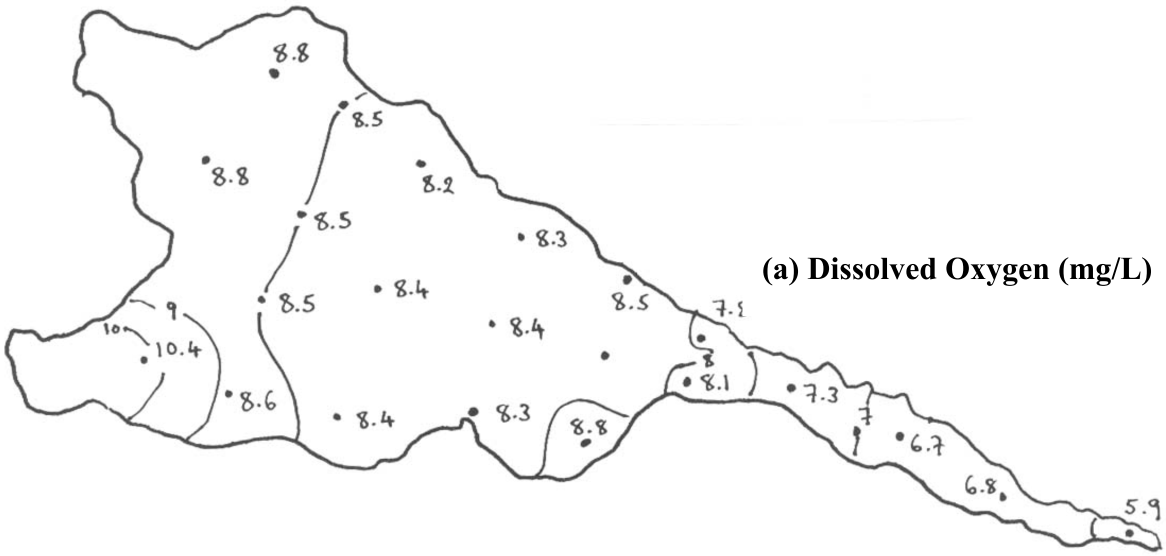
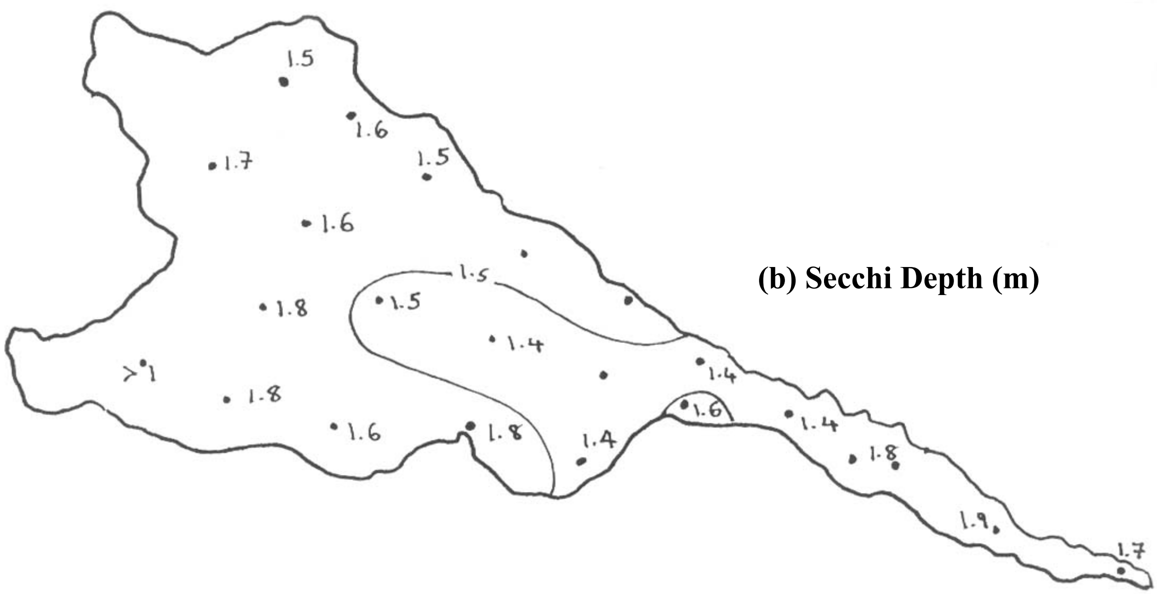


Figure 19 a-b. Synoptic survey of (a) surface dissolved oxygen (mg/L), and (b) water clarity (Secchi disc depth in m) on 21 August 2000 for Bridgeport Reservoir.

