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HORSEHOE LAKE QUALITY, NURTIENT LOADING AND MANAGEMENT

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Water Resources Series
Technical Report No.136
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EXECUTIVE SUMMARY

Horseshoe Lake was studied intensively for one year during 1991 to 1992 to accurately define the state of lake quality and to determine the cause(s) for that quality. The quality of Horseshoe Lake was known to result from the quantity and kind of its algae. The cause for the amount/kind of algae, and a degraded quality state, is usually over supply of the nutrients, nitrogen and phosphorus, so a careful accounting of the input and output of those nutrients was performed. Twenty-two well points were installed around the lake to determine the direction and quantity of ground water flow, and its nutrient content, entering and leaving the lake. Surface water inflows, including the pumped Lewis River water, as well as six stormwater inputs, were also monitored for nutrient content; pumped flow was known and stormwater flow was estimated by a runoff model. Balanced budgets (gains = losses) for water and total phosphorus (TP) were constructed by multiplying TP concentration by flow for each of the inputs and outputs (or losses). TP input from bottom sediments during the summer was determined by difference in the budget.

The lake did not permanently stratify thermally. Lakes with a maximum depth similar to Horseshoe (7.2 m; 24 feet) normally stratify. The reason the lake remained mixed was the large quantity of water entering from the river that was denser than the lake water. The well oxygenated inflow water continuously replaced the lake bottom water preventing stagnation and oxygen depletion, which is the normal summer occurrence in enriched, thermally stratified lakes. As a result dissolved oxygen (DO) remained above 4 mg/L.

The lake during the summer of 1991 was considered slightly eutrophic (literally = good food conditions) with means for TP of 30 μg/L, chlorophyll a of 15 μg/L and water transparency (visibility) of 1.2 m (3.9 feet), compared to threshold values for a eutrophic state of 25 μg/L, 9 μg/L and 1.9 m (6.2 feet), respectively. Quality was slightly poorer

during 1988-1989 with a summer mean transparency of 1.0 m (3.3 feet). The blue-green alga, *Anabaena*, was the most prominent alga during summer 1991. Blue-green blooms are well known in this lake, lasting well into the autumn. The principal bloom came in June, 1991 with no significant bloom lasting into autumn.

River water represented most of the inflow (91%) to the lake, maintaining a high lake level that prevented ground water inflow during the study period. That is, well-point-water levels were always lower than the lake level. Thus, there was always exfiltration via groundwater, unless river flow was higher than 116 m^3/s (4,048 cfs). Consistent with water quantity and direction, most of the TP entering the lake came from the river (63%). Compared to water, the lower TP contribution of the total from the river was due to its relatively low inflow concentration of TP, which was lower than the lake concentration (21 versus 30 μg/L). Also, TP input from stormwater was significant, in spite of relatively low flows, because the stormwater TP concentration was high (270 μg/L). Internal loading of P from bottom sediments during summer contributed only 18% of the total (external plus internal) on an annual basis, but 62% during the summer algal growth period. Thus, the higher lake TP than river inflow TP is a result of the internal loading during the summer period, with the river producing a diluting effect on the lake during summer. That is also true for soluble reactive P (SRP), which is the fraction of TP available to algae. Although river SRP exceeded lake SRP, the quantity coming from internal loading can be considered all SRP.

Internal loading of P probably originates from decomposition of organic matter (settled lake algal and river particles) in the surface sediments, mainly in the northern deep area. The sediments there contain more organic matter, P and lead than sediments from the shallow mid-lake area. That results from "focusing," a sorting effect that concentrates fine particles in the deeper areas of lakes. Internal P loading would be higher if DO were depleted in the bottom waters of the deep area. As is, the internal loading rate, determined by calibrating a predictive P model, is relatively high at 2.44

mg/m$^2\cdot$ day during the summer-autumn period. Such a rate means that each day about 0.7 µg/L of P is added (2.44/3.5 m mean depth) to the lake from sediments (2.3% of the summer mean TP). In contrast, river input adds 0.2 µg/L per day or 0.7% of the mean TP.

Pump enlargement (50%) is recommended to enhance the diluting effect of internal P loading in the long term and to maintain a high lake level minimizing ground water influx during the summer. For the short term, an alum treatment is recommended to reduce internal loading from bottom sediments. Alum should be between 50 to 80% effective and should lower summer mean TP from 30 to 15-20 µg/L and increase transparency to between 1.8 and 2.4 m (5.9-7.9 feet). Pumping is not quite as effective, but is more long lasting. With both measures operating together, the ultimate summer TP should be 13 to 18 µg/L, a 40-56^ improvement. Judging from results in other shallow, western Washington lakes, alum treatment effectiveness should last from five to ten years. The estimated cost for that combined treatment is $218,000.

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INTRODUCTION

Statement of Problem

Horseshoe Lake is located in Woodland, Washington on the Willamette Meridian in Township 5 North. The lake is a cutoff meander of the North Fork of the Lewis River (Figure 1). In 1940, during construction of Highway 99, now Interstate 5, the meander was isolated through diversion of the river channel forming Horseshoe Lake.

Since its origin, water quality has been a central issue in the management of Horseshoe Lake and has been the subject of several studies examining water quality related issues (Canning et. al, 1975; Somers 1989). These studies emphasized bacterial quality of the water primarily and the potential contribution of septic tank leachate and groundwater to that problem. While algal blooms have historically been intense and pervasive during the summer months, their characteristics and potential causes were not part of those investigations. The impetus for this study, therefore, was a growing community concern regarding Horseshoe Lake's high algal concentrations during the summer months, which lowered lake quality and its value as a recreational focal point for the Woodland community.

Historical Perspective and Management of Horseshoe Lake

Soon after its creation in 1940, Horseshoe Lake began to have water quality problems caused by "water stagnation" and loss. The conversion from a river channel to a lake brought unique problems in providing adequate water movement to and from the lake. With no natural inlet or outlet, these had to be engineered to provide adequate circulation.

Prior to its diversion, the North Fork of the Lewis River in the vicinity of Woodland, was used for several commercial purposes; logs from two sawmills were stored along its banks and the river water was withdrawn to irrigate crops in nearby farmlands and to water cattle. Due to these commercial uses as well as recreational and

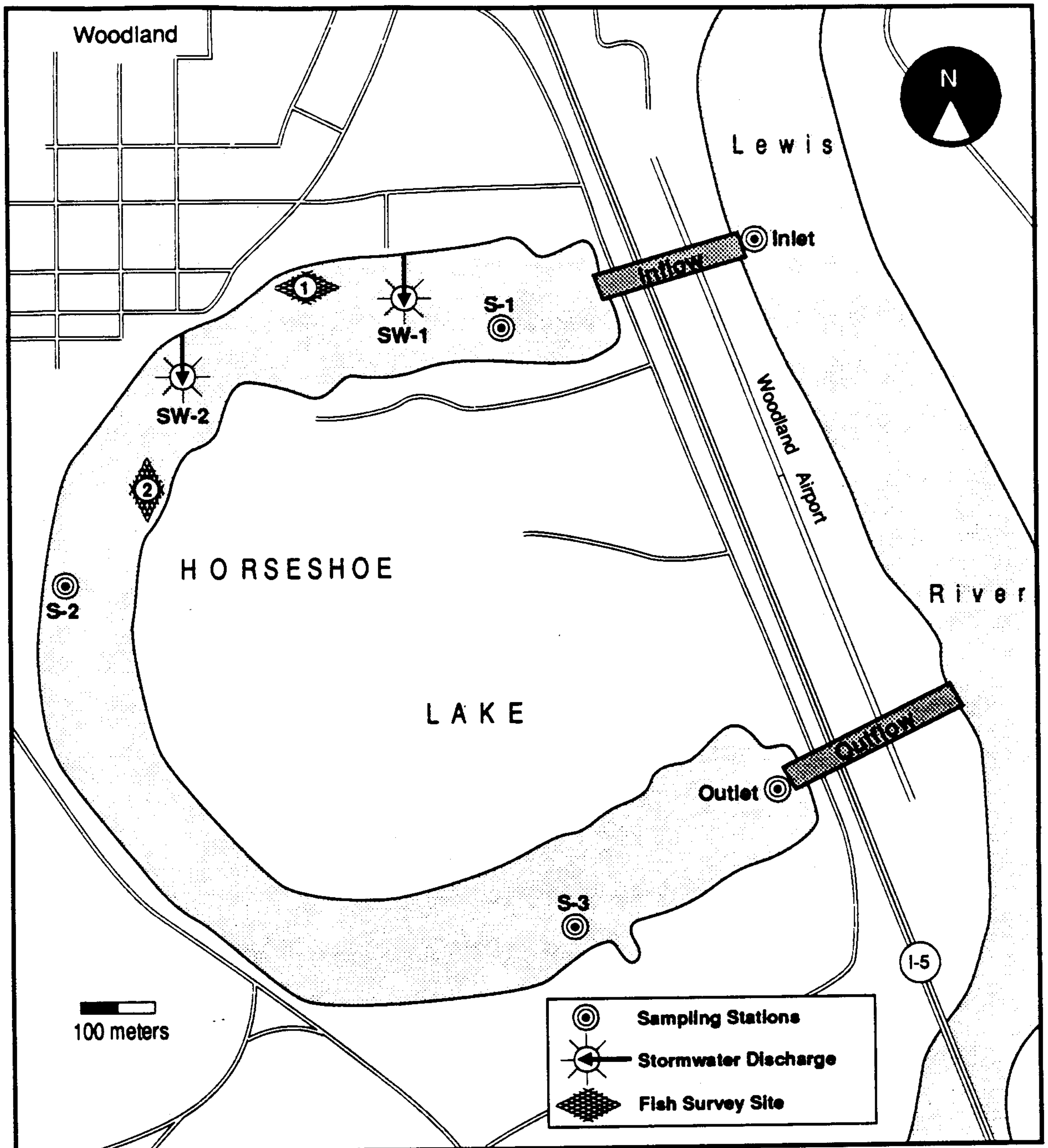


Figure 1. Horseshoe Lake investigation site and sampling stations.

health concerns, a legal agreement was established between the city of Woodland, which had assumed responsibility for the maintenance of the lake, and the Washington State Highway Commission. The agreement stipulated that the lake would not become "stagnant" following the isolation of the meander. Culverts were used to achieve flow between the river (now located 190 meters to the east) and the newly formed Horseshoe Lake. Circulation between the two surface waters was created by changes in river stage due to tidal influences or flow. An outlet was situated at the south arm of the lake. This culvert system was used to provide water inflow to and outflow from the lake for the next 17 years.

However, during this period water quality concerns persisted as the culverts were inadequate in providing sufficient circulation and maintenance of a suitable water level in the lake. The lake's poor water quality was a source of concern to the Cowlitz-Wahkiakum Health District and the Washington State Department of Health (Canning et al., 1975). In an effort to improve circulation and flushing, the Washington State Department of Highways installed a pumping system in 1957 that provided a discharge capacity of 2,850 gallons per minute (gpm) using a 30 horsepower electric motor. Water pumped from the river was fed to a collection box which then flowed by gravity to the north arm of the lake. There was apparently no management scheme for operation of the pumping system to achieve adequate water quality. Although this pumping system was an improvement over the passive culvert-flow scheme, there was still inadequate flow to overcome continued problems with poor water quality.

To further improve circulation through the lake and maintain adequate water levels, the Washington State Department of Highways upgraded the pumping system in 1960 from the combined gravity feed-pump system to a fully pressurized one and deepened the north arm of the lake. Deepening was accomplished by installing a coffer dam and dewatering the north arm. Dredging removed approximately 245,000 cubic meters of bottom sediments (Canning et al., 1975).

In December 1964 a flooding of the Lewis River resulted in the deposition of approximately 50,000 cubic meters of silt and debris into the south arm of the lake. This deposition resulted in a sandy barrier that isolated the south arm at lower water levels.

In 1970, and for the next several years, Horseshoe Lake was declared polluted and unfit for bathing by the Cowlitz-Wahkiakum Health District due to elevated counts of coliform bacteria.

In 1981, the inflow pumping system was again upgraded to what is now the present system. The system consists of a Byron Jackson Type "D" vertical propeller pump, which delivers approximately 3,000 gpm for 55 minutes each hour (Garcia, personal communication). The outflow, located at the east end of the south arm consists of a concrete structure through which water leaves the lake. A gate valve, located at the bottom of the structure, allows lake water to enter the structure. The device was designed to regulate the lake's water level. Water may also enter the outlet structure by flowing over the metal grated top if the water level exceeds 14.6 feet mean sea level. Once into the structure, water flows out through a 36 inch diameter pipe to the river. A backup valve is located midway along the pipeline at the Woodland Airport. Additionally, a flap valve ((tide gate) is located at the river outfall to prevent backflow under flooding conditions.

The management of Horseshoe Lake continues to be the responsibility of the City of Woodland. However, at the request of the City of Woodland, the Washington Department of Transportation's Woodland Station maintains the pumping system. The pumping system is normally operated from March through October. However, during this study, the pump system was operated constantly.

Potential External and Nutrient Sources

Knowledge of present and historical land use within the immediate watershed is important to understand present water quality problems in the lake. Although pumping rates to control lake level and circulation may appear adequate, poor water quality from high algal concentrations may still occur due to excessive nutrient inputs from sources within the watershed. If these nutrient imputs are sufficiently high, they may create an eutrophic state. Although it is a natural process that gradually proceeds over a long period of time, it can be accelerated as a result of man's activities through development in the watershed and is, therefore, termed cultural eutrophication.

Potential external nutrient sources were considered prior to establishing a sampling plan for the Horseshoe Lake study. While there are no point-source discharges, several potentially significant non-point inputs of nutrients are present within the watershed (Figure 1).

Historically, two saw mills were located on the lake and operated from Circa 1946-1970 (Church, personal communication). Former wood debris and sawdust from the mills, if allowed to decompose adjacent to the lake, could have leached nutrients to the lake via transport with groundwater. Logs also were rafted along the shores of the lake prior to cutting and even cut in the water (WPCC, 1950). This method of storage and processing was curtailed in 1970, but could have resulted in the deposition of loose bark and other woody debris to the bottom sediments where it would undergo microbial decomposition and ultimately release nutrients to the lake.

A former landfill is located at the south end of the lake. Depending on the groundwater flow pattern, seepage from the landfill could have found its way to the lake. The landfill was disrupted in 1964, when the Lewis River flooded into Horseshoe Lake.

A cattle yarding area is located along the south end of the lake. Cattle wastes from the area may have leached into the groundwater with its eventual transport into the lake.

Although residences within the Woodland city limits are sewered, a mobile home park with units on the south side of the north arm in Clark County relies on septic tank soil absorption systems for sewage treatment. Failure of septic tank drainfields has often

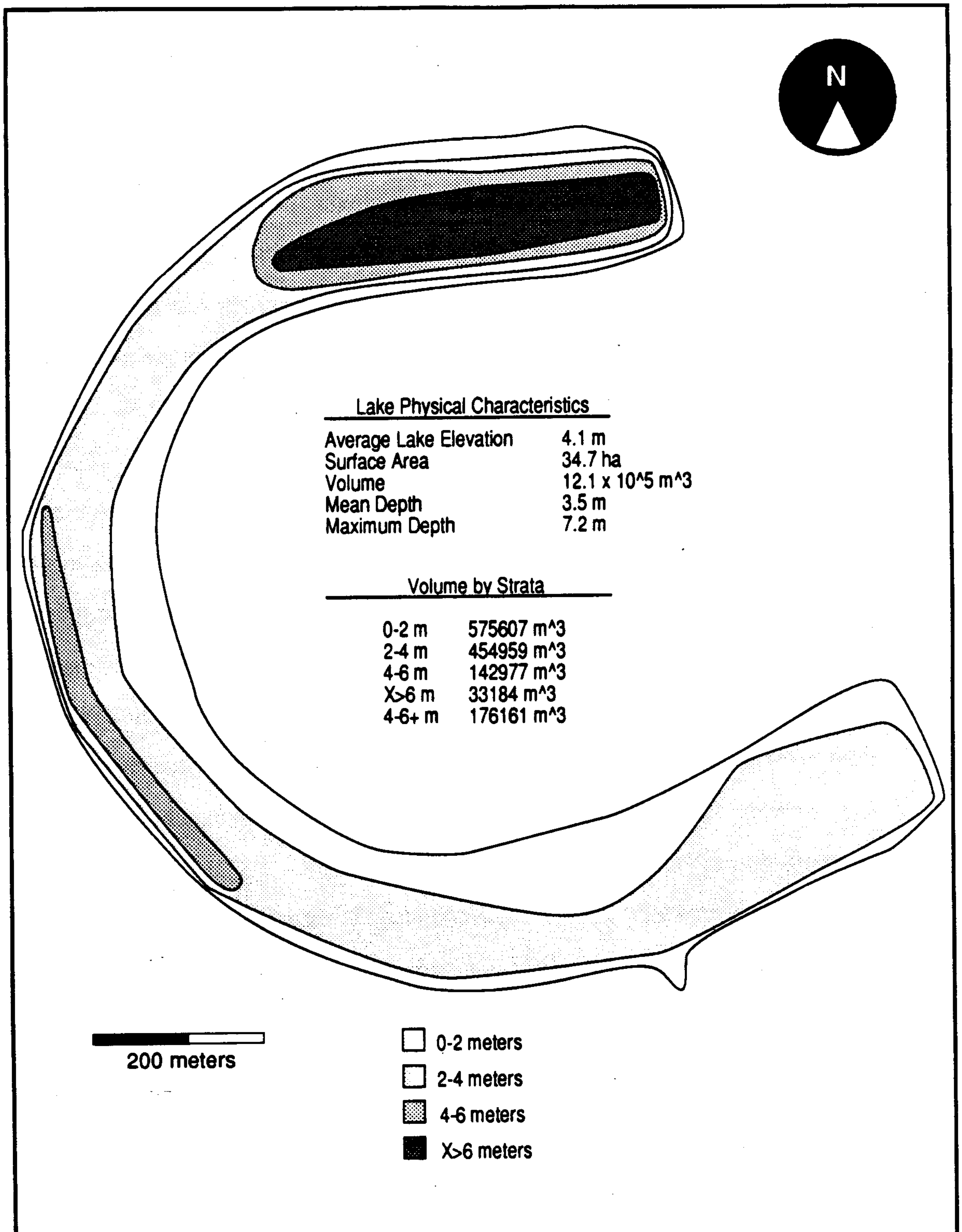


Figure 2. The bathymetry and physical characteristics of Horseshoe Lake.

been implicated as a contributor to the nutrient loading of lakes. These systems were investigated in the past for septic treatment failure. A recent dye test did not detect septic tank effluent leaching to the lake (Somers, 1989). There is the possibility that these systems may drain to the lake only during periods when the watertable is elevated and groundwater flow is directed to the lake.

Significant external nutrient loading could also come from two stormwater inputs that drain the urbanized areas of the city of Woodland. Canning et al. (1975) believed that the stormwater system was not a significant source of nutrients to the lake, although no supporting data were presented. Additionally, there are four other stormdrains located on the lake; two on the end of each arm.

The North Fork of the Lewis River, though diverted in 1940, still forms the major inflow source for the lake. Water quality concerns within the watershed were recently investigated in a report by the Cowlitz County Conservation District (Kahan, 1991). Land use practices within the watershed may be contributing to elevated nutrient levels found in major tributary streams.

Watershed Characteristics

The Horseshoe Lake watershed encompasses approximately 137 hectares (ha), some of which is delineated by dikes. Land use within the watershed includes agricultural, residential housing and business development. The majority of the development within the Horseshoe Lake watershed is situated along the north arm of the lake. Land use along the south arm consists primarily of agricultural and light industrial developments.

The city of Woodland is situated immediately to the north of the lake and stormwater runoff from much of the city (30 ha) is directed to the lake through two stormdrains (SW-1, SW-2; Figure 1). The west side of the lake is bounded by a flood control levee constructed prior to 1940, so little runoff enters the lake from this area.

This main portion of Woodland is sewered (sanitary) and secondary treated effluent is discharged to the Lewis River, downstream from the pump intake.

Land use within this area includes high density residential housing, light industry and business development. Residential development is present along the outer bend of the lake to its midpoint. The city of Woodland maintains the 4 ha Horseshoe Lake Park along the city side of the north arm. Horseshoe Lake Park provides picnic facilities, a bathing beach, a playground, and a boat launching ramp. A 41 hectare woodland area is situated on the inside of the lake. A mobile home park is situated to the north of the wooded area on the south side of the lake.

Knowledge of present and historical land use within the Horseshoe Lake watershed is important to understand the current water quality concerns. Also of significance is that the principal inflow to the lake is from the Lewis River and, therefore, the water quality is closely linked to that of the Lewis River watershed. Water that is pumped from the Lewis River into Horseshoe Lake to maintain circulation and lake level directly influences the lake's water quality. The Lewis River Watershed covers 214,920 ha (530,855 acres), most of which is used for growing forest products. The U.S. Forest Service andWashington State manage 15% of the land with corporate and other private landowners accounting for the remaining 85% (Sommers, 1989).

Lake Characteristics

Horseshoe Lake is relatively small (35 ha) and shallow (mean depth 3.5 m) (Figure 2). Much of the lake's water volume is contained in the north arm, which was dredged in 1960. Besides the dredged area, there is also a deeper trough along the west side (Figure 2). Other physical and morphometric characteristics are listed below.

Lake Physical Characteristics

Average Lake Elevation	4.1 meters
Surface Area	34.7 hectare
Volume	12.1 x 10^5 cubic meters
Mean Depth	3.5 meters
Maximum Depth	7.2 meters
Mean/Maximum Depth	0.49
Shoreline	4.8 kilometers
Shoreline Development	2.3
Relative Depth	1.1%
Maximum Length	814 meters
Maximum Width	636 meters
Mean Width	426 meters

Horseshoe Lake's morphometry, or basin shape, reflects its historical origin as a river channel. The former river channel's meander or "horseshoe" shape gives rise to a relatively high shoreline length in relation to the lake's surface area. The maximum length is a measure of the greatest distance that wind can travel over the surface of the lake. With the predominant wind direction from the west, wind mixes the lake as its force is concentrated on the east-west sections of the lake. The measure of relative depth is taken as the ratio of the maximum depth to the surface area of the lake. The lake has a small relative depth indicating that it should be susceptable to wind mixing.

METHODS AND MATERIALS

Field Sampling

Water quality sampling was conducted twice a month from April, 1991 through October, 1991 and on a monthly basis from November, 1991 through April, 1992. The lake was also sampled again in June, 1992. Water samples were collected from four stations in the lake and one station located at the Lewis River pump (Figure 1). Water quality and biological monitoring included the determination of total phosphorus (TP), soluble reactive phosphorus (SRP), pH, specific conductance, nitrate + nitrite - nitrogen (NO_3 + NO_2 - N), ammonium nitrogen (NH_4 - N), chlorophyll *a* (chl *a*), turbidity, total suspended solids, alkalinity, Secchi disk transparency (SD), dissolved oxygen (DO) and temperature. Water samples were collected for phytoplankton biovolume determination and vertical hauls for zooplankton abundance. On a quarterly basis, lake water samples were collected at two meter depth intervals for the major ions calcium (Ca), magnesium (Mg), sodium (Na), potassium (K), sulfate (SO_4), chloride (Cl), and metals iron (Fe) and total aluminum (Al).

Temperature profiles were determined in situ at one-meter intervals using a Yellow Springs Instrument Company (YSI) Model 57 oxygen and temperature meter. Dissolved oxygen levels were determined in BOD bottles at 2-m intervals using the azide modification of the Winkler method (APHA 1989). Water-column transparency was measured using a 20 centimeter black and white Secchi disk. Total suspended solids measurements were made using the method outlined in standard methods (APHA 1989). Turbidity was determined using a turbimeter.

Water samples for phytoplankton were collected using a two-liter Van Dorn bottle at stations 1, 2 and 3 from the routine depths monitored. Once collected, the subsamples for phytoplankton were placed in acid washed 120 milliliter polyethylene bottles and preserved with 1% Lugol's solution. Zooplankton samples were collected by vertical hauls with a 0.5 meter diameter plankton net with a 76 μm mesh. Replicate samples were collected for zooplankton form a depth of 6 m at Station 1. The samples were transfered to 0.5 L glass bottles and preserved with 50% propanol in the field.

Dredge samples for benthic invertebrates were collected on September 25, 1991 and March 28, 1992 from Stations 1, 2 and 3. Replicate samples were collected with model 196 F10 Wildlife Supply Company 0.25 square foot Ekman grab sampler. Once collected, the samples were transfered to 0.5 L glass bottles and preserved with 50% propanol.

Water samples for chemical constituents were collected from four lake stations (Figure 1). At Station 1, water samples were collected from the surface (0.5 meter), 2, 4 and 6 meter depths. Surface and bottom water samples were collected at both Stations 2 and 3. The bottom depths of station 2 and 3 were 2.5 and 1.5 meters, respectively. In addition, a surface sample was also gathered at the lake's outlet and at the Lewis River pump. All water samples were collected using a 2 L Van Dorn bottle and placed in acid-washed, 2-liter polyethylene bottles which were stored in a cooler for return to the laboratory. The pH was determined in the field at each of the sampling locations using a Cole-Palmer pH meter in a subsample from the Van Dorn sampler. Subsamples were taken from the 2-liter sample in the laboratory for the analysis of chl *a*, alkalinity, pH, specific conductance, TP, SRP, TN, $NO_3 + NO_2$, and NH_4-N.

The samples for SRP, $NO_3 + NO_2$-N, and NH_4-N were filtered through 0.45 μm Millipore filters, placed in 120 ml acid-washed polyethylene bottles and frozen until analyzed. The TP samples were placed in 120 ml acid-washed polyethylene bottles and preserved with two drops of 36 N H_2SO_4. Samples for TN were placed in 120 ml acid-washed polyethylene bottles and frozen until analyzed. Samples for chl *a* were filtered onto a 47 mm glass fiber filter with 2 drops of $MgCO_3$. The filters were then stored frozen in a darkened desiccator until analyzed.

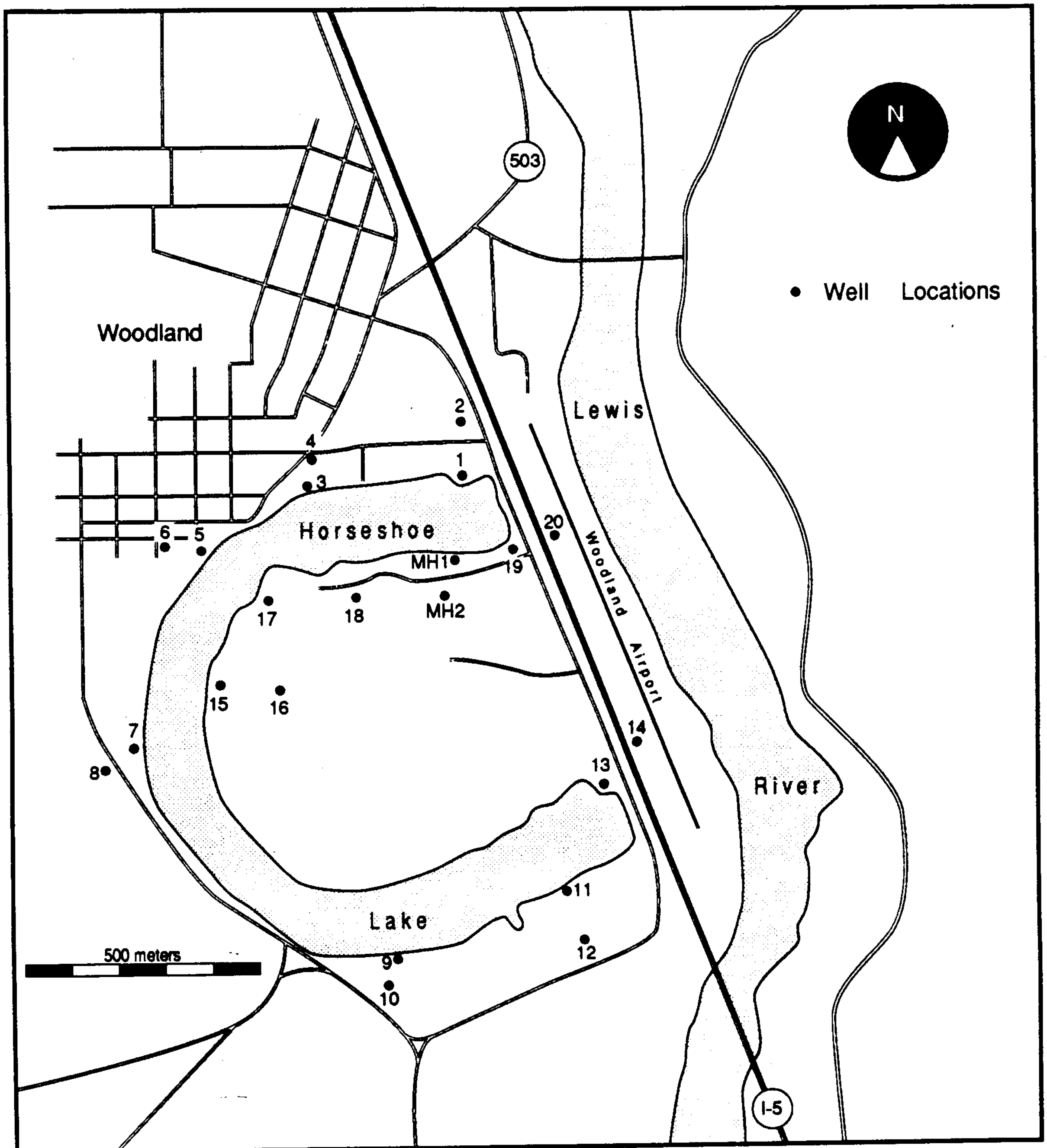


Figure 3. The Horseshoe Lake study area and well locations in Woodland, WA.

The samples for Ca, Mg, Na, K, Fe, and Al were placed in 120 ml acid-washed polyethylene bottles and preserved with 36 N H_2SO_4. The samples for SO_4 and Cl were filtered through pre-soaked 0.45 µm Millipore filters, placed in 120 ml acid-washed polyethylene bottles and frozen until analyzed.

Two sampling locations were used to assess the water quality of stormwater inputs (Figure 1, SW-1 and SW-2). Four other stormwater inflows occurred on the lake, two on each of the arms, however, these locations were found to contribute little stormwater discharge and so sampling was discontinued. During the study, five storms were sampled. During these events grab samples were collected in 1 liter polyethylene acid washed bottles and analyzed for TP, SRP, TN, NH_4-N, NO_3 + NO_2-N using procedures outlined above.

Twenty-two, 1.25 inch diameter wellpoints were installed around the perimeter of the lake and monitored for water quality and water level height (Figure 3). The wells were driven down to a depth of approximately 20 feet below the ground surface. Following installation, the wells were developed using a pitcher pump. Water samples for water quality analysis were collected quarterly in 1-L bottles and analyzed for conductivity, temperature, TP, SRP, NH_4, NO_2 + NO_3, TN, chloride and sulfate. Prior to groundwater sample collection, the wells were purged of approximately ten volumes to remove standing water as well as to remove entrained sediments. Samples were prepared for analysis or storage in the same manner as lake water quality samples.

Water height within the wells was measured each sampling trip using a cork float line. Distance from the top of the well to the water surface was measured to the nearest mm. Three replicate measurements were made at each well per visit and an average distance determined. Well heights were surveyed to mean sea level from the Woodland city benchmark. The distance to water from the well top was subtracted from the well elevation to determine water elevation.

Precipitation samples were gathered using an acid washed polyethylene funnel and a 1 liter polyethylene collection bottle. Both summer and winter composite samples were analyzed for TP, NO_3 + NO_2-N, SRP and TN.

Sediment cores were collected from lake sampling Stations 1 and 2 on September 10, 1991 with a piston-type corer.

On March 25, 1991 two variable mesh, monofilament gill nets were positioned along the north and inner shores of the lake (Figure 1). Brush along the shore provided cover for fish at site 1. Site 2 was relatively shallow (<2 m) and exposed. The nets were fished for approximately 25 hours, which was coincidental with routine Department of Wildlife sampling prior to the recreational fishing season. Entrapped fish were gathered and identified to species, lengths and weights determined, and their stomachs removed and preserved in formaldehyde for later identification of contents.

Laboratory Analysis

SRP was determined with the molybdate blue ascorbic acid method according to standard methods (APHA, 1989). TP was measured by analyzing for SRP in unfiltered water samples after persulfate digestion. Absorbance was determined on a Milton Roy Co. Spectronic 1001 spectrophotometer at 885 nm in a 10 cm cell.

TN was determined with a persulfate digestion in a basic solution, followed by analysis in an ALPKEM RFA-300 continuous flow autoanalyzer according to standard methods (APHA, 1989). NH_4-N was analyzed with indophenol blue color formation produced by the automated phenate method (APHA, 1989), and absorbance read at 630 nm on the autoanalyzer. Nitrate + nitrite - N was determined as nitrite after cadmium reduction and absorbance was read at 520 nm on the autoanalyzer (APHA, 1989).

Alkalinity expressed as $CaCO_3$ was determined by titrating 200 ml of sample to a pH of 4.9 with a standardized 0.02 N H_2SO_4 solution (APHA, 1989). Potentiometric titrations were performed with a Corning glass electrode calibrated at a pH of 4 and 7. Specific conductance was determined in 100 ml samples with a model PM-70CB Sybron/Barnstead conductivity bridge with a cell constant of 1.0.

Water determinations for Zn, Ca, Mg, K, Fe and Al were determined using a Jovin-Yvon Model Jy-50 inductively-coupled argon plasma (ICP) emission spectrophotometer (APHA, 1989). The SO_4 and Cl samples were analyzed on a Dionex QIC ion exchange chromatograph (APHA, 1989).

Zooplankton samples were stained with Iosin Y, and concentrated by filtering (100 μm mesh) prior to counting. The samples were then brought up to 500 or 100 ml with tap water depending on the amount of zooplankton present in the sample. Two 5 ml subsamples were then placed in a counting chamber and counted using a Nikon binocular dissecting microscope. The zooplankton were identified to genera and enumerated as No./L.

A 30 ml subsample from the phytoplankton samples was centrifuged for 20 minutes and aspirated to concentrate the samples ten fold prior to counting. The sample was then mixed and aliquots placed in a 0.1 ml Palmer Maloney counting cell with 50 grids counted per sample. The phytoplankton were identified to genera, enumerated, and biovolume determined as mm^3/L. Chl *a* was determined by grinding the samples in a $MgCO_3$-saturated, 90% acetone solution. Following extraction for 24 hours, absorbance was read at 665 and 750 nm on a Perkin-Elmer Model 3 scanning spectrophotometer (APHA, 1989).

The sediment cores were sectioned at one centimeter intervals. Following the sectioning, the samples were weighted and then dried at 105°C for 24 hours. Percent water content was determined by reweighing the samples. Following this determination, sediment samples were ground and then placed into discrete, sealed, polyethylene capsules. An approximate 300 mg sub-sample of this material was later used to determine the percent of organic material as determined by the weight loss on ignition at a temperature of 550°C.

Approximately 1 gram of the sediment was initially digested in 5 ml of concentrated nitric acid for 24 hours. The samples were then further digested at 150°C for 6 hours in a heater block. After allowing the samples to cool, a 1% nitric acid solution was used to dilute the digestate of 50 ml. The digestate was analyzed for Al, Pb, Zn, Fe and TP. Analysis for the metals was determined following a nitric-acid digestion using a Jovin-Yvon Model Jy-50 inductively-coupled argon plasma (ICP) emission spectrophotometer (APHA, 1989). TP was determined with the molybdate blue ascorbic acid method according to Standard Methods (APHA, 1989).

RESULTS AND DISCUSSION

Water Quality and Physical Characteristics

Temperature: Horseshoe Lake did not permanently stratify thermally in spite of its maximum depth being slightly greater than 6 m (Figure 4). During the summer period, June through August, the difference in temperature between surface and bottom ranged from about 2 to 4 °C. A completely unstable condition occurred in January when water at the bottom was 6 °C warmer, and hence lighter, than surface water. Such a condition probably indicates strong, wind-driven mixing.

Specific conductance, a measure of the total ion or salt content of waters, was also similar between surface and bottom during June through August. The waters differed by < 1 μmho/cm (see Appendix B).

Failure of the lake's deepest water columns to permanently stratify was probably due to the inflow tending to sink and continually replace the "hypolimnion". The river inflow water during June - August averaged 4 μmho/cm more and 4.5 °C less than the lake surface water. Thus, the inflow was much denser than the lake water. Without such replacement, a thermocline would persist blocking any exchange between surface water and bottom (hypolimnetic) water. As a result, oxygen in the bottom water would deplete and the resulting anoxic conditions would permit high rates of phosphorus release from sediments. Thus, the failure of the water to permanently stratify probably prevented worse water quality in the lake than was observed.

