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Shawnigan Lake Water Quality Study

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SUMMARY

This study was undertaken at the request of the Cowichan Valley Regional District (C.V.R.D.) in response to general concerns with deterioration in water quality and effects of increased housing in the watershed on water quality. The C.V.R.D. wanted to use the information to assist with decisions on future planning and zoning within the watershed.

The project required evaluation of two related ecosystems: the aquatic and terrestrial; and this report outlines the result of the investigations of the aquatic environment - the lake and inflow streams. A separate report supplies information on soils, and transfer of nutrients to the lake.

Data were collected from 1977 to 1979 for a variety of parameters of water chemistry and biology at two deep water stations (main basin and south basin), as well as several shallow water stations, and the inflow streams.

Generally it was noted from historical data that minor changes in zooplankton species composition have occurred over the past 40 years, and a significant change in phytoplankton (recorded in the lake sediment) did occur approximately 75 years ago, probably as a consequence of logging and initial settlement in the watershed. The logging and sawmilling activities may also have contributed to an abnormally high hypolimnetic oxygen depletion due to deposition of wood waste on the lake bottom.

Monitoring of inflow streams indicated that the amount of nutrients and other materials generated from different watersheds was apparently proportional to the amount and density of development. The runoff water from the Village area was very poor and significantly different from less disturbed watersheds such as Shawnigan Creek inflow.

One major goal was to document the level of nutrients, their major sources, and the physical processes affecting nutrient distribution. The results indicate that the present concentrations of nutrients and algal

growth are relatively low and the general water quality is good in terms of recreation and water supply (there are a large number of private intakes on the lake in addition to the waterworks). The factors which contribute to this relatively good water quality are:

(1) favourable lake water residence time (one year) which tends to "flush" a portion of nutrients from the lake.
(2) a favourable hydrologic regime - common to many coastal lakes, such that the nutrients from the watershed are supplied to the lake in the autumn in a period when no algal growth is stimulated and much of the nutrients are either lost through sedimentation or outflow.
(3) localized nutrient loading: the highest loadings (on a mass/area basis) occur from the Village area and these nutrients affect a limited area and, being close to the outlet, are partially removed from the lake system.
(4) low nutrient loading: the key nutrient controlling algal growth (phosphorus) is in very low supply because the soils in the watershed have a very high affinity for phosphorus, and bind much of what originates from natural and man made sources. Monitoring of water quality since 1979 indicated that no significant change has occurred in the nutrient concentration of the open water areas since that time.

Another major goal was to investigate the level of contamination by coliform bacteria in the lake. A significant result was elevated fecal coliform concentrations in shallow water areas. The levels of contamination were well within standards for body contact recreation but since there are numerous intakes licenced for domestic water supply the possibility of contaminated drinking water exists. Although surface water for domestic supply must be treated to destroy potential pathogens (by chlorination for instance) many supplies to single family dwellings may not have any treatment capability. No direct evidence as to the source of this contamination was found, but poorly located or poorly maintained older septic tank systems are a likely source. Because of the fluctuating lake levels and high winter water levels, some tile field systems may not provide adequate capability for filtering bacteria and removing nutrients. Other sources of nutrients and suspended sediments would also include road building and clearing and ditching for drainage.

The following provisional water quality objectives are proposed to protect water uses (bulk water supply, water based recreation and fisheries) and prevent eutrophication:

Phosphorus: total phosphorus concentration at spring overturn shall not exceed 8 μg/L. This objective applies to the average of at least three samples taken 1 m below the surface, at mid depth and 1 m above the bottom, at about mid-lake.

Turbidity: the turbidity shall not exceed 5 NTU in any grab sample taken within 10 m of a domestic intake, nor shall the means from at least 10 such samples taken throughout the year exceed 1 NTU.

Fecal Coliform Bacteria:

the fecal coliform density shall not exceed 10 MPN per 100 ML in 90 percent of lake water samples, taken in any consecutive 30-day period, within 10 m of a domestic intake.

Suspended Solids:

In streams flowing into the lake, the suspended solids shall not exceed 25 mg/L in any grab sample.

The objective for total phosphorus was met in 1977-79 throughout the lake. The turbidity objectives were also met although some higher readings were recorded at depths below 15 m. The fecal coliform objective was met in the main body of the lake with some higher values being recorded in the West Arm and northern section near the lake outlet. The objective for suspended solids was met in all streams, except for some occasional high values in the Village inflow.

For protection of water quality, the recommendations are: to implement controls to minimize the amount of land disturbance within the watershed due to more construction, road building and ditching which generate suspended sediments and phosphorus; to consider higher standards for new septic tank installations to reduce phosphorus and bacterial contamination of the lake; and to put in place regulations which would cause upgrading of older, inefficient septic tank tile-field systems.

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1. INTRODUCTION

Shawnigan Lake is a medium size lake located on southern Vancouver Island (Figure 1). The lake for many years was largely a summer resort and retirement community, but in the past 15 years has received substantially increased population growth as residential development occurred throughout the watershed - particularly around the West Arm.

This study was undertaken at the request of the Cowichan Valley Regional District (C.V.R.D.). It attempts to answer general concerns with deterioration in water quality and effects of increased housing in the watershed on water quality. The C.V.R.D. wanted, if possible, to obtain some guidance on planning and zoning within the watershed.