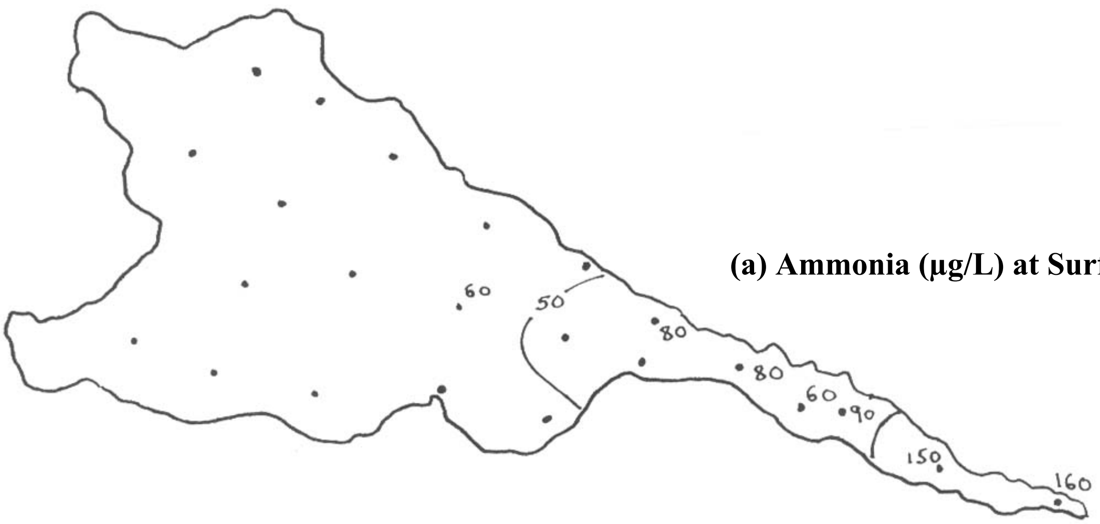


(a) Dissolved Oxygen (mg/L)

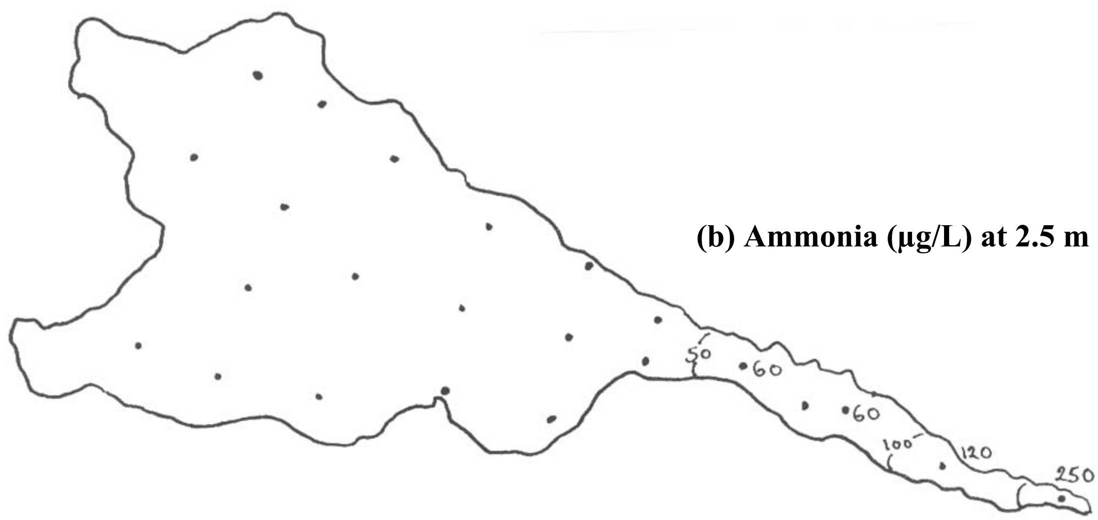


(b) Secchi Depth (m)

Figure 20 a-b. Synoptic survey of ammonia (µg/L) at (a) surface, and (b) 2.5 m, on 21 August 2000 for Bridgeport Reservoir.