Oxygen: The pattern of DO in the lake through the year follows that of temperature and the degree of stratification (Figure 5). DO reached its minimum concentration following the maximum difference between surface and bottom temperature (maximum stratification) and the major algal bloom (see section on phytoplankton). The algal material sinking to the lake bottom provided the organic demand on oxygen at the same time that stratification was greatest, resulting in the

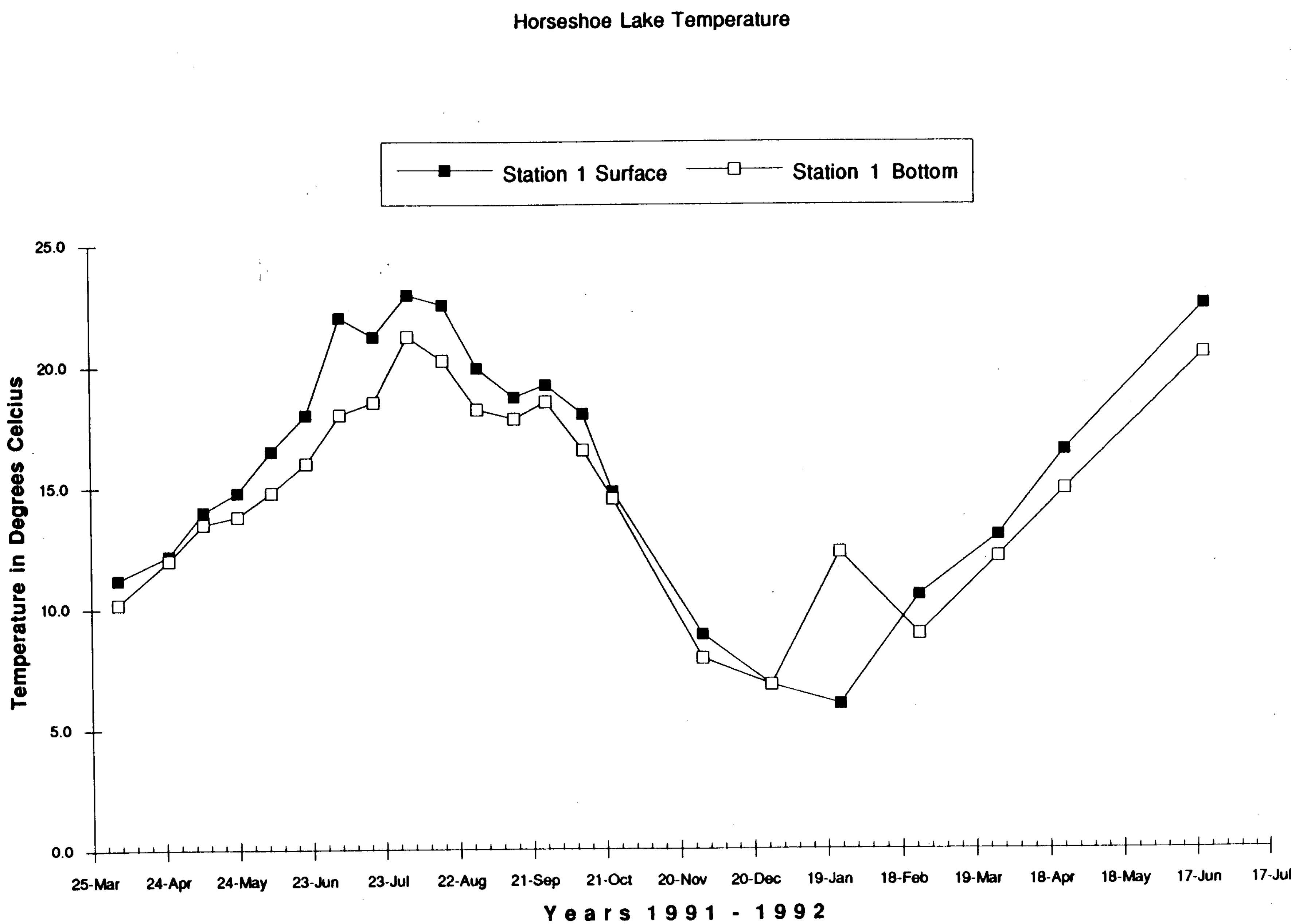


Figure 4. Surface and bottom temperature at the deepest site (station 1) during 1991-1992.

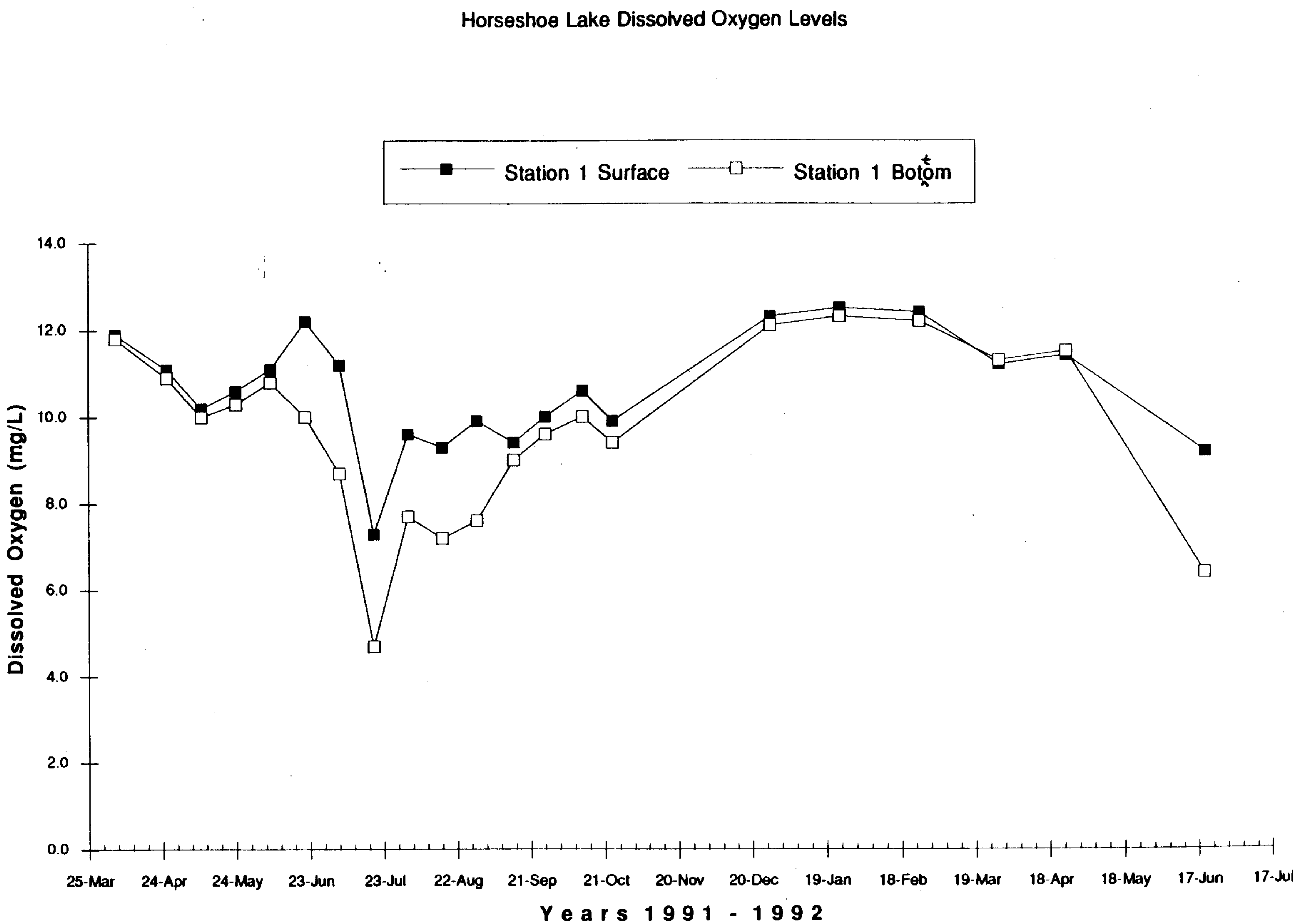


Figure 5. Surface and bottom dissolved oxygen (DO) at the deepest site (station 1) during 1991-1992.

slowest replacement of bottom "hypolimnetic" water. Hence, there was essentially more oxygen demand and less oxygen supply at that time. As the difference between surface and bottom temperature (degree of stratification) declined around August 1 (Figure 4), DO in the bottom water increased (Figure 5). As suggested earlier, if stratification had persisted, DO would probably have reached lower concentrations and may have caused worse (increased phosphorus) water quality conditions.

Nutrients: Total phosphorus (TP) concentrations (whole lake, volume weighted) in the lake ranged from 13.5 µg/L in October to 40.5 µg/L in June. The yearly mean was 24 µg/L and the summer (June-September) mean was 30 µg/L. Summer concentrations were consistently higher than those in winter (Figure 6). The usually high concentrations in summer are probably due to loading from lake bottom sediments as will be discussed further under nutrient budget and phosphorus model.

The trophic state guideline for summer TP beyond which a lake is termed eutrophic is about 25 µg/L (Porcella et al., 1980). Because P is usually the nutrient (rather than nitrogen or carbon) that usually limits plant growth in freshwater, the concentration of TP usually determines the amount of algal biomass that can develop per unit volume of water. The quantity (and production) of algae, in turn, is directly related to the lake's transparency, because the more algal particles present, the less light that will penetrate through the water. Furthermore, hypolimnetic DO will normally decline as algal biomass (oxygen demand) increases (Cornett and Rigler, 1979), as indicated earlier by an analysis of Figures 4 and 5. Higher TP content in lakes usually leads to a greater proportion of the algae being represented by blue greens (Smith, 1990), which are often buoyant and accumulate at the surface forming scums.

Another indication that the higher lake TP concentrations during the summer were due to release from sediments, or internal (versus external) loading, is that river inflow concentrations were nearly always lower than lake concentrations (Figure 7). River inflow contained about 9 µg/L less TP on the average during the summer than the

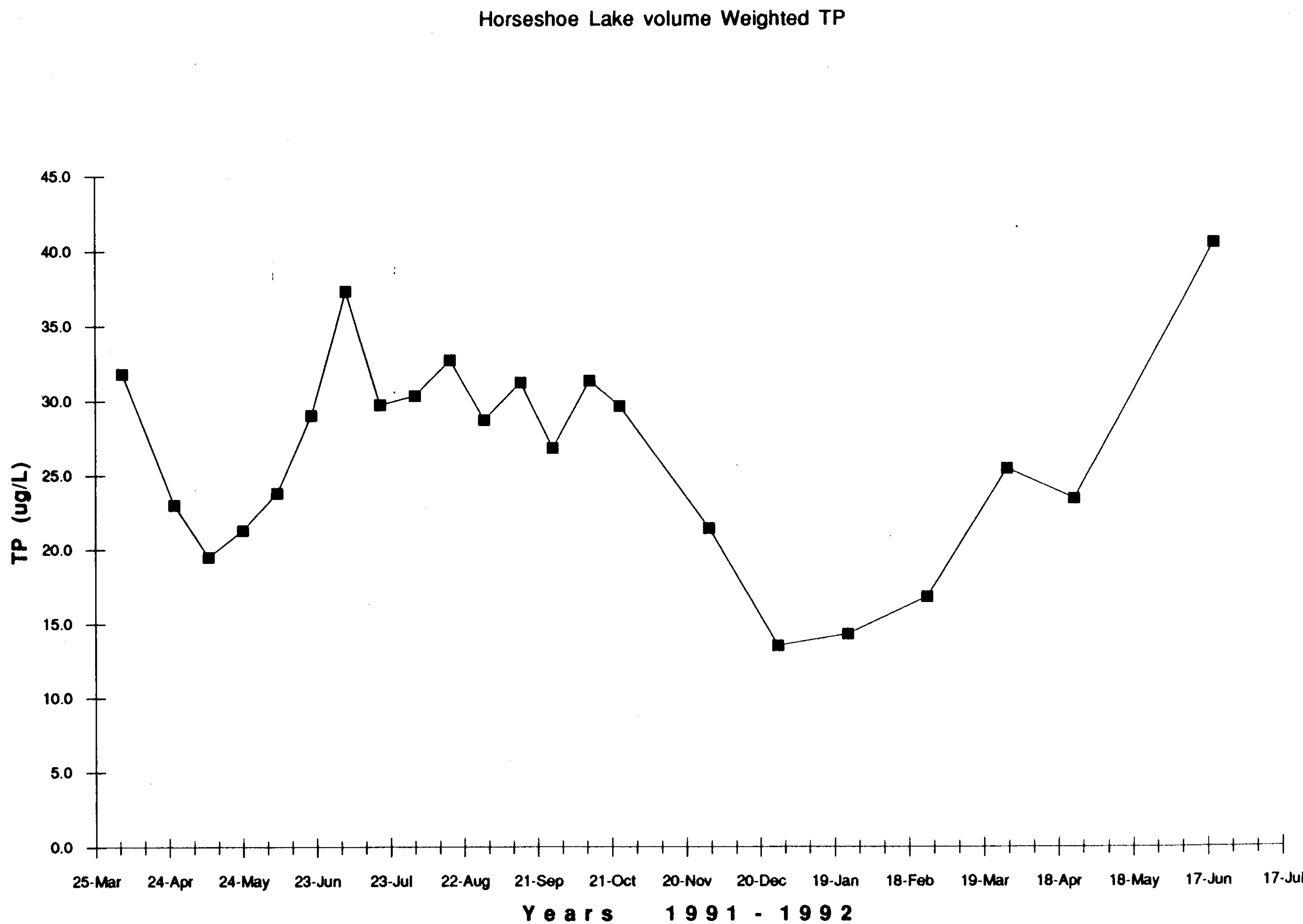


Figure 6. Whole lake, volume weighted mean TP concentration during 1991-1992.

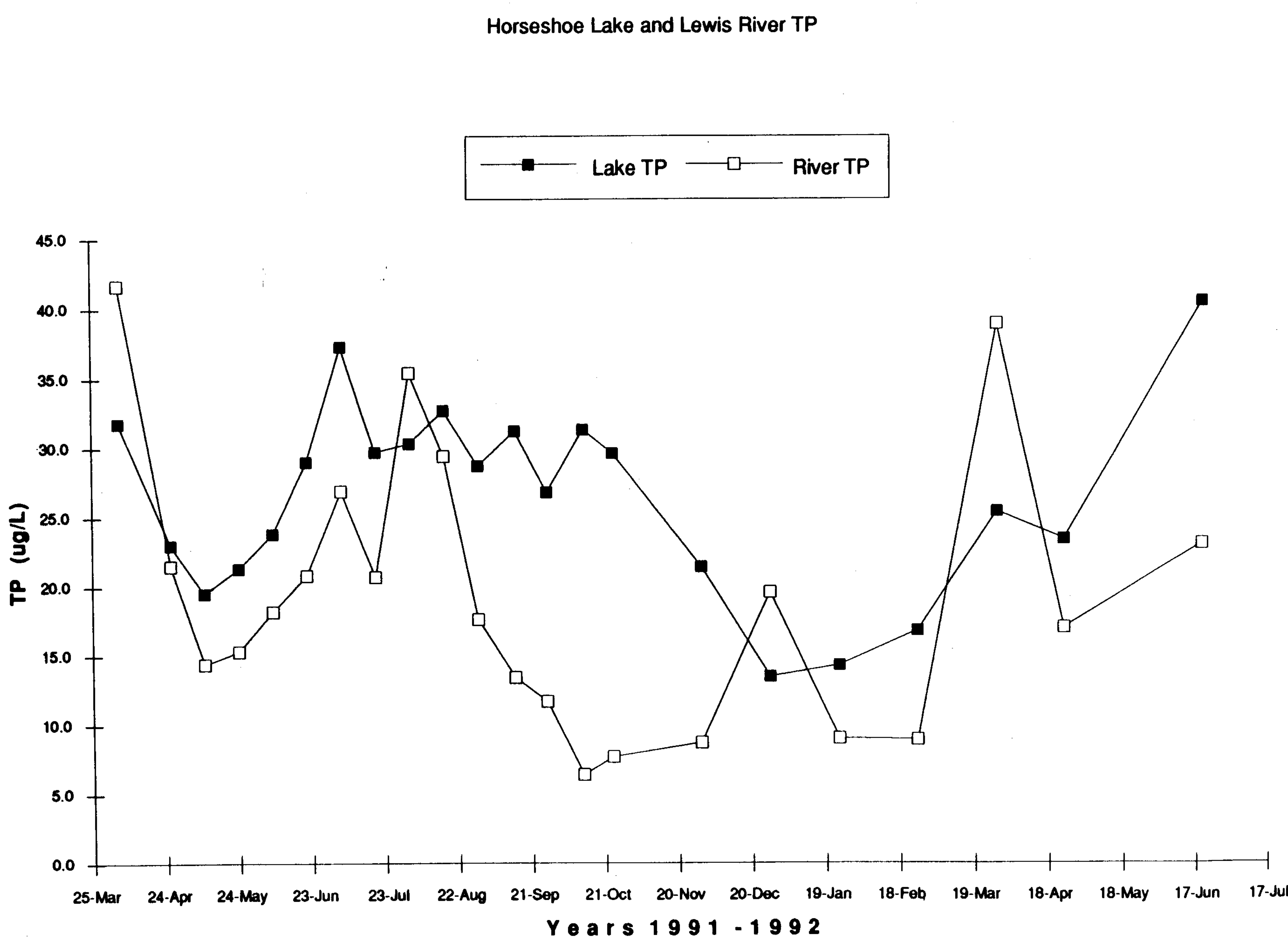


Figure 7. Whole lake, volume weighted mean TP and river inflow TP during 1991-1992.

lake water. Normally, if the principal source of P is the inflow, lake water should have less TP than the inflow due to sedimentation loss of particulate P.

The pattern for soluble reactive P (SRP) was reversed from that of TP, however (Figure 8). On no sampling occasions was inflow SRP less than lake SRP as was the case for TP. And the difference between lake and inflow was usually greater during the summer months. The explanation is that SRP is the form of P available to algae; they can take up soluble inorganic P (SRP) directly from the water, while P from the particulate fraction (=TP minus SRP) must be mineralized by bacterial or algal enzymes, if it is organic, or may not be available at all if it is part of inorganic particulate matter. The lower SRP in lake water than in the inflow probably represents a utilization of most of that difference by algae. Thus, the concentration of SRP in the inflow is really more significant than TP and it tends to increase in significance above a level of about 10 μg/L. The summer average in the inflow was in fact 10 μg/L, reaching maximums of over 15 μg/L both years.

Nitrogen is also an important nutrient for algal growth and can limit production under some conditions, which will be discussed below. Nitrogen can be characterized as total (TN), which includes inorganic nitrate (NO_3^-) and ammonium (NH_4^+) nitrogen as well as particulate organic and inorganic N (=TN minus inorg. N). As with TP, river inflow TN was usually less than lake TN (Figure 9). That means that sources inside the lake, or from sources other than river inflow, were the principle origin for lake TN. During the winter months when biological activity was lowest due to low temperature, lake concentrations roughly equaled those in the inflow.

The highest lake concentrations occurred in July, coincident with the algal bloom (Figure 9). During that time, river inflow TN remained low. The large amount of N associated with the bloom, representing a doubling of lake TN, could have originated from fixation of atmospheric N (N_2) by the blue-green algae present, or as ammonium from sediments, as in the case of phosphorus.

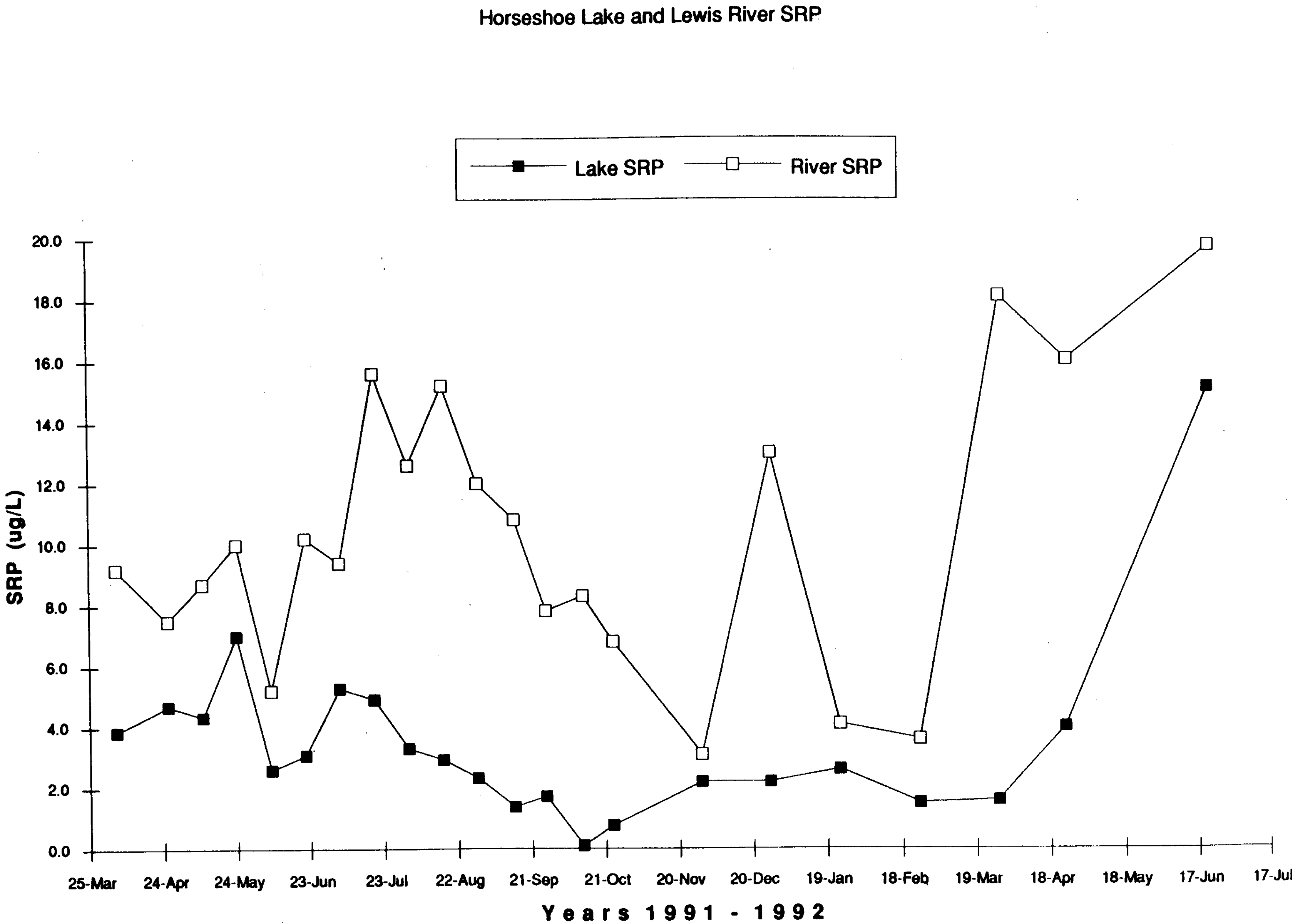


Figure 8. Whole lake, volume weighted mean and river inflow SRP during 1991-1992.

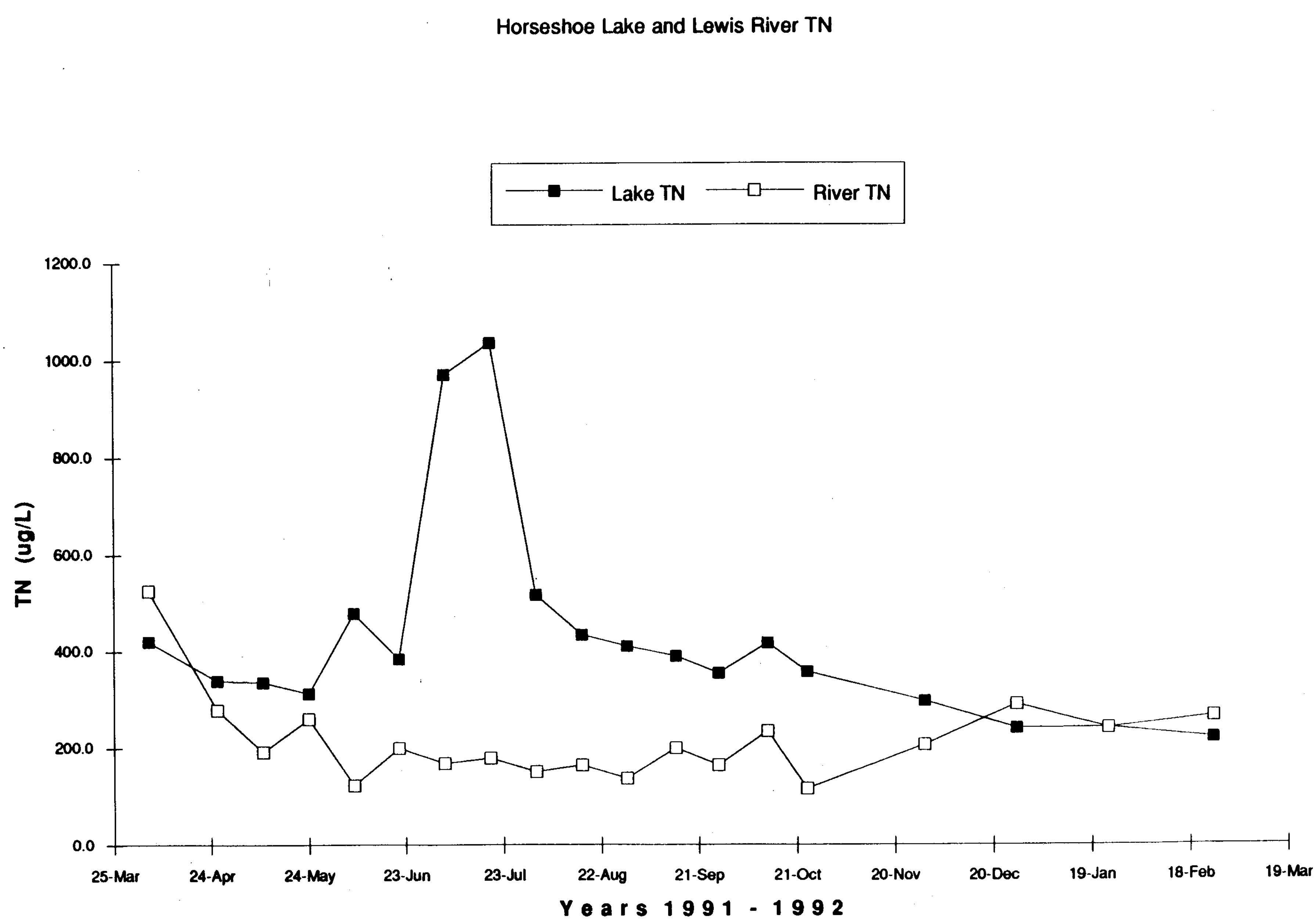


Figure 9. Whole lake, volume weighted mean and river inflow TN during 1991-1992.

Although interesting, the increase in TN was probably not the driving factor causing the algal bloom. The principle cause was probably P and the main source of that P was probably lake bottom sediments, as will be discussed later, with a minor portion from inflow SRP.

One important reason why N was probably not the principle cause for the algal blooms can be illustrated by the TN:TP ratio. If the ratio of TN:TP in water exceeds the ratio required for algal growth (if it were all available and absorbed from water), then P is more likely the growth limiting nutrient rather than N. The nominal value for algal demand is about 10:1, but the required ratio can vary. The TN:TP ratio during the bloom was 30:1 and during the summer it averaged 18:1. Therefore, P should have limited algal bloom development before N and would, therefore, be the nutrient of control in order to control algal biomass, transparency, DO, and the abundance of blue-green algae.

Soluble inorganic N and P data show a different trend than TN and TP, however. Like SRP, soluble inorganic N (IN) concentrations were usually higher in river inflow than in the lake (Figure 10). The difference between inflow and lake IN, on the other hand, appeared to be greater than for SRP, because IN declined to zero in May and June during development of the algal bloom (Figure 10 and see Phytoplankton). Thus, growth rate in the short term appears to have been limited by N rather than P. IN:SRP ratios verify that, because they decreased both in the river from about 16:1 to 8:1 and in the lake from about 2:1 to zero (no N available) during development of the bloom in May and June. Conceivably, the bloom maximum could have been curtailed by the scarcity of N, although N-fixing blue-green algae were present and were probably actively fixing N. That is suggested by the peak in TN during the bloom. Both NO_3^- and NH_4^+ increased to concentrations greater than the river inflow immediately following cessation of the bloom, probably a result of algal cell decomposition and recycling of particulate N to IN.

What is the significance of short term growth rate limitation by N and is it important? Low (or zero) IN:SRP ratios are a normal occurrence in heavily enriched

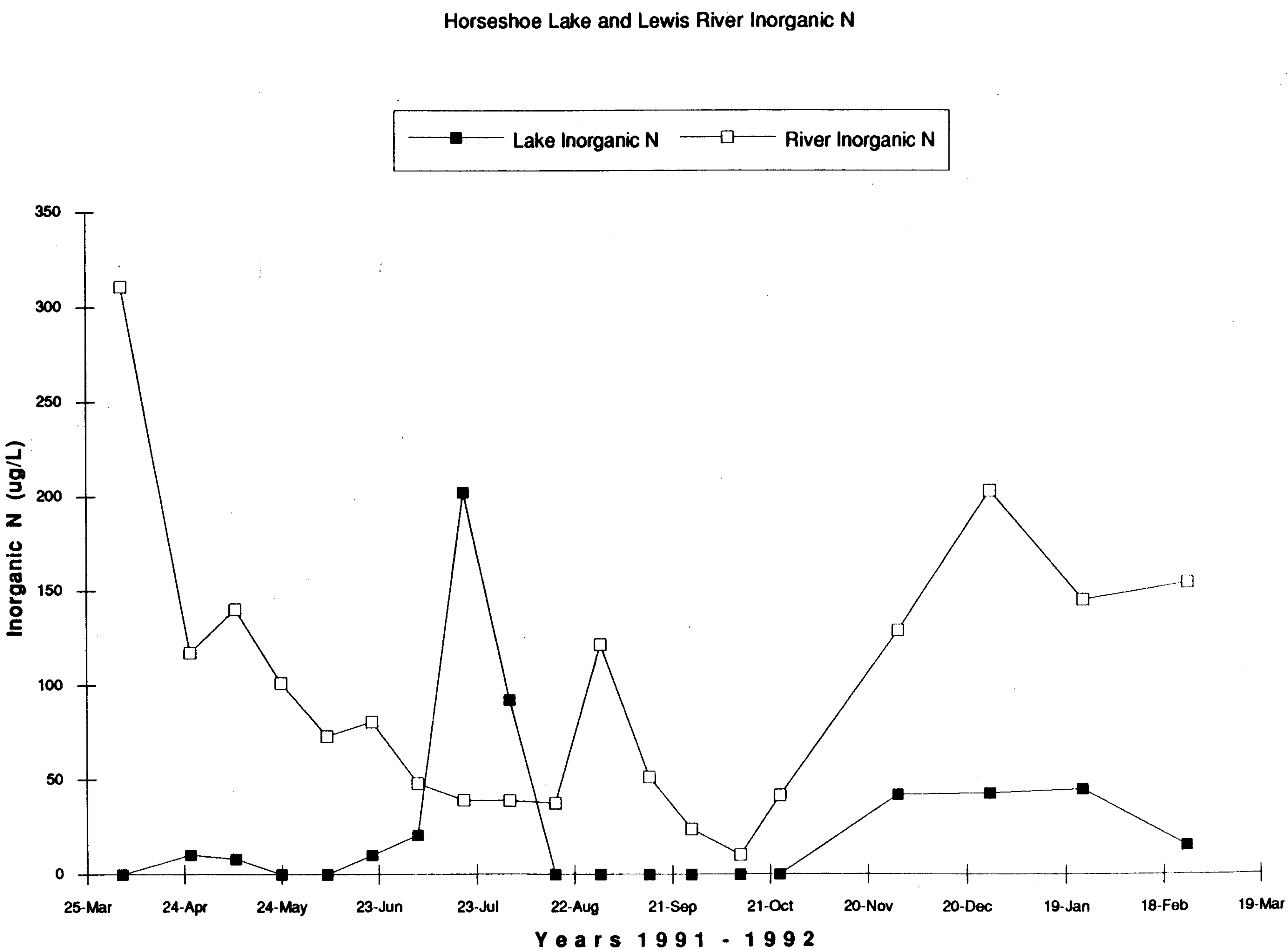


Figure 10. Whole lake, volume weighted mean and river inflow inorganic N (NO_3^- + NH_4^+) during 1991-1992.

lakes and even occur in moderately enriched lakes. While such low IN:SRP ratios that are observed in Horseshoe Lake are probably limiting short term growth rate, the resulting algal biomass is more likely dependent on the concentration of available P. Although IN is low or even zero, N is made available for growth from fixation of atmospheric N, which is an unlimited reservoir. The two-fold increase in TN during the bloom in May-June is evidence that a large quantity of N appeared in the lake. Since there was no indication of other internal or external sources at that time, and because it appeared coincident with the bloom, the origin must have been N-fixation.

Blue-green algae are in fact favored by low N:P ratios (Smith, 1986). Moreover, lake fertilization with N has been shown to reduce blue-greens (Stockner and Shortreed, 1988). However, the conventional approach is to control P, thereby not only resulting in an increased N:P ratio discouraging blue-greens, but also causing a reduction in total algal biomass (Sas et al., 1989).

Phytoplankton and Transparency: Two relatively large algal blooms were observed during June of each year (Figure 11). The 1991 early-summer bloom was rather short-lived, lasting less than one month; longevity of the 1992 early-summer bloom is unknown, however, because sampling ceased. Casual observations suggest that high algal concentrations persisted longer, well into fall, during the previous two years (1989-1990). The long, persistent blooms were instrumental in initiating the quest for Centennial funds to conduct this study.

Why the bloom was short-lived in 1991, but persisted in previous years is unclear. The possible influence of low IN was discussed previously, but so long as P is available, N-fixation can proceed and supply the needed N to maintain a bloom even if the growth rate is curtailed. However, N-fixation is a rather slow process, allowing blue-greens, such as *Aphanizomenon*, to grow at only about 5% per day (Horne and Goldman, 1972). Normally, plankton algae can grow up to 100% per day, not counting losses through sinking, grazing, and washout. If water exchange rate were substantial, say ≥

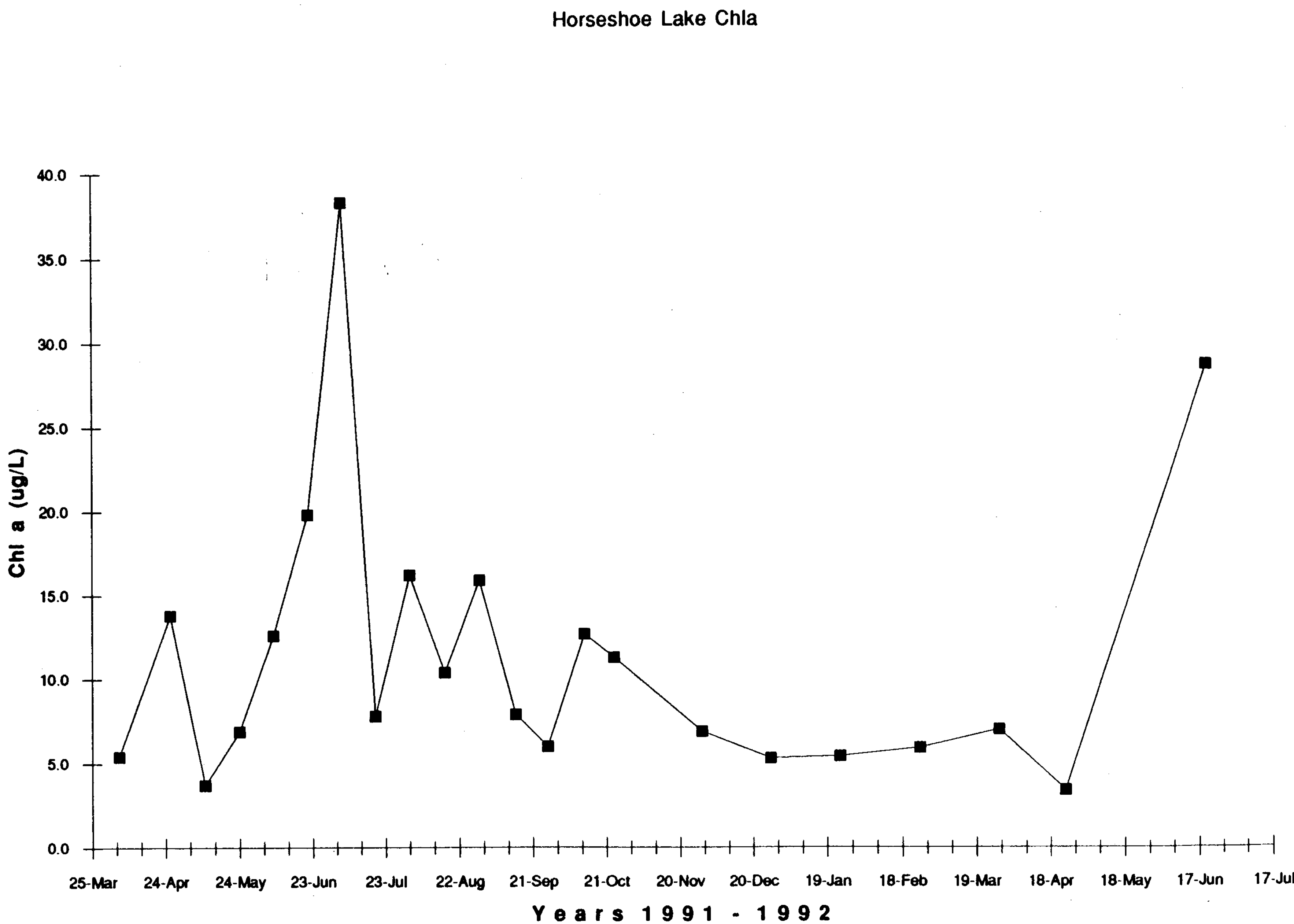


Figure 11. Whole lake, area weighted chl a concentration in the upper 4 m during 1991-1992.

10% per day, then the slow N-fixation dependent growth rate could result in cell washout through the lake outflow faster than cell increase through growth. However, that is probably not the case in Horseshoe Lake, through which water exchange is only 1.3% per day as a result of the inflow pumping rate.

The other possibility for the relatively low algal biomass during the late summer-fall period is wind mixing. As discussed previously (Figure 4), the bloom occurred during the most stratified period during mid-June to mid-July (maximum surface-to-bottom temperature difference). After that, the lake tended to be relatively well mixed (Figure 4). Mixing has been shown to discourage bloom persistence of blue-green algae (Shapiro et al., 1982), and, therefore, as a possible benefit of artificial circulation (Cooke et al., 1986).

Chl a averaged 15 µg/L during the summer. That is about 6 µg/L higher than the suggested threshold indicating an eutrophic state (Porcella et al., 1980). Although the lake is known for its large, persistent algal blooms, the lake state was eutrophic, based on TP and chl a during 1991, but not strongly so.

Transparency, as determined by a Secchi disc, averaged 1.2 m (3.9 feet) during the summer. The maximum (3.2 m) occurred in January and the minimum in early July coincident with the peak of the algal bloom (Figure 12). Transparency was generally lowest during summer-fall, when algae was most abundant, and highest in winter when algae content (chl a) was lowest. The eutrophic threshold is ≤ 1.9 m, so transparency indicated that Horseshoe Lake was eutrophic during the summer of 1991. Average transparency during the summer months of 1988-1989 was 1.0 m, slightly less than in 1991, corroborating the apparent greater and more persistent algal abundance during earlier years (Somers, 1989).