The project required input from two disciplines - water quality and soils - in order to consider the link between the terrestrial ecosystem (the watershed) and the aquatic system (the lake). This report deals largely with the lake and water quality. The studies related to soils of the Shawnigan Lake watershed are referred to throughout the text and are cited in the bibliography.

The water quality investigations were carried out in 1977-1979 and reported to the C.V.R.D. in a series of meetings shortly afterwards. A basic water quality monitoring program has been continued since then to document any major changes which might be taking place.

No published report of the water quality work was prepared. However, public requests to have the data and interpretation made available, and continuing public concern for the lake water quality prompted preparation of this report.

2. BACKGROUND INFORMATION AND PREVIOUS REPORTS

There have been a number of studies carried out related to the water quality of Shawnigan Lake. Those prior to 1976 were reviewed by Nordin (1977). The most important of these reports were by Stonehouse (1969) who described sewage disposal and coliform bacterial concentrations in the lake; and Carl (1940) who described some aspects of the lake biota. The lake has been examined three times in projects undertaken by the University of Victoria: McKinnell (1978) examined the deposition of diatoms in the sediments; Black et al. (1977) made a general survey of water chemistry and coliforms; and Lucey and Jackson (1983) compared a number of attributes of Shawnigan Lake to the much more eutrophic Langford Lake.

The characteristics of the watershed (geology, soils, land use) are dealt with in detail in the report of the terrestrial portion of this investigation (Wiens and Nagpal, 1984).

A narrative of the social and settlement history of the Shawnigan Lake area is provided by Gibson (1967) and provides a review of settlement and human activities which could influence water quality.

3. LAKE MORPHOMETRY

Shawnigan Lake is located on southern Vancouver Island. The lake is medium sized (537 ha surface area) composed of one major basin and several smaller basins in the south part of the lake (Figure 2). The lake has a maximum depth of 50 m and a mean depth of 12 m. Other data regarding the morphometry of the lake are summarized in Table 1.

The surficial geology and soil characteristics of the watershed are described in Wiens and Nagpal (1983). For this study, the watershed was divided into a number of sub-watersheds in order to estimate water and nutrient inputs to the lake from different parts of the watershed.

4. METHODS

The investigation of water quality has taken place under two separate tasks, one, monitoring inflow streams to assess their effects on the lake, and the second task, monitoring the lake itself.

4.1 PROJECT GOALS

For Shawnigan Lake, the overall intent of the monitoring program was threefold. First, nutrient inputs to the lake were to be quantified from all the major sources (agriculture, sewage, streamflow, groundwater, aerial deposition). Their evaluation (in cooperation with the terrestrial portion of the project) would determine the principal contributors to the lake and suggest which if any could be minimized to prevent lake eutrophication. Second, any parameters which were changing as a consequence of development of the watershed and thus might indicate that certain uses of the water were being compromised were to be identified. Third, a clear baseline data set was to be provided which could be used to evaluate the water quality of Shawnigan Lake at some future date so that long term, subtle changes in water quality might be identified.

4.2 STREAM MONITORING

Ten inflow streams plus one outlet stream were monitored during 1977, 1978 and 1979 in an attempt to estimate the net inflow of materials, particularly nutrients, to the lake. The inflows were monitored (water quality and flow) monthly during most of the period of study, but more frequently during high flow (winter) periods. The stream stations are shown in Figure 1. Loadings of nutrients were calculated by multiplying concentration by stream flow. The parameters which were measured were pH, temperature, total residue 105°C (total solids), residue filterable 105 (dissolved solids), residue non-filterable 105 (suspended solids), conductivity, turbidity, colour, organic carbon, chloride, calcium, magnesium, nitrate, ammonia, organic nitrogen, ortho phosphorus, total dissolved phosphorus, total phosphorus.

4.3 LAKE MONITORING

The lake was monitored at five stations from autumn 1976 to spring 1979. The station locations are shown in Figure 1 and Table 2. At monthly intervals, temperature and dissolved oxygen profiles were taken (YSI model 57 calibrated with each use), light extinction measured (Secchi disc and Protomatic™ underwater photometer), and samples for water chemistry taken at the surface, five metres (epilimnion) and 20 metres (hypolimnion) depth at the two deep water stations (1199901 and -02 labelled 1 and 2 in Figure 1). At the shallow water stations (1199903, -04 and -05) samples for surface water chemistry and for the presence of coliform bacteria were taken. All of the water samples were kept cold in coolers and shipped to the Environmental Laboratory of the B.C. Ministry of Environment in Vancouver. Transit time to the lab was less than 24 hours and nutrient analyses were carried out within 72 hours.

Other sampling included phytoplankton identification and counts, zooplankton identification and counts (vertical net tow, 15 cm mouth size Wisconsin net), benthos and sediment sampling (6 cm by 6 cm Ekman grab), and a lake sediment core taken with a Phleger corer.

All water chemistry data are stored on the B.C. Ministry of Environment EQUIS data storage base.

5. WATER USES

Evaluation of water quality and protection from deterioration must be keyed to present water uses. In some cases this can be extended to potential water uses, although there are difficulties projecting both quantities and type of use. Evaluation of uses in this case is confined to present uses.

5.1 WATER BASED RECREATION

A variety of activities dependent upon water are important for both residents and visitors. Swimming, canoeing, water skiing and other leisure or vacation activities are important uses. Quantifying water uses is difficult. Water quality plays an important part since water clarity and the amount of algae or suspended sediments could help determine both the types and amounts of recreational activities.