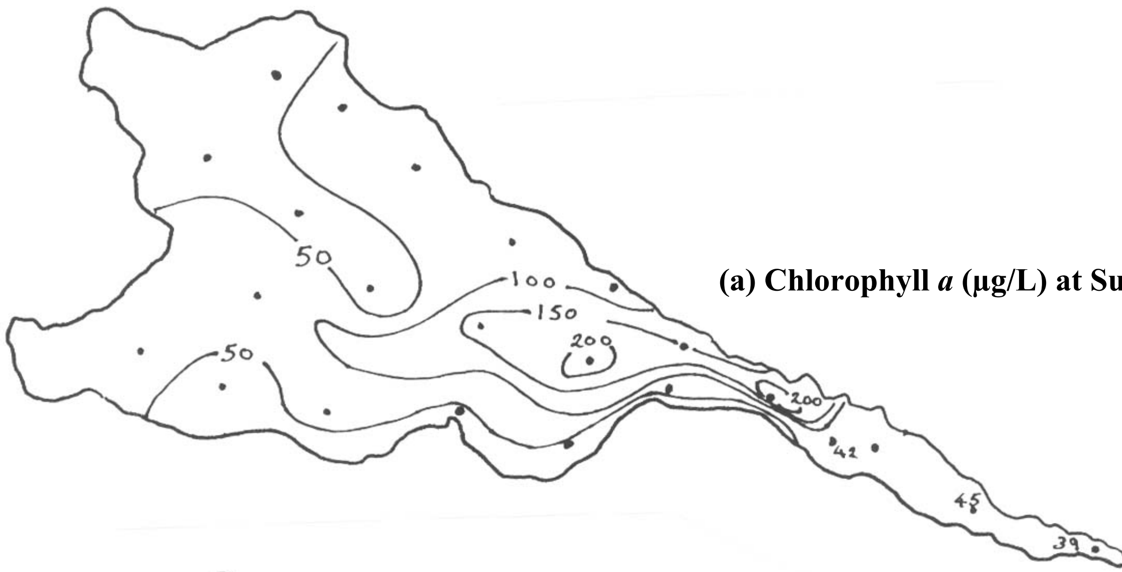


(a) Ammonia (µg/L) at Surface

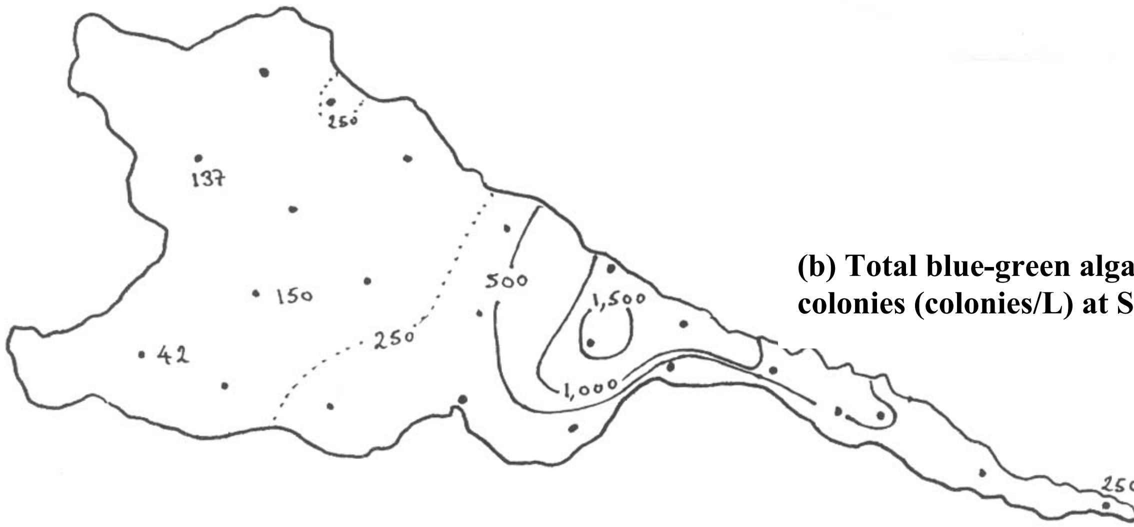


(b) Ammonia (µg/L) at 2.5 m

Figure 21 a-b. Synoptic survey of (a) surface chlorophyll *a* (µg/L), and (b) total surface blue-green algal colonies-per-liter on 21 August 2000 for Bridgeport Reservoir.



(a) Chlorophyll *a* (µg/L) at Surface



(b) Total blue-green algal colonies (colonies/L) at Surface

Figure 22 a-c. Synoptic Survey of surface colonies per liter for (a) *Gleotrichia*, (b) *Aphanizonmenon*, and (c) *Anabaena* on 21 August 2000 for Bridgeport Reservoir.