Transparency was also inversely related to TP concentration during the year (Figure 13). One would normally expect a closer indirect relation between Secchi and chl *a* than between Secchi and TP, because chl *a* should be more directly related to

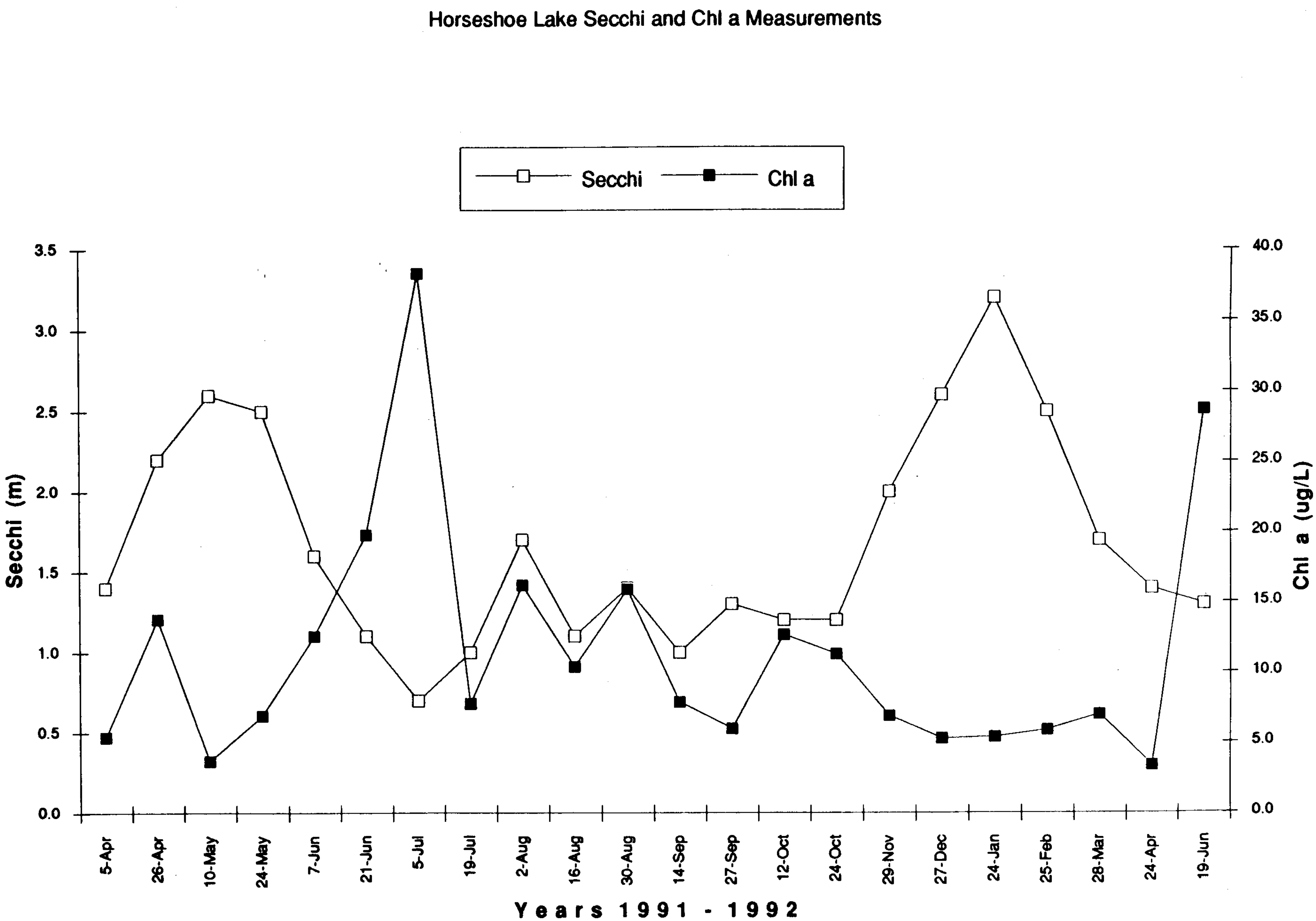


Figure 12. Whole lake, area weighted Secchi disc transparency and chl a in the top 4 m during 1991-1992.

particulate algae, which is the principle cause for light attenuation in this lake. In this case, however, most P may be tied up in algal cells causing TP to be the better index.

That lake turbidity was caused primarily by in-lake growth of algae, and not by suspended particles entering with the river water, is illustrated in Figure 14. Values in the lake were much higher than in the inflow during the summer-fall period. Also, turbidity, measured as the amount of light scattered by suspended particles, showed about the same seasonal pattern as did chl a (Figure 11).

Zooplankton: Abundance of all zooplankton as well as the species of the large cladoceran zooplankter, *Daphnia*, reached a maximum the first week in May. The levels for total zooplankton and *Daphnia* were both relatively high at 160 and 80/L, respectively (Figure 15; Appendix E). There was a general downward trend in zooplankton during summer and fall with the initial decrease being rather abrupt. The abrupt decrease (about 75% in a month) in *Daphnia*, as well as total zooplankton, occurred coincident with the increase in algae, which reached a maximum around the end of June.

Herbivorous zooplankton, especially *Daphnia*, are effective grazers of plankton algae. Levels of *Daphnia* in Lake Washington, which were only one-half those in Horseshoe Lake, were found to effectively increase water transparency by their grazing activity (Edmondson and Litt, 1982). While *Daphnia* are effective at removing plankton algae of relative small size, their grazing activity is inhibited by the presence of filamentous blue-green algae, which are difficult for them for filter from the water. The filamentous blue-green, *Oscillatoria*, inhibited the filtering process of *Daphnia* in Lake Washington (Infante and Abella, 1985). Only after *Oscillatoria* declined did *Daphnia* populations increase and improve water transparency.

A blue-green, primarily the filamentous form *Anabaena*, was the primary bloom found in Horseshoe Lake in June, representing 94% of the algal biomass. Thus, there is reason to speculate that the presence of a large population of the blue-green *Anabaena*

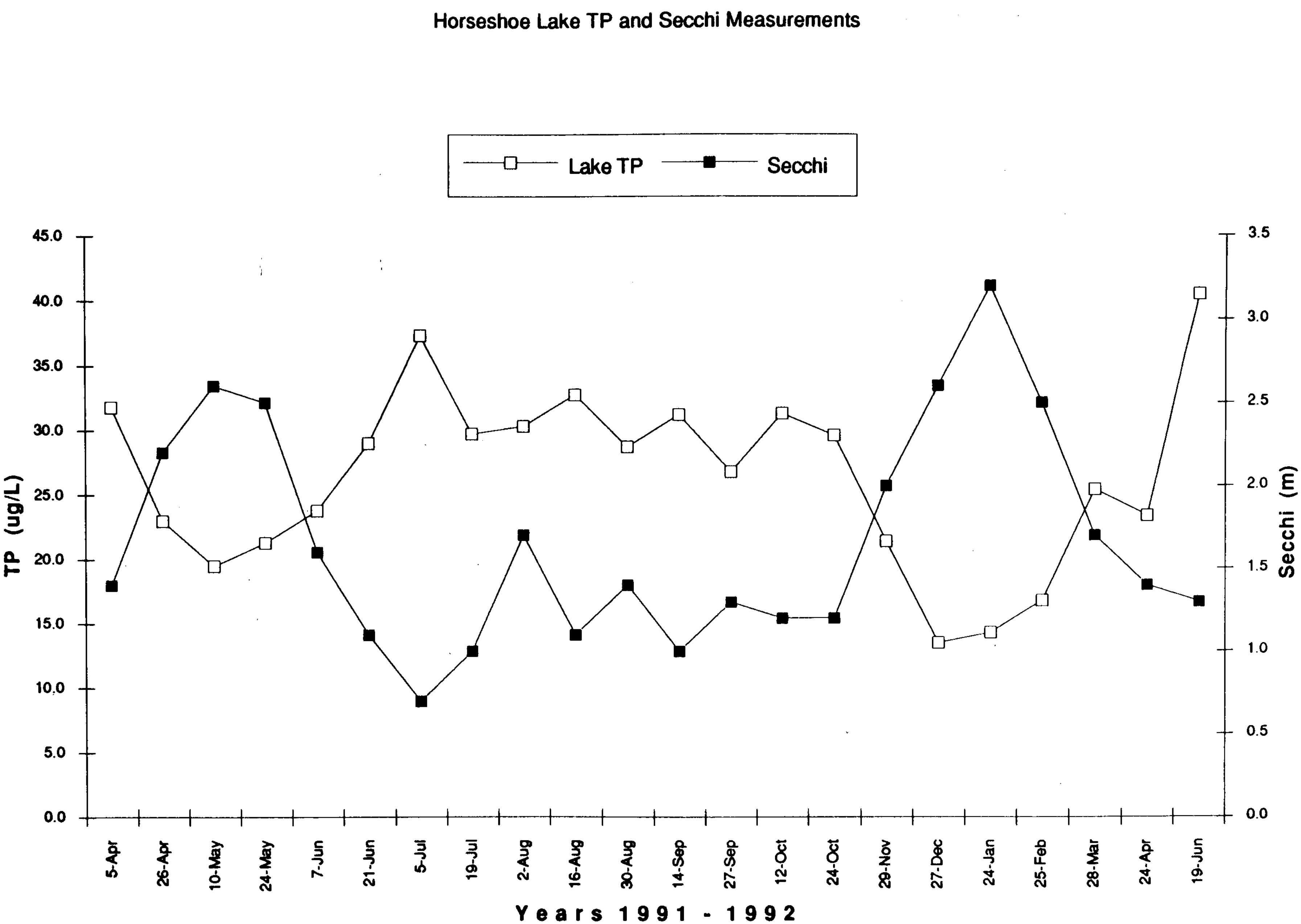


Figure 13. Whole lake, volume weighted TP and area weighted Secchi transparency during 1991-1992.

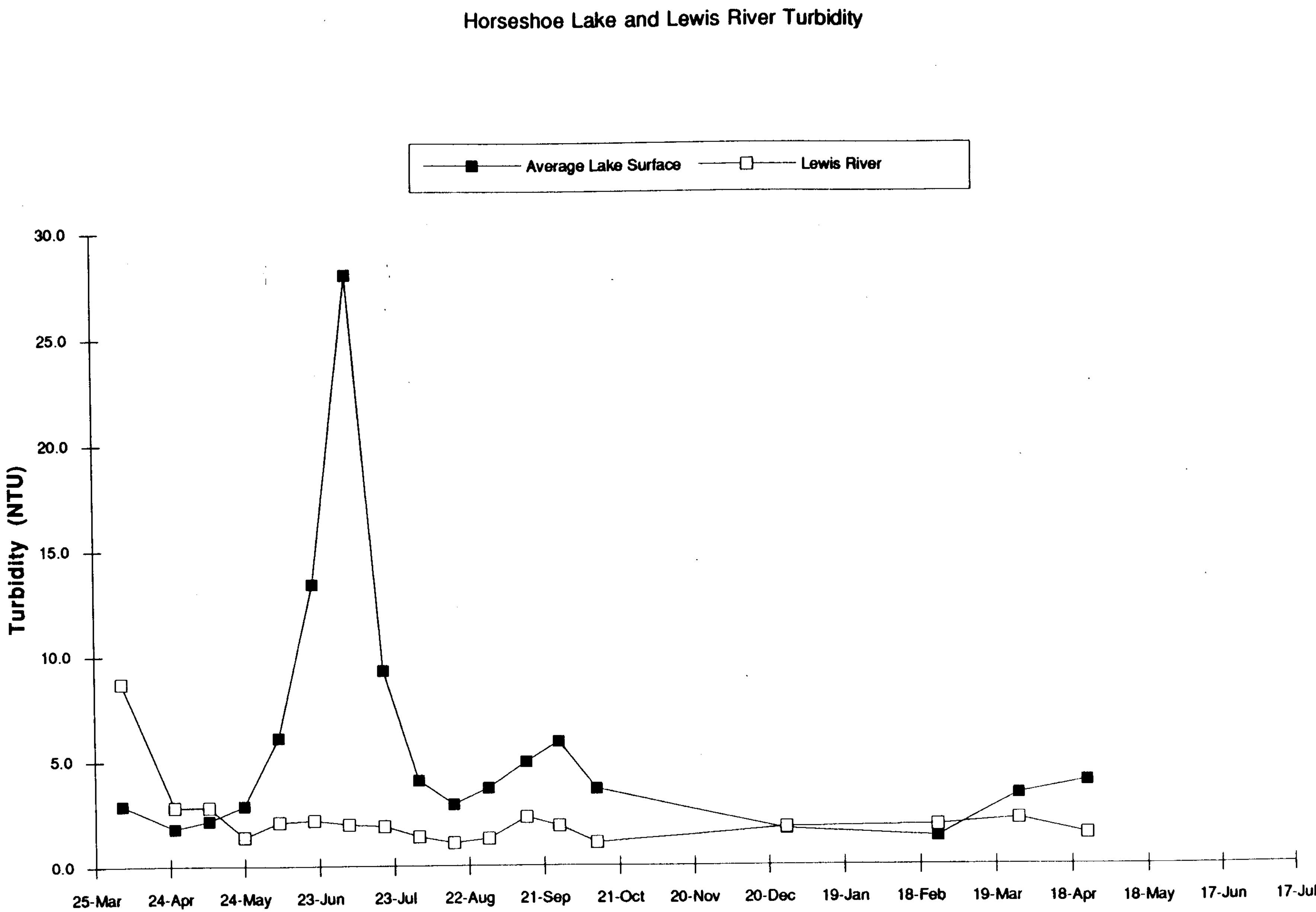


Figure 14. Whole lake, area weighted turbidity in the lake compared to river inflow turbidity during 1991-1992.

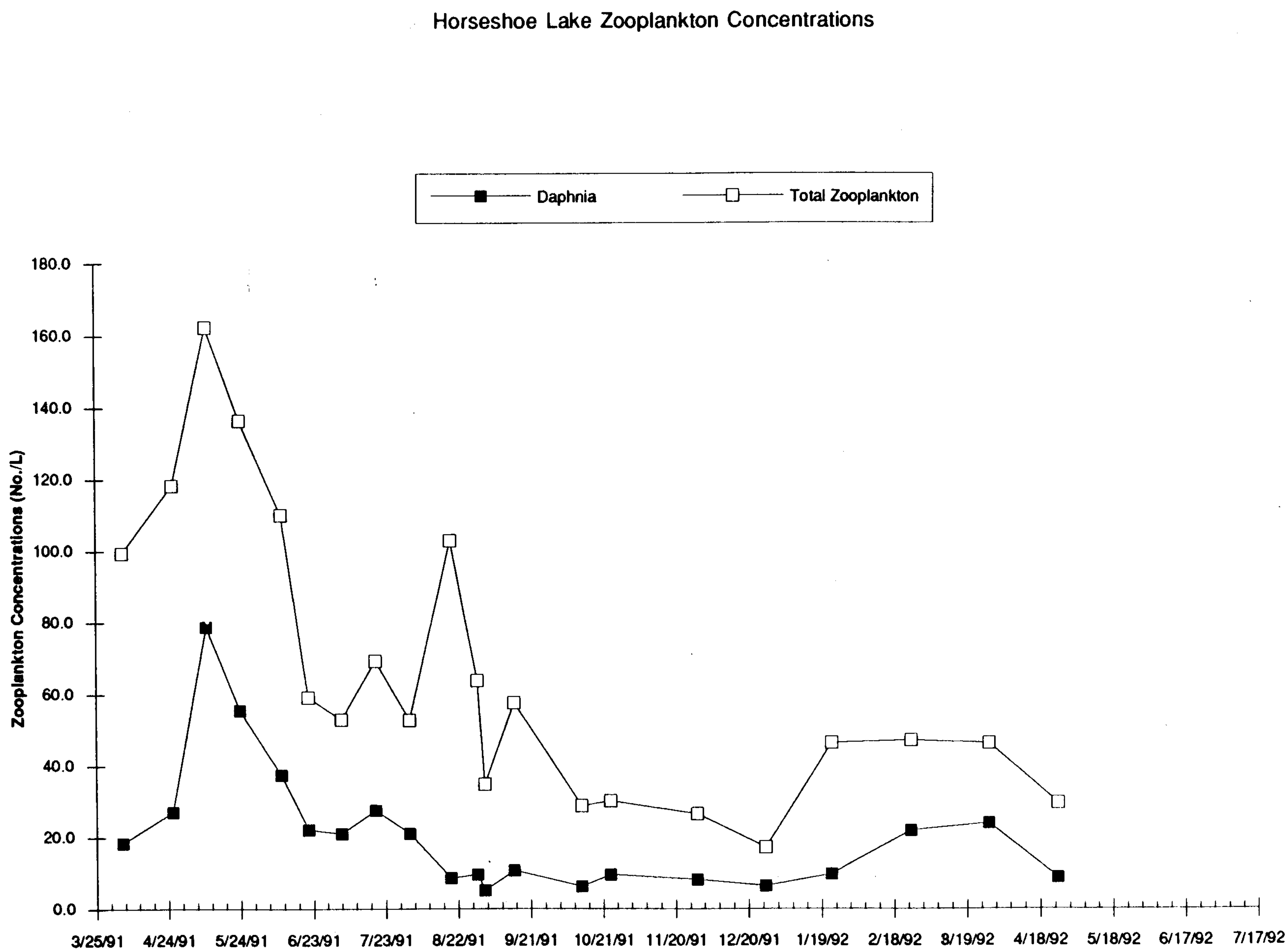


Figure 15. Abundance of cell zooplankton and _Daphnia_ from vertical net hauls at station 1.

was a deterrent to the *Daphnia* populations in Horseshoe Lake. Furthermore, if blue-green blooms are reduced, by limiting the P supply, *Daphnia* may persist at greater abundance throughout the spring-summer period. In this way, reduction of blue-green algae could benefit fish growth, since the large Daphnids are a preferred food source.

Predation by fish can also effectively reduce zooplankton, especially *Daphnia*, because it is large. The practice of annually planting large numbers of hatchery rainbow trout in western Washington lakes prior to the opening of the fishing season can drastically lower *Daphnia* abundance, thereby reducing the control on algae by grazing. There were 21,570 catchable size trout planted during April, 1991. That is 620 fish per ha. The trends evident in Figures 11 and 15 suggest that the algal bloom was a result of reduced grazing caused by fish predation on *Daphnia*. *Daphnia* and other zooplankton began declining in early May, following the April fish planting. Moreover, the trout stomachs examined contained *Daphnia*, although there were other food items present as well.

In summary, *Daphnia* are effective grazers of plankton algae; they can consume algae, if of edible size, faster than the algae can grow (Welch, 1992). In that way, they can maintain a relative clear-water condition. However, for *Daphnia* to persist, the plankton algae should not be dominated by filamentous blue-green algae and the lake should not be overpopulated by planktivorous (*Daphnia* eaters) fishes.

Fish: Fish taxa represented in the catch from the nets included, brown trout, rainbow trout, large mouth bass and suckers. All taxa were present at site 1, while only suckers were found at site 2. Surveys conducted in 1984, 1989 and 1990 found similar taxa present. Results from the March, 1992 survey follow.

Recreationally important species in the lake include rainbow trout, brown trout and large mouth bass. Presently, the Department of Wildlife stocks the lake with brown

Table 1. Fish catch results

Site 1	Weight g	Length mm
Rainbow Trout (*Oncorhynchus mykiss*)	298	293
Brown Trout *Salmo trutta*)	290	315
	240	287
Large Mouth Bass (*Micropterus salmoides*)	250	251
Sucker (*Catostomus macnocheilus*)	613	361

Site 2	Weight g	Length mm
Sucker	913	439
	535	365
	687	381
	971	431
	1366	490
	1146	460
	1042	450
	1058	452

and rainbow trout. In 1992 approximately 33,250 rainbow trout juveniles, 17,000 legal length rainbow trout and 150 rainbow broodstock were stocked in the lake from April to July. In addition, approximately 10,000 legal length brown trout were stocked during April-May (Lucas, personal communication). Historically, large mouth bass were planted in the lake and those caught in the net sets were probably recruitment from that initial stocking.

A survey conducted in 1984 using an electroshocker provided more thorough information on the extent of fish diversity present in the lake. In addition to the recreationally important species mentioned above, the largescale sucker, brown and yellow bullhead, carp, squawfish, goldfish, sculpin, and yellow perch were collected in the survey.

Horseshoe Lake is a popular sport fishing location within the Woodland area. This is due to the abundance of sport fish present during the spring months as well as the lake's accessibility. Much of the shoreline is accessible to fishing. In addition, a boat launch is present at Horseshoe Lake Park.

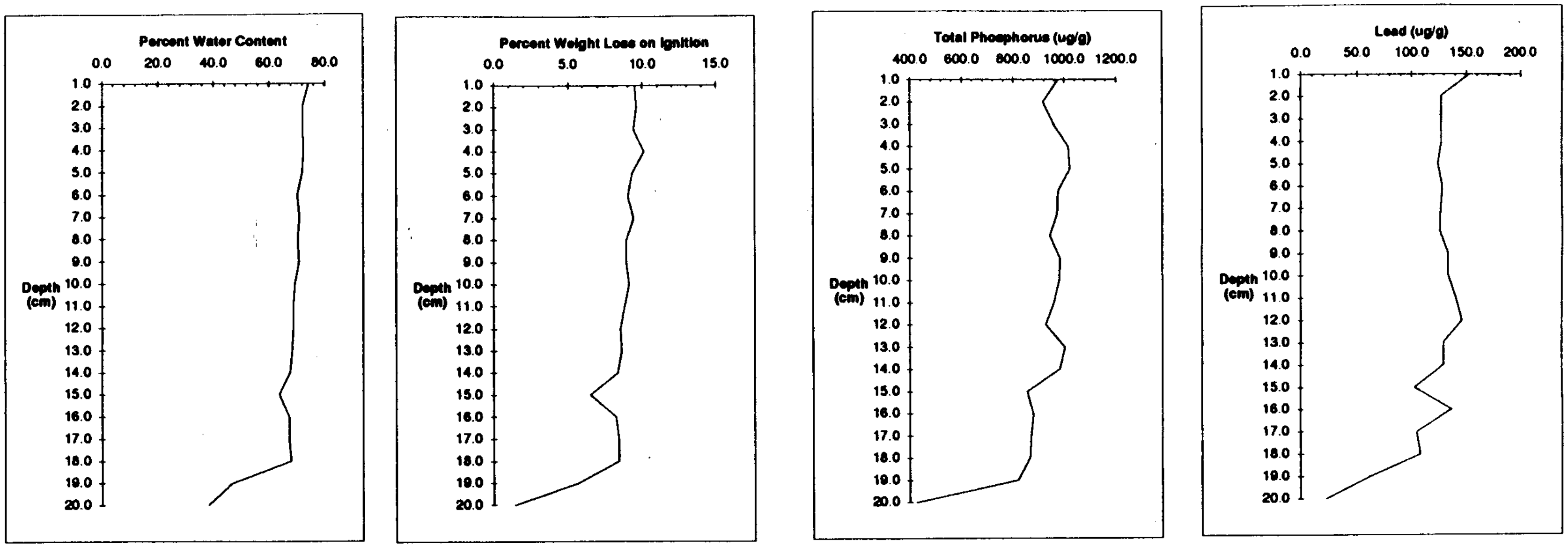
Lake Sediment Characteristics and Chronology

Results of analysis of four sediment cores, two from each of two stations located at the north arm (Station 1) and half way between north and south arms (Station 2) are presented in Figure 16 and Table 2 (for one core from each station) and Appendix D for replicate cores. The profiles of water and organic matter contents at Stations 1 (deepest portion of the lake, 7 m) and 2 (about 4 m) reveal two different types of sediments and sedimentation regime). As expected, sediments at the deep station have higher organic matter and water contents than the sediments at the shallower station. These differences are also reflected in the bulk densities (dry weight per 1-cm deep sections of 9.32 cm^2 cross-sectional area). Thus, with the exception of the bottom 2 cm of the core from Station 1, bulk densities at Station 2 are nearly 1.5 to 2 times higher than those measured at Station 1. Although sediments at both stations show large autochthonous inputs (primarily inorganic), sediments at Station 1 appear to be appreciably modified by autochthonous inputs, resulting in higher organic contents, 9% versus 6% at Station 2. The 9% organic matter contents of sediments at Station 1 are not far below those measured in recently deposited sediments in large and small lowland lakes, such as Lakes Washington, Sammamish and Meridian.

Phosphorus contents of Horseshoe Lake sediments are low (below 0.1%) remaining nearly constant with depth, about 0.1% P at Station 1 versus .07% at Station 2. There are no pronounced increases in P concentrations at the top few cm as is normally seen in lakes with pronounced oxygen stratification. Upward diffusion of P in anaerobic sediments results in elevated P content in the top 2-5 cm. The surface values are often 50 to 100% higher than those at depths greater than 5 cm. The absence of distinctly higher P contents at the surface sediments in Station 1 may also indicate that there is not extensive focusing of fine sediment particles rich in easily decomposable organic matter which will induce anaerobic conditions in sediments. Alternatively, fine detritus material may be washed away by the pumped in river water.

The profiles of stable lead at both sediment stations remain practically unchanged with depth, in mark contrast to Pb profiles from other lowland lakes, such as Silver Lake (Welch et al., 1989). There are not distinct sections in the Pb profiles with elevated Pb

Station 1



Station 2

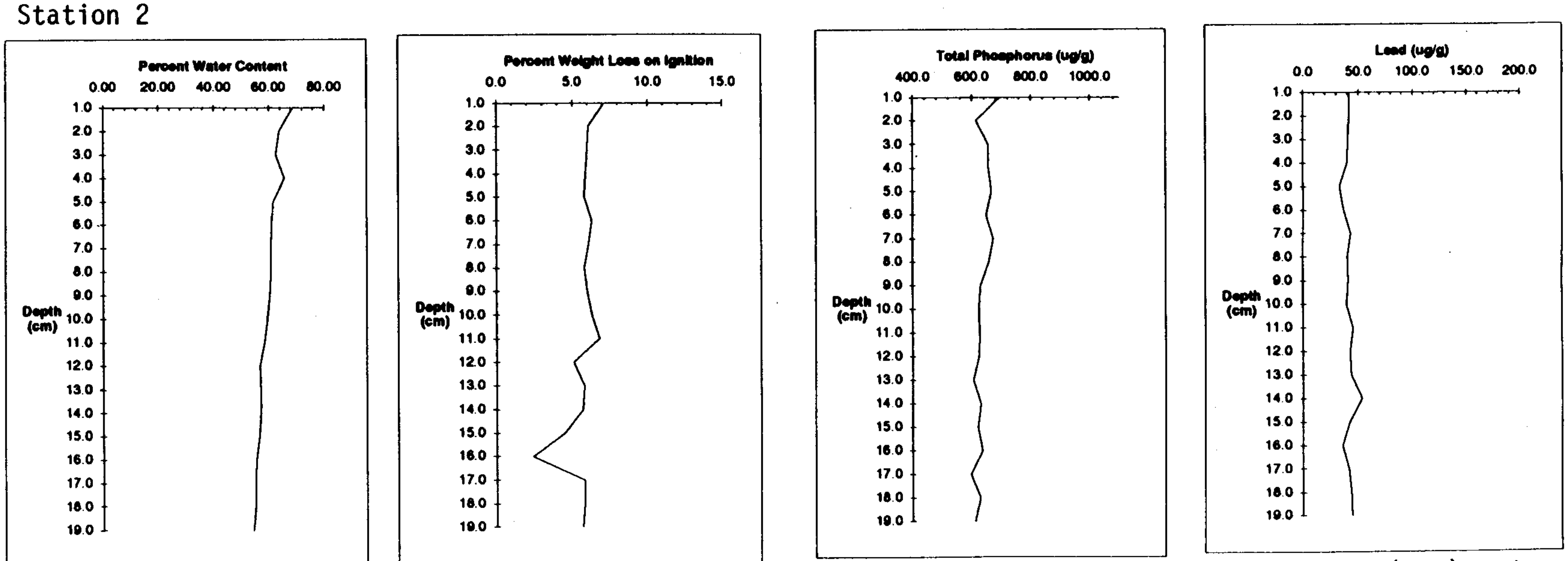


Figure 16. Comparison of % H_2O, weight loss on ignition, P and Pb of sediment from Station 1 (7 m) and Station 2 (3.5 m) collected September 1991.

Table 2. Sediment Characteristics

STATION 1

Depth cm	Water %	Organic Content % *a	Dry Weight g *b	TP ug/g	Pb ug/g	Zn ug/g	Al %	Fe %
1	74.29	9.5	0.7859	974.5	152.4	178.4	2.05	3.60
2	72.11	9.6	1.3288	918.2	127.1	146.6	1.75	2.51
3	72.10	9.4	1.8073	959.7	127.1	141.6	1.76	2.64
4	72.51	10.1	1.8302	1014.9	127.5	146.3	1.84	2.75
5	72.05	9.3	2.0514	1022.6	124.2	140.4	1.80	2.69
6	70.28	9.0	1.9818	974.6	128.7	145.7	1.81	2.71
7	70.96	9.4	2.2171	971.7	126.8	144.5	1.84	2.67
8	70.42	8.9	1.9420	942.4	126.1	146.5	1.83	2.60
9	70.85	8.9	2.0762	983.2	133.4	158.1	2.22	2.94
10	69.34	9.1	2.1741	979.2	133.4	158.8	2.16	2.92
11	68.80	8.8	2.3091	959.0	140.5	164.4	2.22	2.98
12	68.74	8.5	2.3535	927.9	146.1	170.6	2.36	3.14
13	68.35	8.6	2.5691	1003.2	129.4	153.2	2.07	2.82
14	67.55	8.3	2.4467	982.1	129.3	153.6	2.04	2.77
15	63.69	6.5	3.0611	854.0	102.1	124.0	1.68	2.32
16	67.16	8.2	2.6350	877.6	136.7	168.8	2.39	3.03
17	67.16	8.4	2.7107	868.7	103.9	127.1	1.69	2.12
18	67.95	8.4	2.7308	864.3	107.7	128.7	1.71	2.14
19	46.77	5.7	4.3669	817.5	62.4	89.6	1.25	2.00
20	38.30	1.4	8.7598	426.0	22.0	40.7	0.59	1.00

* a Percent weight lost on ignition at 550 degrees celcius

* b Dry weight per section (A=9.32 cm^2)

STATION 2

Depth cm	Water %	Organic Content % *a	Dry Weight g *b	TP ug/g	Pb ug/g	Zn ug/g	Al %	Fe %
1	68.63	7.1	2.4722	693.6	41.3	111.6	1.43	2.43
2	63.67	6.1	3.2296	613.3	41.6	82.7	1.45	2.07
3	62.53	6.0	3.2771	655.0	40.7	83.8	1.40	2.12
4	65.63	5.9	3.4964	655.8	39.8	85.6	1.37	2.18
5	61.58	5.8	3.5005	666.2	33.0	74.5	1.25	1.96
6	60.90	6.3	3.5325	647.0	36.5	81.7	1.28	2.07
7	60.78	6.1	3.7151	671.3	43.3	90.1	1.49	2.35
8	60.76	5.8	3.7494	656.4	40.3	87.7	1.50	2.33
9	60.22	6.0	3.7572	627.6	41.1	87.5	1.43	2.28
10	59.31	6.3	3.9139	621.7	39.1	86.6	1.42	2.31
11	58.29	6.8	4.0768	625.6	45.2	94.9	1.61	2.57
12	56.70	5.1	4.3359	622.4	42.3	86.3	1.48	2.40
13	56.97	5.8	4.4058	603.9	43.7	86.3	1.44	2.31
14	56.91	5.7	4.3472	629.7	53.5	106.0	1.82	2.89
15	56.30	4.5	4.4660	619.7	42.2	84.1	1.44	2.30
16	55.34	2.4	4.6019	635.5	35.5	77.6	1.22	2.00
17	55.02	5.8	4.7800	595.6	41.9	88.5	1.42	2.29
18	54.99	5.8	4.7029	627.2	44.1	88.9	1.47	2.34
19	54.27	5.7	4.7900	610.1	44.9	90.9	1.51	2.43

* a Percent weight lost on ignition at 550 degrees celcius

* b Dry weight per section (A=9.32 cm^2)

contents corresponding to large Pb inputs from peak consumption of tetraethyl leaded gasoline in the 1950-1970 period. Likewise, nearly complete elimination of leaded gasoline since early 1980 is not reflected in lower Pb contents of surface sediments. This lack of stable Pb markers in recent sediments (top 20 cm or so) is not readily explainable. Dredging of the deep (Station 1) sediments in 1960 revealed sediments with Pb contents of about 22 ppm at the 20 cm section which are more than double of Pb contents measured in the background sediments of the lowland lakes of Western Washington. Comparison of Pb profiles from Stations 1 and 2 shows that sediments at Station 1 are 2 to 3 times richer in Pb, indicating perhaps that the source of Pb is runoff material rich in Pb entering the lake at the north arm. The proximity of the freeway may be responsible in producing storm runoff rich in Pb-containing material.

The Zn sediment profiles are similar to the Pb profiles showing no distinct patterns, appearing nearly uniform throughout the core. Zn contents at Station 1 are 1.5 to 2 times higher than those at Station 2. Interestingly, Zn/Pb ratios 2 to 1 in sediments from Station 2 are nearly double of those observed at Station 1 core for the 0-18 cm section. Beginning with the 19 cm section the 20 cm section has Zn/Pb ratio just below 2, nearly identical to that observed at Station 2.

The HNO_3 extracted Fe and Al represent only a portion of their total sediment contents, for HNO_3 does not dissolve aluminosilicate minerals. There appears that more Al was extracted from Station 1 sediments and to a lesser degree this is also true for Fe. This may reflect the finer nature of the material deposited at Station 1. There is some evidence of Fe mobilization at the top 1 cm sediments from Station 1. Sediments from this section have Fe contents 50% higher than those in the 2-10 cm section.

The presence of coarser sediments, described by the lower organic composition, higher bulk density and lower P, Pb and Zn contents, clearly delineates the 19-20 cm core section from the shallower depths (0-18 cm). It is assumed that these sediments were deposited after the 1960 dredging of the bottom sediments. If this assumption is true, then the sedimentation in the north arm of Horseshoe Lake is large, averaging about 0.6 cm/y. A comparable sedimentation rate (~0.6 cm/y) is obtained if we assume that the slightly elevated Pb contents of sediments at 11-12 cm section of the core from Station 1 are coincident with peak gasoline consumption in about 1970-1976. Based on this rate and actual dry weight measurements of the core sediment, a deposition rate of 0.131 g dry sediment/cm^2-y is calculated for the post 1960 deposited material. Multiplying this deposition rate by the average P contents (0.091%) of the 0-18 cm sediments and expressing the results per m^2 produces a P deposition rate of about 1.19 g P/m^2-y which is nearly three times larger than independently measured and estimated surface P loadings. Correcting the 1.19 g P/m^2-y deposition rate for sediment focusing will produce a more realistic value. Given the shape and morphology of the lake, it would be difficult to arrive at a permanent sedimentation area. Assuming a permanent sedimentation area about equal to 1/3 of lake surface area (a rather conservative estimate), it will produce a P deposition rate of $1.19 \times 1/3 \simeq 0.4$ g/m^2-y.

Water Budget

The water budget covered the period from April 1991 to March of 1992. The budget included water inflows and outflows from the lake, taking into consideration their effect on the lake's volume. The budget was completed by balancing the following equation:

$$\Delta \text{Lake Volume} = \Sigma \text{ Inflows} - \Sigma \text{ Outflows}$$

where Σ = sum.

Description of Inflows/Outflows: The principal sources of inflow to the lake include water pumped from the Lewis River, precipitation falling directly on the lake surface, and stormwater runoff from within the watershed, primarily the City of Woodland.

Discharge to the lake from the Lewis River pumping system was assumed to be constant at a rate of 15,008 m^3/day. The pump was operated throughout the study period, 55 minutes of each hour. A five minute backflow period each hour allowed for clearing the pipe system of debris. The discharge enters the lake at the eastern edge of the north arm (Figure 1).

The pumping rate is a conservative estimate and is based on discharge measurements made in 1989 and discussion with Washington State Department of Transportation personnel (Garcia 1992). However, deviations in discharge occur with changes in the water elevation between the lake and river. Flow in the river is controlled primarily by discharge from Merwin Dam, a hydroelectric facility operated by Pacific Power and Light. Daily fluctuations in river flow (and head difference) can be significant. In addition, the river is tidally influenced within the reach of the pumping station, causing elevation changes of up to several feet.

During the study period, the pump operated continuously beginning in February, 1991. This deviated from the usual management practice which typically began pumping operation in March and was halted in October.

Daily rainfall information was obtained from the Bonneville Power Administration, which has a rainfall monitoring station in Woodland. The volume entering the lake from precipitation was derived from multiplying the rainfall depth by the area of the lake.

Stormwater runoff volume entering the lake was determined from relationships derived by the United States Soil Conservation Service (SCS). These relationships are based on soil characteristics and the level and type of development within the watershed. Six storm drains enter the lake; two on the eastern edge of both arms which drain the siding road, one located at Horseshoe Lake Park and another drain located on the north shore, west of the Park (Figure 1). The latter two drains receive stormwater runoff from the business and residential areas of Woodland. In total, the lake receives stormwater runoff from an approximate area of 31.2 ha, with the majority being directed to the north arm of the lake. Runoff is minimal from the remainder of the watershed due to the highly porous soils.

Inflow may occur from groundwater from the inner land area of the lake (wells 15, 16, 17, 18) and through seepage (well 9), although the amount is assumed to have been insignificant in comparison to the other sources (see Figure 3).

The majority of the outflow from the lake occurs through a gate valve and overflow structure and through groundwater exfiltration. Also, approximately 17,262 m^3 of water was withdrawn from the lake during the summer months for irrigation. A small amount of water is also withdrawn and cycled through a refrigeration system in a slaughterhouse located on the south arm of the lake.

The outlet gate valve did not function during the study period. The valve was locked into a partially opened position allowing water to leave the lake. A backup valve system located at the Woodland airport was used to regulate the outlet flow. As a result, outlet flow measurements were not possible. That necessitated grouping the outflow volume through the structure with that via groundwater loss during April through June and December through March. The backup outlet valve was closed from mid June to late December, so during that period all outflow was considered to be through groundwater loss and evaporation from the lake surface.

Evaporation accounted for a significant part of water loss from the lake. Estimates were based on ten-year averages from the Puyallup weather station. A coefficient of 0.7 was used to correct pan evaporation rates to lake conditions.