5.2 FISH

Water quality requirements for fish and hence the relative value of recreational fishing are important aspects of water use. Deterioration of water quality manifested as hypolimnetic oxygen depletion, or changes in phytoplankton or zooplankton species composition can lead to losses in fish habitat or available food supply.

The species which are most important for sport fishing are cutthroat and rainbow trout, and it is estimated that 1 500-2 000 angler days per year are spent by fishermen. (Hay, pers. comm.). This is considered to be a low to moderate fishing pressure. The lake is usually stocked annually with 10 000 to 20 000 rainbow trout yearlings (depending on availability).

5.3 WATER SUPPLY

Shawnigan Lake serves as an important source of domestic water supply for two waterworks and numerous private water intakes from the lake. Figure 3 shows the location and type of licenced water withdrawals. There may be a number of unlicenced withdrawals as well. Waterworks and schools or hotels (listed as industrial licences) withdraw the largest volumes of water but domestic use accounts for the large majority of withdrawals. The total volumes of water withdrawn by various types of users are shown in Table 3.

6. HYDROLOGY

Two methods were used to calculate the stream flow at the outlet of Shawnigan Lake. From January 1978 to December 1979, a stream flow gauge (08HA004) was operated by the Water Management Branch of the B.C. Ministry of Environment.

Calculation of lake outflow for the other years listed in Table 4 , used the stream flow measurements taken on Shawnigan Creek (08HA033) near Mill Bay. The flow of Shawnigan Creek near Mill Bay was correlated with the 1978 and 1979 stream flow at the lake outlet. Correlations were developed for winter runoff (soils are saturated), summer runoff (evaporation exceeds precipitation), and fall runoff (soil recharge). Three equations were developed to predict, with a high degree of accuracy, the monthly stream outflow from Shawnigan Lake. They are:

Winter flow (dam^3) = -5.2 + 0.944 x Monthly flow at Mill Bay (dam^3)

Summer flow (dam^3) = -0.095 + 0.88 x Monthly flow at Mill Bay (dam^3)

Fall flow (dam^3) = -0.19 + 0.31 x Monthly flow at Mill Bay (dam^3)

Flows for the month of December should be calculated using the fall flow equation if below normal precipitation in October and November causes incomplete recharge, or using the winter flow equation if normal or above normal precipitation in October and November completes recharge, and saturates the soils.

Figure 4 and Tables 4 and 5 summarize the monthly stream flows at the outlet of Shawnigan Lake, and the monthly water exchange rates. Future outflow volumes can be estimated using the above equations. This method will only be valid as long as stream flow measurements are taken at Mill Bay. Discontinuation of these flow measurements will necessitate the correlation of stream flow and the most representative precipitation gauge. The correlation between runoff and precipitation will not be as accurate as the method outlined above.

The one-in-ten low flow estimates were completed by the Surface Water Section of the Water Management Branch (Barr, 1983). The frequency of low flow was calculated using the 1975-1983 data summarized in Table 4.

The high flows in late fall/early winter are typical of coastal watersheds and contrast markedly with most hydrologic patterns in North America where highest flows (freshet) occur in spring or early summer and are a reflection of snow melt and spring rainfall. The pattern of runoff is significant to nutrient loading and will be discussed below.

The water exchange rates are relatively high (1.06 times per year) with the majority of input coming in the November to February period.

Variation in monthly lake levels from 1977-1982 are presented in Figure 5. Although there is considerable annual variation, maximum lake levels occur from December through March. Decreasing rainfall and increased evaporation, reduce watershed runoff from April through October which causes the lake level to drop at a rate of 0.1 m/month.

Water level information should be a key consideration in the siting of sewage disposal systems. Disposal of effluent into saturated soils provides poor filtration of pathogens and transmission of high proportions of nutrients to the lake. This may be the case with some older sewage disposal installations on lakefront lots.

Estimated flows at scattered times are given in the section dealing with watershed loadings and are more relevant in that context. Estimates for sub-basins are given there.

7. WATER QUALITY

7.1 WATER CHEMISTRY

7.1.1 LAKE WATER QUALITY

To examine the chemical water quality of the lake, two stations were established, one in the main basin and one in the south basin. Three shallow stations were established to monitor changes in the lake water adjacent to the shore. The three shallow stations were established in areas, where from preliminary examination, effects might be expected because of proximity to developed areas. The deepwater stations were sampled to obtain information on the physical structure of the lake and spatial and temporal variation of key parameters such as nutrients and oxygen. The deep water stations are also for long term monitoring, to provide records of changes to the lake as a whole over long periods. The following discussion is based on data from the deepwater stations.

General Ions

The two deep water stations 1199901 (main basin) and 1199902 (south basin) were very similar in their water chemistry characteristics. The data for these two stations are summarized in Table 6. For general water quality, at both stations the water can be characterized as neutral (pH 7.1-7.2), with low dissolved material (filterable residue ~39 mg/L, specific conductance ~43 µS/cm, hardness ~17 mg/L, alkalinity ~16 mg/L), clear (generally <1 NTU, Secchi disc 6 m) with low colour in the surface water (some elevation in concentration in the deepest samples).

Phosphorus

The nutrient variables were all at very low concentrations. Neither station had an ortho-phosphorus concentration above 0.003 mg/L at any depth anytime of the study. Mean total dissolved phosphorus and total phosphorus were very low at 0.004 and 0.006 mg/L respectively.