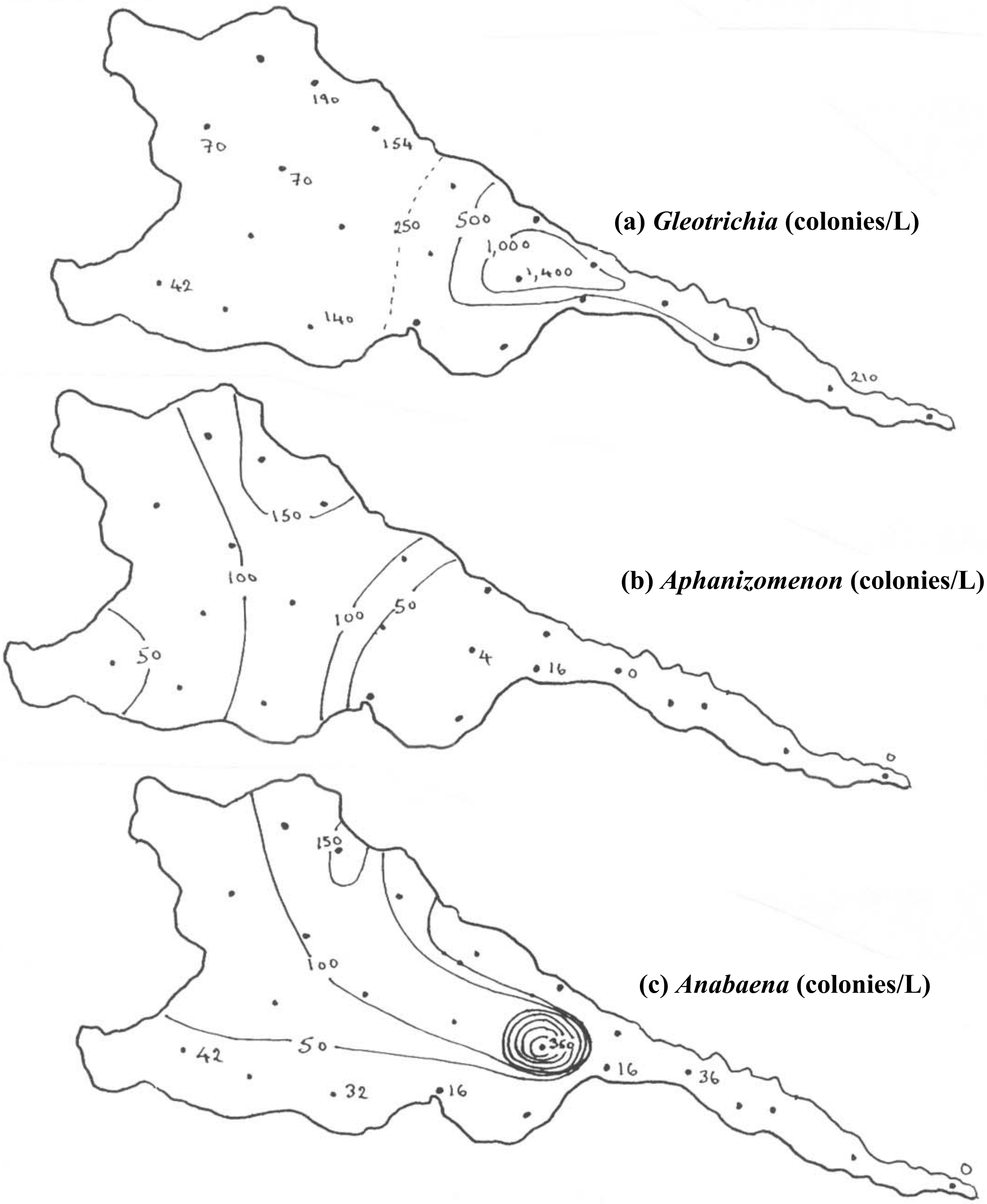
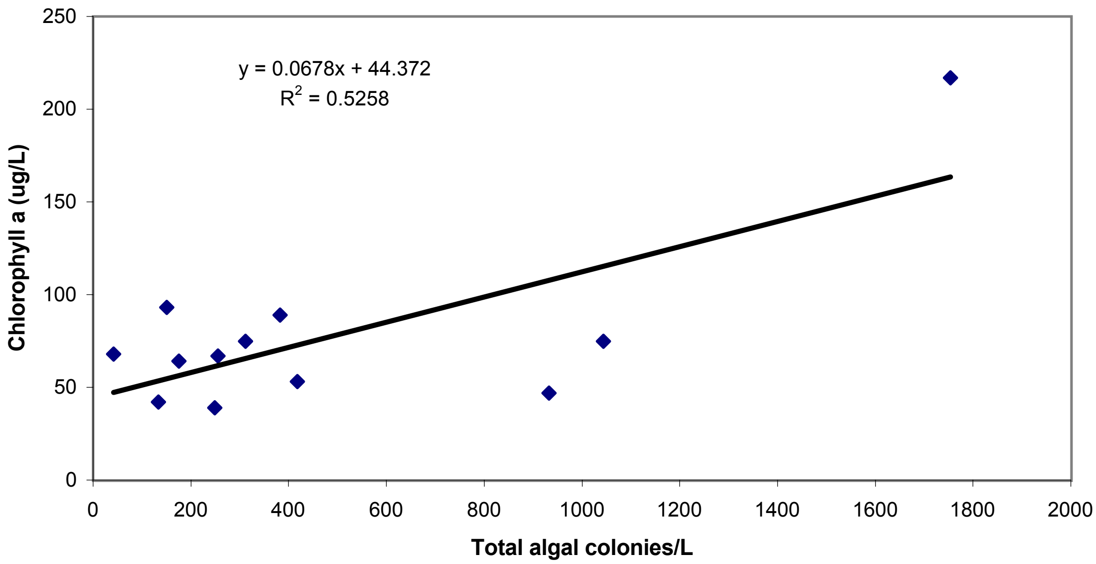


Figure 23a-b. Relationship between chlorophyll *a* and (a) total algal colonies, and (b) *Gleotrichia* for the Bridgeport Reservoir synoptic survey on 21 August 2000.