Computed Water Budget: Table 3 provides a monthly accounting of the water inflows and outflows to and from the lake. From the yearly flow totals, several hydraulic characteristics for the lake can be determined: hydraulic load (q_s=Qo/A) was 16.8 m/yr;

Table 3. The water budget for Horseshoe Lake during April, 1991 through March, 1992

	Horseshoe Lake Inflow					Horseshoe Lake Outflow	
	Lewis River Inflow	Precipitation	Storm Water	Total Inflow	Lake Volume Change	Evaporation	Grdwtr + Outlet
	cubic meters	cubic meters	cubic meters	cubic meters	cubic meters	cubic meters	cubic meters
Apr-91	464685	61944	35796	562425	21814	16268	524343
May-91	464685	30562	11787	507034	-15148	26041	496141
Jun-91	449695	19983	7242	476920	-71500	29814	518605
Jul-91	464685	1262	446	466393	22722	34021	409650
Aug-91	464685	9960	2830	477474	-53322	30680	500116
Sep-91	449695	0	0	449695	-45445	17814	477325
Oct-91	464685	19503	8835	493023	-1515	7052	487486
Nov-91	449695	91571	51995	593261	99979	3711	489571
Dec-91	464685	43337	18026	526048	9468	4021	512559
Jan-92	464685	43665	21057	529406	12346	5753	511308
Feb-92	434705	38653	18082	491440	105432	5381	380627
Mar-92	464685	15275	4167	484126	-35220	10144	509201
Yearly Total	5501267	375715	180262	6057244	49611	190701	5816932

water residence time (T_w=V/Qo) was 0.22 yr or 78 days; and flushing rate ($\rho = \frac{1}{T_w}$) was 4.8/yr.

Water pumped from the river provided the majority of the inflow. That source accounts for approximately 91% of the inflow total during the budget period (Table 3). Precipitation and stormwater accounted for 6% and 3% of the total inflow, respectively.

Precipitation and stormwater inflows peaked in November. During that month 91,571 m^3 of precipitation and 51,995 m^3 of stormwater entered the lake. This accounted for approximately 24% of the yearly precipitation and 29% of the yearly stormwater inflow. Figure 17 shows the distribution of precipitation and stormwater runoff during the year. The majority of the inflow from these sources occurred in the spring months of April and May and during the late fall and winter.

Total outflow (including groundwater and evaporation) exceeded total inflow to the lake during most of the period from May through October (Figure 18). The decreased outflow in July was caused by closure of the outlet valve in mid June. Closing the outlet valve caused the lake volume to increase. That increase occurred in July as a net positive change in lake volume (Figure 19). Net water loss from the lake again occurred by the middle of July and continued to October (Figure 18). The major loss of water from the lake during the period was through groundwater flow to the Lewis River.

Distinguishing water loss due to flow through the outlet from that through groundwater was not possible. However, the rate of groundwater loss could be estimated during the period following outlet closure in June until its opening in mid December, with an accounting for losses due to evaporation. From July through October the groundwater loss averaged 15,265 m^3/day, a loss rate that was greater than the pumped Lewis River inflow, as shown by the comparison of inflow and outflow in Figure 18 and the negative change in lake volume in Figure 19. During the month of September when

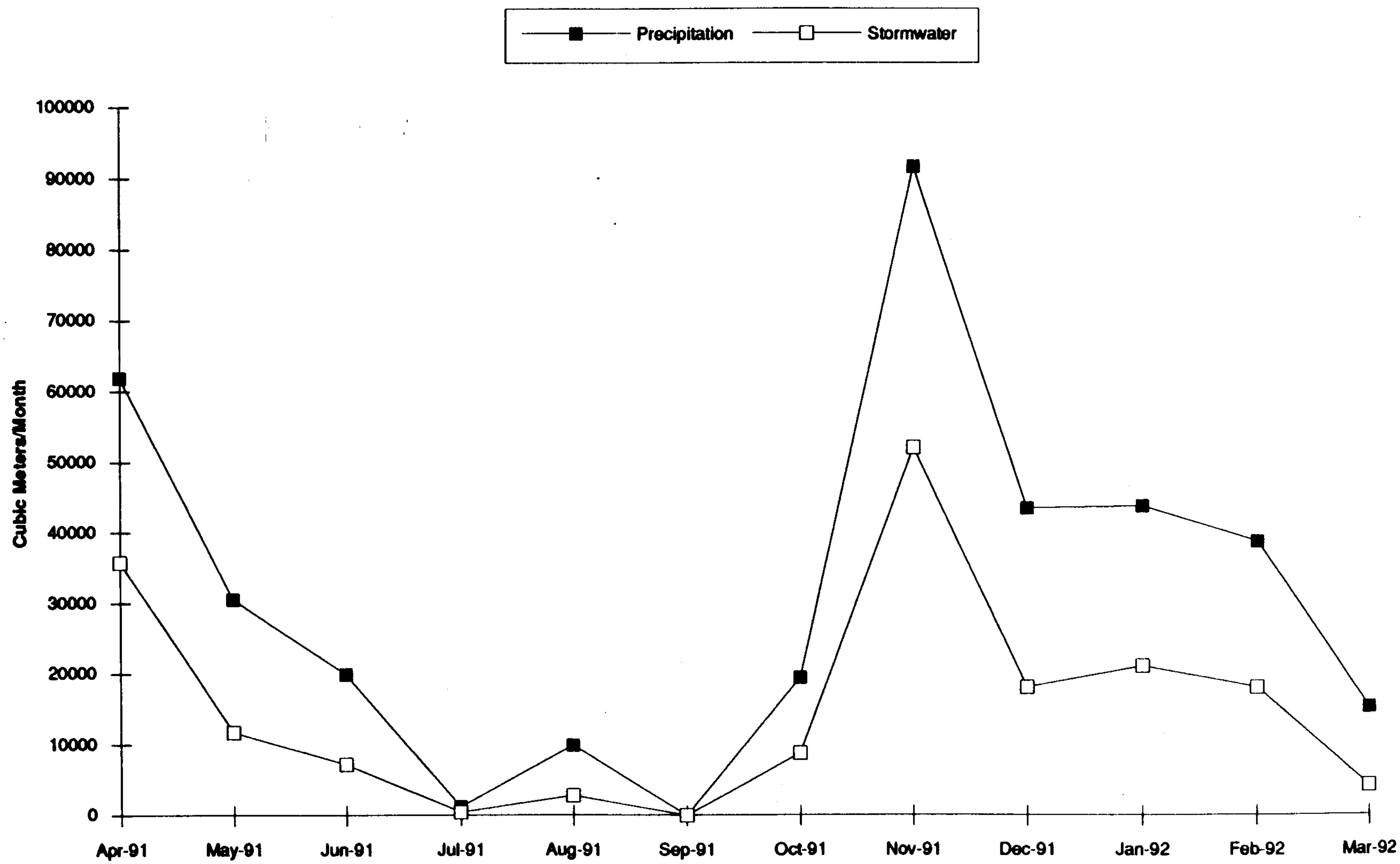


Figure 17. Distribution of precipitation and stormwater inflow to the lake during April, 1991 through March, 1992

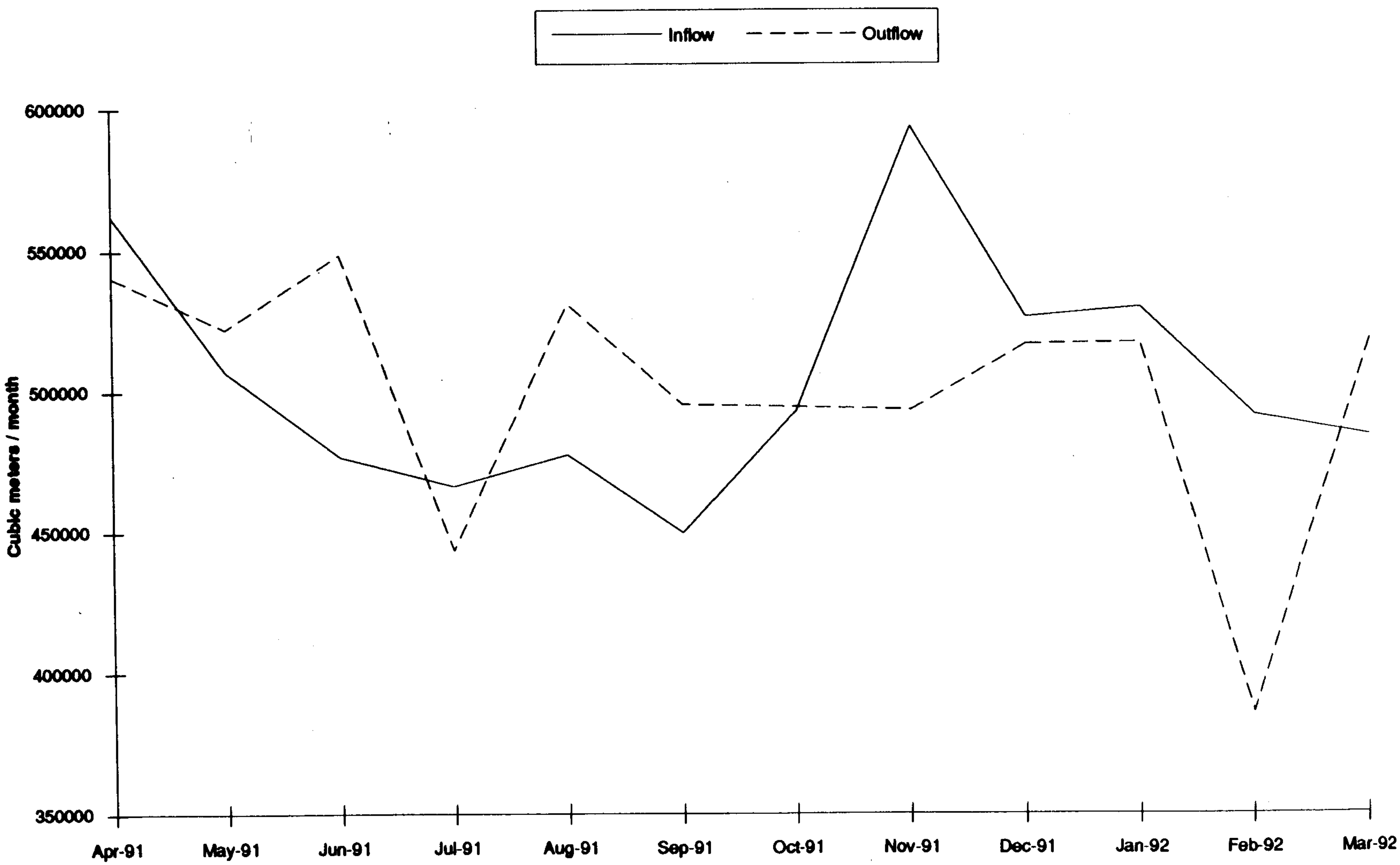


Figure 18. Total inflow and outflow (evaporation + groundwater) from Horseshoe Lake from April, 1991 to March, 1992

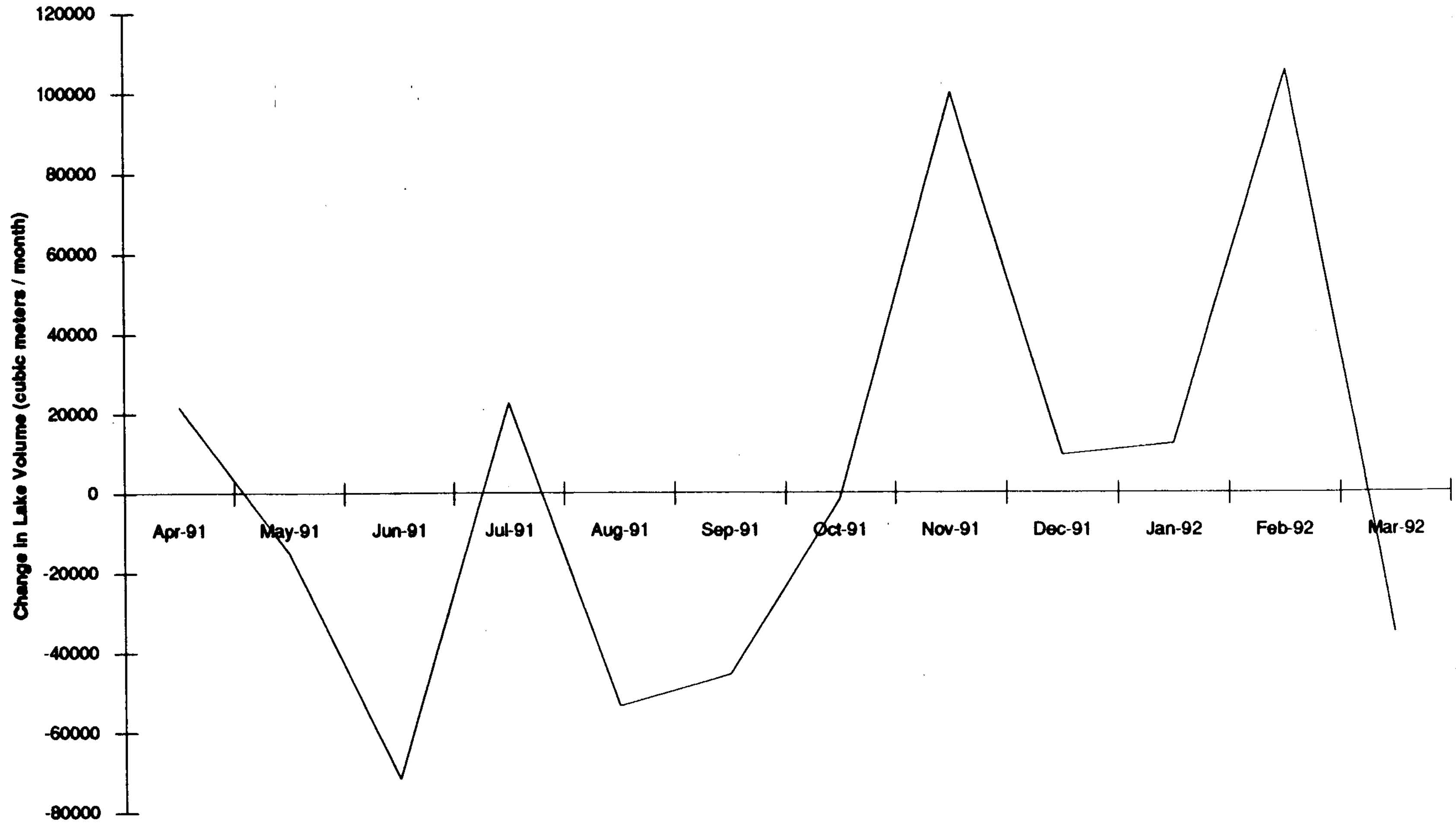


Figure 19. Net change in lake water during April, 1991 to March, 1992

there was no precipitation or stormwater, 15,911 m^3/day of water left the lake via groundwater (Figure 17).

Figure 20 shows a flow net that describes groundwater exfiltration from the lake to the Lewis River for September, 1991. This flow net displays a typical flow pattern for the hydraulic connection between Horseshoe Lake and the Lewis River. The areas of fill for the highways along the eastern edge of the two arms of the lake appear to provide a conduit for groundwater flow.

The inflow during the period from July through October (including stormwater and precipitation) averaged 15,722 m^3/day, while total outflow (including evaporation) averaged 16,368 m^3/day. This net loss resulted in a lake volume reduction of 19,390 m^3/month (Figure 19).

The groundwater elevations observed at wellpoints positioned adjacent to the lake depict the flow direction of local groundwater (Figure 21). Groundwater table elevations as determined from water levels from within the wells show a pattern of consistently lower hydraulic elevation eastward from the lake toward the Lewis River. With few exceptions, water table elevations from wellpoints (see locations in Figure 3) were lower than the lake (Figure 21). That is especially clear from water table elevations of wells 13 and 19 nearest the lake's east side, compared with that of paired wells 14 and 20 located farthest from the lake's eastside, and the lake elevation (Figure 22). The hydraulic gradient was from Horseshoe Lake to the Lewis River throughout the year. These data taken together indicate that the lake was acting as a recharge source to the groundwater with the majority of the recharge directed to the Lewis River. That behavior may be different from typical groundwater flow patterns in previous years when pumping was shut off during the winter months. In that case, lake level may have been low enough and river and water table elevation high enough for groundwater to enter the lake. But, during the 1991-1992 study period, that did not happen. Thus, groundwater was not a source of nutrients to the lake.

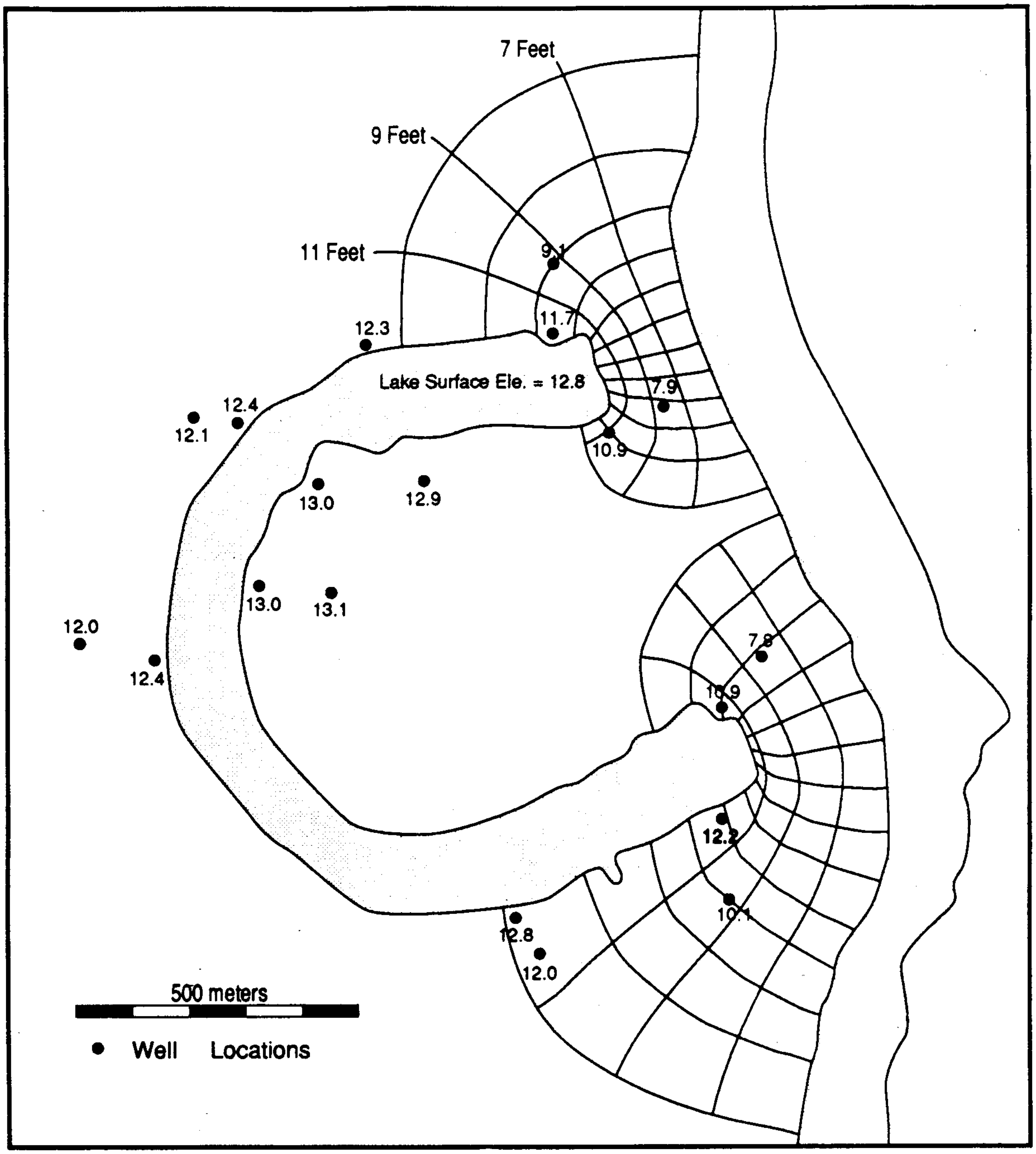


Figure 20. A flownet constructed for the water table surrounding the lake during September, 1991. Well elevations are shown in feet.

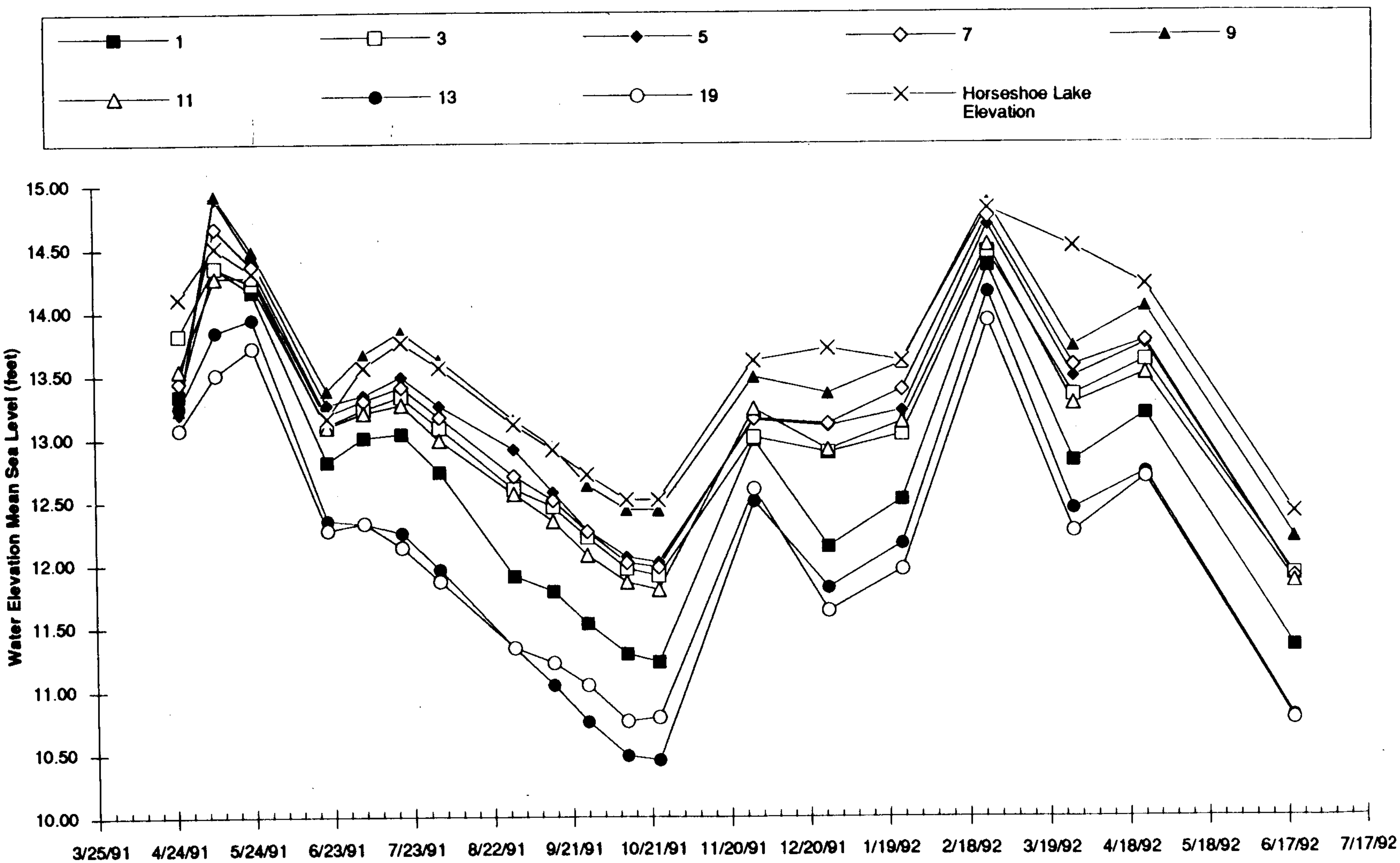


Figure 21. Water levels in well points surround Horseshoe Lake during April, 1991 to March, 1992. See Figure 3 for well point location.

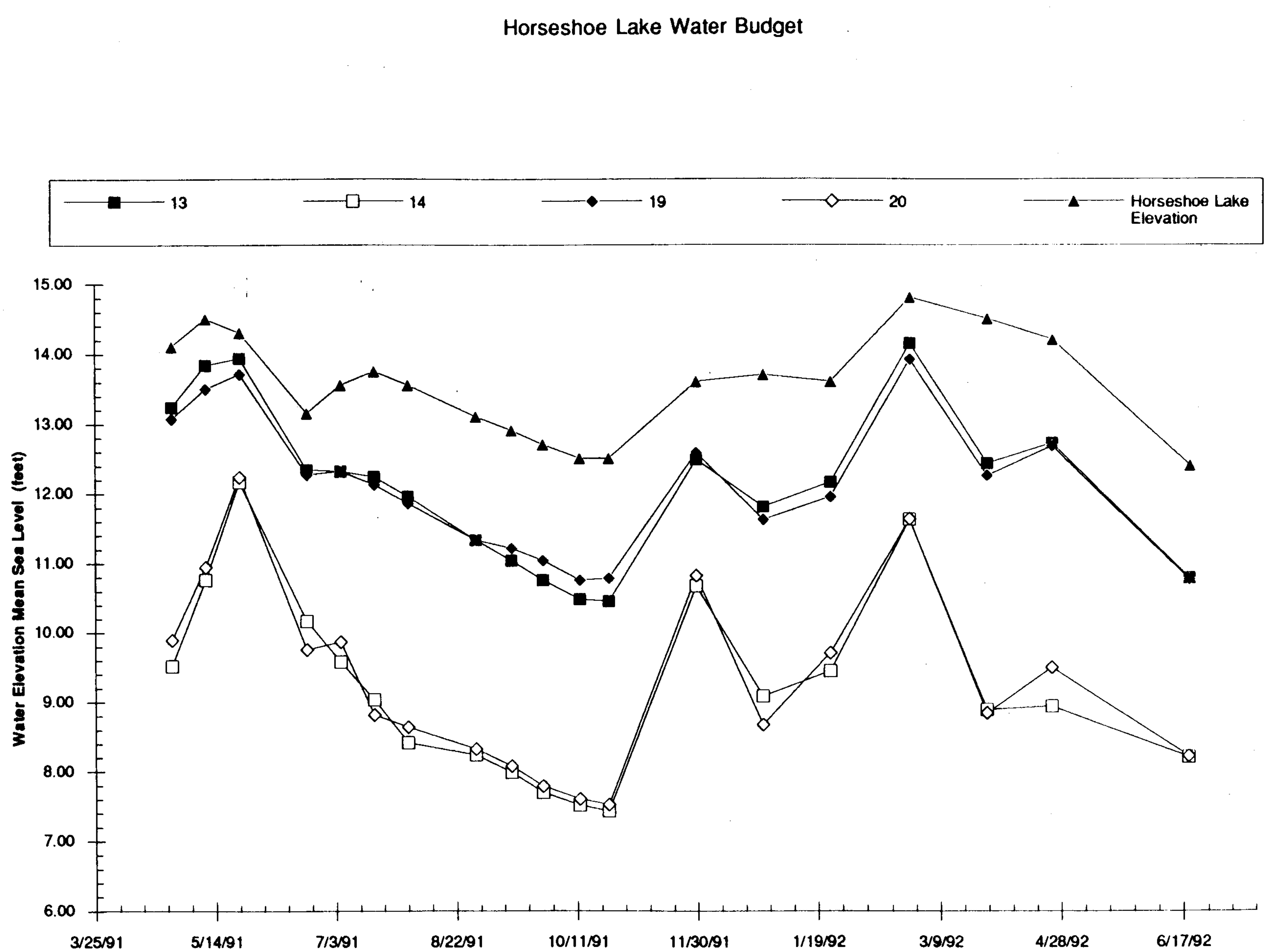


Figure 22. Water levels in well points lying directly east of Horseshoe Lake toward the river during April, 1991 to March, 1992. See Figure 3 for well point locations.

During the winter of 1989 (November-February), when the pump was not operating, the lake lost approximately 270,397 m^3 of water (2,253 m^3/day). The majority of that may be assumed to have been via groundwater (Somers, 1989). However, as lake elevation subsequently dropped during the winter months and if precipitation kept watertable elevations high, groundwater may have entered the lake.

The large gain in lake volume for the month of November (see Figures 18 and 19) came largely from the increase in precipitation and stormwater inputs (Figure 17). However, the decrease in groundwater loss and increase in lake volume for the month of February (Figure 19) cannot be explained from inflow sources. In spite of decreasing precipitation and stormwater inflows during February (Figure 17), the lake gained volume due to the decrease in groundwater discharge (Figure 19; Table 3). The decrease in outflow through groundwater can be explained only by changes occurring in the lake's greater watershed. Because the principal direction of groundwater discharge from Horseshoe Lake is toward the Lewis River, that hydraulic connection was explored further.

When the change in lake volume is compared with changes in the mean monthly flow of the Lewis River, reasons for the lowered groundwater discharge become evident. Large deviations in lake volume occurred during late June when the outlet was closed and in December when the outlet was opened. To lower this disturbance factor, values for the months of June and August were averaged for July's lake volume change and November and January values were averaged to obtain the change in lake volume for December. These corrected volume changes are displayed in Figure 23 and show that the lake volume change in February and reduced loss of groundwater was due to increased river flow. That connection is shown further by a regression of Horseshoe Lake volume change versus the Lewis River mean monthly flow (Figure 24). The increase in flow of the Lewis River accounts for the decrease in groundwater discharge from the lake. From the regression, an approximate Lewis River flow of 116 m^3/s (4,048 cfs) is needed to

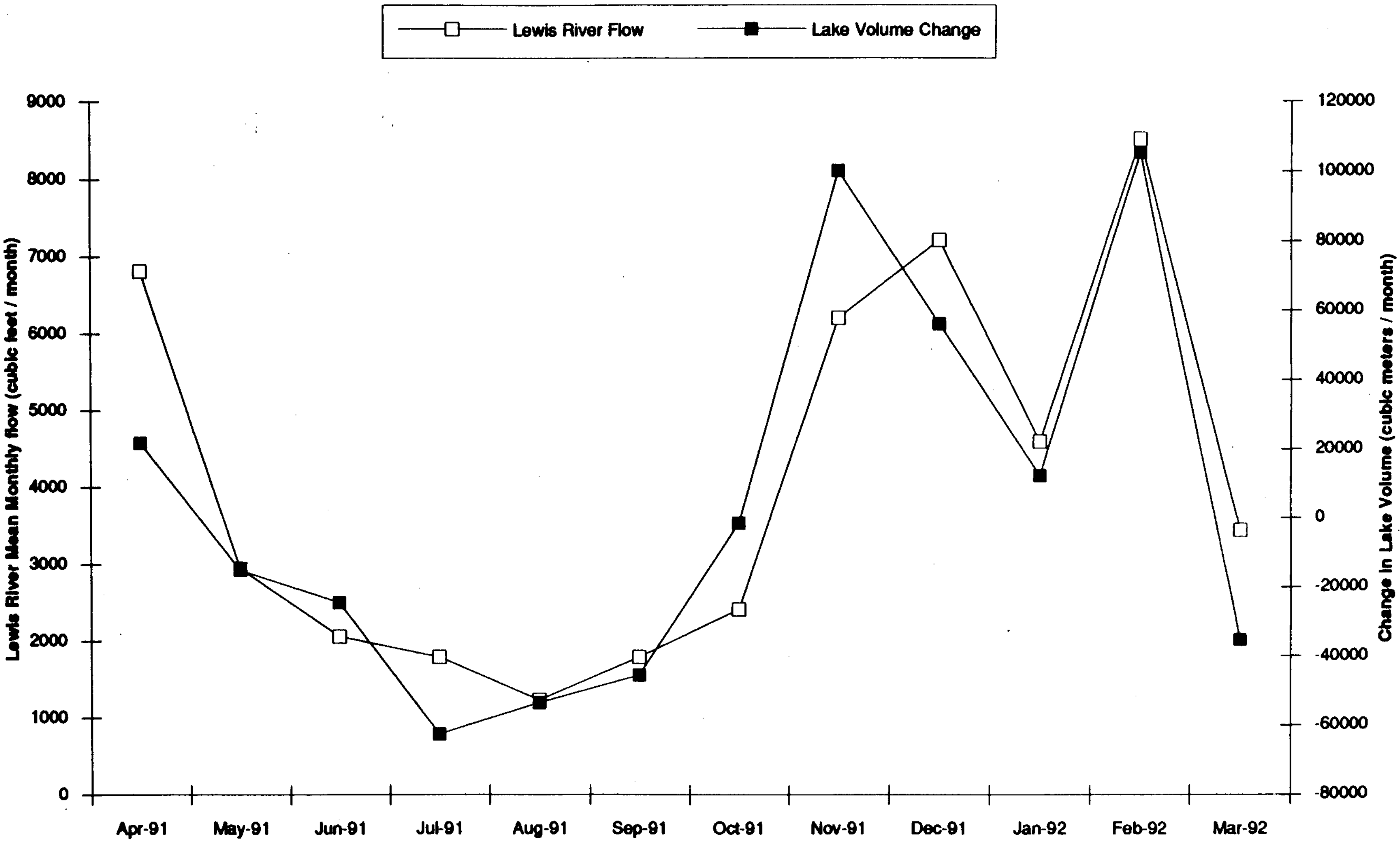


Figure 23. River flow and lake volume, corrected for July and December (see text), during April, 1991 to March, 1992

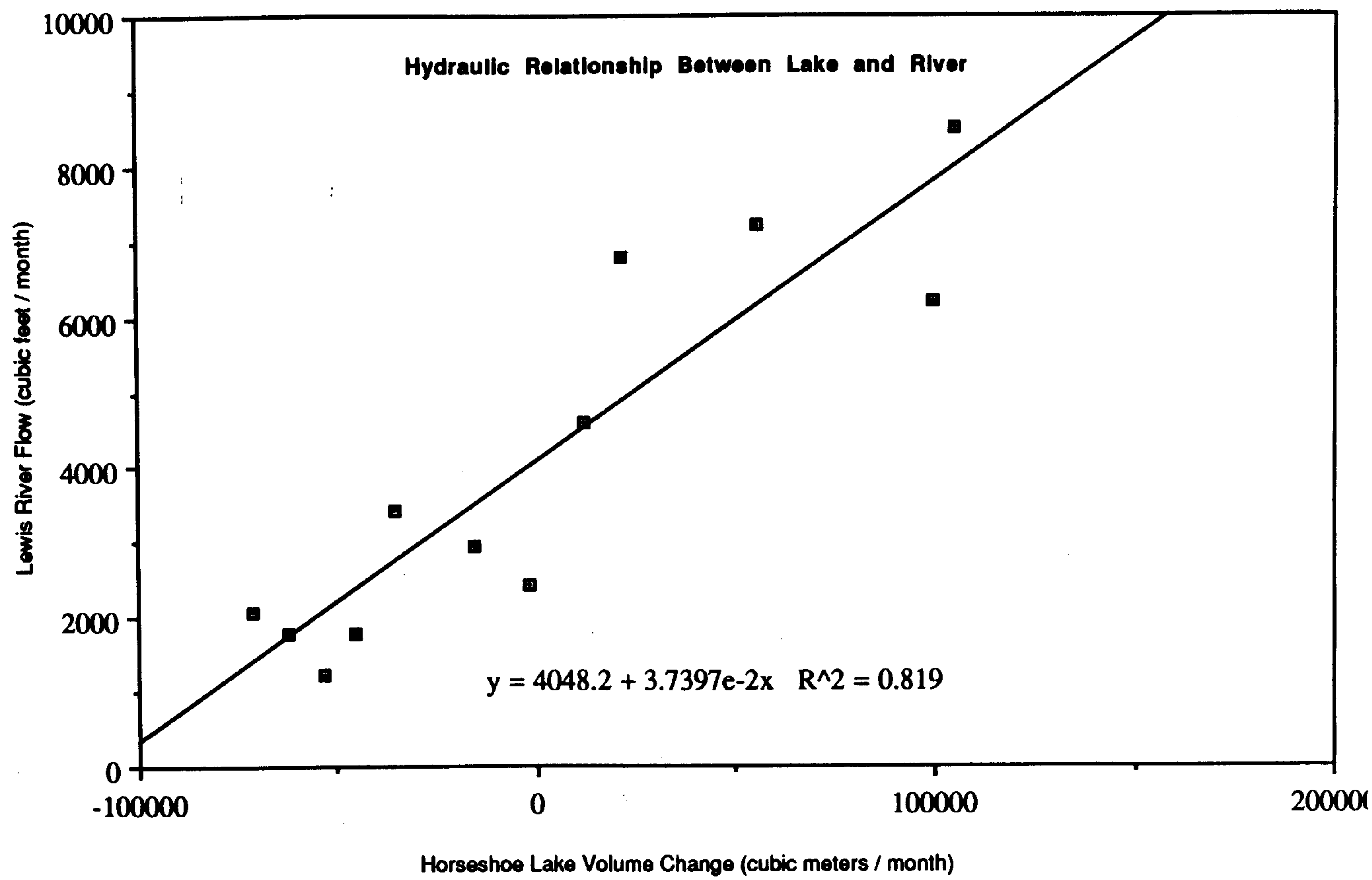


Figure 24. Relationship between monthly lake volume change and monthly average river flow.

reduce groundwater loss and maintain the lake volume at a steady state. River flows below this level create net water loss from the lake with the present pumping scheme.

Phosphorus Budget

The major inflows and outflow to the lake were analyzed for all forms of P and N. Loading analyses were performed for TP and SRP only. That is because P is the most important nutrient driving productivity in this lake. From flows determined in the water budget and the concentrations of TP, a budget for P was constructed. Such a P budget is an accounting of the movement of P mass into, within, and out of the lake system. Although much of these data exist on a twice-monthly basis, the data were grouped on a monthly basis to construct the annual P budget based on the following mass balance:

$$\Delta P = \text{Lex} - \text{Lout} \pm (-\text{Sedimentation} + \text{Internal Loading}).$$

where ΔP = change in lake TP mass

Lex = TP mass from external sources

Lout = TP mass lost through outflow

Sedimentation = TP mass lost due to settling to the bottom sediments

Internal = input of P due to release from the bottom sediments

The sedimentation and internal loading terms are both unknowns and only the net effect of the two processes can be calculated. For that calculation the equation is solved for each period by calculating the total residual in brackets. If the residual is negative, the quantity is attributed to net sedimentation. If positive, it is attributed to net internal loading. Estimates of gross sedimentation and gross internal loading will be made separately later by calibrating a non-steady state model, but at this point only net exchange rates across the sediment-water interface will be determined.