In examining spatial and temporal changes of total phosphorus in the lake, it appears that total phosphorus showed little variation (Figure 6). There appears to be only a minor trend in concentration increases with depth with the deep water samples having somewhat higher concentrations. This is likely a consequence of sedimentation of plankton from the euphotic zone although one would expect that deep water concentrations would display a seasonal pattern, being higher at depth in summer when plankton biomass (and sedimentation) were high, but this was not the case. An obvious seasonal pattern for total phosphorus in the surface water is also difficult to discern. Concentrations of phosphorus in many monomictic lakes are highest in November to February when the lake is well mixed, little biological production is occurring and sedimentation is low. However in Shawnigan the decrease in summer was very small - generally 0.001 or 0.002 mg/L, matching the increase in hypolimnetic concentration.

Spring overturn phosphorus has been used as a useful index of phosphorus supply to a lake since it correlates with summer algal biomass and water clarity. The concentration at spring overturn then represents an important indicator of nutrient status of the lake and provides a useful measurement which can be used for management decisions (see Appendix 2). Spring phosphorus values for coastal lakes should ideally be obtained in February since the lakes are rarely ice covered and biological production (and hence uptake of nutrients) generally begins in late February or March; much earlier than the initiation of thermal stratificaton. During the period of this study water samples were not obtained at this "ideal" time but representative samples (of "overturn") indicated total phosphorus concentrations of 0.006 mg/L for 1977 and 1978. Sampling has been conducted irregularly since the study to ascertain if any long term changes have occurred. The sampling has been keyed to phosphorus concentrations during the fall/winter/spring unstratified period.

Station 1199902 at the end of January had a mean water column concentration of 0.006 mg/L total phosphorus and samples for early March averaged 0.008 mg/L for both the main basin and the south basin. At the end of October the main basin mean water column concentration was just less than 0.007 mg/L. The south basin had a relatively high concentration at 15 m likely due to incomplete mixing at that time. In February 1980 the main basin phosphorus was 0.0087 mg/L and in the fall (late October) the concentration was 0.0063 for the main basin and 0.008 mg/L for the south basin. The only sample in 1982 was not taken at a very desirable time (early May) but measured 0.007 mg/L (main basin). In 1983 a sample taken in early March had a mean water column concentration of 0.007 mg/L.

From these measurements taken since 1978, there seems some evidence that concentrations in the lake have increased. However it is extremely difficult to determine the exact amount of the increase, because concentrations fluctuate from year to year, with hydrologic exchange (flushing rate) being an extremely important factor governing spring (and winter) phosphorus concentration.

Nitrogen

Nitrogen also displayed low concentrations for all forms. The mean ammonia nitrogen concentration for the main basin was 0.011 mg/L, for nitrate nitrogen 0.047 mg/L and organic nitrogen 0.154 mg/L. Spatial and temporal patterns were more evident with some forms than others. Total nitrogen like total phosphorus showed only minor changes through the year and with depth. There appears to be no regular annual cycle of concentration in the surface waters but with depth there appears to be a general increase at all times of the year, as with phosphorus probably a reflection of sinking plankton and detritus (Figure 7).

Ammonia nitrogen (Figure 8) also showed a pattern of increased concentration with depth but no particular systematic cycle through the year in either the surface or bottom waters. One unusual occurrence was the presence of ammonia-nitrogen in surface waters at all times of the year.

The reason for this is unclear. Ortho-phosphorus and nitrate (see below) are absent from surface water during all the periods when phytoplankton growth was occurring, so the presence of biologically available inorganic ammonia nitrogen is paradoxical. The presence of elevated ammonia in deeper waters is probably a consequence of decomposition of organic materials in the water column and the sediments.

Nitrate is only present in surface waters in winter and early spring (Figure 9). This is a result of two factors. First, the input of nitrate to the lake was almost entirely in winter (Section 7.1.2) and during winter very little biological uptake of nitrate occurs. Second, during spring, a rapid decrease in concentration occurs as a result of biological uptake. In Figure 9 a decrease from 0.100 mg/L is shown in February to 0.080 mg/L in March to 0.040 mg/L in April to less than 0.020 mg/L in May.

An important consideration in determining possible future effects of nutrient enrichment is the ratio of nitrogen to phosphorus which are available for algal growth. Algae normally require nitrogen to phosphorus in the ratio of 5-10:1 (by weight). At N:P ratios greater than 12 or 15:1 phosphorus is generally considered limiting (Golterman, 1975, Dillon and Rigler, 1974). When N:P ratios are less than 5:1, it is generally considered that nitrogen is the limiting element (Shindler, 1977).

In Shawnigan Lake the ratio of total nitrogen to total phosphorus was approximately 212:6 or 35:1. The ratio of inorganic nitrogen to total dissolved phosphorus was 28:1 at spring overturn, and 15:1 for an annual mean. What these ratios imply is that the supply of phosphorus is extremely small in relation to nitrogen and is likely to be the factor limiting algal growth in Shawnigan Lake. This is corroborated by the extremely low concentrations of ortho phosphorus (mineral dissolved phosphorus) in the lake at all times of the year and by the low concentrations in the inflow streams.