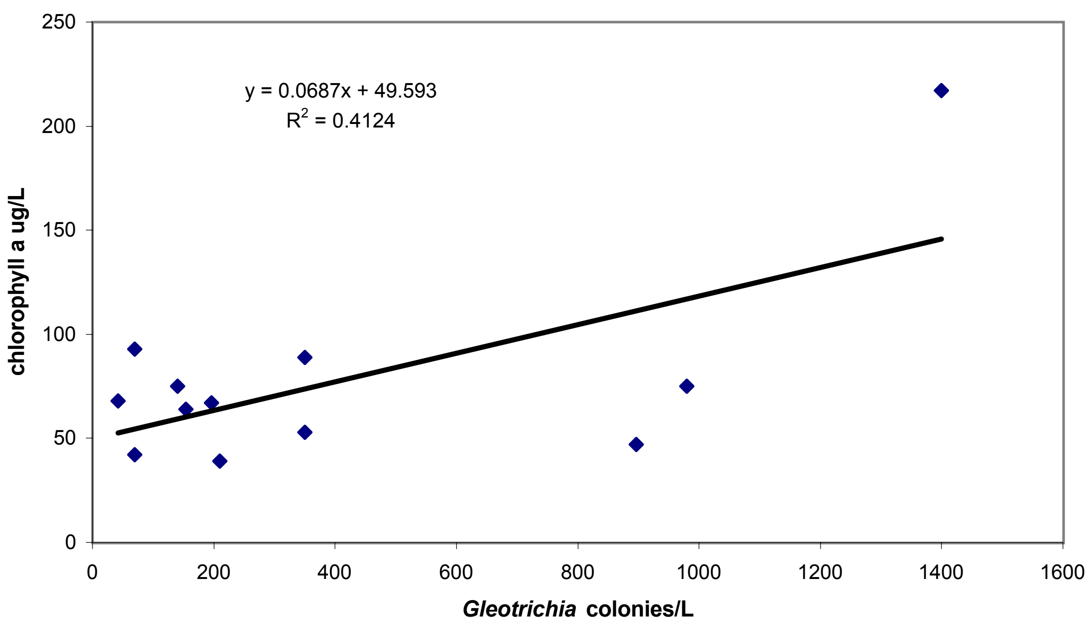
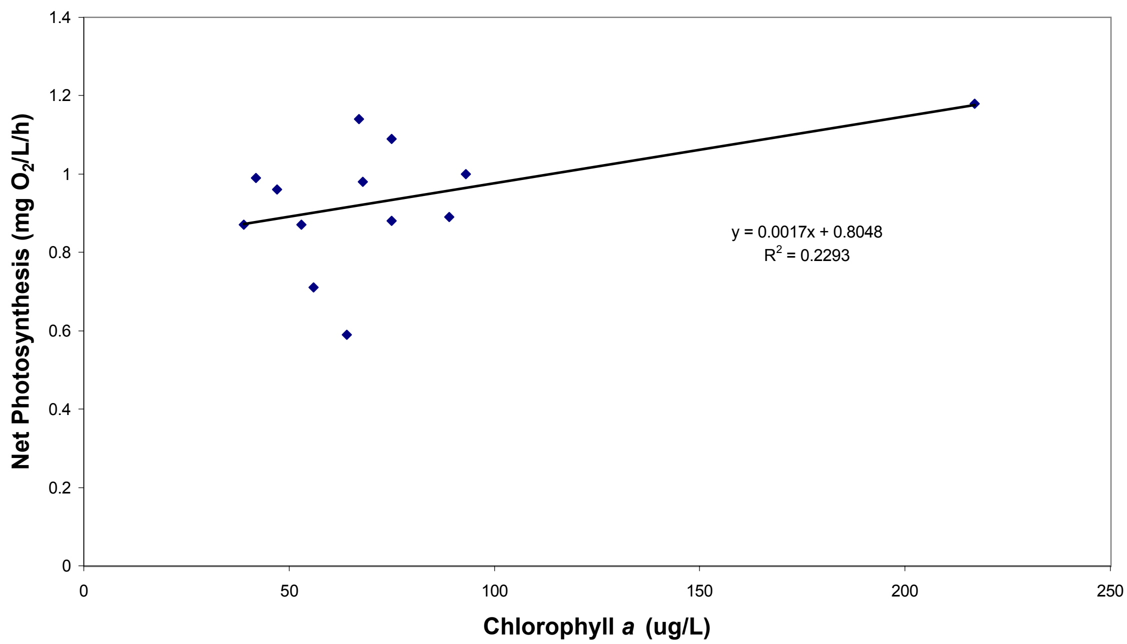


Figure 24a-b. Relationship between chlorophyll *a* and (a) primary production, and (b) respiration for the Bridgeport Reservoir synoptic survey on 21 August 2000.