The mass nutrient loading was determined by multiplying the determined nutrient concentration (mass/volume) by the input volume of water. An average TP concentration of 270 μg/L was used to estimate loading from stormwater. The TP concentration for both fall and spring composites of precipitation were both 4.7 μg/L, which was multiplied by the depth of precipitation. A volume weighted lake TP concentration was multiplied by outflow rates and grouped whether via the surface or through groundwater exfiltration, to give the mass of TP leaving the lake system.

External Loading: Of the principal external loading sources, the Lewis River, stormwater, and precipitation, the Lewis River provided the greatest mass of phosphorus input to the lake. During the study year, 96.2 kg of TP were loaded to the lake from this source (Table 4). Stormwater, due to its high TP concentration, contributed 53.5 kg for the year. Stormwater runoff contributed only about 3% of the water input to the lake, but 35% of the annual TP load. The Lewis River inflow provided 91% of the water input and 63% of the yearly TP loading; the reverse of stormwater, because TP concentration was much lower in the river than in stormwater. Precipitation provided only 1.8 kg of TP to the lake, representing approximately 2% of the annual load. The total annual external loading was 151.5 kg or on an areal basis, 437 $mg/m^2 \cdot y$.

Figure 25 shows the trends in sources of external loading of TP for the year. Stormwater loading was high during the spring, late fall and winter months. During the months of April and November 10.6 kg and 15.4 kg of TP, respectively, entered the lake from stormwater. The combined TP mass from these two months comprised 49% of the yearly total from stormwater. Increased precipitation levels in April and May resulted in increased loading from both stormwater and the Lewis River. During the summer and early fall the Lewis River provided the major source of external loading. The increase in rainfall during November brought a dramatic increase in stormwater loading to the lake. The November rainfall alone resulted in an external loading peak of 19.4 kg TP.

Horseshoe Lake Nutrient Budget

	Lewis River	Stormwater	Rainwater	Total External Loading	Outflow TP	Delta Lake TP	Sedimentation	Internal Loading
	Kg/mo.	Kg/mo.	Kg/mo.	Kg/mo.	Kg/mo.	Kg/mo.	Kg/mo.	Kg/mo.
Apr-91	16.3	10.6	0.3	27.3	15.5	-13.6	-25.5	0.0
May-91	7.3	3.5	0.1	10.9	10.2	3.8	-2.3	5.4
Jun-91	8.8	2.1	0.1	11.1	13.5	15.9	-0.7	19.1
Jul-91	11.7	0.1	0.0	11.8	13.3	-7.0	-8.1	2.6
Aug-91	13.3	0.8	0.0	14.2	15.6	-1.5	-3.7	3.6
Sep-91	6.2	0.0	0.0	6.2	14.1	-0.3	-1.2	8.7
Oct-91	3.6	2.6	0.1	6.3	14.6	-3.2	-1.5	6.6
Nov-91	3.6	15.4	0.4	19.4	13.5	-8.3	-14.3	0.0
Dec-91	5.7	5.3	0.2	11.2	9.9	-6.8	-8.2	0.0
Jan-92	7.0	6.2	0.2	13.5	7.1	1.7	-5.6	0.8
Feb-92	3.9	5.4	0.2	9.4	5.6	4.5	-2.9	3.7
Mar-92	8.6	1.2	0.1	9.9	10.0	6.9	-2.8	9.7
Yearly Total Kg/yr	**96.2**	**53.5**	**1.8**	**151.4**	**142.9**	**-8.0**	**-76.8**	**60.2**

Table 4. Annual budget for TP during April 1991 to March 1992

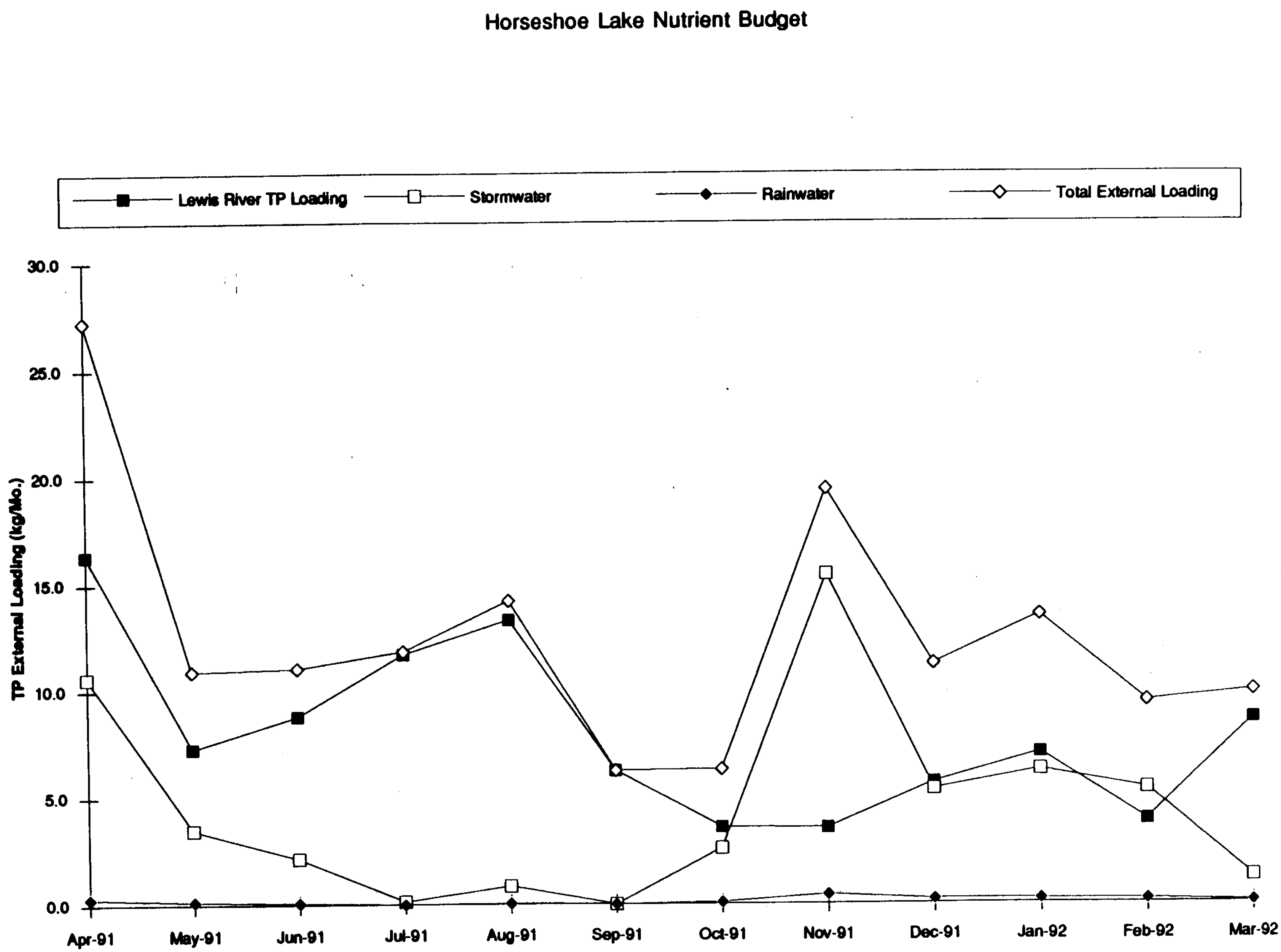


Figure 25. Sources of external TP loading to the lake during April, 1991 to March, 1992

External loading reached its lowest level during the months of September and October when 6.2 and 6.3 kg TP entered, respectively. During the critical summer months, June through September, external loading from the Lewis River comprised 40 kg or 92% of the external loading.

Internal Loading: From May through October external loading of TP was less than that lost from the lake via the outflow (Figure 26). This period coincides with the time of maximum algal growth. Thus, external loading of TP apparently provided only a part of the P needed for algal production during that period. Also, the lake was obviously yielding P. The balance of the P needed to sustain algal production originated via internal loading. Internal loading is the release of phosphorus, primarily in a soluble form (SRP), to the water column from the bottom sediments. Internal loading can result from several processes, which cannot be distinguished by the mass balance. All that is known is that the P must have originated internally, assuming there is no significant external source omitted.

Internal loading contributed the majority of SRP to Horseshoe Lake during the summer months, June through September, when there was a net release of 34 kg to the lake (Figure 27). External sources accounted for approximately 21 kg during the same period. The maximum net internal loading peaked in June when 19.1 kg was contributed to the water column (Table 4). This release was coincident with the algal bloom.

As with TP loading, the Lewis River contributed the majority of external SRP loading during the summer months, comprising 98% of the total summer external SRP loading (21 kg). Stormwater contributed only 0.5 kg SRP during this period.

However, a further examination of the relation between the external and net internal sources of SRP and how they affected the change in SRP mass in the lake is important in distinguishing the primary contributor for the algal blooms. During the periods of high algal biomass, May through September, SRP input to the lake from

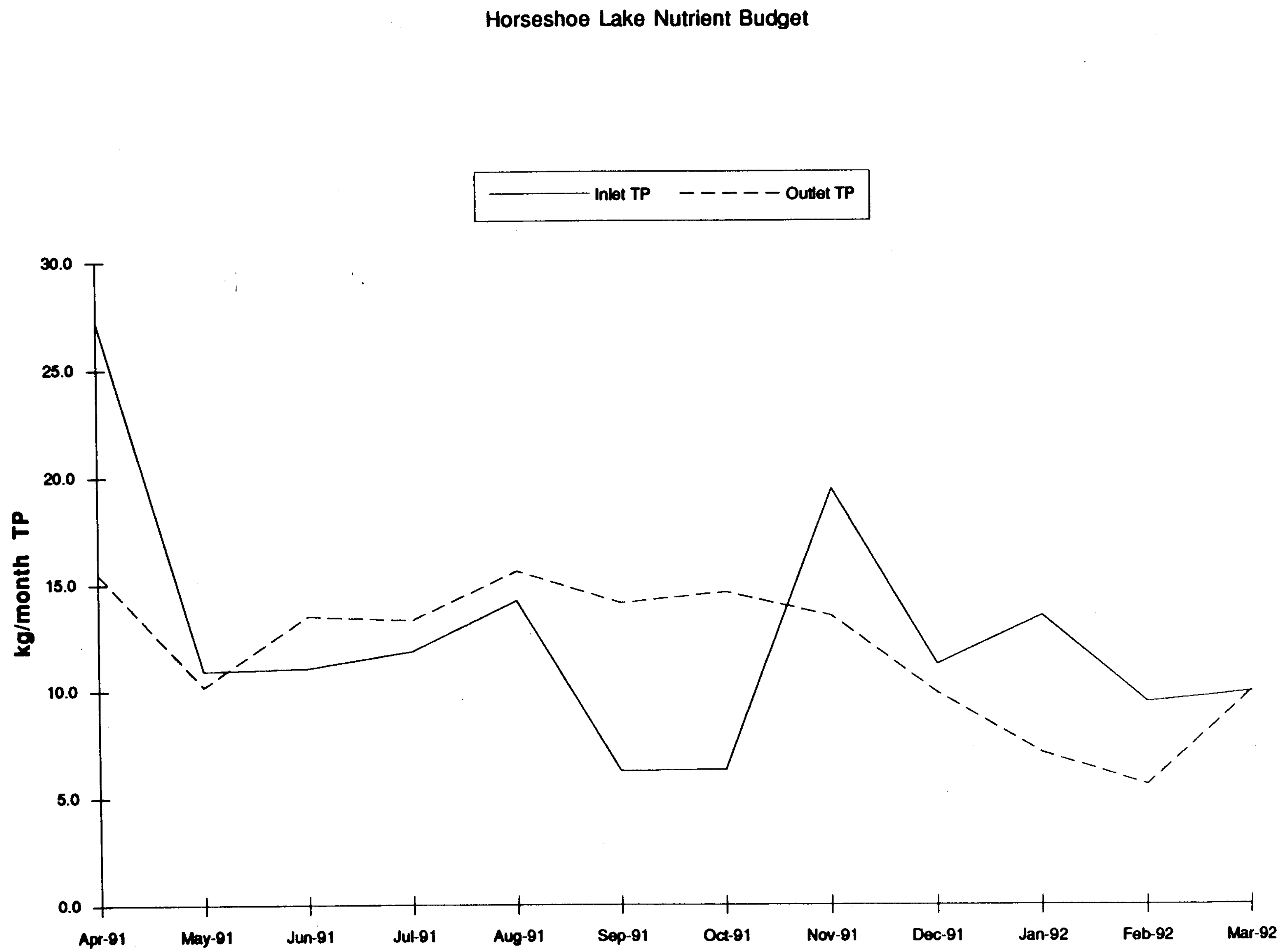


Figure 26. TP external loading compared to loss of TP through the outlet during April, 1991 through March, 1992

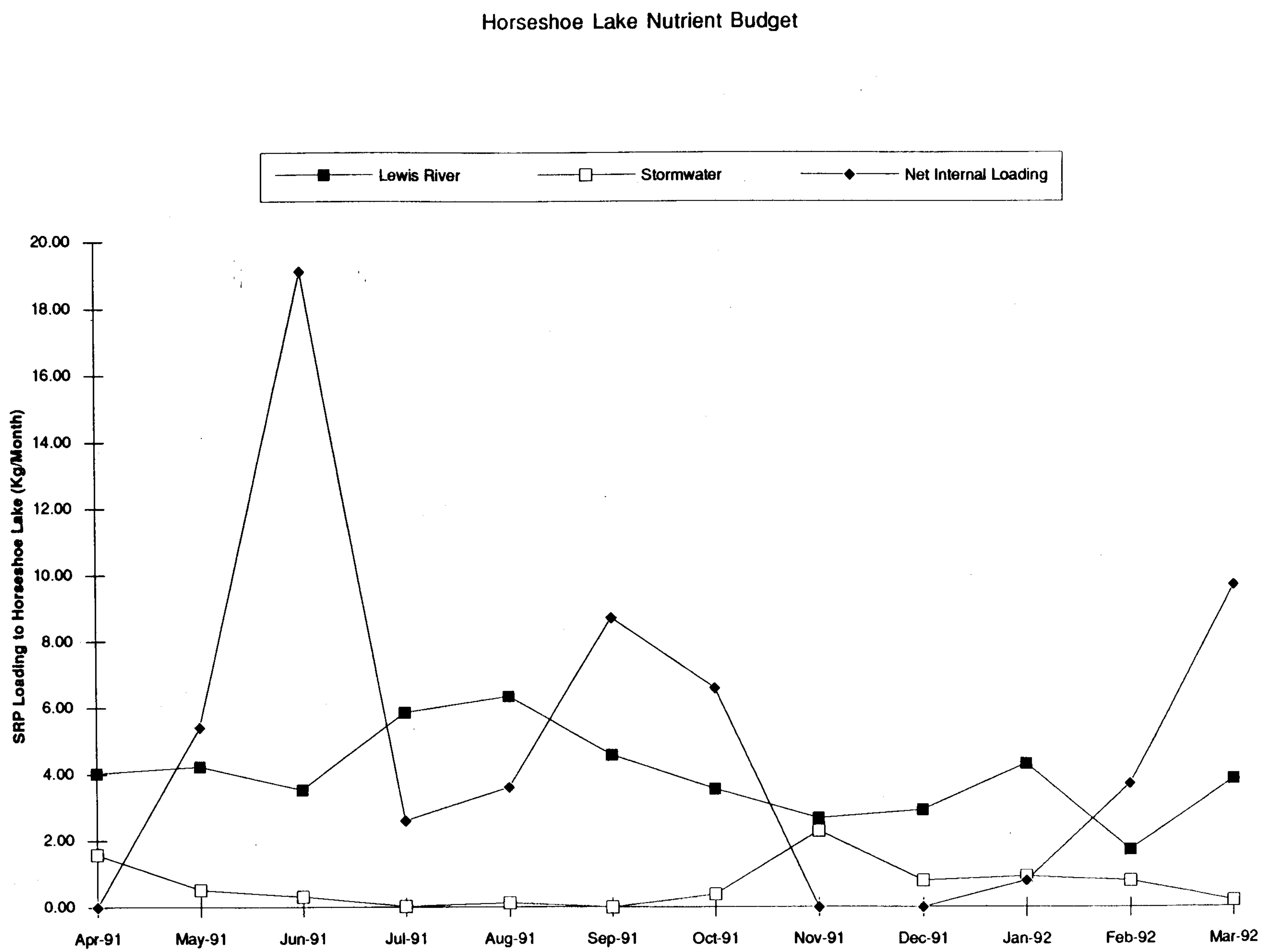


Figure 2 7. Comparison of external and internal sources of SRP during April, 1991 through March, 1992 internal loading had the greatest effect on the SRP mass in the lake (Figure 28). Because SRP is the form that is readily assimilated by algae, internal loading was apparently more important than SRP in the entering Lewis River water even though that inflow concentration averaged 10 μg/L. Thus, controls for algal blooms and lake quality should be aimed at reducing or mitigating internal loading.

Phosphorus Model

Formulation, Assumptions, and Calibration: A predictive capability is needed to determine the effects of proposed restoration to improve the quality of Horseshoe Lake. A dynamic, or non-steady state, mass balance P model was calibrated for that purpose. The model is formulated as follows:

$$\frac{dTP}{dt} = \frac{L_{ex}}{\bar{z}} - \rho TP - \sigma TP + \frac{L_{int}}{\bar{z}}$$

Where TP = lake TP concentration in mg/m^3(= μg/L), L_{ex} is external loading of TP (mg/m^2-day), ρ is flushing rate or lake inflow rate/lake volume, σ is the sedimentation rate coefficient, L_{int} is internal loading of TP (sediment release), and $\bar{z}$ is mean depth. To apply the model from year to year, σ was replaced with ρ^x. The known values are lake TP, L_{ex}, $\bar{z}$, and ρ. L_{int} and σ must be estimated. The important assumptions for the model are 1) constant inflow and outflow, 2) constant volume and 3) completely mixed.

To estimate the sedimentation rate coefficient, σ, the model was calibrated to lake TP during the winter when there was no appreciable L_{int} occurring. The ρ^x producing the best fit for predicted TP with observed TP was then calibrated to the whole-year TP data to obtain an estimate of L_{int} during the summer-fall period. L_{int} was adjusted to obtain a best fit of predicted with observed TP. The results of the calibration exercise are shown in Figure 29 and actual values are given in Appendix F. The squared correlation

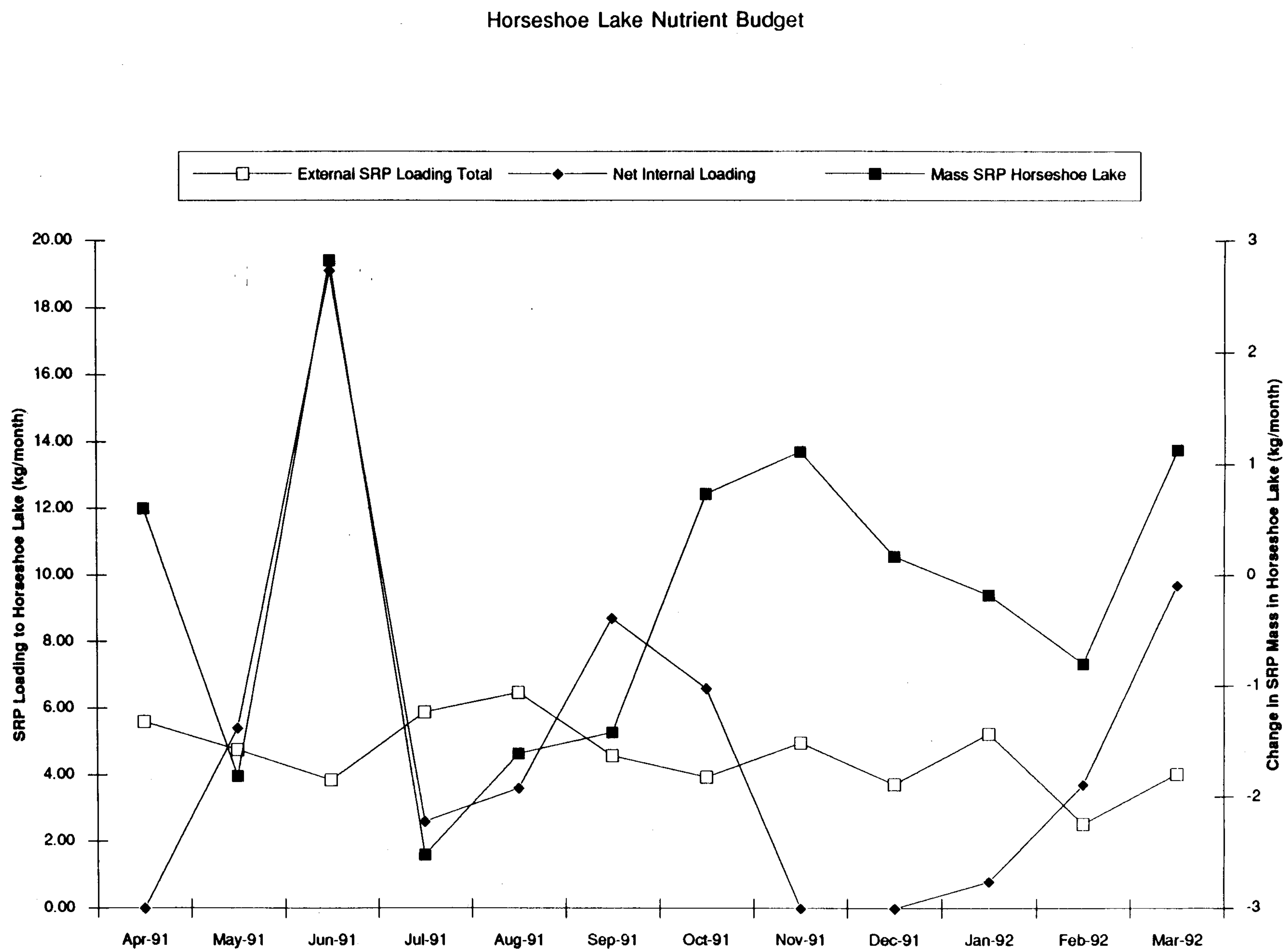


Figure 28 External and net internal SRP loading compared to SRP change in the lake during April, 1991 through March 1992

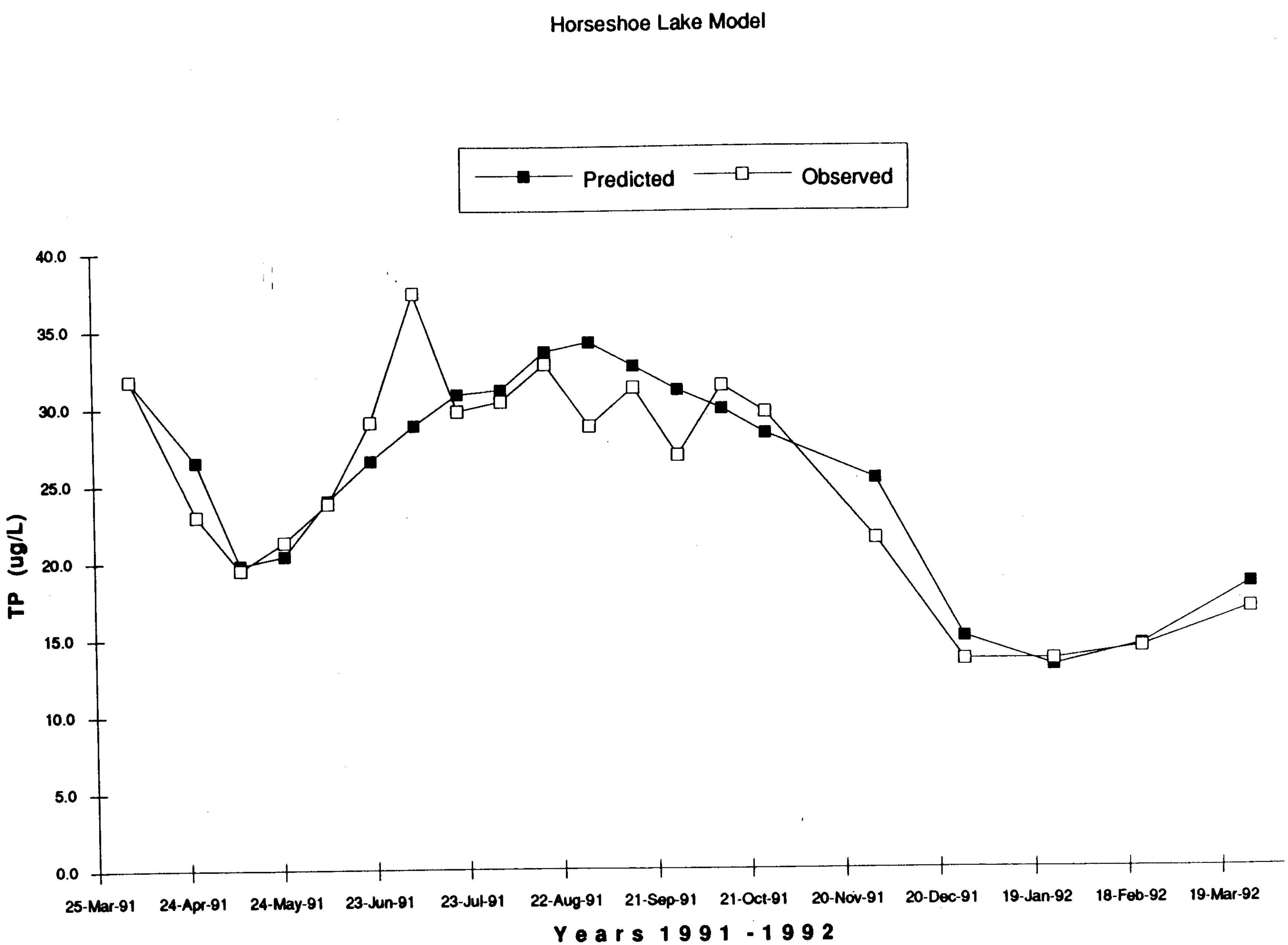


Figure 29. Whole lake, volume weighted TP concentration compared with predicted TP from a calibrated non-steady state model.

coefficient (r^2) for the fit is 0.83 (ie. 83% of the variance in the lake TP can be accounted for by the model).

To simplify the calibration, a constant value for the exponent to flushing rate ($X = 0.85$) was used for the sedimentation rate coefficient, as well as a constant L_{int} during the summer-fall of 2.44 mg/m^2-day. Such constant values are not expected to occur either in magnitude or in distribution with time, but the purpose is to approximate the level of TP that may occur in response to different management alternatives. Although the model calibration resulted in a good fit of the data for 1991-1992 (r^2 = 0.83), there is no way of knowing how well it would predict lake TP in other years without an additional year's data for verification. The exchange of P between sediment and overlying water is affected by wind, temperature, etc., and thus L_{int} would be expected to vary in magnitude and timing from year to year. Nevertheless, the model can be used to estimate potential effects of treatments on lake quality, recognizing that the estimates will only approximate the actual conditions.

Results of Manipulations: For example, the summer mean TP of 30 µg/L in 1991 is predicted to decrease if the river inflow rate were increased. An increase in flow of 50% would be expected to lower lake TP to an average of about 25 µg/L (Figure 30). This trend to decrease the lake TP concentration with increased river inflow results from the much lower concentration in river water than in the lake (see Figure 7). Thus, the river water dilutes the lake water TP, counteracting the enrichment by internal P loading. Lake TP can be more effectively lowered if internal P loading is reduced by, for example, inactivating sediment P (Figure 31).

The sensitivity of the lake to a reduction in stormwater input is illustrated in Figure 32. A 100% reduction in stormwater P loading would reduce the summer lake TP concentration by only about 1 µg/L. Thus, the summer lake TP concentration is rather insensitive to the direct effect of stormwater inputs. That is understandable, because stormwater inputs occur largely during the winter and by the time summer comes,

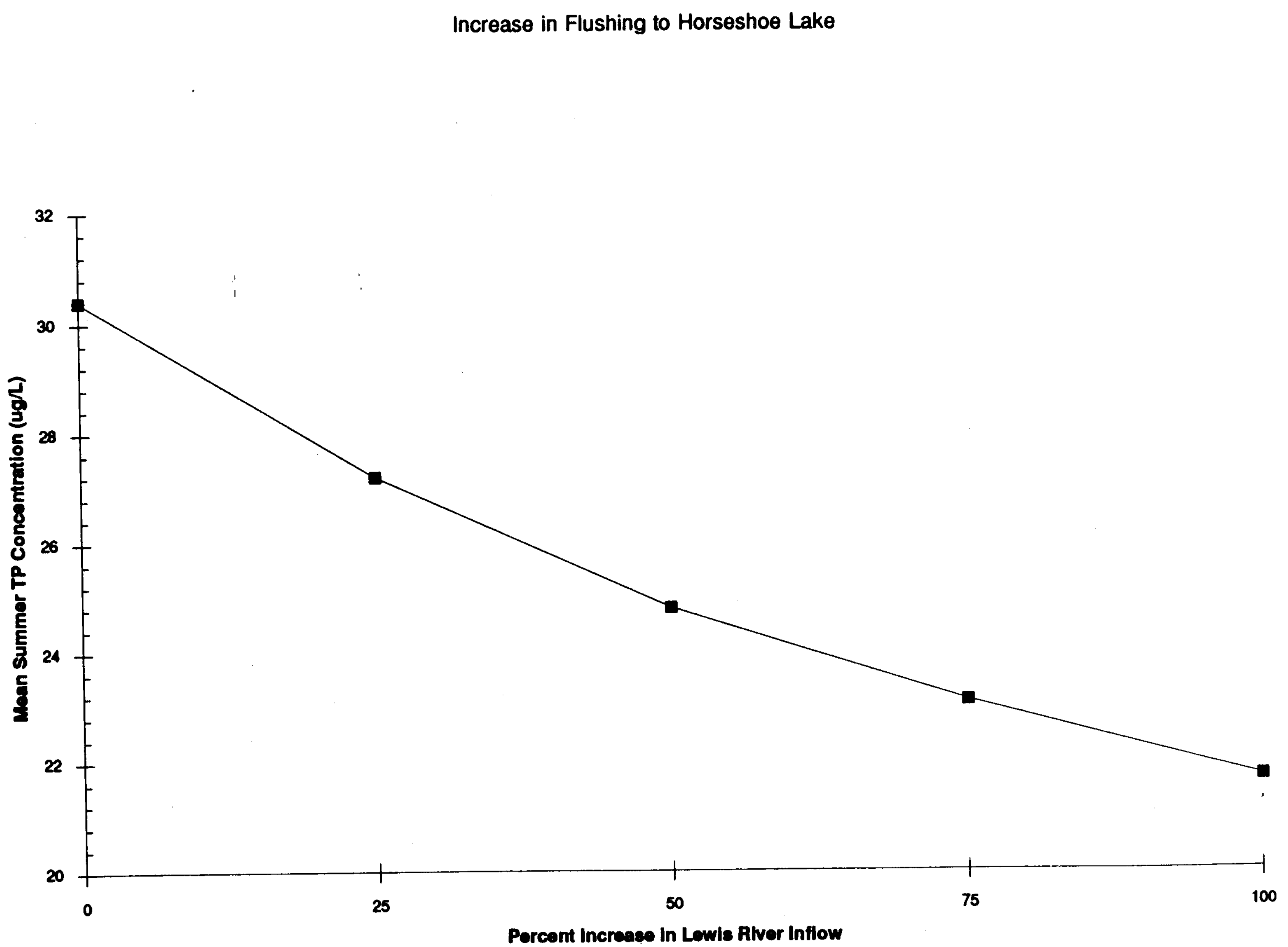


Figure 30. Predicted summer lake TP concentration related to percent increase in pumped river inflow.

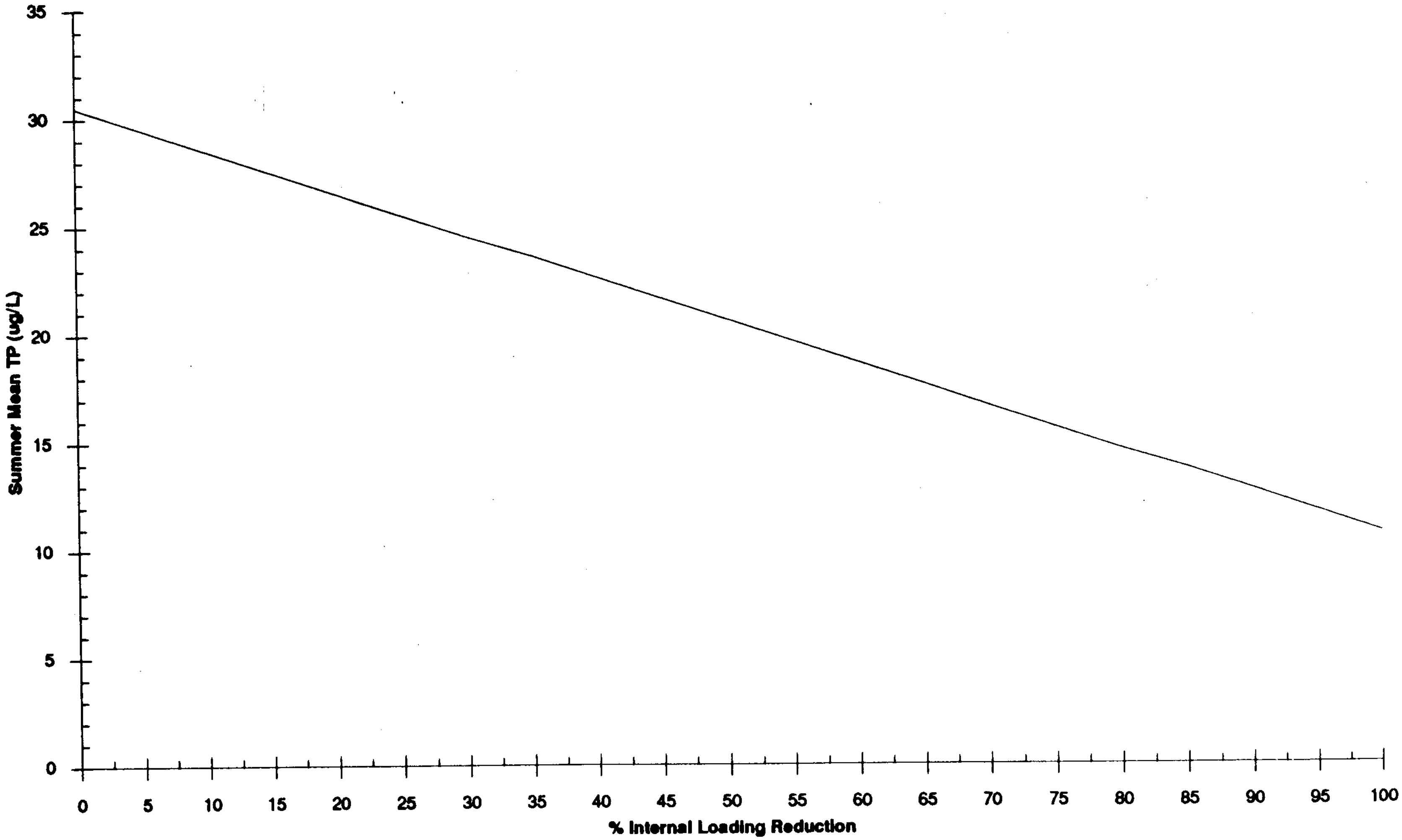


Figure 31 . Predicted summer TP concentration related to percent reduction in summer internal P loading.

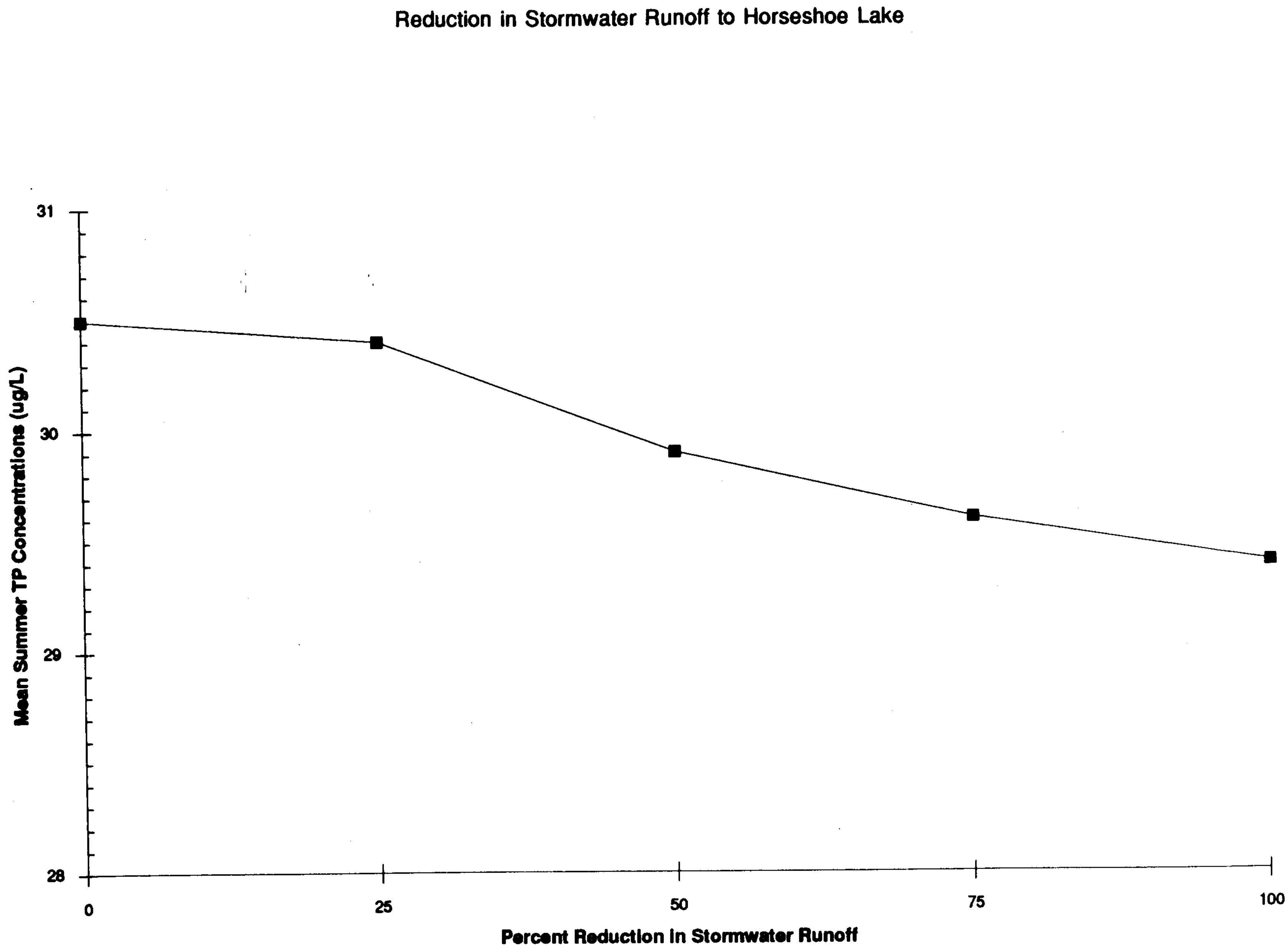


Figure 32. Predicted summer TP concentration related to decreased stormwater input

stormwater TP has either left via the lake outflow or sedimented to the bottom. On the other hand, stormwater P enriches the bottom sediment contributing in some unknown amount to internal P loading.