Wiens and Nagpal (1983) have noted the high affinity of many of the soil types in the Shawnigan Lake watershed for phosphorus from sewage disposal systems. The implication of this is that if nutrient-related changes to the lake (water clarity, algal blooms) and water treatment by waterworks are to be minimized, then efforts to reduce nutrient input should be keyed to controlling phosphorus movement to the lake. Controls would be required for sewage disposal, land clearing, drainage of low or swampy areas, road building and ditching and application of fertilizers. Recommendations to minimize phosphorus input to the lake are given in detail in Section 8.2.

Data for the shallow water stations are shown in Tables 6 and 7. What is notable is the similarity in water quality between these stations (West Arm, near outlet, near Railway station) and the deep water stations (1199901 and 02). In examining either individual samples or the summaries of the samples, it is impossible to differentiate between the shallow and deep water analyses on the basis of general ions or nutrients. However there is a major difference when coliform bacteria concentrations are considered (Section 7.3.6), with significantly higher concentrations in the near shore areas as compared to the deep water stations.

7.1.2 INFLOW STREAM WATER QUALITY SAMPLING

A great deal of effort was directed toward measuring input of nutrients and other materials to Shawnigan Lake. The intent was to estimate loadings from the watershed, to determine if different areas of the watershed contributed more materials to the lake than others, and what the timing of the input was. Ten streams in all were measured, but some have only limited data records. Two sub-watersheds were chosen for specific examination because they had streams which drained a substantial portion of the watershed. Other sub-watersheds did not have definite surface drainage patterns and these streams may not have provided water quality representative of the land use of the watershed.

McGee Creek on the west shore (Figure 1) drains an area of 9.09 km^2, and Shawnigan Creek drains the large southern section of the watershed which has an area of 26.27 km^2. The remaining streams had small watersheds which were not easily delineated. As a result, the small streams were grouped into three areas (Figure 1). Flows of all surface inflows were estimated in the field (cross-section and velocity), but the streams characteristically had measurable flows only in the November to May period and some of the smaller streams for an even shorter period.

7.1.2.1 Shawnigan Creek (Inflow)

Shawnigan Creek water quality (Table 8) in general was very good. There is a very low density of residential housing in this watershed but some agricultural activity. The water quality characteristics which were monitored indicated little elevation above what would be considered normal for most Vancouver Island streams and only slightly above the lake concentrations for the same variable. Conservative parameters like total residue (stream 41.4, lake 40 mg/L) specific conductance (stream 50.8, lake 43.0 µS/cm), calcium (stream 4.2, lake 6.0 mg/L) showed little sign of deterioration in water quality. Variables which are indicative of disturbance (suspended sediments, turbidity, colour) were generally at levels similar to those in the lake, although colour was significantly higher (10.7 color units) in the stream than the lake (5.8 units). The variables of particular interest were nutrients.

In Shawnigan Creek, the nitrate nitrogen was significantly higher than in the lake (0.225 mg/L versus 0.043 mg/L) but organic nitrogen and ammonia nitrogen in the stream were lower than in the lake. The nitrate concentration was unusually high for a coastal stream but as is noted below, most of the watersheds around Shawnigan Lake had elevated nitrate concentrations.

The different fractions of phosphorus showed no noticeable elevation above what would have been expected, and were similar to the lake concentration.

The pattern of discharge of nitrogen is shown in Figure 10. There was little change in concentrations over the winter and no obvious peaks as was the case in many other sub-watersheds. This may be a reflection of the large size of the drainage area and the relative lack of disturbance of vegetation and topography. What is significant is that the highest concentration occurred with the first winter rains. There appears to be the highest loss of nitrogen from the sub-watershed at this time and a general decrease through the winter and spring.

7.1.2.2 Roundhouse Creek

Roundhouse Creek is located in a relatively well developed area of the watershed and might have been expected to show some changes in the water quality. However, this was not the case. Except for nitrate nitrogen (a very high mean concentration of 0.405 mg/L) all other variables showed no particular deviation from what can be considered as background either in concentration or periodicity of flow. The pattern of flow for nitrogen

(Figure 11) showed extremely elevated concentrations in the beginning of the hydrologic year (Nov.-Dec.) and the "first flush" effect is very obvious for this sub-watershed. The high concentrations in December (1.4 mg/L) decreased to less than 0.2 mg/L by March.

7.1.2.3 Unnamed Creek - First North of Roundhouse Creek

The first Creek north of Roundhouse Creek was very similar to Roundhouse Creek for all parameters. Nitrate nitrogen was less than either Roundhouse Creek or Shawnigan Creek but still unusually high (0.152 mg/L).

7.1.2.4 McGee Creek

McGee Creek is also located in an area which has significant residential development. Unlike Roundhouse Creek, there were changes to the water quality. In comparison to the lake, the Creek had elevated total residue (59.9 versus 40 mg/L), specific conductance (80.2 versus 43 µS/cm), alkalinity (26.9 versus 16.2 mg/L), chloride (4.4 versus 2.5 mg/L), calcium (10.4 versus 6.0 mg/L), magnesium (2.0 versus 1.0 mg/L) and colour (14.0 versus 5.8 TAC units). Phosphorus and nitrogen were not elevated except nitrate nitrogen which averaged 0.253 mg/L and was measured as high as 0.830 mg/L (Table 9).

McGee Creek like many others where a water quality effect was evident, had high concentrations of nitrogen in the period after the fall rains began (November and December), and declining concentrations thereafter with minimum concentrations before the creeks dried up (May, June, July). This pattern appears again to be due to transport of nitrate out of the soils by the heavy rains. The materials which accumulate during the summer in the soils, and which are not bound but mobile in soils (i.e., nitrate) are moved through the watershed to the streams. The most likely source of the nitrate (Figure 12) is sewage effluent.