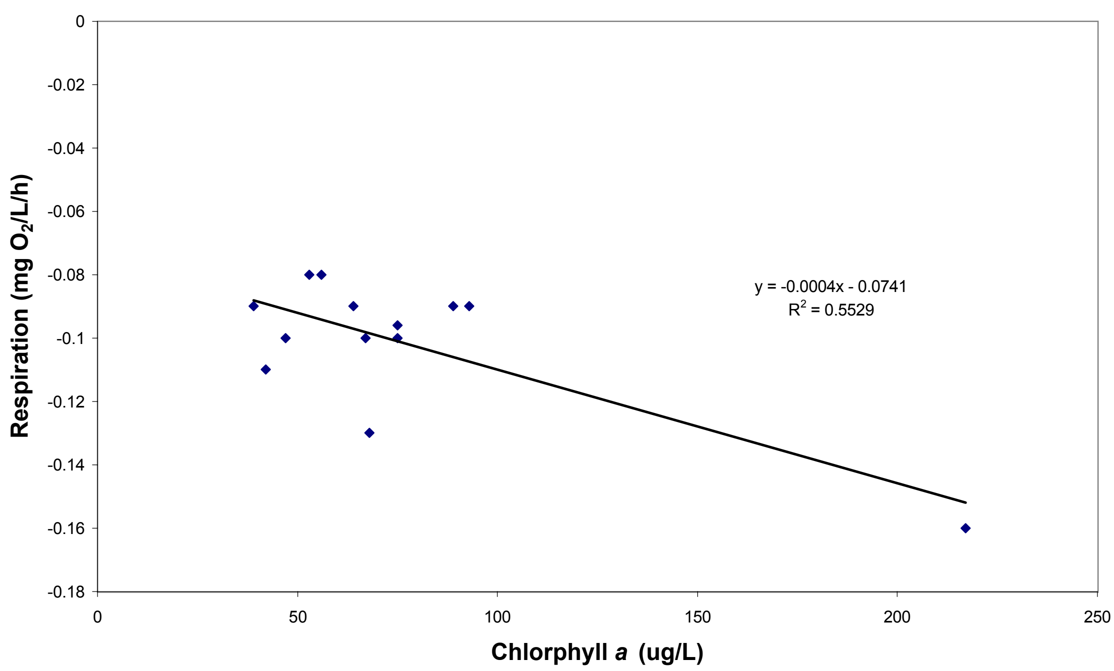
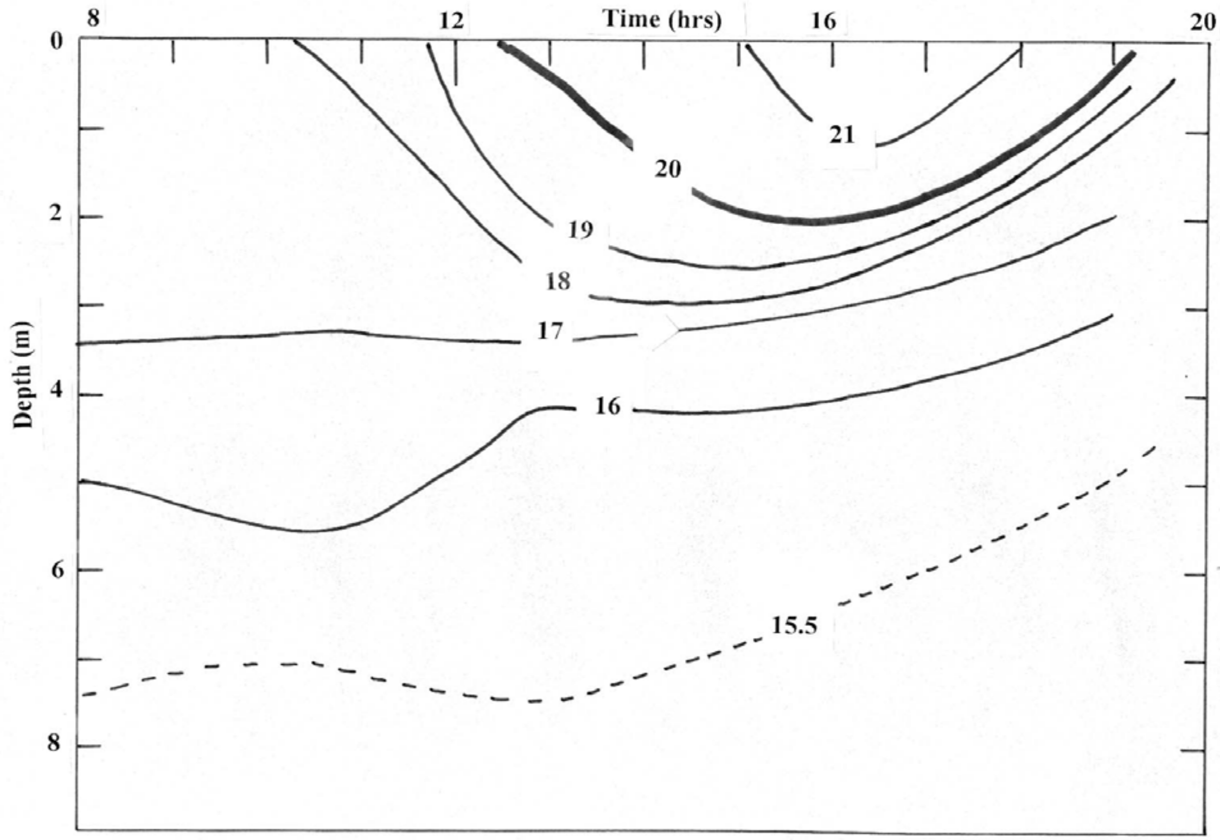


Figure 25 a-b. Diurnal study of Bridgeport Reservoir on 11 September 2000. (a) temperature isotherms, and



(b) dissolved oxygen isopleths.