MANAGEMENT ALTERNATIVES

This section is restricted to a discussion and evaluation of the treatment alternatives that are pertinent to the control of blooms of plankton algae and related quality characteristics.

Watershed Source Controls

Stormwater contains high concentrations of nutrients and represents a significant nutrient load, relative to the river (see Figure 25). However, as pointed out earlier, stormwater enters during the winter and early spring, but inflows during the dry summer are usually insignificant. As a result, stormwater P is usually no longer present in the lake in summer when the algae bloom and poor water quality exist. Figure 32 shows that reduction of stormwater inputs through diversion or treatment would have only a slight direct effect on summer TP concentration. Thus, stormwater treatment or diversion cannot be defended as a cost-effective measure to improve summer lake quality in the short term.

Stormwater may be contributing indirectly to internal loading through enrichment of bottom sediments. That is evident from the high TP content in stormwater (mean 270 µg/L). Results from other lakes has shown that reduction in external P loading will eventually lead to reduced internal loading, but the time delay for such recovery is uncertain (Welch et al., 1986; Sas et al., 1989).

Dilution/Flushing

Lake quality appears to be rather sensitive to water inflow rate (Figure 30). As pointed out earlier, that sensitivity is due to the lower TP concentration in river inflow than the lake concentrations, which during summer results largely from internal P loading. The maximum effect of increased addition of river inflow would be to bring the summer (June-Sept.) lake TP concentration (30 μg/L) down to near the average concentration.

The question is, how much should inflow be increased beyond the current 3,000 gpm to achieve what lake TP concentration and lake quality? As shown in Figure 30, a 50% increase in river infow rate should bring lake concentration to about 25 μg/L TP. That should lower summer average chl *a* from 15 μg/L to about 11 μg/L (corrected equation from Jones and Bachmann, 1976) and summer mean transparency from 1.2 m to about 1.5 (equation from Carlson, 1977). So for a 17% reduction in TP, transparency should increase by 25%.

As already mentioned, the TP model is calibrated for one year and year-to-year variation in climatic conditions affect mixing conditions in shallow lakes. Varied mixing has been shown in Moses Lake to cause ± 100% variation in internal P loading. Therefore, a goal of 25 μg/L TP for Horseshoe Lake may be reasonable to prevent significant blooms of blue green algae and their persistence, but an increase of 50% in river inflow rate suggested by the model to achieve that level may be too much some years and not enough other years.

The above description of the effect of increased river inflow is one of dilution, i.e. reduce lake TP concentration by adding water of lower TP concentration. However, there is another effect of pumped river inflow to Horseshoe Lake that was alluded to in the discussion of temperature and mixing. The denser river water (higher conductance and lower temperature) apparently sinks through the less dense lake water and exchanges the bottom water. Without that inflow, the deep section of the lake would probably become more thermally stratified and the bottom water more depleted of oxygen. There was a tendency in that direction in July, but was subsequently prevented, apparently by increased mixing. Therefore, an increased input rate of denser river water would apparently reduce the residence time of bottom water, thereby preventing DO depletion and P regeneration.

There may be short circuiting of inflow water, because much of the outflow leaves via groundwater through the north end of the lake. To increase the effectiveness of dilution water, the inflow pipe could be extended to depth and farther into the lake.

P Inactivation

This technique involves the use of alum (aluminum sulfate) to reduce internal P loading. Results from investigations of six shallow Washington lakes indicate that alum treatments are 50-80% effective at reducing internal P loading and that effect should last at least 5 years and maybe up to 10 years, so long as macrophytes are not present.

Horseshoe Lake is largely devoid of macrophytes in spite of its shallowness. Its coarse, relatively compact sediments (see low water content, Table 2) may discourage the development of macrophytes. Although transparency is not high (mean 1.2 m), light should still be sufficient for colonization at depths less than about 2.0 m, which represents much of the lake area. Therefore, the sediments are probably unsuitable for colonization or macrophytes would be present.

The effect of sediment P inactivation is illustrated in Figure 21 with predictions of lake TP resulting from various levels of reduction in internal P loading. The model results indicate that an alum treatment should be highly effective at reducing the lake TP content during summer assuming a 50-80% effectiveness. At those levels of effectiveness, predicted summer lake TP would be about 20-15 µg/L. Chl *a* concentration and transparency would be expected to be about 8.3-5.4 µg/L and 1.8-2.4 m.

Artificial Circulation

This technique is used to keep lakes completely destratified. The benefits are increased contact of lake water with the atmosphere, eliminating waters with low DO and the problems resulting from such anoxia. Further, the increased mixing can reduce the concentration of algae by reducing the amount of light available to mixing algal cells and decrease the amount of blue-green algae by neutralizing the advantage afforded by their buoyancy.

The technique is not recommended for Horseshoe Lake, because DO is not low enough to cause problems. Also, the lake is not deep enough for complete mixing to reduce total algae by light limitation. Whether reduced amounts of blue-green algae would result from complete circulation is uncertain. There is some evidence that the occurrence of the algal bloom and its subsequent crash during July, 1991 was associated with increased stratification and subsequent mixing, respectively (see discussion on phytoplankton). Although maintaining complete mixing could deter blue-green blooms, there is simply not enough evidence on a practical scale to confidently recommend the measure, especially since other, more certain techniques exist.

Hypolimnetic Aeration

This technique is designed to increase DO and reduce the problems associated with anoxia in the hypolimnion of stratified lakes. Because Horseshoe Lake does not permanently stratify, there is no reason to pursue the application of this technique further.

Hypolimnetic Withdrawal

The lack of permanent stratification also applies here making this technique impractical.

Dredging

Sediments are removed to various depths in lakes, depending on sediment nutrient content, in order to reduce internal loading of nutrients. As described in the sediment section, the P content of Horseshoe Lake sediment does not increase toward the more recent sediments. Moreover, the content is not high (0.9 mg/g) relative to sediment in other lakes. Therefore, sediment removal to some cost-practical depth would probably not be effective in reducing P internal loading and is not recommended for that purpose.

However, another purpose for dredging is lake deepening. The south end of the lake is very shallow, which resulted largely from flood waters entering at that end in 1964, depositing large amounts of sediment. Deepening all or part of the south end would improve the recreational potential, especially during the late summer when lake level declines (~ 1 m in 1991) due largely to the dropping level of the river. Sediment removal to provide at least a meter of water throughout the south end during low lake level may be a desirable goal in the future.

The effect of such deepening on water quality, once complete, may be minor. Such a depth increase is relatively small compared to what would be necessary to begin to limit algal abundance by mixing processes described earlier. However, a greater depth should have a diluting effect on sediment-released P. Also, as depth increases, the effect of wind at entraining surficial sediments into the water column should diminish, resulting in greater sediment retention of P and, hence, clearer water with less algae. There are too few data available to predict how much improvement would result, but a significant beneficial effect on water quality, in terms of algae, cannot be expected with a depth increase of only 1 m.

Treatment Cost Effectiveness

Of the restoration alternatives considered, dilution and phosphorus inactivation through alum treatment appears the most feasible for Horseshoe Lake. Table 5 includes a cost analysis of the several remediation scenerios based on these methods.

The most cost effective treatment would apparently be alternative 3, alum treatment to inactivate sediment P and reduce internal loading by an estimated 50-80%, together with a 50% increase in pumping rate. There is not much advantage in increasing pumping rate above 50% increase (4,500 gpm). That provides a drop of

Table 5. Comparison of treatment cost effectiveness.

Alternatives	Description	Cost*	Summer TP µg/L
Current State			30
1	Alum with Sodium Aluminate or Soda Ash	$138,000	15-20
2A	Pumping: x 1.5 (4500 gpm)	80,000	25
2B	Pumping: x 1.75 (5250 gpm)	84,000	23.5
2C	Pumping: x 2.0 (6000 gpm)	98,000	22.0
3	Alum plus pumping: x 1.5	218,000	13-18
4A	Pumping: x 1.5, with diffuser pipe	570,000	25**
4B	Pumping: x 1.75, with diffuser pipe	606,000	23.5
4C	Pumping: x 2.0, with diffuser pipe	614,000	22.0
5A	Pumping: x 1.5, diffuser pipe plus alum injection	956,000	10-15***
5B	Pumping: x 1.75, diffuser pipe plus alum injection	1,018,000	10-15
5C	Pumping: x 2.0, diffuser pipe plus alum injection	1,054,000	10-15

*Total project cost, which is 2x construction cost.
**These estimates are the same as without diffuser because P model assumes complete mixing, which diffuser pipe would ensure.
***No test results on P removal from inflow water, but would probably reduce lake TP lower than alum or pumping alone and similar to alum and pumping together.

5 μg/L in lake TP, but pump increments above that decrease lake TP by only 1.5 μg/L. Effectiveness is much improved by adding alum along with pumping, since there would be less internal loading to dilute with river water.

The added 2,000 foot long (24" diameter) diffuser pipe to disperse the inflow would double the cost. Whether it would double effectiveness is doubtful, although there is little information to judge how much more such a pipe would disperse river inflow water. The P model assumes complete mixing. Releasing the water at the lake edge, as is done now, may lose effectiveness if much of it is short circuited out of the lake via ground water exfiltration from that end of the lake (see water budget).

Monitoring results from the lake after the alum treatment and river flow increase would provide evidence on the effectiveness of treatment without the diffuser pipe. If data indicated short circuiting, then installation of a diffuser pipe could be considered. The evidence at hand does not justify doubling the cost.

Alum injection appears too costly for the added benefits. Also, continuous addition of alum (6 months/year), in proportion to flow, could present unanticipated problems from accumulated floc in the lake and elevated aluminum concentrations. In only 6 months, the quantity added would be 3-4 times more than with the whole-lake alum treatment alone.

Watershed Management Plan

The effect on lake TP concentration from controlling stormwater input (diversion, treatment, etc.) should be relatively small, compared to other alternatives (e.g., increased pumping and alum). Therefore, installation of a detention pond, soil infiltration or diversion of stormwater from the lake is not considered cost effective at this time. Nevertheless, precautions can be taken on a day-to-day basis to minimize sediment and nutrient transport to the lake resulting from activities in the lake's watershed.

The Horseshoe Lake watershed is rather small and the stormwater is primarily concentrated within the city north of the lake (Figure 33). Precautions have already been taken to reduce runoff from the cattle-holding area on the south shore by holding the cattle out of the watershed. Other precautions that can be taken include:

1. Require an erosion control plan to be filed for all construction projects larger than some minimum size. Review the erosion control plan to ensure that the proper controls are provided for the site characteristics, coverage is complete, and designs are appropriate. Inspect the sites to ensure that erosion control plans are properly implemented.
2. For any new developments, besides erosion control precautions, require wet detention ponds, followed by vegetated drainage courses, to treat runoff by particle settling and retention in sediment/soil.
3. Institute mechanical street sweeping and stormwater trap cleaning on a routine basis.
4. Investigate inputs of storm and waste water to the river upstream from the pump intake. The river level of SRP should be kept as low as possible to ensure the greatest effectiveness of the pumped water to maintain high lake quality.
5. Adopt non-structural measures, such as adopting regulations, inspection and enforcement programs to put the various recommendations in force. Establish a facilities maintenance program based on recommendations above. Establish a complaint and response program. Establish a comprehensive education program for watershed residents. This program should include appropriate signs in public areas conveying ecological messages, displays and publications, educational activities, special training, and measures to discourage significant pollutant releases (e.g., catch basin stencils and used oil recycling).

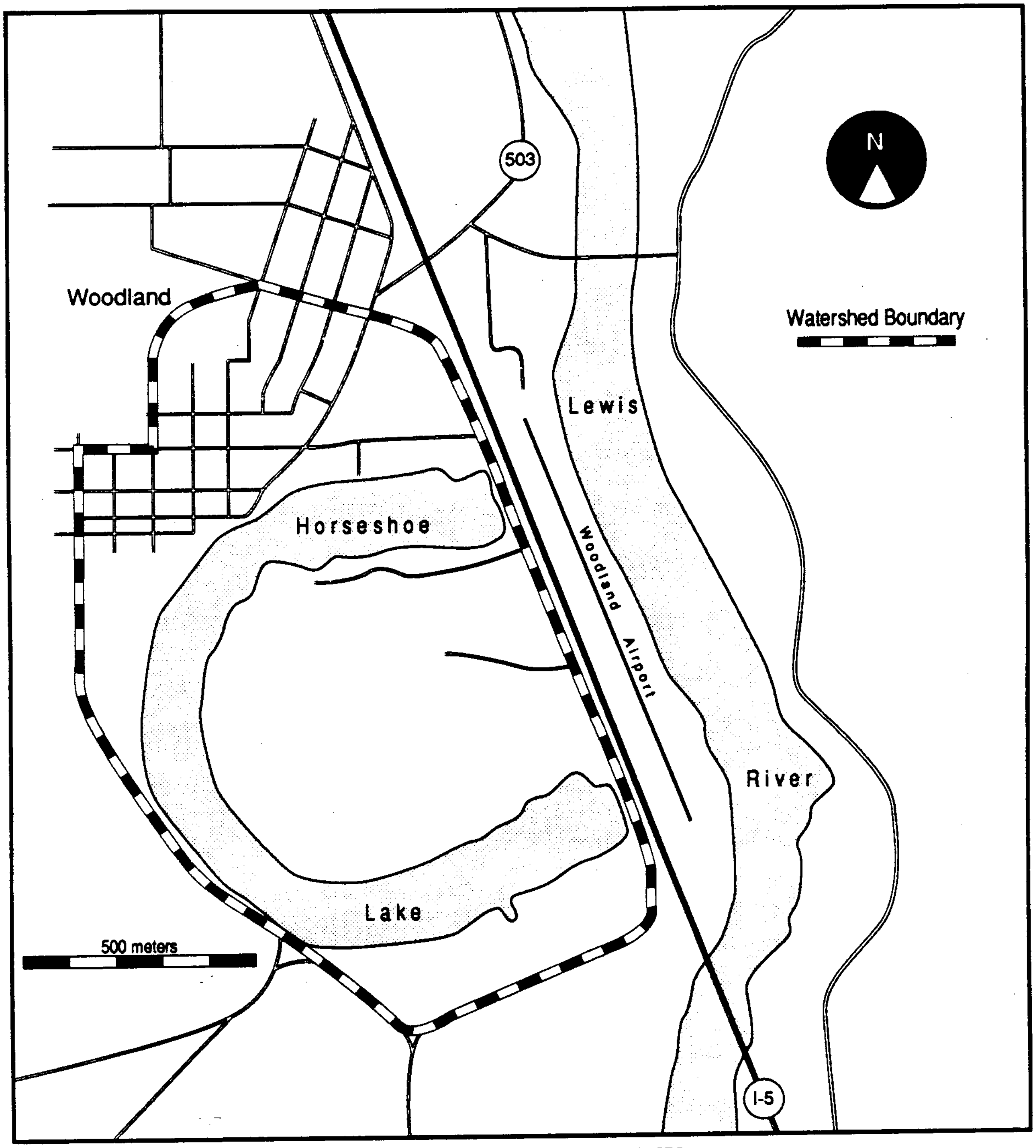


Figure 33. The Horseshoe Lake Watershed in Woodland, Wa.

SUMMARY AND CONCLUSIONS

1. The lake did not permanently stratify thermally (i.e., it remained mixed), probably because the river water was denser (higher conductance and lower temperature) than the lake water and added at a high rate. That prevented oxygen depletion in the bottom water, which would have allowed a greater rate of phosphorus release from bottom sediments. Dissolved oxygen at the bottom never dropped below 4 mg/L.
2. Summer TP concentrations were higher than those in winter indicating an internal source of P. The whole-lake summer mean was 30 μg/L, which exceeded the guidelines for an eutrophic lake (>25 μg/L). The river TP concentration averaged 9 μg/L less than the lake concentration in summer, again indicating an internal source of P. That was corroborated by river TN also being lower than lake TN.
3. River SRP concentration averaged 10 μg/L (with a maximum of 15 μg/L) in summer and represents the usable form for growth.
4. Inorganic N:SRP ratios in the lake indicated that N may have limited growth rate of algae in the short term. However, the high TN:TP ratio (summer mean = 18:1) suggests that P was the more important limiting nutrient in the long term and should be the nutrient to control and to manage water quality.
5. Algae reached bloom proportions during June of 1991 and 1992, but was surprisingly short lived in 1991. Observations by residents and the author indicated that blooms (surface water and colored water) persisted longer during summer and fall in previous years. The mean chl a in summer 1991 was 15 μg/L, substantially above the eutrophic treshold of 8.7 μg/L. Blue-green algae, primarily *Anabaena*, was the dominant taxon. Summer transparency averaged 1.2 m, slightly greater than in 1988-1989 (1.0 m).

6. Zooplankton, especially *Daphnia*, were very abundant in the early spring, but declined following the stocking of nearly 21,500 catchable rainbow trout. The blue-green algal bloom followed the reduction in zooplankton. That chain of events could have been causally linked, because trout select *Daphnia*, which are effective grazers of suitable sized algae.

7. Phosphorus concentrations in sediment samples were low ($X < 0.1\%$ P) and of uniform profile. Concentrations of the constituents measured were greater at Station 1 than Station 2 reflecting the greater external loading to the north arm basin. Uniform phosphorus profiles may indicate efficient nutrient cycling from the sediments to the water column enhanced by Lewis River inflow scour.

8. River water was pumped to the lake continuously during the study period amounting to 91% of the total, with the balance coming from stormwater and precipitation. Outflow, including groundwater and evaporation, exceeded inflow during the summer-fall period due to groundwater exfiltration.

9. Due to the continuous pumping, at no time did groundwater flow into the lake as indicated by water elevations in well points being lower than the lake level elevation. There was reason to suspect that in previous years groundwater may have flowed into the lake during periods of no pumping and lake level lowering.

10. Lake level and groundwater exfiltration are affected by river level. River flows below about 116 m^3/s (4,048 cfs) presently result in net water loss from the lake.

11. Consistent with water inflow volume, most (63%) of the external loading of TP came from the river. The contribution of TP from stormwater was substantial, in spite of the low volume flow, because the TP concentration was high. However, stormwater entered during winter and early spring and was probably unavailable for the algal growth season.

12. Internal loading from lake sediments provided a net release of 34 kg during June through September, about one quarter of the annual external load. Internal

loading represented the major contribution to lake P concentration and algal growth, whether TP or SRP is considered.

13. To reduce internal P loading and control summer algal blooms, sediment P inactivation with alum is recommended, together with increased pumping of river water to dilute lake TP, as the most cost effective solution. Alum alone is expected to be 50-80% effective, which according to a calibrated mass balance model, would lower mean summer TP from 30 to 15-20 µg/L. That should result in a summer chl a decrease from 15 µg/L to 5.4-8.3 µg/L and a transparency increase from 1.2 m to 1.8-2.4 m. Pumping (50% increase) alone would lower lake TP from 30 µg/L to 25 µg/L. Both treatments together would lower lake TP to 13-18 µg/L.

14. Sediment P inactivation is more cost effective than pump enlargement, but pumping increase has other benefits in maintaining high lake level to prevent groundwater influx and increased lake mixing to prevent stratification.

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APPENDIX A

Quality Assurance

Horseshoe Lake Quality Control

	Field Samples	Quality Control Samples		Sediment Samples	
	Range of % RSD	% RSD	% Recovery	% RSD	% Recovery
TP	4.9 - 27.0	3.4	98.9	2.8	96.7
SRP	2.7 - 37.7	1.8	101.5		
NO3-N	3.6 - 10.7	2.3	98.4		
NH4-N	9.6 - 16.2	4.6	92.4		
TN	1.9 - 8.9	8.7	88.4		
Chl a	6.9 - 78.6	3.2	88.9		
ALK	0.5 - 3.8	3.2	98.5		
pH	0	2.8	98.3		
T(C°)	0				
Conductivity	0.3 - 4.1				
Turbidity	1.2 - 6.7				
Seston	15.7 - 33.7				
K		1.4	98.7		
Ca		6.3	89.5		
Mg		1.7	92.8		
Al		0.0	90.9	3.2	94.3
Fe		10.5	95.0	5.4	96.8
Mn		3.2	96.4		
Na		3.6	92.1		
Cl		0.6	97.3		
SO4		2.4	94.6		
Pb				6.4	89.2
Zn				4.1	94.3

Precision and accuracy of water and sediment analyses expressed as % relative standard deviation (RSD) and % recovery.

APPENDIX B-1

Whole Lake, Volume Weighted Mean and

River Inflow Water Quality Constituents

Horseshoe Lake Water Quality Data

Sampling Date	TP ug/L	SRP ug/L	TN ug/L	NO3 ug/L	NH4 ug/L	Inorganic N ug/L	TN/TP Ratio	Inorganic N/SRP Ratio	Secchi m	Chla ug/L
5-Apr	31.8	3.9	420.7	0.0	0.0	0.0	13.2	0.0	1.4	5.4
26-Apr	23.0	4.7	339.6	10.3	0.0	10.3	14.8	2.2	2.2	13.8
10-May	19.5	4.3	336.6	8.0	0.0	8.0	17.3	1.8	2.6	3.7
24-May	21.3	7.0	313.3	0.0	0.0	0.0	14.7	0.0	2.5	6.9
7-Jun	23.8	2.6	478.6	0.0	0.0	0.0	20.1	0.0	1.6	12.6
21-Jun	29.0	3.1	384.6	10.2	0.0	10.2	13.3	3.3	1.1	19.8
5-Jul	37.3	5.3	971.0	20.6	0.0	20.6	26.0	3.9	0.7	38.3
19-Jul	29.7	4.9	1036.1	18.9	183.1	202.0	34.9	41.0	1.0	7.8
2-Aug	30.3	3.3	517.1	20.7	71.6	92.3	17.1	28.0	1.7	16.2
16-Aug	32.7	2.9	433.6	0.0	0.0	0.0	13.3	0.0	1.1	10.4
30-Aug	28.7	2.3	411.2	0.0	0.0	0.0	14.3	0.0	1.4	15.9
14-Sep	31.2	1.4	390.2	0.0	0.0	0.0	12.5	0.0	1.0	7.9
27-Sep	26.8	1.7	355.6	0.0	0.0	0.0	13.3	0.0	1.3	6.0
12-Oct	31.3	0.1	417.9	0.0	0.0	0.0	13.4	0.0	1.2	12.7
24-Oct	29.6	0.8	358.2	0.0	0.0	0.0	12.1	0.0	1.2	11.3
29-Nov	21.4	2.2	297.9	6.5	35.6	42.1	13.9	19.1	2.0	6.9
27-Dec	13.5	2.2	241.3	42.6	0.0	42.6	17.9	19.4	2.6	5.3
24-Jan	14.3	2.6	243.3	45.0	0.0	45.0	17.0	17.3	3.2	5.4
25-Feb	16.8	1.5	221.5	15.2	0.0	15.2	13.2	10.1	2.5	5.9
28-Mar	25.4	1.6			0.0				1.7	7.0
24-Apr	23.4	4.0							1.4	3.4
19-Jun	40.5	15.1							1.3	28.7
Summer Averages	29.9	3.1	553.1	7.8	28.3	36.1	18.3	8.5	1.2	15.0
Yearly Averages	24.0	2.9	396.3						1.8	9.3

Lewis River Water Quality Data

Sampling Date	TP ug/L	SRP ug/L	TN ug/L	NO3 ug/L	NH4 ug/L	Inorganic N ug/L	TN/TP Ratio	Inorganic N/SRP Ratio
5-Apr-91	41.7	9.2	525.3	310.8	0.0	310.8	12.6	33.8
26-Apr-91	21.5	7.5	279.3	117.1	0.0	117.1	13.0	15.6
10-May-91	14.4	8.7	192.3	140.1	0.0	140.1	13.4	16.1
24-May-91	15.3	10.0	260.4	101.1	0.0	101.1	17.0	10.1
7-Jun-91	18.2	5.2	123.6	62.5	10.3	72.8	6.8	14.0
21-Jun-91	20.8	10.2	200.2	68.0	12.6	80.6	9.6	7.9
5-Jul-91	26.9	9.4	168.8	36.4	11.5	47.9	6.3	5.1
19-Jul-91	20.7	15.6	179.3	19.4	19.9	39.3	8.7	2.5
2-Aug-91	35.4	12.6	151.4	23.9	15.0	38.9	4.3	3.1
16-Aug-91	29.4	15.2	165.4	27.3	10.3	37.6	5.6	2.5
30-Aug-91	17.6	12.0	137.5	121.4	0.0	121.4	7.8	10.1
14-Sep-91	13.4	10.8	200.2	34.1	17.3	51.4	14.9	4.8
27-Sep-91	11.7	7.8	165.4	23.9	0.0	23.9	14.1	3.1
12-Oct-91	6.4	8.3	235.0	10.4	0.0	10.4	36.7	1.3
24-Oct-91	7.7	6.8	116.6	31.4	10.3	41.7	15.1	6.1
29-Nov-91	8.7	3.1	207.2	128.7	0.0	128.7	23.8	41.5
27-Dec-91	19.6	13.0	290.8	191.2	11.5	202.7	14.8	15.6
24-Jan-92	9.0	4.1	242.0	145.1	0.0	145.1	26.9	35.4
25-Feb-92	8.9	3.6	266.4	154.4	0.0	154.4	29.9	42.9
28-Mar-92	38.9	18.1		72.4	0.0		0.0	4.0
24-Apr-92	17.0	16.0					0.0	0.0
19-Jun-92	23.1	19.7					0.0	0.0

APPENDIX B-2

Water Quality Data for Lake and Groundwater by Station and Date

HORSESHOE LAKE S1-S

Station	Date	Secchi m	TP ug/L	SRP ug/L	TN ug/L	NO3 ug/L	NH4 ug/L	Chl a ug/L	Alk. mg/L CaCo3	DO mg/L	Temp. C	Conduct. umhos	pH	Turbidity NTU	Lake Level ft. (m. s. l.)
S1-S	5-Apr-91	1.4	23.1	3.6	260.4	x<3	x<10	6.7	17.4	11.9	11.2	40.7	7.3	2.3	14.0
S1-S	26-Apr-91	2.2	17.8	7.2	268.0	15.2	x<10	9.5		11.1	12.2	40.1	7.3	1.0	14.1
S1-S	10-May-91	2.6	12.5	3.9	283.1	17.2	x<10	3.6	16.4	10.2	14.0	40.4	7.1	1.8	14.5
S1-S	24-May-91	2.9	17.7	5.9	260.4	x<3	x<10	3.3	16.4	10.6	14.8	40.1	7.3	2.3	14.3
S1-S	7-Jun-91	1.7	23.0	2.3	559.0	x<3	x<10	2.6		11.1	16.5	39.5	8.3	5.8	13.7
S1-S	21-Jun-91	1.1	26.1	3.2	416.2	3.6	x<10	33.6	14.9	12.2	18.0	40.1	8.6	13.6	13.2
S1-S	5-Jul-91	0.8	31.3	4.3	1119.9	18.1	x<10	32.0	14.6	11.2	22.0	37.6	9.4	20.0	13.6
S1-S	19-Jul-91	1.1	25.5	6.0	970.1	13.3	132.0	18.0	13.5	7.3	21.2	40.6	6.9 *	9.4	13.8
S1-S	2-Aug-91	2.2	15.5	3.7	513.7	19.7	82.5	6.4	15.0	9.6	22.9	42.4	7.7	2.5	13.6
S1-S	16-Aug-91	1.4	19.2	2.3	398.8	x<3	x<10	0.0	13.4	9.3	22.5	40.1	8.5	2.3	13.4
S1-S	30-Aug-91	1.6	29.3	2.4	388.3	4.4	x<10		14.1	9.9	19.9	42.7	8.4	3.4	13.1
S1-S	14-Sep-91	1.1	19.0	1.3	336.1	4.4	10.3	6.8	14.0	9.4	18.7	43.0	7.6	3.9	12.9
S1-S	27-Sep-91	1.9	22.2	1.7	388.3	x<3	x<10	0.0	14.6	10.0	19.2	43.6	7.9	4.1	12.7
S1-S	12-Oct-91	1.5	25.7	0.0	311.7	x<3	x<10	10.9	14.2	10.6	18.0	40.6	7.9	3.4	12.5
S1-S	24-Oct-91	1.4	30.3	1.8	318.6		x<10	13.5	14.5	9.9	14.8	45.1	7.7		12.5
S1-S	29-Nov-91	2.1	26.6	2.0	308.2	23.9	31.3	2.6			8.9	50.5	7.5		13.6
S1-S	27-Dec-91	2.2	16.3	2.5	262.9	80.2	x<10	6.4	14.0	12.3	6.8	40.0	7.8	2.6	13.7
S1-S	24-Jan-92	2.9	10.7	2.2	231.5	75.4	x<10	1.6	13.8	12.5	6.0	44.4	7.7		13.6
S1-S	25-Feb-92	2.5	11.5	1.1	179.3	37.2	x<10	1.3	13.3	12.4	10.5	39.3	7.9	1.2	14.8
S1-S	28-Mar-92	2.0	19.2	0.0		x<10	x<10	7.4	13.9	11.2	13.0	40.5	7.5	3.1	14.5
S1-S	24-Apr-92	1.6	21.6	3.2		x<10	x<10	3.8	14.8	11.4	16.5	43.3	8.3	4.2	14.2
S1-S	19-Jun-92	1.5	37.1	18.2					16.5	9.2	22.5	45.8	7.3	4.5	12.4

HORSESHOE LAKE S1-2

Station	Date	TP ug/L	SRP ug/L	TN ug/L	NO3 ug/L	NH4 ug/L	Chl a ug/L	Alk. mg/L CaCo3	DO mg/L	Temp. C	Conduct. umhos	pH	Turbidity NTU
S1-2	5-Apr-91	34.1	3.5	347.4	x<3	x<10	4.7	15.26	12.0	10.6	40.1	7.0	2.3
S1-2	26-Apr-91	21.0	3.3	343.7	13.2	x<10	12.5		11.1	12.2	40.1	6.9	1.8
S1-2	10-May-91	22.8	4.2	373.9	19.2	12.4	2.0	17.44	10.1	14.0	40.4	7.3	1.8
S1-2	24-May-91	20.8	8.0	252.8	x<3	x<10	10.0	16.90	10.8	14.1	40.1	7.3	2.2
S1-2	7-Jun-91	22.6	2.8	492.8	x<3	x<10	19.2		11.8	15.9	39.8	8.2	6.3
S1-2	21-Jun-91	32.0	2.3	419.7	6.0	x<10	4.8	14.62	12.0	16.7	39.5	8.8	11.7
S1-2	5-Jul-91	33.7	3.9	1039.8	23.0	x<10	31.2	14.13	11.5	20.0	37.2	9.2	13.0
S1-2	19-Jul-91	26.4	3.5	1022.4	20.6	144.9	12.4	13.30	7.0	20.9	43.0	7.1	9.6
S1-2	2-Aug-91	33.7	3.7	374.4	16.3	79.0	18.3	14.77	9.5	22.5	44.4	7.8	2.9
S1-2	16-Aug-91	27.1	3.0	416.2	x<3	x<10	5.3	13.50	9.6	21.5	40.8	8.9	2.2
S1-2	30-Aug-91	21.1	2.7	363.9	x<3	x<10	6.4	13.84	10.0	19.5	43.7	8.1	3.6
S1-2	14-Sep-91	24.1	1.7	370.9	x<3	12.6	3.2	14.33	9.7	18.1	41.8	7.6	4.7
S1-2	27-Sep-91	21.4	1.6	336.1	x<3	x<10	6.0	14.23	10.2	19.0	42.7	7.9	3.8
S1-2	12-Oct-91	28.9	0.0	381.4	x<3	x<10	7.4	13.94	10.7	17.6	42.0	8.2	2.0
S1-2	24-Oct-91	37.6	1.5	346.5		x<10	12.2	14.18	9.9	14.6	46.4	8.3	
S1-2	29-Nov-91	18.9	2.6	339.5	x<10	39.4	10.9	10.25		8.9	43.5	7.6	
S1-2	27-Dec-91	12.3	2.5	259.4	73.1	x<10	10.3	14.69	12.3	6.8	40.2	7.7	2.9
S1-2	24-Jan-92	24.8	1.8	249.0	74.2	x<10	6.1	14.43	12.3	6.0	37.6	7.7	
S1-2	25-Feb-92	17.3	1.4	207.2	x<10	x<10	10.6	14.03	12.6	9.2	39.1	7.9	1.5
S1-2	28-Mar-92	27.4	1.9		x<10	x<10	5.8	13.97	11.3	12.5	40.2	7.6	3.1
S1-2	24-Apr-92	24.5	5		x<10	x<10	3.2	14.74	11.7	15.5	43.3	8.3	3.9
S1-2	19-Jun-92	50.3	18.3				25.6	16.12	8.7	22	47.1	7.3	4.8

HORSESHOE LAKE S1-4

Station	Date	TP ug/L	SRP ug/L	TN ug/L	NO3 ug/L	NH4 ug/L	Chl a ug/L	Alk. mg/L CaCo3	DO mg/L	Temp. C	Conduct. umhos	pH	Turbidity NTU
S1-4	5-Apr-91	29.8	5.1	487.5	5.2	x<10	7.3	16.4	12.0	10.5	40.7		2.8
S1-4	26-Apr-91	18.1	3.6		15.2	x<10	11.0		10.7	12.0	40.1	7.3	1.8
S1-4	10-May-91	16.0	4.3	275.5	16.2	11.2	3.2	16.9	10.0	13.8	40.7	7.0	1.8
S1-4	24-May-91	18.2	8.0	283.1	3.2	x<10	6.4	16.4	10.7	13.9	40.1	7.3	2.4
S1-4	7-Jun-91	21.1	2.7		x<3	x<10	13.4		11.1	14.9	39.8	7.9	5.5
S1-4	21-Jun-91	32.8	2.6	353.5	13.3	x<10	25.6	14.7	10.7	16.0	39.5	8.4	11.5
S1-4	5-Jul-91	28.4	4.2	785.5	23.0	x<10	8.8	14.0	10.2	18.5	36.3	8.9	15.0
S1-4	19-Jul-91	26.3	4.2	876.1	23.0	139.3	15.2	13.7	5.7	18.9	42.8	6.8	6.2
S1-4	2-Aug-91	30.2	3.5	482.4	23.1	82.5	17.0	14.7	8.6	22.0	45.5	7.6	3.7
S1-4	16-Aug-91	39.8	2.3	391.8	x<3	10.3	11.7	14.0	8.5	20.8	40.8	8.4	3.8
S1-4	30-Aug-91	26.5	2.2	339.5	x<3	24.3	21.6	14.5	8.2	18.3	45.0	7.6	4.2
S1-4	14-Sep-91	38.7	1.8	367.4	x<3	x<10	10.4	14.4	8.6	17.9	44.2	7.5	7.4
S1-4	27-Sep-91	24.2	1.5	294.3	x<3	x<10	6.4	13.9	9.7	18.7	43.1	7.8	4.3
S1-4	12-Oct-91	30.5	0.0	423.2	x<3	x<10	12.5	14.6	10.7	17.0	42.2	8.0	3.0
S1-4	24-Oct-91	28.7	1.3	325.6		x<10	8.3	14.1	9.8	14.5	46.0	7.8	
S1-4	29-Nov-91	20.0	2.4	304.7	23.3	40.6	11.5	2.2		8.9	42.7	7.6	
S1-4	27-Dec-91	18.6	2.4	255.9	68.3	15.6	0.0	13.8	12.7	6.8	40.1	7.7	2.9
S1-4	24-Jan-92	13.5	2.9	231.5	71.9	x<10	1.9	13.8	12.3	6.0	39.1	7.7	
S1-4	25-Feb-92	22.1	1.2	266.4	31.4	x<10	5.1	13.2	12.3	9.0	39.1	7.8	1.8
S1-4	28-Mar-92	25.0	1.4		x<10	x<10	6.4	14.5	11.3	12.2	40.3	7.6	3.3
S1-4	24-Apr-92	20.0	4.1		x<10	x<10	2.9	14.6	11.7	15.1	43.3	8.2	4.8
S1-4	19-Jun-92	36.4	19.2				29.2	15.7	6.7	21.0	48.7	7.1	4.8