7.1.2.5 Unnamed Creek North of McGee Creek

This creek was only sampled in February and March 1979. Other than the elevated nitrate, few other water quality variables were notable.

7.1.2.6 West Arm Inflow

The West Arm inflow is similar to McGee Creek in that there were clear elevations in a number of variables. The changes, however, were even more marked. The general variables (total residue, specific conductance, alkalinity, calcium, magnesium) were all elevated and the variables more keyed to disturbance (non-filterable residue (suspended sediment), turbidity, colour, organic carbon) were also higher than in the less disturbed streams, providing clear evidence of disturbance. The nutrient parameters were all generally at higher levels. Ortho phosphorus, total dissolved phosphorus and total phosphorus were all higher than in previously discussed streams and indicate a significant supply to the lake. All the nitrogen concentrations were higher than the lake concentrations and nitrate nitrogen again in particular was very high with a mean concentration of 0.475 mg/L and a maximum concentration of 1.840 mg/L. The fall-early winter nitrogen concentrations (Figure 13) were very high and appear to reflect the "first flush" phenomenon apparent in the data for streams discussed earlier.

7.1.2.7 Landfill Inflows 1 and 2

This small surface flow originates in the extreme north east part of the watershed in the area of an old landfill. The two stations are located upstream and downstream from a boggy area. The general area had been disturbed. The water quality data indicate that this area generates higher concentrations of the variables measured than any of the previous streams.

The total residues averaged 153 mg/L at landfill inflow number 1 versus 60 mg/L for McGee Creek or 40 mg/L for the lake. The conductance was high 218 μS/cm (against 43 μS/cm for the lake). Many of the indicators of land disturbance showed high values: suspended solids 3.2 mg/L, turbidity 1-25 NTU, color units 10.6, chloride 21.3 mg/L. There were elevated concentrations for both nitrogen and phosphorus and nitrate nitrogen was especially notable at 1.14 mg/L. The effect of this poor water quality on the lake was somewhat reduced by the small flow and proximity to the Shawnigan Creek outflow, so loading was relatively low but the overall water quality from this area is still poor. Samples were taken here for heavy metals (cadmium, copper, chromium, lead, arsenic, manganese) but no elevated concentrations were found.

7.1.2.8 Village Inflow

The drainage from this area, which is the most densely developed in the watershed, was of extremely poor quality, and was consistently so through the study. The poor water quality was represented in every variable measured and levels of many variables were four to five times higher than in the least disturbed stream (Shawnigan Creek inlet) or the lake. Nitrate and ammonia were more than an order of magnitude higher. The nitrate nitrogen averaged 1.8 mg/L over the period of sampling and the maximum concentration was 3.0 mg/L. This stream and the landfill inflow were the only locations where nitrite (indicative of reducing i.e., deoxygenated conditions) was encountered.

7.1.2.9 Summary of Inflow Streams

In general there appears to be a strong correlation between the amount of development in the sub-watersheds around Shawnigan Lake and the high levels of a number of water quality variables. The influx of high concentrations of materials was largely from areas with relatively low flows so the overall effect on the water quality of the lake was small. However, it is apparent that development of larger areas of the watershed and higher housing density could lead to a deterioration of water quality.

7.1.3 NUTRIENT LOADING

Nutrient loading to a lake can be estimated in a number of ways. The most accurate but most expensive, is to measure, on a frequent basis, all the inputs to the lake. These would include stream concentration and flow, groundwater concentration and flow, aerial loading (dustfall and precipitation), sewage or industrial input, internal loading and other factors. Some effort was expended in this study on measuring some of these inputs, but sufficient data could not be collected to provide accurate estimates for all components. The stream nutrient inputs are discussed below. Groundwater sampling was attempted in 1978 using the method of Lee (1977). In the three areas where the samplers were installed no groundwater flow could be measured. This may have been a reflection of several factors. Groundwater flow to the lake may not have been significant in these areas and the time of year was likely (in retrospect) one in which very low groundwater flow would occur (May through September). Because of rising water levels in the fall, there was some hesitancy in installing samplers in fall or early winter since winter water levels would have made the samplers difficult to find, much less to operate.

Measurement of aerial loading was not done in this study but a number of studies from the literature can be referred to which can allow estimates of aerial loading. As well a number of studies in the area (Truscott, 1981, Nordin _et al_. 1983) provide data on loading estimates. According to these studies, the amount of aerial loading is likely to be relatively small (probably ~20 $kg/km^2/yr$). Aerial input also represents a loading which cannot be reduced by any normal management techniques - in comparison to inputs from the watershed.

Wiens and Nagpal (1984) have spent considerable effort on quantifying the amount of nutrient originating from domestic sewage disposal, and their results are outlined in their report. They have also estimated inputs from agricultural land uses in the watershed.

There was no evidence from the lake nutrient water quality data of any internal loading taking place. The detection and conditions allowing the release of nutrients from the lake sediments are discussed in Nordin and McKean (1983).