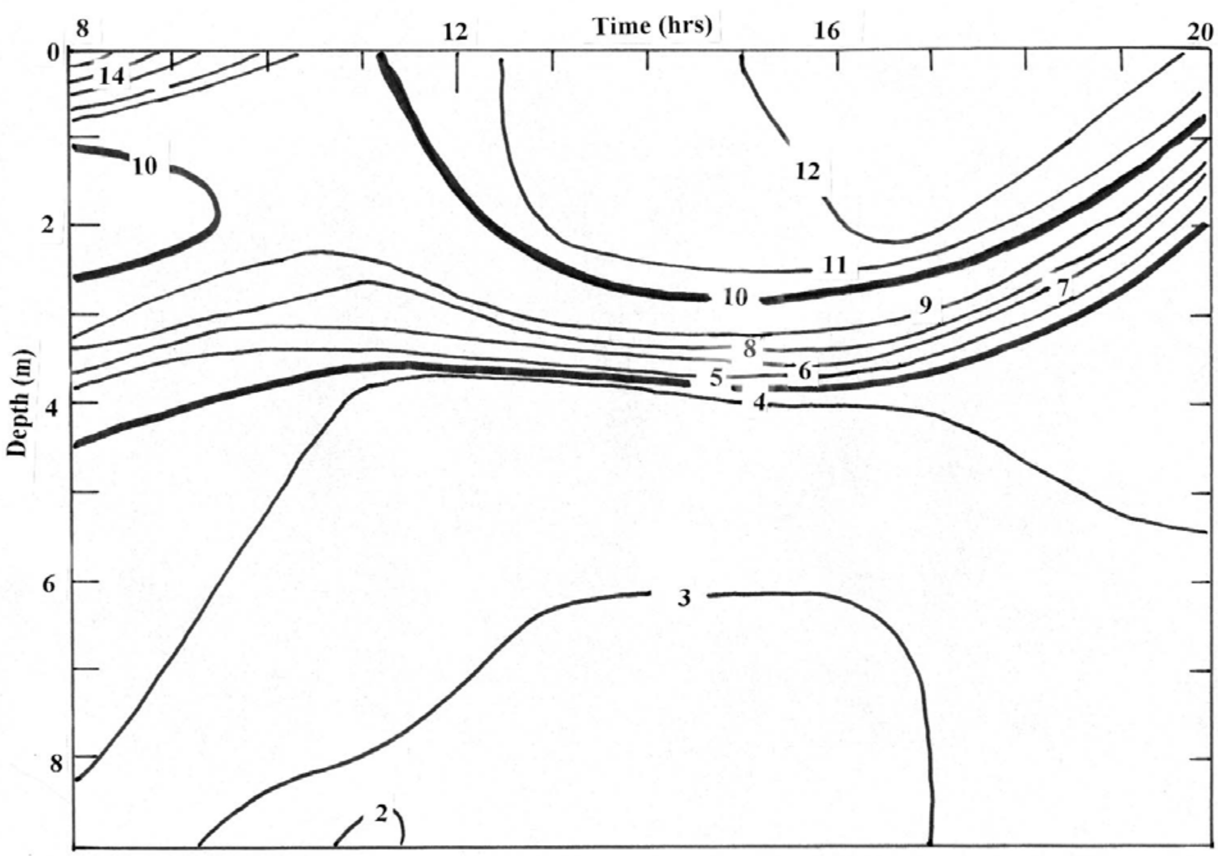
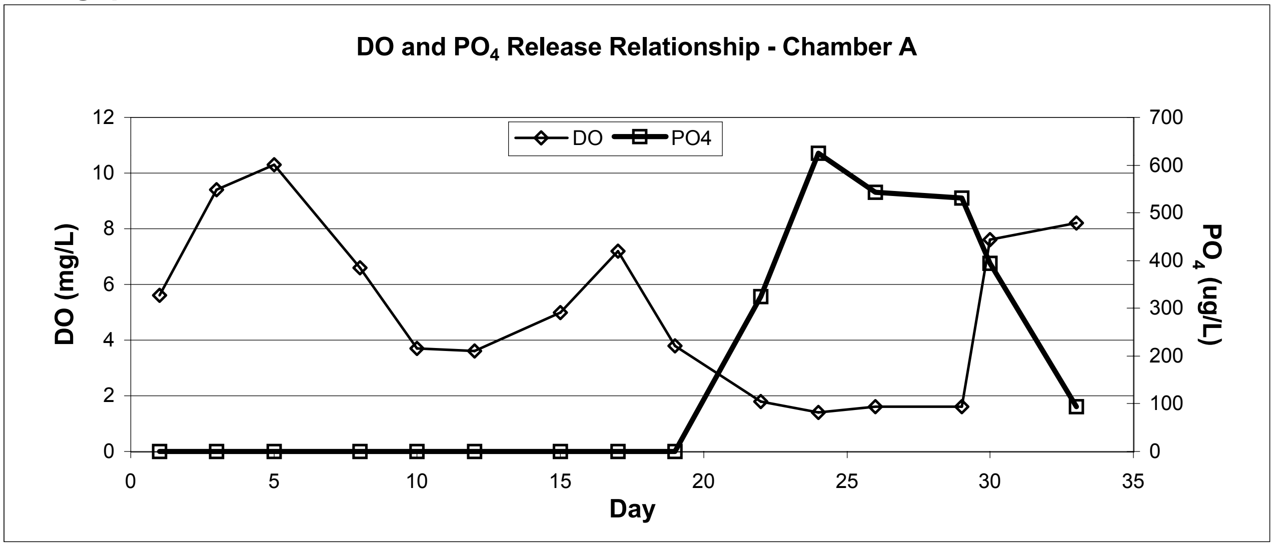
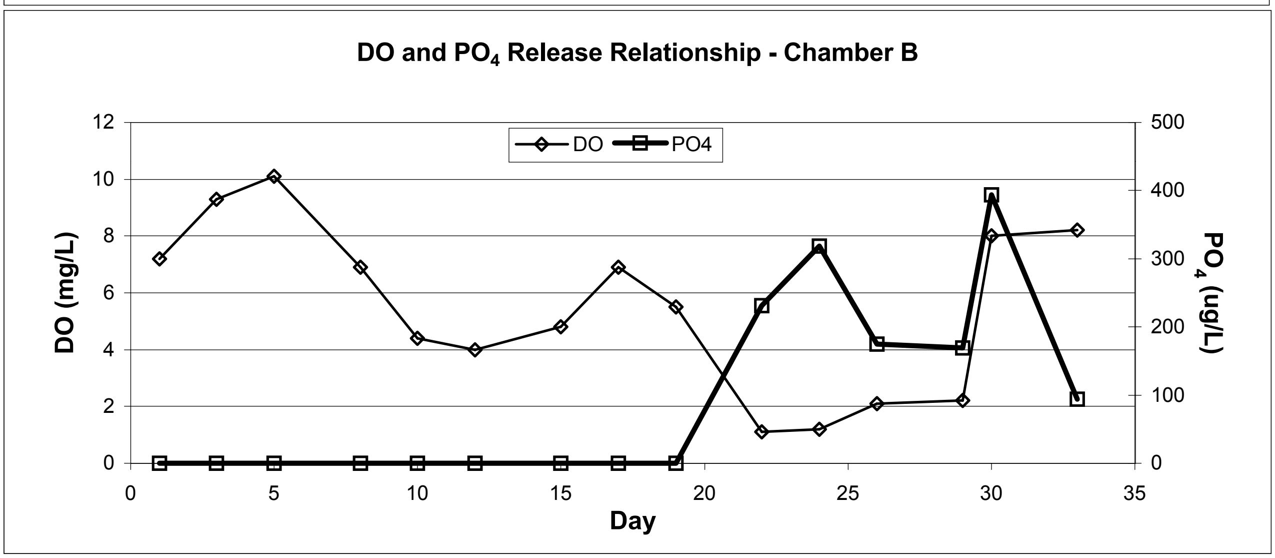