HORSESHOE LAKE S1-6

Station	Date	TP ug/L	SRP ug/L	TN ug/L	NO3 ug/L	NH4 ug/L	Chl a ug/L	Alk. mg/L CaCo3	DO mg/L	Temp. C	Conduct. umhos	pH	Turbidity NTU
S1-6	5-Apr-91		3.6	593.5	21.2	x<10	7.3	15.3	11.8	10.2	40.1	7.2	
S1-6	26-Apr-91	20.9	4.6		13.2	x<10	13.0		10.9	12.0	40.1	7.3	2.2
S1-6	10-May-91	14.3	4.1	404.2	19.2	13.6	2.0	16.9	10.0	13.5	40.4	7.5	1.7
S1-6	24-May-91	19.0	7.2	305.8	3.2	x<10	6.5	16.7	10.3	13.8	40.1	7.1	2.9
S1-6	7-Jun-91	29.7	1.9	332.6	7.2	10.3	5.3		10.8	14.8	40.1	7.6	5.6
S1-6	21-Jun-91	31.1	2.9	318.6	18.1	x<10	6.4	14.3	10.0	16.0	40.1	7.9	11.3
S1-6	5-Jul-91	29.2	7.6	548.6	30.3	x<10	4.8	14.3	8.7	18.0	36.9	8.3	6.0
S1-6	19-Jul-91	40.8	6.5	869.1	10.8	277.2	4.4	14.7	4.7	18.5	42.0	7.0	6.1
S1-6	2-Aug-91	24.5	4.0	513.7	23.9	110.5	21.5	14.6	7.7	21.2	42.2	7.4	4.3
S1-6	16-Aug-91	25.0	4.4	447.5	x<3	x<10	10.7	14.5	7.2	20.2	41.5	7.8	4.5
S1-6	30-Aug-91	24.4	4.0	374.4	x<3	x<10	22.1	14.6	7.6	18.2	45.0	7.3	4.9
S1-6	14-Sep-91	27.8	1.6	409.2	x<3	x<10	9.2	13.8	9.0	17.8	43.8	7.5	6.5
S1-6	27-Sep-91	29.1	1.7	308.2	x<3	x<10	2.4	14.4	9.6	18.5	42.8	7.7	4.8
S1-6	12-Oct-91	29.7	1.8	391.8	x<3	x<10	14.7	14.1	10.0	16.5	42.1	7.9	3.5
S1-6	24-Oct-91	28.4	1.6	308.2		x<10	17.3	13.9	9.4	14.5	44.5	7.6	
S1-6	29-Nov-91	19.9	2.1	297.7	25.6	33.6	13.1			7.9	43.7	7.5	
S1-6	27-Dec-91	18.3	2.2	235.0	70.7	x<10	2.2	13.8	12.1	6.8	40.1	7.6	3.1
S1-6	24-Jan-92	11.0	9.4	235.0	71.3	x<10	3.5	14.2	12.3	6.0	42.1	7.7	
S1-6	25-Feb-92	25.6	1.5	228.1	37.2	x<10	7.4	13.4	12.2	8.9	38.1	7.6	2.0
S1-6	28-Mar-92	31.0	1.7		x<10	x<10	6.7	14.2	11.3	12.1	40.8	7.6	3.1
S1-6	24-Apr-92	17.1	5.7		x<10	x<10	2.2	14.6	11.5	14.9	42.1	7.7	4.7
S1-6	6/19/92	45.4	36.0					12.8	6.4	20.5	45.3	7.0	4.9

HORSESHOE LAKE S2-S

Station	Date	Secchi m	TP ug/L	SRP ug/L	TN ug/L	NO3 ug/L	NH4 ug/L	Chl a ug/L	Alk. mg/L CaCo3	DO mg/L	Temp. C	Conduct. umhos	pH	Turbidity NTU
S2-S	5-Apr-91	1.4	26.8	3.2	373.9	x<3	x<10	5.3	17.3	11.6	11.2	40.4	7.3	2.8
S2-S	26-Apr-91	2.2	21.0	7.8	336.1	16.2	x<10	8.7				40.4	7.4	1.8
S2-S	10-May-91	2.6	18.0	4.0	260.4	x<3	x<10	3.2	17.4	10.0	14.0	40.7	7.5	2.3
S2-S	24-May-91	2.4	18.2	5.9	328.5	x<3	x<10	6.4	16.0	10.4	15.5	40.4	7.3	2.4
S2-S	7-Jun-91	1.6	22.1	2.6	492.8	x<3	10.3	17.6		11.1	17.0	39.8	8.2	5.7
S2-S	21-Jun-91	1.1	27.8	2.6	346.5	x<3	x<10	20.8	14.6	12.4	16.9	39.5	8.6	13.3
S2-S	5-Jul-91	0.6	30.2	4.4	1046.8	15.7	17.1	12.0	13.7	10.9	21.5	35.4	9.3	21.0
S2-S	19-Jul-91	1.0	33.5	3.9	1032.8	21.8	181.9	8.0	13.6	7.1	21.2	41.2	7.3	8.7
S2-S	2-Aug-91	1.4	35.2	3.4	513.7	23.1	66.2	18.6	14.2	9.7	23.5	42.3	8.1	5.4
S2-S	16-Aug-91	0.9	34.8	1.9	482.4	x<3	x<10	4.8	13.3	9.1	23.9	40.4	9.0	3.8
S2-S	30-Aug-91	1.3	21.9	2.0	433.6	x<3	x<10	15.1	13.8	9.8	20.7	42.3	8.4	3.6
S2-S	14-Sep-91	0.9	26.4	1.1	416.2	x<3	x<10	9.6	13.7	9.3	19.1	41.5	7.7	4.8
S2-S	27-Sep-91	1.2	20.8	1.5	374.4	x<3	x<10	1.6	12.5	10.1	20.1	40.3	8.1	6.6
S2-S	12-Oct-91	1.1	26.5	0.0	437.1	x<3	x<10	12.8	13.0		17.5	41.4	8.4	3.4
S2-S	24-Oct-91	1.2	24.4	1.3	388.3		x<10		12.7		14.0	44.5	7.8	
S2-S	29-Nov-91	2.0	17.1	1.8	290.8	x<10	25.5	5.4	0.6		7.9	43.3	7.5	
S2-S	27-Dec-91	2.8	9.5	1.3	207.2	23.4	x<10	0.0	13.4	12.7	6.3	38.8	8.0	1.3
S2-S	24-Jan-92	3.7	10.6	1.9	221.1	33.8	x<10	5.4	14.0	12.5	5.8	37.0	7.6	
S2-S	25-Feb-92	2.6	11.5	1.0	262.9	49.0	x<10	4.8	12.8	12.3	11.0	38.7	7.6	1.3
S2-S	28-Mar-92	1.5	23.0	1.2		x<10	x<10	6.4	13.3	11.3	13.2	38.2	7.6	3.3
S2-S	24-Apr-92	1.4	24.0	3.7		x<10	x<10	0.6	14.8	11.0	17.0	40.5	8.1	4.2
S2-S	19-Jun-92	1.2	34.0	10.6				29.8	16.6			47.3		5.7

HORSESHOE LAKE S2-B

Station	Date	TP ug/L	SRP ug/L	TN ug/L	NO3 ug/L	NH4 ug/L	Chl a ug/L	Alk. mg/L CaCo3	DO mg/L	Temp. C	Conduct. umhos	pH	Turbidity NTU
S2-B	5-Apr-91	38.7	3.3	419.4	x<3	x<10	6.7	15.3	11.4	10.6	40.7	7.4	6.5
S2-B	26-Apr-91	22.1 #	5.2	457.2	21.2	x<10	14.5				40.7	7.3	2.2
S2-B	10-May-91	19.6	4.3	373.9	x<3	x<10	4.4	17.4	10.0	14.0	40.4	7.2	2.3
S2-B	24-May-91	21.4	5.9	313.4	x<3	x<10	6.0	16.4	10.5	14.8	40.7	7.4	2.9
S2-B	7-Jun-91	26.3	2.6	454.5	x<3	10.3	15.5		10.9	15.2	39.8	7.9	6.4
S2-B	21-Jun-91	28.7	3.8	374.4	7.8	27.2	16.0	14.3	11.5	16.3	39.5	8.6	12.6
S2-B	5-Jul-91	45.3	4.3	966.6	18.1	x<10	56.9	14.1	9.0	19.8	37.6	9.2	21.0
S2-B	19-Jul-91	33.5	4.8	1050.3	15.7	180.2	1.6	13.7	6.8	20.5	42.3	7.5	9.2
S2-B	2-Aug-91	28.1	3.5	621.7	23.9	70.9	27.2	13.9	9.6	22.0	43.3	8.2	6.8
S2-B	16-Aug-91	40.1	2.5	478.9	x<3	x<10	19.8	12.8	9.2	20.9	40.4	8.5	4.3
S2-B	30-Aug-91	38.1	2.1	458.0	x<3	x<10		13.5	8.1	18.5	42.4	7.8	7.4
S2-B	14-Sep-91	40.0	1.1	437.1	x<3	x<10	14.0	13.0	9.1	17.9	40.6	7.7	10.5
S2-B	27-Sep-91	31.2	1.5	388.3	x<3	x<10	10.8	13.2		18.9	40.8	8.0	6.6
S2-B	12-Oct-91	33.6	0.0	454.5	x<3	x<10	15.4	13.1		17.0	41.2	8.2	4.3
S2-B	24-Oct-91	31.4	0.0	412.7		x<10	10.3	12.7		13.9	43.9	7.8	
S2-B	29-Nov-91	20.2	2.4	318.6	x<10	30.1	12.5	2.6		7.9	42.3	7.5	
S2-B	27-Dec-91	13.7	3.6	207.2	25.8	x<10	2.6	13.3	12.7	6.2	38.8	7.9	1.9
S2-B	24-Jan-92	11.8	1.3	228.1	22.1	x<10	4.5	15.0	12.5	5.8	37.0	7.6	
S2-B	25-Feb-92	16.5	1.4	210.6	x<10	x<10	8.0	13.0	12.2	9.2	37.8	7.6	2.2
S2-B	28-Mar-92	18.3	1.9		x<10	x<10	10.4	13.7	10.9	12.5	38.0	7.6	3.6
S2-B	24-Apr-92	22.2	3.5		x<10	x<10	4.0	14.1	11.2	16.8	41.7	7.9	4.2
S2-B	19-Jun-92	47.9	10.3				41.3	14.8			49.4		8.7

HORSESHOE LAKE S3-S

Station	Date	Secchi m	TP ug/L	SRP ug/L	TN ug/L	NO3 ug/L	NH4 ug/L	Chl a ug/L	Alk. mg/L CaCo3	DO mg/L	Temp. C	Conduct. umhos	pH	Turbidity NTU
S3-S	5-Apr-91	1.4	29.3	3.6	396.6	x<3	x<10	0.0	13.1	11.4	11.8	40.7	7.5	3.0
S3-S	26-Apr-91	2.2	24.6	7.0	199.8	x<3	x<10	10.0				39.8	7.3	2.0
S3-S	10-May-91	2.6	23.7	4.5	373.9	x<3	x<10	2.8	16.9	10.0	14.5	40.7	7.3	2.3
S3-S	24-May-91	2.2	18.7	7.3	279.3	x<3	x<10	7.6	16.4	10.3	15.5	39.8	7.4	2.5
S3-S	7-Jun-91	1.6	21.7	2.3	471.9	x<3	x<10	13.9		11.0	16.2	40.1	8.1	6.3
S3-S	21-Jun-91	1.1	25.1	2.6	402.3	7.2	x<10	28.8	14.6	11.6	18.2	39.5	8.7	13.1
S3-S	5-Jul-91	0.6	36.1	6.2		17.5	x<10	64.1	13.7	11.5	23.2	37.3	9.8	32.0
S3-S	19-Jul-91	0.9	30.1	3.4	1182.6	20.0	190.3	3.2	13.6	7.3	22.5	41.2	7.6	9.2
S3-S	2-Aug-91	1.5	28.5	2.8	597.4	32.4	52.2	12.2	13.6	9.7	24.2	43.1	8.7	4.7
S3-S	16-Aug-91	0.9	26.9	2.0	409.2	x<3	x<10	8.0	13.2	9.1	24.3	39.0	9.0	3.8
S3-S	30-Aug-91	1.1	27.0	1.9	458.0	x<3	x<10	12.8	13.6	9.3	20.2	41.7	7.9	4.7
S3-S	14-Sep-91	0.9	26.8	1.1	402.3	x<3	x<10	4.4	12.7	9.5	20.2	40.1	7.8	5.8
S3-S	27-Sep-91	1.0	28.2	0.0	381.4	x<3	x<10	5.6	12.5		20.1	39.7	8.1	6.5
S3-S	12-Oct-91	1.0	26.0	0.0	451.0	x<3	x<10	12.8	12.3		17.0	38.9	8.2	3.3
S3-S	24-Oct-91	1.1	23.8	0.0	370.9		x<10	13.5	12.8		14.2	43.6	7.8	
S3-S	29-Nov-91	2.0	19.3	1.7	304.7	x<10	19.6					41.6	7.4	
S3-S	27-Dec-91	1.9	11.5	3.2	231.5	16.3	x<10	4.8	13.3	12.7	6.0	38.1	7.9	1.6
S3-S	24-Jan-92	2.6	9.5	1.6	245.5	18.6	x<10	9.0	13.4	12.5	5.5	39.1	7.6	
S3-S	25-Feb-92	2.6	11.7	1.4	200.2	25.5	x<10	3.8	13.0	12.1	11.0	37.2	7.6	1.2
S3-S	28-Mar-92	1.6	25.0	1.5		x<10	x<10	0.0	13.0	10.8	14.0	38.8	7.6	3.7
S3-S	24-Apr-92	1.3	25.0	4.6		x<10	x<10	2.9	13.6	11.0	17.0	41.7	8.0	3.4
S3-S	19-Jun-92	1.3	35.4	8.7				19.2	16.4			47.7		5.6

HORSESHOE LAKE S3-B

Station	Date	TP ug/L	SRP ug/L	TN ug/L	NO3 ug/L	NH4 ug/L	Chl a ug/L	Alk. mg/L CaCo3	DO mg/L	Temp. C	Conduct. umhos	pH	Turbidity NTU
S3-B	5-Apr-91		3.7	525.3	x<3	x<10	4.0	17.4	11.2	11.5	40.1	7.5	6.8
S3-B	26-Apr-91	30.2	3.4	252.8	x<3	x<10	28.7				40.4	7.1	2.4
S3-B	10-May-91	22.5	4.8	396.6	x<3	x<10	4.4	17.4	10.0	14.2	41.6	7.3	2.5
S3-B	24-May-91	24.0	7.0	355.0	x<3	x<10	8.4	15.8	10.3	15.0	40.4	7.3	2.9
S3-B	7-Jun-91	22.7	2.5	499.8	x<3	x<10	2.7		11.2	16.0	40.1	8.2	6.4
S3-B	21-Jun-91	27.6	2.8	388.3	4.2	x<10	11.2	14.6	11.5	16.4	39.5	8.7	13.5
S3-B	5-Jul-91	45.3	8.2		13.3	x<10	49.7	13.7	11.1	21.5	34.5	9.7	32.0
S3-B	19-Jul-91	33.5	7.3	1099.9	20.0	221.1	5.3	13.3	6.9	21.5	41.7	7.6	9.4
S3-B	2-Aug-91	36.7	2.8	548.6	15.4	55.7	8.0	14.1	9.9	24.0	43.3	8.5	4.5
S3-B	16-Aug-91	40.4	4.6	458.0	x<3	x<10	11.7	12.7	8.9	21.9	40.2	9.0	4.8
S3-B	30-Aug-91	36.4	1.7	485.9	x<3	x<10	18.6	13.6	9.2	18.9	42.8	7.9	6.1
S3-B	14-Sep-91	35.0	1.2	395.3	x<3	16.1	4.0	12.1		18.7	41.0	8.0	6.6
S3-B	27-Sep-91	37.5	2.6	370.9	x<3	x<10	5.6	13.3	9.6	19.0	42.0	8.1	6.5
S3-B	12-Oct-91	40.6	0.1	458.0	x<3	x<10	13.8	13.3		17.0	40.5	8.1	3.8
S3-B	24-Oct-91	39.8	0.1	360.5		x<10	10.6	12.0		14.0	42.5	7.7	
S3-B	29-Nov-91	26.7	2.2	235.0	x<10	47.6	4.2	6.7		9.1	43.0		
S3-B	27-Dec-91	12.3	1.2	255.9	16.3	12.1	6.7	13.7	12.7	6.0	38.5	7.9	1.5
S3-B	24-Jan-92	25.0	1.5	266.4	23.3	x<10	7.0	13.2		5.5	37.0	7.6	
S3-B	25-Feb-92	18.0	1.5	231.5	x<10	x<10	7.0	13.8		9.5	38.4	7.7	1.6
S3-B	28-Mar-92	34.4	1.9		x<10	x<10	5.8	13.8		13.1	38.6	7.6	6.0
S3-B	24-Apr-92	28.8	3.7		x<10	x<10	0.3	14.0		16.8	41.7	7.9	4.2
S3-B	19-Jun-92	38.5	7.8				23.1	16.3			45.8		5.7

HORSESHOE LAKE OUTLET

Station	Date	Secchi m	TP ug/L	SRP ug/L	TN ug/L	NO3 ug/L	NH4 ug/L	Chl a ug/L	Alk. mg/L CaCo3	DO mg/L	Temp. C	Conduct. umhos	pH	Turbidity NTU	Seston mg/L
Outlet	5-Apr-91		32.1	5.1	351.2	x<3	x<10		17.4	11.8	11.9	40.1	7.6	3.5	3.0
Outlet	26-Apr-91		28.9	4.2	355.0	x<3	x<10					39.8		2.4	3.3
Outlet	10-May-91		19.7	4.6	215.0	x<3	x<10	6.4	15.8	10.1		40.4	7.0	2.3	4.0
Outlet	24-May-91		33.1	7.3	487.5	x<3	x<10		16.4	10.4	15.0	39.8	7.4	4.3	6.3
Outlet	7-Jun-91		29.8	3.4	503.3	x<3	10.3			11.2	16.2	40.1	8.1	6.6	5.6
Outlet	21-Jun-91		25.1	6.5	405.7	4.8	x<10		14.3	11.1	17.0	40.1	8.7	13.6	Outlet Closed
Outlet	5-Jul-91		48.2	5.3		12.7	10.4		13.7	12.2	25.2	35.6	9.7	39.0	Outlet Closed
Outlet	19-Jul-91		26.7	4.5	1224.4	18.1	254.8	5.6	13.1	6.8	23.0	42.7	7.6	9.9	Outlet Closed
Outlet	2-Aug-91	1.5	28.2	2.9	562.5	17.1	60.4	5.9	13.3	9.6	25.0	42.3	8.7	3.6	Outlet Closed
Outlet	16-Aug-91		23.7	2.0	412.7	x<3	x<10	9.6	12.4	8.6	25.2	40.0	9.0	1.8	Outlet Closed
Outlet	30-Aug-91	1.3	26.1	3.0	374.4	x<3	12.6	8.0	13.0	9.2	20.5	40.7	8.0	3.1	Outlet Closed
Outlet	14-Sep-91	0.9	31.3	1.1	381.4	x<3	x<10	6.0	11.9	9.2	21.0	46.9	8.0	5.2	Outlet Closed
Outlet	27-Sep-91		19.9	1.8	370.9	x<3	x<10	9.6	11.9		20.2	38.7	8.1	6.3	Outlet Closed
Outlet	12-Oct-91	1.1	29.7	0.0	402.3	12.1	x<10	14.1	11.8	10.0	18.5	39.8	8.1	4.5	Outlet Closed
Outlet	24-Oct-91	1.2	18.9	0.0	367.4		x<10	11.9	12.3		14.2	41.7	7.7		Outlet Closed
Outlet	29-Nov-91	2.1	21.7	2.0	283.8	x<10	33.6	6.4	2.9			40.5			
Outlet	27-Dec-91	x>2.6	9.0	1.0	231.5	6.3	x<10	2.9	12.8	12.7	6.0	37.8	8.0	1.4	2.2
Outlet	24-Jan-92	x>2.8	9.8	4.0	262.9	11.6	x<10	5.8	13.7	12.6	5.5	36.8	7.6		0.8
Outlet	25-Feb-92	2.0	13.6	3.0	186.3	25.5	x<10	6.1	12.7		10.6	38.5	7.8	1.7	3.4
Outlet	28-Mar-92	1.5	20.0	1.4		x<10	x<10	8.7	12.8	10.6	14.2	38.6	7.6	3.4	Outlet Closed
Outlet	24-Apr-92	1.2	21.6	3.1		x<10	x<10	12.8	13.9	11.2	17.2	37.9	8.0	4.2	Outlet Closed
Outlet	6/19/92	1.4	29.6	19.7					16.4			47.8		4.1	Outlet Closed

Station	Date	TP ug/L	SRP ug/L	TN ug/L	NO3 ug/L	NH4 ug/L	Chl a ug/L	Alk. mg/L CaCo3	DO mg/L	Temp. C	Conduct. umhos	pH	Turbidity NTU	Seston mg/L
Inlet	5-Apr-91	41.7	9.2	525.3	310.8	x<10			11.9	7.0	41.0	7.7	8.7	15.5
Inlet	26-Apr-91	21.5	7.5	279.3	117.1	x<10					39.5		2.8	2.7
Inlet	10-May-91	14.1	8.7	192.3	140.1	x<10	0.4	16.9	11.7		41.0	7.4	2.8	1.6
Inlet	24-May-91	15.3	10.0	260.4	101.1	x<10		16.4	11.3		39.5	6.7	1.4	
Inlet	7-Jun-91	18.2	5.2	123.6	62.5	10.3			11.4	12.0	39.8	7.4	2.1	2.5
Inlet	21-Jun-91	20.8	10.2	200.2	68.0	12.6		17.4	11.3	11.9	40.7	7.3	2.2	2.2
Inlet	5-Jul-91	26.9	9.4	168.8	36.4	11.5		16.3	11.1	16.5	38.7	7.9	2.0	1.6
Inlet	19-Jul-91	20.7	15.6	179.3	19.4	19.9		16.6	10.5	16.5	47.2	7.6	1.9	2.7
Inlet	2-Aug-91	35.4	12.6	151.4	23.9	15.0	0.0	17.1	10.0	19.5	49.2	7.7	1.4	5.2
Inlet	16-Aug-91	29.4	15.2	165.4	27.3	10.3	1.1	17.3	9.9	19.3	48.3	7.7	1.1	2.6
Inlet	30-Aug-91	17.6	12.0	137.5	121.4	x<10		16.4	10.2	15.9	47.7	7.7	1.3	0.5
Inlet	14-Sep-91	13.4	10.8	200.2	34.1	17.3		16.2	9.6	17.1	47.0	7.5	2.3	4.4
Inlet	27-Sep-91	11.7	7.8	165.4	23.9	x<10		16.5	9.8	16.1	46.4		1.9	3.0
Inlet	12-Oct-91	6.4	8.3	235.0	10.4	x<10		17.6		16.3	49.2	7.8	1.1	0.8
Inlet	24-Oct-91	7.7	6.8	116.6		10.3		16.1	9.5	14.0	49.6	7.0		
Inlet	29-Nov-91	8.7	3.1	207.2	128.7	x<10				8.2	47.3	7.5		
Inlet	27-Dec-91	19.6	13.0	290.8	191.2	11.5		17.1	10.8	8.8	45.5	7.4	1.8	1.2
Inlet	24-Jan-92	9.0	4.1	242.0	145.1	x<10		14.3	12.0	7.8	38.7	7.8		1.7
Inlet	25-Feb-92	8.9	3.6	266.4	154.4	x<10		13.8	12.8	11.5	41.2	7.2	1.9	1.4
Inlet	28-Mar-92	38.9	18.1		72.4	x<10	0.6	17.3	11.6	10.0	48.8	7.3	2.2	2.0
Inlet	24-Apr-92	17.0	16.0		47.1	x<10	0.6	15.5	12.0	12.0	44.8	7.3	1.4	2.6
Inlet	19-Jun-92	29.6	19.7					17.1			45.1		1.1	1.0

19-Sep-91

Well No.	pH	Conductivity	Chloride	SO4	TP	SRP	TN	NO3	NH4
		umhos	ug/L	ug/L	ug/L	ug/L	ug/L	ug/L	ug/L
1	6.7	48.3	Not Determined	Not Determined	9.4	7.6	300	100	x<10
2	6.3	337.4	32.9	8.3	15.6	5.4	29300	27100	38.0
3	7.0	70.1	1.4	0.2	97.4	10.2	900	100	128.0
5	6.7	54.8	1.8	0.4	63.9	17.6	200	x<100	x<10
6	9.3	106.0	1.0	2.5	482.3	12.1	Not Determined	Not Determined	Not Determined
7	6.8	Not Determined	1.7	0.0	237.4	Not Determined	x<100	100	296.0
8	6.4	165.0	1.0	1.1	61.7	25.2	900	x<100	15.0
9	6.5	298.0	2.1	0.9	1082.7	1772.8	900	100	156.0
10	6.8	146.0	5.1	0.0	117.3	26.0	Not Determined	Not Determined	Not Determined
11	6.6	64.9	1.7	0.0	220.3	16.3	3900	x<100	122.0
12	6.6	361.0	5.3	2.2	125.6	4.1	600	x<100	610.0
13	6.7	59.3	1.7	0.0	68.2	17.6	100	x<100	38.0
15	6.5	70.1	1.7	0.7	14.8	14.6	100	x<100	122.0
16	6.9	94.7	3.4	1.4	34.1	18.5	2500	2700	x<10
17	6.6	103.3	1.8	0.9	63.5	13.2	100	x<100	15.0
18	6.4	166.0	10.4	0.7	25.1	9.1	500	x<100	10.0
19	6.8	52.0	1.7	1.4	11.5	9.8	500	x<100	83.0
20	6.6	78.5	1.8	1.1	13.3	14.3	100	x<100	x<10

28-Dec-91

Well No.	pH	T	Conductivity	Chloride	SO4	TP	SRP	TN	NO3	NH4
		C	umhos	ug/L	ug/L	ug/L	ug/L	ug/L	ug/L	ug/L
1	7.0	6.5	46.1	1.7	1.9	25.9	3.5	100	x<100	x<10
2	6.4	14.5	376.0	24.8	5.3	41.2	9.0	28800	27500	38.0
3	6.9	9.5	58.9	1.8	1.9	175.9	17.0	100	x<100	83.0
5	6.8	9.5	68.0	2.6	4.3	50.3	7.0	100	x<100	15.0
7	7.0	9.5	91.0	1.9	0.5	340.3	27.8	500	x<100	10.0
8	6.5	13.0	133.0	1.7	4.9	567.9	30.9	4100	4100	1560.0
9	7.0	10.0	62.7	34.5	4.5	4003.5	965.8	1300	x<100	150.0
10	6.9	14.0	162.0	7.2	0.4	370.8	17.0	500	x<100	542.0
11	6.6	15.0	67.9	1.9	0.0	417.5	27.2	100	x<100	38.0
12	6.8	13.5	225.0	3.2	0.0	131.0	4.1	500	x<100	122.0
13	6.8	7.0	52.8	1.9	2.1	220.6	12.2	500	100	95.0
14	6.9	14.0	254.0	4.7	10.8	55.9	2.9	9800	9200	10.0
15	7.1	14.0	65.5	1.8	0.4	69.3	21.6	500	300	10.0
16	7.1	11.0	71.2	1.4	1.4	21.6	19.4	1900	2300	10.0
17	6.8	11.5	126.0	2.1	0.8	138.4	8.1	x<100	x<100	x<10
18	6.9	13.0	134.0	3.8	2.6	52.4	15.3	x<100	100	77.0
19	7.1	7.0	45.0	1.8	2.3	11.1	4.4	500	200	15.0
20	6.9	13.0	82.0	2.2	4.1	38.4	9.8	1600	1700	15.0

28-Mar-92

Well No.	pH	T	Conductivity	Chloride	SO4	TP	SRP	TN	NO3	NH4
		C	umhos	ug/L	ug/L	ug/L	ug/L	ug/L	ug/L	ug/L
1	7.0	12.0	46.0	1.8	1.8	35.2	2.5	100	100	38
2	6.5	14.5	225.0	30.9	7.1	59.0	16.7	23900	10800	27
3	7.2	8.0	62.8	1.8	1.9	151.9	39.1	900	x<100	100
5	7.0	8.5	86.7	3.9	6.5	62.8	9.8	200	x<100	271
6	Not Determined	Not Determined	142.0	Not Determined	Not Determined	518.5	4.4	900	300	10
7	7.0	9.5	145.0	2.4	1.9	385.1	23.5	900	100	251
8	6.5	13.5	134.0	4.6	4.9	497.0	41.6	3900	2200	21
9	6.7	9.0	557.0	22.5	12.7	312.5	126.4	Not Determined	28900	335
10	7.1	14.5	127.0	7.4	1.4	443.9	20.7	200	200	60
11	6.8	12.0	55.9	2.0	0.0	636.0	87.2	500	100	531
12	6.9	14.0	279.0	3.9	1.1	394.9	5.0	300	100	49
13	6.7	10.5	51.1	1.9	0.3	212.4	20.3	2800	100	144
14	Not Determined	Not Determined	171.0	3.0	6.2	138.1	18.4	5800	4100	32
15	6.6	12.5	95.8	2.0	0.2	77.6	11.3	x<100	x<100	21
16	7.1	11.0	86.1	2.6	1.7	55.4	23.1	9200	6200	49
17	6.9	14.0	129.0	2.2	0.6	115.4	16.6	x<100	x<100	38
18	6.9	12.2	109.5	3.7	3.2	77.7	28.1	300	300	27
19	6.6	13.5	38.5	1.6	1.6	21.0	4.8	x<100	x<100	27
20	6.7	13.0	82.0	Not Determined	Not Determined	Not Determined	Not Determined	300	400	x<10
MH1	7.1	13.0	53.2	1.9	1.2	159.8	2.0	200	x<100	217.0
MH2	6.7	17.0	117.0	7.9	5.8	237.0	33.1	8100	5500	105.0

26-Jun-91

Well No.	pH	T	Conductivity	TP	SRP	TN	NO3	NH4
		C	umhos	ug/L	ug/L	ug/L	ug/L	ug/L
1	7.0	21.0	49.0	28.1	3.7	1600	100	10
2	6.3	13.5	310.3	81.5	10.6	26700	26600	116
3	6.8	19.0	73.1	221.7	36.2	900	x<100	133
5	6.7	19.0	64.0	87.7	7.7	900	x<100	x<10
6	6.8	15.0	190.5	318.3	6.5	1600	200	21
7	6.9	17.5	78.3	458.1	156.8	1300	x<100	442
8	6.3	14.5	132.7	311.7	69.3	2400	400	10
9	6.6	15.0	312.4	1508.8	1614.0	5000	x<100	111
10	6.4	16.0	139.1	514	11.4	900	x<100	161
11	6.7	14.0	61.2	138.4	22.6	1600	x<100	660
12	6.6	15.0	412.0	220	11.5	1300	x<100	150
13	6.6	19.0	73.2	193.8	3.1	900	x<100	206
14	6.4	15.0	233.3	54	8.8	2000	700	27
15	6.7	13.5	73.0	93	36.4	2200	x<100	21
16	7.1	11.5	76.7	87.5	49.9	4700	400	21
17	6.7	14.8	157.3	144.2	16.8	1300	x<100	27
18	6.8	14.0	135.9	124.1	21.3	900	x<100	15
19	6.7	21.0	52.0	27.4	24.8	900	x<100	43
20	6.9	17.0	79.9	39.2	5.4	900	x<100	38
MH1	6.7	21.0	52.6	84.5	12.3	900	x<100	234
MH2	6.7	12.5	108.1	20.1	35.9	3500	3200	27

APPENDIX C

Sediment Replicate Data

STATION 1 **Rep. 1**

Depth cm	Water %	Organic Content % *a	Dry Weight g *b	TP ug/g	Pb ug/g	Zn ug/g	Al %	Fe %
1	71.4	9.1	0.9342	929.7	128.9	176.9	2.41	3.43
2	73.3	9.4	1.5765	1098.5	110.8	152.5	1.95	2.80
3	72.6	9.2	1.7469	1098.5	88.0	108.8	1.61	2.24
4	71.3	9.9	1.7786	1015.9	87.1	105.9	1.59	2.26
5	70.4	9.5	1.8494	964.8	95.9	112.1	1.69	2.43
6	70.2	9.1	2.2719	1008.8	92.8	112.5	1.67	2.42
7	69.2	9.4	2.1006	1116.2	116.3	140.7	2.15	3.12
8	69.2	8.6	2.2275	933.8	105.6	125.8	1.89	2.67
9	69.1	8.7	2.2682	1118.1	99.1	121.2	1.82	2.61
10	68.9	9.1	2.3089	972.1	89.9	109.5	1.64	2.34
11	69.5	8.5	2.5323					
12	67.0	9.1	2.3658					
13	66.2	8.9	2.4889					
14	67.1	9.0	2.7076					
15	65.5	8.9	2.6443					
16	65.3	10.8	2.9562					
17	65.0	8.4	2.8198					
18	58.4	6.1	4.2572					
19	70.9	8.3	2.2880					
20	55.6	6.0	4.3538					

* a Percent weight lost on ignition at 550 degrees celcius

* b Dry weight per section (A=9.32 cm^2)

STATION 2 **Rep. 1**

Depth cm	Water %	Organic Content % *a	Dry Weight g *b	TP ug/g	Pb ug/g	Zn ug/g	Al %	Fe %
1	67.48	6.3	2.3781	699.0	37.2	89.4	1.41	2.24
2	63.32	5.9	3.3714	673.1	40.5	89.9	1.64	2.14
3	62.33	5.8	3.3981	671.2	37.0	86.2	1.59	2.20
4	62.18	5.7	3.3837	672.7	40.4	92.4	1.72	2.41
5	62.03	5.9	3.5051	668.4	40.4	89.9	1.62	2.38
6	61.63	5.5	3.4761	647.2	37.0	87.5	1.59	2.31
7	60.66	6.0	3.6611	648.6	45.0	96.2	1.78	2.55
8	58.75	5.7	3.9442	648.3	37.8	89.5	1.63	2.36
9	57.79	5.7	4.2959	613.9	39.2	85.4	1.51	2.21
10	56.77	5.3	4.4052	614.2	36.8	85.9	1.49	2.15
11	55.45	5.5	4.5218					
12	55.53	5.3	4.6901					
13	53.98	5.3	4.8138					
14	55.64	5.7	4.4932					
15	55.35	5.3	4.7132					
16	54.91	5.6	4.6401					
17	54.25	5.4	4.7808					
18	53.11	5.3	5.0165					
19	53.37	5.1	5.2117					

* a Percent weight lost on ignition at 550 degrees celcius

* b Dry weight per section (A=9.32 cm^2)

APPENDIX D

Phytoplankton Taxa and Biovolume

Zooplankton Taxa and Enumeration

Benthic Invertebrate Taxa and Enumeration

Station 1 Surface

Algal Genus	4/5/91	4/26/91	5/10/91	5/24/91	6/7/91	6/21/91	7/5/91	7/19/91	8/2/91	8/16/91	8/30/91	9/14/91	9/27/91	10/12/91	10/24/91	11/29/91	12/27/91	1/24/92	2/25/92	3/28/92	4/26/92	6/19/92
Anabaena		4.04	0.66	1.01	3.90	6.82	6.43	1.34			0.94	0.16	5.62	5.49	2.39	1.10					10.99	0.88
Ankistrodesmus																						
Aphanizomenon									0.08					0.10						0.03	0.55	
Asterionella		0.08									2.24	0.94						0.49		1.27	6.73	
Ceratium																			2.24			
Chlamydomonas							0.01										0.02	0.06		0.01		0.03
Chroococcus											0.01											
Coelosphaerium								1.54		1.80				4.11	18.47				4.36			16.93
Cosmarium																						
Dinobryon		0.02		0.04					0.20									0.35				
Fragillaria	0.33				1.35	0.24	0.20	0.29	2.00	1.67	1.80	0.45	0.20				0.69	3.59	1.02	11.27	1.96	5.43
Gleocystis										0.05	0.10			0.04			0.05		0.19			
Kirchneriella																						
Melosira	10.44	4.09	0.82	0.38	0.72		0.67	0.24		0.72	6.40	2.17	0.63	4.76	2.74	1.83	1.88	0.91	0.38	1.73	2.02	1.40
Oocystis																						
Pediastrum		6.91										4.46										25.86
Scenedesmus					0.08						0.14	0.18							0.27			0.14
Spirogyra											17.35	5.63		13.60								8.91
Stephanodiscus			4.54	2.27								6.82	2.27		2.27	24.99	59.07	2.27	2.27	2.27	13.63	9.09
SFG				0.03																		
Staurastrum															1.63				1.63			
Tetrahedron				0.07																		
Ulothrix	1.74			0.31																		

	4/5/91	4/26/91	5/10/91	5/24/91	6/7/91	6/21/91	7/5/91	7/19/91	8/2/91	8/16/91	8/30/91	9/14/91	9/27/91	10/12/91	10/24/91	11/29/91	12/27/91	1/24/92	2/25/92	3/28/92	4/26/92	6/19/92
Total Volume mm3/L	12.51	15.14	6.02	4.11	6.05	7.06	7.31	3.41	2.28	4.24	28.98	20.81	8.72	28.10	27.50	27.92	61.71	7.67	12.36	16.58	35.88	68.67
Volume Blue-Green	0.00	4.04	0.66	1.01	3.90	6.82	6.43	2.88	0.08	1.80	0.95	0.16	5.62	9.70	20.86	1.10	0.00	0.00	4.36	0.03	11.54	17.81
Percent Blue-Green	..	26.69	10.89	24.53	64.49	96.60	87.96	84.47	3.51	42.45	3.29	0.79	64.43	34.51	75.85	3.95	0.00	0.00	35.28	0.18	32.16	25.94

Station 1 2 m

Algal Genus	4/5/91	4/26/91	5/10/91	5/24/91	6/7/91	6/21/91	7/5/91	7/19/91	8/2/91	8/16/91	8/30/91	9/14/91	9/27/91	10/12/91	10/24/91	11/29/91	12/27/91	1/24/92	2/25/92	3/28/92	4/26/92	6/19/92
Anabaena	0.16	0.65	2.18	0.16	1.63	5.74	12.10	4.58	0.21				5.96	0.74	2.18	5.12			0.58		2.39	0.51
Ankistrodesmus																						
Aphanizomenon																			0.14	0.23	0.13	
Asterionella	0.04	0.12		1.47					0.53		1.06								0.29		5.22	
Ceratium							2.24						2.24									
Chlamydomonas			0.02	0.02					0.01	0.01						0.02	0.02	0.06	0.01		0.01	0.03
Chroococcus										0.03	0.07	0.03										
Coelosphaerium									6.41	13.86												12.32
Cosmarium																						
Dinobryon				0.20														0.63	0.76	0.04		
Fragilaria	0.24			0.45	0.49	0.16	0.41		2.37	3.10	0.29	0.12	0.20			2.78	0.24	1.51	3.27	17.35		0.86
Gleocystis	0.09							0.18	0.12	0.08												0.06
Kirchneriella																						
Melosira	0.34	0.43		0.77	1.06	0.19	0.34		1.78	0.82	1.97	2.02	0.63	3.95	3.61	5.05	0.48	0.67	0.87	2.21	1.15	1.11
Oocystis									0.55													
Pediastrum															5.57							10.48
Scenedesmus					0.10				0.12							0.16						
Spirogyra						4.69																22.51
Stephanodiscus	2.27	9.09	6.82		2.27					9.09			4.54		9.09	22.72	93.15	2.27	2.27	9.09	4.54	
SRG																						
Staurastrum						1.63			4.90	11.43												
Tetrahedron																						
Ulothrix	2.10																					
Total Volume mm3/L	5.24	10.29	9.02	3.07	5.55	12.41	15.09	4.76	17.00	38.42	3.39	2.17	13.57	4.69	20.45	35.85	93.89	5.14	8.19	28.92	13.44	47.88
Volume Blue-Green	0.16	0.65	2.18	0.16	1.63	5.74	12.10	4.58	6.62	13.89	0.07	0.03	5.96	0.74	2.18	5.12	0.00	0.00	0.72	0.23	2.52	12.83
Percent Blue-Green	3.11	6.35	24.13	5.23	29.39	46.27	80.18	96.22	38.94	36.15	2.06	1.38	43.91	15.70	10.66	14.28	0.00	0.00	8.77	0.80	18.76	26.79