7.1.3.1 Stream Loading

Loading estimates were made by matching the runoff estimates used in Section 6 with nutrient concentrations in the streams for the corresponding period. The estimates are approximate since they were made using mean monthly flows and only one to three stream concentrations. The Shawnigan Lake watershed was divided into five sub-watersheds as shown in Figure 1. The runoff from each area was presumed to represent an amount proportional to the area of the watershed. The results of the calculated loadings for five variables (total phosphorus, total dissolved phosphorus, total nitrogen, nitrate and suspended sediments) are shown in Tables 11 through 15. In Table 11, the estimated phosphorus loading varies considerably in the two years. For the hydrologic year 1977-78, the total phosphorus entering the lakes from the streams was estimated at 337 kg, and for 1978-79 it was 228 kg. These estimates do not include direct surface runoff from land adjacent to the lake or groundwater input. It is difficult to estimate what proportion of the phosphorus originates from "natural" sources (loss from soils, decay of vegetation) and what proportion is from human activity (sewage, fertilizer, erosion, etc.). Another difficulty is that the stream loading estimates likely do not include the relatively high loading which occurs in storm events, particularly with winter storms when significant inputs may occur during only short periods.

The other Tables (Tables 12-15) also estimate the amount of dissolved phosphorus, total and nitrate nitrogen, and suspended sediments which enter the lake. These data show that the amount of nitrogen supplied to the lake in comparison to the amount of phosphorus is relatively high (61:1 for TN:TP, and 52:1 for nitrate N:dissolved P). This implies that phosphorus is the more important element in determining algal production in the lake, i.e., it is the "limiting" element. A similar conclusion is reached when the lake water chemistry data are examined.

An important evaluation which can also be made is the amount of materials (nutrients, sediments) which originate from different sub-watersheds with different levels of development. This information is given in Tables 11-15. It shows that sub-watersheds such as the West Arm (Area B) or the sub-basin on the east side of the Lake (Area C) produce substantially more nutrients and sediments than a relatively undisturbed watershed such as Shawnigan Creek (inflow).

An alternative way of estimating phosphorus input to the whole lake is to use some equations which have been developed to relate the factors of loading, hydrology and lake concentration for phosphorus (Table 16). In these simple models, the loading is estimated from spring overturn concentration and the annual (runoff). The results of these estimates show a range of values, some of which appear to be due to peculiarities of individual equations. The mean loading value for L1 (Vollenweider) is 0.169 $g/m^2/yr$ and for L2 is 0.161 $g/m^2/yr$. The mean value for L3 (0.232) is a reflection of an unreasonably high value for 1979 which was a year with little water input (L1 also has a relatively high value for 1979 but not as extreme). The mean value for L3 without the 1979 value is 0.180 $g/m^2/yr$, which is reasonably close to the other two estimates.

On the basis of these estimates, the total phosphorus loading to the lake is about 0.17 $g/m^2/yr$ or 913 kg/yr. This would represent an "average" annual loading. In comparison to the measured loadings discussed earlier, these calculated loadings would represent total loading to the lake whereas the earlier estimates were only for the stream flows and would not include groundwater, overland flow, aerial loading and internal loading.

7.1.4 PHOSPHORUS BUDGET

It is useful to estimate the various inputs and outputs of phosphorus in order to evaluate the relative contributions, particularly of inputs. The basic information which is available is: the total annual phosphorus input, probably about 900 kg/yr; the stream input (average for the study period but probably underestimated), about 270 kg/yr; the aerial input, probably about 100 kg/yr; leaving the majority of the loading unaccounted for. The remaining input would likely come via groundwater, overland flow, and unmeasured small streams, with the phosphorus originating from sewage or fertilizers. The most obvious area where lack of data is apparent is groundwater. This constitutes a major shortcoming in the understanding of the origins of phosphorus from the watershed.

In considering the phosphorus output, the two major mechanisms are sedimentation and outflow. The phosphorus sedimentation coefficient used was 0.49, meaning approximately 50% of the phosphorus input would be lost to the sediments. The measured phosphorus outflow for the study averaged 424 kg/yr. So if half the input (~450 kg/yr) is lost to sedimentation, the loss of 424 kg/yr through the outlet supports indirectly the estimate of total phosphorus loading (900 kg/yr) calculated by models.

As noted above, the most serious problem in considering the phosphorus budget is the lack of data on phosphorus inputs from specific sources, particularly the proportion of the total loading contributed by anthropogenic sources (the most important of which is thought to be sewage). This information is important if effective management priorities are to be set. However, the Wiens and Nagpal report estimates this component of phosphorus loading as part of their investigations.

7.2 LIMNOLOGICAL CHARACTERISTICS

A number of considerations must be made of the characteristics of the lake itself which have an effect on many other aspects (water chemistry, aquatic biology).

7.2.1 LAKE TEMPERATURE STRATIFICATION

The lake progresses through a seasonal cycle. During the winter, the lake, except during times of ice cover, is isothermal, and apparently well mixed at 4 - 6°C. Ice cover occurs occasionally on Shawnigan and was present during the study period from late December 1978 through February 1979. Under ice cover the lake surface water cooled more than normal (Figure 17). As spring and summer progress, the input of heat causes the surface waters to become warmer and a temperature stratification to occur. The stratification is well established by May and the temperature gradient increases during the summer, reaching a maximum in August. The maximum surface water temperature is generally 22-25°C. The thermocline is generally centred at 8-10 m depending on the time of the year. The thermocline gradually becomes deeper through the summer and begins to be eroded in September as the surface waters cool and the epilimnion becomes deeper. The breakdown of the thermocline is gradual and the gradient becomes weaker until the lake becomes isothermal and begins the winter vertical mixing, generally in late November or early December. Lake water temperatures for the two deep water stations (main basin and south basin) 1977-79 are given in four time-depth diagrams (Figures 14-17). Some minor differences exist between the main and south basins, largely due to the differences in morphometry of the two basins. In the south basin the water is somewhat warmer and corresponding isotherms are slightly deeper. The maximum surface water temperatures are marginally higher, the fall overturn occurs slightly earlier and the hypolimnion temperatures are higher.