Figure 26. Summary of phosphate releases from experimental chambers under oxic and anoxic conditions using sediment from the deep section of Bridgeport Reservoir in summer 2000.





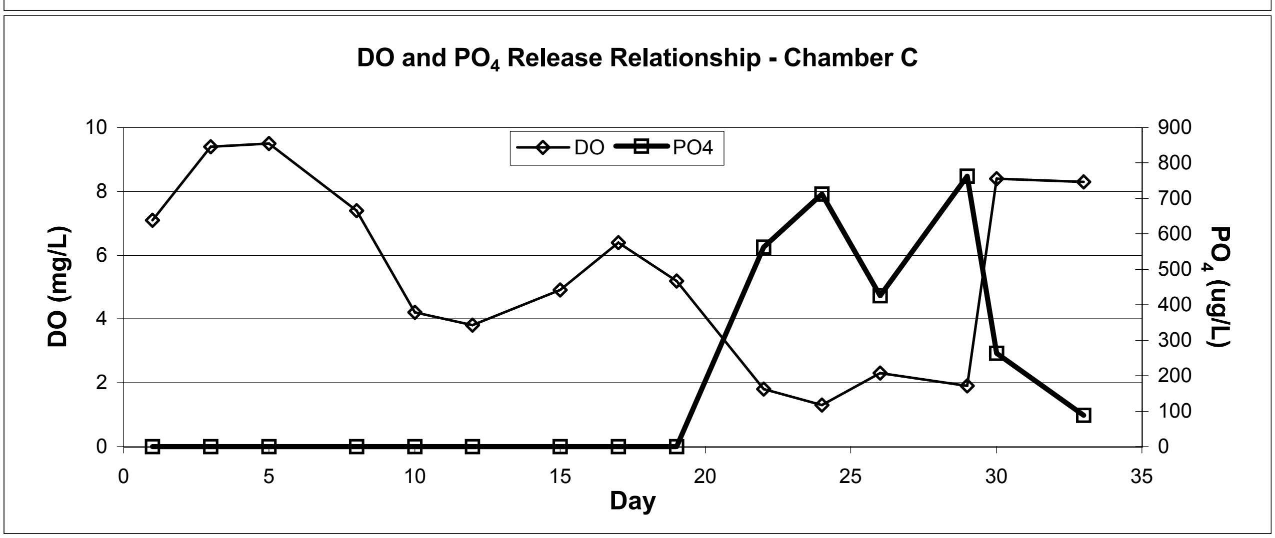


Figure 27. Details of the three replicates used to determine phosphate releases from experimental chambers under oxic and anoxic conditions using sediment from the deep section of Bridgeport Reservoir in summer 2000.