Station 1 6 m

Algal Genus	4/5/91	4/26/91	5/10/91	5/24/91	6/7/91	6/21/91	7/5/91	7/19/91	8/2/91	8/16/91	8/30/91	9/14/91	9/27/91	10/12/91	10/24/91	11/29/91	12/27/91	1/24/92	2/25/92	3/28/92	4/26/92	6/19/92
Anabaena		1.26	0.99	2.48	1.54	2.45	6.66	4.77		4.02					1.13	3.49					2.02	0.49
Ankistrodesmus																						
Aphanizomenon																			0.10		0.25	
Asterionella	0.24	0.53				0.04				0.49								0.24	1.31		10.78	
Ceratium						4.49				4.49									2.24			
Chlamydomonas		0.01		0.02												0.02		0.02				0.01
Chroococcus																						
Coelosphaerium	15.01																		4.36			
Cosmarium																					0.04	
Dinobryon	0.16																	1.04	0.31		0.47	
Fragilaria	4.33	0.37		0.24	1.59	0.24	0.20	0.49							0.69	3.55	2.45	4.16	3.39		3.96	1.02
Gleocystis										0.05	0.21											
Kirchneriella																						
Melosira	3.32	3.27	0.48	1.44	0.19	1.11	0.96	0.72		1.11	3.37				3.03	3.22	1.11	0.67	1.11		4.19	4.23
Oocystis																						
Pediastrum	6.46						6.24	10.70			6.46					5.57						
Scenedesmus																						
Spirogyra	3.75					5.63				5.63	2.27					5.63						
Stephanodiscus	20.45			2.27			11.36	4.54								86.34	11.63	4.54	2.27		15.90	18.18
SRG																						
Staurastrum																						
Tetrahedron																						
Ulothrix	1.01																					
Total Volume mm3/L	54.73	5.44	1.47	6.45	3.32	13.96	25.42	21.22	0.00	15.79	12.31	0.00	0.00	0.00	4.85	107.82	15.19	10.67	15.09	0.00	37.61	23.93
Volume Blue-Green	15.01	1.26	0.99	2.48	1.54	2.45	6.66	4.77	0.00	4.02	0.00	0.00	0.00	0.00	1.13	3.49	0.00	0.00	4.46	0.00	2.27	0.49
Percent Blue-Green	27.43	23.12	67.41	38.45	46.31	17.54	26.19	22.47	0.00	25.44	0.00	0.00	0.00	0.00	23.28	3.24	0.00	0.00	29.56	0.00	6.05	2.05

Station 2 Surface

Algal Genus	4/5/91	4/26/91	5/10/91	5/24/91	6/7/91	6/21/91	7/5/91	7/19/91	8/2/91	8/16/91	8/30/91	9/14/91	9/27/91	10/12/91	10/24/91	11/29/91	12/27/91	1/24/92	2/25/92	3/28/92	4/26/92
Anabaena	0.73	1.68	1.29	0.78	3.01	1.04		4.34		0.26		0.66	9.92	1.75	1.94						2.19
Ankistrodesmus		0.04															0.24				
Aphanizomenon																			0.66		3.06
Asterionella	0.12							0.08				1.55	6.73					0.38	0.33	0.54	10.00
Ceratium																					
Chlamydomonas			0.03					0.01	0.02	0.01							0.03	0.03		0.02	
Chroococcus										0.09											
Coelosphaerium								1.28	9.37	1.80		12.32		16.81							
Dinobryon	0.02	0.08		0.04															0.59		
Fragillaria				0.69				0.86	4.98	2.86	0.41	0.73		0.73	0.24	0.02	0.61	3.02	3.43	0.69	3.27
Gleocystis	0.02			0.04					0.23		0.03										0.08
Kirchneriella											0.06	0.03									
Melosira		2.26	0.53	1.83	1.68			0.29	2.94	1.92	2.17	3.51	0.77	3.18	2.12	1.40	0.87	0.63	0.34		4.52
Pediastrum	2.02												13.15	0.22							13.82
Scenedesmus					0.08				0.04	0.16				0.02	0.12						
Spirogyra					7.03							14.54		5.63	2.81						
Stephanodiscus	9.09	4.54		4.54	2.27			9.09			2.27	4.54	2.27	6.82	6.82	15.90	59.06			2.27	9.09
SRG																					
Staurastrum									14.69	1.63											
Ulothrix	1.41																				
Total Volume mm3/L	13.41	8.60	1.85	7.92	14.07	1.04	0.00	15.95	32.27	8.73	4.94	37.88	32.84	35.16	14.05	17.32	60.81	4.06	5.35	3.52	46.03
Volume Blue-Green	0.73	1.68	1.29	0.78	3.01	1.04	0.00	5.62	9.37	2.15	0.00	12.98	9.92	18.56	1.94	0.00	0.00	0.00	0.66	0.00	5.25
Percent Blue-Green	5.42	19.55	69.65	9.90	21.38	100.00	0.00	35.22	29.04	24.66	0.00	34.26	30.21	52.79	13.83	0.00	0.00	0.00	12.34	0.00	11.40

Station 2 Bottom

Algal Genus	4/5/91	4/26/91	5/10/91	5/24/91	6/7/91	6/21/91	7/5/91	7/19/91	8/2/91	8/16/91	8/30/91	9/14/91	9/27/91	10/12/91	10/24/91	11/29/91	12/27/91	1/24/92	2/25/92	3/28/92
Anabaena	0.64	0.95		1.68	3.31	15.87	12.30	2.34	0.34	0.18		0.89	4.38	5.49	2.15		0.81	0.68		
Ankistrodesmus		0.05																		
Aphanizomenon																		0.26	0.12	0.15
Asterionella	0.04			0.08		0.16					0.78	0.94		0.24	0.24		0.61			2.16
Ceratium						2.24			2.24				2.24							
Chlamydomonas				0.01				0.01	0.02	0.01		0.01					0.02	0.03		
Chroococcus										0.06										
Coelosphaerium								2.18	9.11			9.37								
Cosmarium							0.04													
Dinobryon				0.16															0.04	0.20
Fragillaria	2.69			0.45	1.92			0.37	3.67	2.41	1.18	0.78		1.67	1.43	0.57		1.14		3.06
Gleocystis			0.03						0.16	0.06	0.36	0.11			0.04					
Kirchneriella																				
Melosira	1.97	1.01	0.24	1.44	0.43	1.59	2.69	0.48	3.75	2.17	3.22	7.60	2.26	5.10	4.91	2.26	1.68	1.73	1.83	3.22
Oocystis					0.16															
Pediastrum				6.46			6.46				6.02	6.24								
Scenedesmus					0.16	0.08	0.08		0.04	0.53		0.10								
Spirogyra	6.10			3.75							6.10	12.19			3.28					
Stephanodiscus	4.54	2.27	11.36	2.27	4.54	2.27	9.09		6.82	2.27		4.54	9.09	4.54	13.63	43.17	31.81	9.09	2.27	11.36
SRG																				
Staurastrum	2.35								2.45	0.96										
Total Volume mm3/L	18.33	4.28	11.63	16.30	10.52	22.21	30.66	5.38	28.60	8.65	17.66	42.77	17.97	17.04	25.68	46.00	34.93	12.93	4.26	20.15
Volume Blue-Green	0.64	0.95	0.00	1.68	3.31	15.87	12.30	4.52	9.45	0.24	0.00	10.26	4.38	5.49	2.15	0.00	0.81	0.94	0.12	0.15
Percent Blue-Green	3.52	22.28	0.00	10.30	31.48	71.46	40.12	84.00	33.05	2.82	0.00	23.99	24.39	32.20	8.38	0.00	2.33	7.28	2.82	0.74

Station 3 Surface

Algal Genus	4/5/91	4/28/91	5/10/91	5/24/91	6/7/91	6/21/91	7/5/91	7/19/91	8/2/91	8/16/91	8/30/91	9/14/91	9/27/91	10/12/91	10/24/91	11/29/91	12/27/91	1/24/92	2/25/92	3/28/92	4/26/92
Anabaena		0.72	1.78	3.20	5.34	7.68	14.02	2.16	3.38			1.39	5.38		2.15	1.92	1.63	4.96			0.47
Ankistrodesmus													0.03								
Aphanizomenon																0.08		0.30			
Asterionella											0.20	0.37	0.12								7.39
Ceratium							6.73														
Chlamydomonas				0.02							0.01				0.02	0.01	0.03	0.06	0.02		
Chroococcus										0.06											
Coelosphaerium									10.90	1.28		1.54	14.11								
Cosmarium																					
Dinobryon																		0.37	3.10		
Fragillaria						0.24	1.18	0.57	8.29	1.39	0.29	2.24	1.51			0.73	1.71	3.22			2.73
Gleocystis							0.12		0.57	0.06	0.06	0.06									
Kirchneriella										0.09	0.04										
Melosira	2.12	1.01	0.77	1.11	0.24	1.15	1.20	3.46	1.97	1.30	2.21	5.15	3.08			6.35	0.72	1.01	0.77		1.88
Oocystis							0.37														
Pediastrum									7.13			10.25									
Scenedesmus							0.29			0.61		0.16									
Spirogyra					5.16					2.81						7.03					
Stephanodiscus						2.27	15.90	6.82	4.54			2.27	6.82		6.82	77.25	63.62	6.82			15.90
SRG																					
Staurastrum									4.90	3.27											
Ulothrix	1.79																				
Total Volume mm3/L	3.91	1.73	2.55	4.33	10.74	11.34	39.81	13.01	41.68	10.87	2.81	23.43	31.05	0.00	8.99	93.37	67.71	16.74	3.89	0.00	28.37
Volume Blue-Green	0.00	0.72	1.78	3.20	5.34	7.68	14.02	2.16	14.28	1.34	0.00	2.93	19.49	0.00	2.15	2.00	1.63	5.26	0.00	0.00	0.47
Percent Blue-Green	0.00	41.56	69.75	73.90	49.74	67.73	35.21	16.60	34.27	12.33	0.00	12.51	62.77	0.00	23.93	2.14	2.40	31.43	0.00	0.00	1.67

Station 3 Bottom

Algal Genus	4/5/91	4/26/91	5/10/91	5/24/91	6/7/91	6/21/91	7/5/91	7/19/91	8/2/91	8/16/91	8/30/91	9/14/91	9/27/91	10/12/91	10/24/91	11/29/91	12/27/91	1/24/92	2/25/92	3/28/92	4/26/92
Anabaena		1.48		0.83	3.33	6.26	14.02	3.44					3.99	1.81	2.20	1.65	4.38	2.34			1.15
Ankistrodesmus																	0.19				
Aphanizomenon						0.09			0.09							0.03					0.89
Asterionella	0.08			0.73	0.12						0.49	1.14									
Ceratium							6.73				2.24										
Chlamydomonas				0.01		0.01			0.02									0.06			
Chroococcus											0.01										
Coelosphaerium												1.54	2.18	16.81							
Cosmarium														0.04							
Dinobryon									0.78									1.04			
Fragilaria	1.88		0.86	1.06	1.76	0.49	1.18	0.9		2.2		0.49	0.2			0.24	1.76	6.08	1.35		2.20
Gleocystis							0.12			0.18	0.12	0.12							0.08		
Kirchneriella								0.13			0.02										
Melosira	2.69	2.65	2.17	0.63	0.58		1.2	2.21	1.68	5.1	7.03	4.38	4.19	2.89	2.6	2.84			1.11		2.17
Oocystis							0.37														
Pediastrum											18.5	46.36	12.48	0.22							
Scenedesmus	0.24			0.08			0.29	0.24		0.2			0.08	0.02							
Spirogyra				8.91				9.85		7.5		4.69		5.63							6.57
Stephanodiscus	6.82	11.36		2.27	4.54		6.82	6.82		6.82	2.27	2.27	4.54	9.09		65.89	40.9	6.82			2.27
SRG				0.03																	
Staurastrum						1.63				8.98											
Ulothrix	1.67																				
Total Volume mm3/L	13.38	15.49	3.03	14.55	10.33	8.48	30.73	23.59	2.57	30.98	30.68	60.99	27.66	36.51	4.80	70.65	47.23	16.34	2.54	0.00	15.25
Volume Blue-Green	0.00	1.48	0.00	0.83	3.33	6.35	14.02	3.44	0.09	0.00	0.01	1.54	6.17	18.62	2.20	1.68	4.38	2.34	0.00	0.00	2.04
Percent Blue-Green	0.00	9.54	0.00	5.72	32.22	74.87	45.62	14.58	3.50	0.00	0.03	2.53	22.31	51.00	45.88	2.38	9.28	14.30	0.00	0.00	13.39

HORSESHOE LAKE ZOOPLANKTON

		4/5/91	4/26/91	5/10/91	5/24/91	6/10/91	6/21/91	7/5/91	7/19/91	8/2/91	8/19/91	8/30/91	9/2/91	9/14/91	10/12/91	10/24/91
		No./L	No./L	No./L	No./L	No./L	No./L	No./L	No./L	No./L	No./L	No./L	No./L	No./L	No./L	No./L
Cladocera																
	Daphnia	18.5	27.2	78.8	55.4	37.4	22.1	21.0	27.5	21.0	8.7	9.6	5.1	10.7	6.3	9.7
	Bosmina	0.1	1.1	0.1	0.5	0.1	0.3	0.6	0.6	0.3	0.0	0.2	0.0	0.3	0.1	0.1
	Ceriodaphnia	0.4	16.3	19.2	9.6	7.9	2.9	2.9	5.4	5.3	8.5	11.5	2.7	6.7	2.1	2.2
	Eurycercus	0.8	2.0	3.2	.1.0	0.1	0.1	0.2	0.0	0.0	0.0	0.1	0.0	0.0	0.2	0.0
	Chydorus	0.0	0.3	0.0	0.0	0.1	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	other	0.0	0.0	0.0	0.0	0.1	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Copepoda																
	Calanoid	3.1	4.0	6.1	3.2	2.3	1.4	1.0	1.4	1.2	9.3	5.4	10.9	10.0	11.6	14.7
	Cyclopoid	66.2	44.1	45.3	50.7	52.0	29.6	25.4	31.2	15.3	32.0	34.9	11.3	18.7	6.9	3.2
	Nauplia	10.4	22.0	9.1	16.0	9.9	2.4	1.8	3.2	9.7	44.5	2.2	4.7	11.2	0.6	0.1
Others																
	Ostracod	0.1	1.4	0.5	0.1	0.2	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.1	0.1

		11/29/91	12/27/91	1/23/92	2/25/92	3/28/92	4/25/92
		No./L	No./L	No./L	No./L	No./L	No./L
Cladocera							
	Daphnia	8.3	6.5	9.7	21.8	24.0	8.8
	Bosmina	0.6	0.6	1.1	0.4	0.4	0.0
	Ceriodaphnia	1.4	0.0	0.2	0.4	0.0	0.4
	Eurycercus	0.0	0.0	0.0	0.0	0.0	0.1
	Chydorus	0.0	0.0	0.0	0.0	0.0	0.0
	other	0.0	3.5	0.0	0.0	0.0	0.0
Copepoda							
	Calanoid	11.0	6.5	5.4	3.0	1.0	0.5
	Cyclopoid	5.1	0.3	14.0	21.4	20.8	14.5
	Nauplia	0.1	0.0	15.9	0.1	0.1	5.3
Others							
	Ostracod	0.1	0.0	0.0	0.0	0.0	0.0

Horseshoe Lake Benthic Invertebrates

Sampling Date 9/25/91

Station 1

	Replicate 1	Replicate 2
Chironomidae	3	2
Oligochaeta	107	58

Station 2

	Replicate 1	Replicate 2
Ceratopogonidae	1	
Chaoboridae		1
Chironomidae	12	10
Oligochaeta	107	97

Station 3

	Replicate 1	Replicate 2
Chironomidae	55	32
Ephemeridae	1	1
Oligochaeta	30	17

Sampling Date 3/28/92

Station 1

	Replicate 1	Replicate 2
Chironomidae	7	7
Oligochaeta	56	29

Station 2

	Replicate 1	Replicate 2
Chironomidae	4	10
Elmidae		1
Oligochaeta	16	4
Sphaeriidae		3

Station 3

	Replicate 1	Replicate 2
Chironomidae	5	3
Ephemeridae	2	2
Oligochaeta		10
Sericostomatidae		2
Sphaeriidae	1	

APPENDIX E

Lake Ionic Concentrations

HORSESHOE LAKE IONS

		Cl	SO4	Ca	Mg	K	Na	Fe	Al	Mn
		mg/L	mg/L	ug/L	ug/L	ug/L	ug/L	ug/L	ug/L	ug/L
Quarter 1	Surface	1.6	1.5	3480.0	980.0	1250.0	2880.0	130.0	110.0	5.0
	2 meter	1.5	1.4	3650.0	1050.0	800.0	2940.0	120.0	100.0	8.0
	4 meter	1.2	1.2	4640.0	1120.0	550.0	3110.0	150.0	140.0	8.0
	6 meter	1.6	1.5	3650.0	950.0	620.0	2660.0	130.0	120.0	8.0
	Mean	**1.5**	**1.4**	**3855.0**	**1025.0**	**805.0**	**2897.5**	**132.5**	**117.5**	**7.3**
Quarter 2	Surface	1.8	1.8	2400.0	550.0	500.0	1490.0	70.0	70.0	5.0
	2 meter	1.8	1.9	1990.0	500.0	410.0	1390.0	80.0	60.0	5.0
	4 meter	1.7	1.8	1030.0	230.0	150.0	590.0	60.0	50.0	4.0
	6 meter	1.7	1.9	3390.0	970.0	530.0	2710.0	130.0	100.0	9.0
	Mean	**1.8**	**1.8**	**2202.5**	**562.5**	**397.5**	**1545.0**	**85.0**	**70.0**	**5.8**
Quarter 3	Surface	1.9	1.9	3080.0	780.0	680.0	2070.0	340.0	280.0	9.0
	2 meter	1.4	1.6	2820.0	940.0	344.0	2690.0	170.0	140.0	9.0
	4 meter	1.7	2.0	3460.0	970.0	421.0	2750.0	120.0	110.0	9.0
	6 meter	1.3	1.5	820.0	230.0	n.d.	450.0	67.0	70.0	3.0
	Mean	**1.6**	**1.7**	**2545.0**	**730.0**	**481.7**	**1990.0**	**174.3**	**150.0**	**7.5**
Quarter 4	Surface	1.8	1.9	3620.0	1030.0	320.0	2830.0	100.0	90.0	8.0
	2 meter	1.7	1.8	1860.0	440.0	30.0	1170.0	57.0	60.0	4.0
	4 meter	1.7	1.8	3550.0	1010.0	480.0	2850.0	120.0	100.0	9.0
	6 meter	1.7	1.8	1480.0	450.0	230.0	1110.0	56.0	70.0	5.0
	Mean	**1.7**	**1.8**	**2627.5**	**732.5**	**265.0**	**1990.0**	**83.3**	**80.0**	**6.5**

APPENDIX F

P Model Input Output

P model input output

Week Ending	External Loading Kg/wk	Model Flushing Kg/wk	Model Sedimentation Kg/wk	Model Internal Loading Kg/wk	Model TP Predicted ug/L Observed	Lake TP Observed ug/L Predicted
6-Apr-91	12.0	4.4	5.9	0.0	31.3	31.8
13-Apr-91	7.2	3.4	4.8	0.0	29.1	31.8
20-Apr-91	4.5	2.6	3.7	0.0	26.5	23.0
27-Apr-91	2.6	2.4	3.5	0.0	22.8	23.0
4-May-91	2.3	2.0	2.9	0.0	19.8	19.5
11-May-91	3.9	2.1	3.0	0.0	18.0	19.5
18-May-91	1.8	1.6	2.3	5.9	20.4	21.3
25-May-91	2.2	1.9	2.7	5.9	22.5	21.3
1-Jun-91	1.9	2.0	2.9	5.9	24.0	23.8
8-Jun-91	1.9	2.1	3.0	5.9	25.4	23.8
15-Jun-91	2.4	2.4	3.4	5.9	26.5	29.0
22-Jun-91	3.8	2.7	3.8	5.9	28.1	29.0
29-Jun-91	2.2	2.4	3.5	5.9	28.8	37.3
6-Jul-91	2.8	2.5	3.6	5.9	29.9	37.3
13-Jul-91	2.8	2.6	3.8	5.9	30.8	29.7
20-Jul-91	2.2	2.7	3.9	5.9	30.9	29.7
27-Jul-91	2.2	2.7	3.9	5.9	31.1	30.3
3-Aug-91	4.0	2.8	4.0	5.9	32.5	30.3
10-Aug-91	3.8	2.9	4.1	5.9	33.5	32.7
17-Aug-91	3.1	2.9	4.2	5.9	33.8	32.7
24-Aug-91	3.1	2.9	4.2	5.9	34.1	28.7
31-Aug-91	2.6	3.2	4.6	5.9	33.2	28.7
7-Sep-91	1.8	2.9	4.2	5.9	32.6	31.2
14-Sep-91	1.4	2.8	4.1	5.9	31.7	31.2
21-Sep-91	1.4	2.8	4.0	5.9	31.0	26.8
28-Sep-91	1.2	2.7	3.9	5.9	30.4	26.8
5-Oct-91	1.2	2.6	3.8	5.9	29.8	31.3
12-Oct-91	0.7	2.6	3.7	5.9	28.9	31.3
19-Oct-91	0.7	2.5	3.7	5.9	28.2	29.6
26-Oct-91	3.2	3.0	4.2	5.9	28.5	29.6
2-Nov-91	1.1	2.6	3.7	5.9	28.1	29.6
9-Nov-91	4.4	3.2	4.5	5.9	29.0	29.6
16-Nov-91	1.5	2.8	3.9	5.9	28.4	29.6
23-Nov-91	6.6	3.7	5.0	0.0	25.3	21.4
30-Nov-91	6.5	3.2	4.4	0.0	23.1	21.4
7-Dec-91	3.3	2.5	3.5	0.0	19.9	21.4
14-Dec-91	1.5	1.9	2.7	0.0	16.6	21.4
21-Dec-91	3.1	1.8	2.5	0.0	15.0	13.5
28-Dec-91	2.1	1.3	1.9	0.0	13.5	13.5
4-Jan-92	2.7	1.3	1.8	0.0	12.6	13.5
11-Jan-92	2.8	1.2	1.7	0.0	12.0	13.5
18-Jan-92	2.6	1.1	1.6	2.0	13.1	13.5
25-Jan-92	1.4	1.2	1.7	2.0	13.0	14.3
1-Feb-92	6.1	1.6	2.1	2.0	15.9	14.3
8-Feb-92	1.1	1.4	2.0	2.0	15.1	14.3
15-Feb-92	1.3	1.4	2.0	2.0	14.5	14.3
22-Feb-92	4.6	1.7	2.3	2.0	15.9	14.3
29-Feb-92	1.6	1.5	2.1	4.0	16.9	16.8
7-Mar-92	1.5	1.6	2.3	4.0	17.6	16.8
14-Mar-92	0.9	1.5	2.2	4.0	17.9	16.8
21-Mar-92	1.5	1.7	2.4	4.0	18.4	16.8
28-Mar-92	4.1	1.6	2.3	4.0	21.2	25.4
4-Apr-92	4.6	2.0	2.8	4.0	23.5	25.4
Yearly Total Kg/yr	154.0	123.0	174.8	195.3		

APPENDIX G

KCM Cost Estimates

Memorandum

Date: 10/28/92

To: Eugene Welch, University of Washington

From: Harry Gibbons, Jeff Lykken

Subject: Horseshoe Lake / 22911-01
Treatment Alternative Cost Estimates

INTRODUCTION

In October 1992, the Department of Environmental Engineering & Science of the University of Washington authorized Kramer, Chin & Mayo, Inc. to provide preliminary cost estimates for 10 alternatives intended to enhance the water quality of Horseshoe Lake in Woodland, Washington. The alternatives consist of increasing the circulation of water within the lake and alum treatment during the spring and summer.

Circulation is currently accomplished by a pumping system which conveys approximately 3000 gallons per minute from the Lewis River to the inlet of the lake. The system is comprised of a 16" vertical propellor pump with a 30 horsepower motor and an 18-inch diameter steel forcemain that runs under the Woodland Airport and Interstate 5 approximately 600 feet to Horseshoe Lake. The forcemain terminates approximately 10 feet from the lakeshore.

The ten alternatives for which cost estimates were prepared are as follows:

- **Alternative 1A:** Increase pumping rate to 4500 gallons per minute.
- **Alternative 1B:** Increase pumping rate to 5250 gallons per minute.
- **Alternative 1C:** Increase pumping rate to 6000 gallons per minute.
- **Alternative 2A:** Increase pumping rate to 4500 gallons per minute.
  Add diffuser pipe along lake bottom.
- **Alternative 2B:** Increase pumping rate to 5250 gallons per minute.
  Add diffuser pipe along lake bottom.
- **Alternative 2C:** Increase pumping rate to 6000 gallons per minute.
  Add diffuser pipe along lake bottom.
- **Alternative 3A:** Increase pumping rate to 4500 gallons per minute.
  Add diffuser pipe along lake bottom.
  Spring and summer alum injection.

- **Alternative 3B:** Increase pumping rate to 5250 gallons per minute.
  Add diffuser pipe along lake bottom.
  Spring and summer alum injection.

- **Alternative 3C:** Increase pumping rate to 6000 gallons per minute.
  Add diffuser pipe along lake bottom to the north end of the lake.
  Spring and summer alum injection.

- **Alternative 4:** Alum treatment with sodium carbonate or sodium aluminate as buffer.

The estimates for Alternatives 1A, 1B and 1C are simply replacement costs for the existing pump and motor to increase the pumping rate to the lake as noted for each alternative.

Alternatives 2A, 2B and 2C include replacement costs for the pump and motor as well as the addition of a 2000 foot 24-inch diameter high-density polyethylene diffuser pipe in Horseshoe Lake.

Alternatives 3A, 3B and 3C include the same components as Alternative 2A-2C with the addition of alum injection to the lake during the spring and summer. Alum injection for each alternative is proportional to the proposed pumping rate. Alum quantities are based on an injection period of 6 months.

One assumption common to the cost estimates for each of the pumping alternatives (Alternatives 1A-3C) is the existing 600 foot 18-inch diameter force main will not need to be replaced. The cost of replacing the pipe under Interstate 5 and the airport would be quite expensive since it would probably have to be installed by jacking and boring rather than the conventional open cut trench method. We estimate the construction cost for replacing the forcemain would be an additional $200,000.

CONSTRUCTION COST AND TOTAL PROJECT COST ESTIMATES

Construction cost and total project cost estimates for the ten alternatives are presented in Table 1. Unit construction costs for the alternatives are based on comparable work prices obtained from available sources. Manufacturers, suppliers of materials and equipment, and local contractors were sources of information for specific questions. Cost data for pipelines and pump installation were derived from recent project experience.

Unit prices were developed for the construction components in each alternative based on current October, 1992 prices (Seattle ENR–5335). Total project costs were estimated by multiplying construction costs by a factor of 2.0. This factor accounts for a 35 percent construction contingency, engineering and design fees of 35 percent, a sales tax of 8.2 percent, construction administration fees of 10 percent, and 10 percent for inflation.

TABLE 1.
ESTIMATED CONSTRUCTION AND TOTAL PROJECT COSTS

Alternative	Description	Estimated Construction Cost	Estimated Total Project Cost[a,b]
1A	Increase pumping rate to 4500 gpm	$40,000	$80,000
1B	Increase pumping rate to 5250 gpm	$42,000	$84,000
1C	Increase pumping rate to 6000 gpm	$49,000	$98,000
2A	Increase pumping rate to 4500 gpm Add diffuser pipe along lake bottom	$285,000	$570,000
2B	Increase pumping rate to 5250 gpm Add diffuser pipe along lake bottom	$303,000	$606,000
2C	Increase pumping rate to 6000 gpm Add diffuser pipe along lake bottom	$307,000	$614,000
3A	Increase pumping rate to 4500 gpm Add diffuser pipe along lake bottom Spring and summer alum injection	$478,000	$956,000
3B	Increase pumping rate to 5250 gpm Add diffuser pipe along lake bottom Spring and summer alum injection	$509,000	$1,018,000
3C	Increase pumping rate to 6000 gpm Add diffuser pipe along lake bottom Spring and summer alum injection	$527,000	$1,054,000
4	Alum treatment with sodium carbonate or sodium aluminate as a buffer	$69,000	$138,000

a. Total project costs include construction contingencies, design fees, construction administration, sales tax and inflation; and were estimated by multiplying construction costs by a factor of 2.0.

b. Monitoring not included in estimated costs.

Additional items that could increase costs include replacement of the existing forcemain from the river to the lake and structural improvements to the existing pier at the Lewis River where the pump is mounted. No additional costs or contingencies have been included for these items.

Unit construction costs for each alternative are included in the Appendix.

HORSESHOE LAKE
ALTERNATE 1A CONSTRUCTION COST
INCREASE PUMP RATE TO 4500 GPM

Level of Est.: PRELIMINARY | DATE 10/27/92
Location: WOODLAND, WASHINGTON | PROJECT NO. 22911

	Description	Quant.	Unit	Unit Price	Total
1	MOBILIZATION @10%	LS	LS	$3,600	$3,600
2	PUMP AND MOTOR ASSEMBLY	1	EA	$30,000	$30,000
3	MISCELLANEOUS VALVES, FITTINGS AND PIPING	LS	LS	$6,000	$6,000
	TOTAL				$40,000

Prepared by KCM Consulting Engineers
Seattle ENR: 5335 - October, 1992

HORSESHOE LAKE
ALTERNATE 1B CONSTRUCTION COST
INCREASE PUMP RATE TO 5250 GPM

Level of Est.: PRELIMINARY | DATE 10/27/92
Location: WOODLAND, WASHINGTON | PROJECT NO. 22911

	Description	Quant.	Unit	Unit Price	Total
1	MOBILIZATION @10%	LS	LS	$3,800	$3,800
2	PUMP AND MOTOR ASSEMBLY	1	EA	$32,000	$32,000
3	MISCELLANEOUS VALVES, FITTINGS AND PIPING	LS	LS	$6,000	$6,000
	TOTAL				$42,000

Prepared by KCM Consulting Engineers
Seattle ENR: 5335 - October, 1992

HORSESHOE LAKE
ALTERNATE 1C CONSTRUCTION COST
INCREASE PUMP RATE TO 6000 GPM

Level of Est.: PRELIMINARY | DATE 10/27/92
Location: WOODLAND, WASHINGTON | PROJECT NO. 22911

	Description	Quant.	Unit	Unit Price	Total
1	MOBILIZATION @10%	LS	LS	$5,200	$5,200
	PUMP AND MOTOR ASSEMBLY	1	EA	$44,000	$44,000
	MISCELLANEOUS VALVES, FITTINGS AND PIPING	LS	LS	$8,000	$8,000
	TOTAL				$49,000

Prepared by KCM Consulting Engineers
Seattle ENR: 5335 - October, 1992

HORSESHOE LAKE
ALTERNATE 2A CONSTRUCTION COST
INCREASE PUMP RATE TO 4000 GPM AND ADD DIFFUSER

Level of Est.: PRELIMINARY | DATE 10/28/92
Location: WOODLAND, WASHINGTON | PROJECT NO. 22911

	Description	Quant.	Unit	Unit Price	Total
1	MOBILIZATION @10%	LS	LS	$25,950	$25,950
2	PUMP AND MOTOR ASSEMBLY	1	EA	$30,000	$30,000
3	MISCELLANEOUS VALVES, FITTINGS AND PIPING	LS	LS	$8,000	$8,000
4	DIFFUSER PIPE-24-IN DIA HDPE	2000	LF	$100	$200,000
5	54-INCH DIAMETER MANHOLE	1	EA	$3,500	$3,500
6	HDPE PIPE ANCHORS	100	EA	$200	$20,000
	TOTAL				$285,000

HORSESHOE LAKE
ALTERNATE 2B CONSTRUCTION COST
INCREASE PUMP RATE TO 5250 GPM AND ADD DIFFUSER

Level of Est.: PRELIMINARY | DATE 10/28/92
Location: WOODLAND, WASHINGTON | PROJECT NO. 22911

	Description	Quant.	Unit	Unit Price	Total
1	MOBILIZATION @10%	LS	LS	$27,550	$27,550
2	PUMP AND MOTOR ASSEMBLY	1	EA	$44,000	$44,000
3	MISCELLANEOUS VALVES, FITTINGS AND PIPING	LS	LS	$8,000	$8,000
4	DIFFUSER PIPE-24-IN DIA HDPE	2000	LF	$100	$200,000
5	54-INCH DIAMETER MANHOLE	1	EA	$3,500	$3,500
6	HDPE PIPE ANCHORS	100	EA	$200	$20,000
	TOTAL				$303,000

HORSESHOE LAKE
ALTERNATE 2C CONSTRUCTION COST
INCREASE PUMP RATE TO 6000 GPM AND ADD DIFFUSER

Level of Est.: PRELIMINARY | DATE 10/28/92
Location: WOODLAND, WASHINGTON | PROJECT NO. 22911

	Description	Quant.	Unit	Unit Price	Total
1	MOBILIZATION @10%	LS	LS	$27,950	$27,950
2	PUMP AND MOTOR ASSEMBLY	1	EA	$48,000	$48,000
3	MISCELLANEOUS VALVES, FITTINGS AND PIPING	LS	LS	$8,000	$8,000
4	DIFFUSER PIPE-24-IN DIA HDPE	2000	LF	$100	$200,000
5	54-INCH DIAMETER MANHOLE	1	EA	$3,500	$3,500
6	HDPE PIPE ANCHORS	100	EA	$200	$20,000
	TOTAL				$307,000

Prepared by KCM Consulting Engineers
Seattle ENR: 5335 - October, 1992

HORSESHOE LAKE
ALTERNATE 3A CONSTRUCTION COST
INCREASE PUMP RATE TO 4000 GPM
ADD DIFFUSER PIPE
WITH SPRING AND SUMMER ALUM INJECTION

Level of Est.: PRELIMINARY
Location: WOODLAND, WASHINGTON

DATE 10/28/92
PROJECT NO. 22911

	Description	Quant.	Unit	Unit Price	Total
1	MOBILIZATION @10%	LS	LS	$43,450	$43,450
2	PUMP AND MOTOR ASSEMBLY	1	EA	$30,000	$30,000
3	MISCELLANEOUS VALVES, FITTINGS AND PIPING	LS	LS	$6,000	$6,000
4	DIFFUSER PIPE-24-IN DIA HDPE	2000	LF	$100	$200,000
5	54-INCH DIAMETER MANHOLE	1	EA	$3,500	$3,500
6	HDPE PIPE ANCHORS	100	EA	$200	$20,000
7	ALUM	300	TON	$250	$75,000
8	ALUM INJECTION	LS	LS	$100,000	$100,000
	TOTAL				$478,000

HORSESHOE LAKE
ALTERNATE 3B CONSTRUCTION COST
INCREASE PUMP RATE TO 5250 GPM
ADD DIFFUSER PIPE
WITH SPRING AND SUMMER ALUM INJECTION

Level of Est.: PRELIMINARY
Location: WOODLAND, WASHINGTON

DATE 10/28/92
PROJECT NO. 22911

	Description	Quant.	Unit	Unit Price	Total
1	MOBILIZATION @10%	LS	LS	$46,300	$46,300
2	PUMP AND MOTOR ASSEMBLY	1	EA	$44,000	$44,000
3	MISCELLANEOUS VALVES, FITTINGS AND PIPING	LS	LS	$8,000	$8,000
4	DIFFUSER PIPE-24-IN DIA HDPE	2000	LF	$100	$200,000
5	54-INCH DIAMETER MANHOLE	1	EA	$3,500	$3,500
6	HDPE PIPE ANCHORS	100	EA	$200	$20,000
7	ALUM	350	TON	$250	$87,500
8	ALUM INJECTION	LS	LS	$100,000	$100,000
	TOTAL				$509,000

HORSESHOE LAKE
ALTERNATE 3C CONSTRUCTION COST
INCREASE PUMP RATE TO 6000 GPM
ADD DIFFUSER PIPE
WITH SPRING AND SUMMER ALUM INJECTION

Level of Est.: PRELIMINARY
Location: WOODLAND, WASHINGTON

DATE 10/28/92
PROJECT NO. 22911

	Description	Quant.	Unit	Unit Price	Total
1	MOBILIZATION @10%	LS	LS	$47,950	$47,950
2	PUMP AND MOTOR ASSEMBLY	1	EA	$48,000	$48,000
3	MISCELLANEOUS VALVES, FITTINGS AND PIPING	LS	LS	$8,000	$8,000
4	DIFFUSER PIPE-24-IN DIA HDPE	2000	LF	$100	$200,000
5	54-INCH DIAMETER MANHOLE	1	EA	$3,500	$3,500
6	HDPE PIPE ANCHORS	100	EA	$200	$20,000
7	ALUM	400	TON	$250	$100,000
8	ALUM INJECTION	LS	LS	$100,000	$100,000
	TOTAL				$527,000

Prepared by KCM Consulting Engineers

HORSESHOE LAKE
ALTERNATE 4 CONSTRUCTION COST
ALUM TREATMENT WITH SODIUM ALUMINATE BUFFER

Level of Est.: PRELIMINARY
Location: WOODLAND, WASHINGTON

DATE 10/27/92
PROJECT NO. 22911

	Description	Quant.	Unit	Unit Price	Total
1	MOBILIZATION @10%	LS	LS	$6,250	$6,250
2	ALUM	82	TON	$250	$20,500
3	SODIUM ALUMINATE	35	TON	$1,200	$42,000
	TOTAL				$69,000

Prepared by KCM Consulting Engineers
Seattle ENR: 5335 - October, 1992