7.2.2 DISSOLVED OXYGEN

The two major factors affecting dissolved oxygen are the temperature stratification pattern and the amount of phytoplankton photosynthesis. The temperature stratification divides the lake into two separate non-mixing layers and prevents any oxygen from being introduced into the hypolimnion from April or May through October or November. Phytoplankton are largely confined to the epilimnion and can cause periods of oxygen supersaturation in the epilimnion. This phenomenon is of no particular concern to water quality and has no negative environmental effects. The annual fluctuations of dissolved oxygen with time and depth are shown in Figures 18 to 21. Periods of oxygen supersaturation (greater than 100%) are present in May through August and correspond to periods of high phytoplankton growth. In the hypolimnion during the corresponding period, a gradual decrease of oxygen takes place with a gradient of concentration decreasing with depth and decreasing over the summer. This oxygen depletion begins with the onset of stratification and persists until late fall when overturn occurs.

Phosphorus concentration and/or loading (and hence phytoplankton production) is a key factor in determining the hypolimnetic oxygen depletion (Cornett and Rigler 1979, Walker 1979, Welch and Perkins 1979). As a lake becomes more productive (more algal growth), concentrations of dissolved oxygen in the hypolimnion decrease due to decaying algae and this has a number of consequences. Decreased oxygen concentrations can limit habitat for fish, specifically in summer months when sport fish may require the cooler water temperatures found at depth. However if deeper waters have low dissolved oxygen fish can no longer use this area of the lake as a temperature refuge or for feeding and must continue to use warm epilimnetic water as habitat, which may cause decreases in growth rates or survival. Decreases in oxygen may also lead to increases in nutrient release from lake sediments which can accelerate the eutrophication process or allow liberation of materials which cause deterioration of drinking water (manganese, iron, hydrogen sulphide) into the water column.

Shawnigan Lake has a hypolimnetic oxygen depletion which is unexpectedly large. From the data on nutrients and phytoplankton, the lake would clearly be considered unproductive (oligotrophic). The oxygen depletion is out of character with the other limnological parameters for the lake, and is an area of particular concern for the general condition of the lake. Table 17 shows oxygen depletion rates for four years and despite the variability the oxygen depletion would indicate mesotrophy. Table 17A gives calculated oxygen depletion rates for three phosphorus concentrations. The data show that the present depletion rate is more comparable to a phosphorus concentration of approximately 0.010 mg/L phosphorus.

The most plausible explanation for this anomalous depletion possibly lies with the settlement history around the lake. From the 1890's to 1940's a log dump and sawmill were major features of the lake and the handling and sawing of logs generated a large amount of wood waste (bark, sawdust, etc.) much of which was deposited in the lake. During sampling of the lake sediments wood waste was encountered in many areas, particularly near the old log dump (on the west side) and in the area where the mills were located (on the east side). Another possibility is input of organic materials from the watershed (sewage, land disturbance) which require oxygen for decomposition.

Since decomposition takes place very slowly in fresh water, it would be expected that the wood waste would take many years to decompose (or be covered by sedimentation). Thus the wood waste seems the likely source of this unusual oxygen deficit. A similar oxygen content for Shawnigan Lake in the 1930's was reported by Carl (1940) so the phenomenon is not a recent one. The south basin has a more severe depression of oxygen than the main basin (Figure 22) probably because of a relatively small hypolimnion (compared to its epilimnion and to the morphometry of the main basin).

7.2.3 WATER CLARITY

Water clarity is a key parameter both as a limnological characteristic and also as one of the most obvious features which the general public uses as a measure of acceptability of a lake for recreational use.

The measurements were taken both with the standard Secchi disc and with an underwater photometer. The data can be used to note trends in water clarity over the year or over longer periods. Measurements were taken at the two main sampling stations in the north and south basins (Table 18). The data show good water clarity in both areas with the north basin having slightly clearer water over the long term (mean 6.2 m Secchi) than the south basin (5.8 m). This difference is also apparent in a number of other characteristics which are discussed below. No measurements for water clarity have been made since 1979, but the 1976-79 data form a good basis of comparison should another examination of the lake be carried out. Clarity is a parameter which could easily be measured by a citizen's association or residents group who wish to carry out one significant aspect of water quality monitoring at minimal cost and effort.

7.3 BIOLOGICAL PARAMETERS

7.3.1 PHYTOPLANKTON

Phytoplankton (planktonic algae) provide useful information on several aspects of water quality. Changes in species composition provide a sensitive indicator of changes in water chemistry, and more importantly nutrients. Knowledge of species composition (Table 19) can be used to evaluate if organisms are present which might cause problems with water use. Such problems include imparting taste or odour to drinking water, clogging filters and pumps, or simply causing a decrease in water clarity and in the aesthetic attractiveness of the water.

The phytoplankton biomass was measured as chlorophyll *a*. As well, samples were taken and organisms identified and counted.

In 1977, the May samples were dominated by diatoms (*Asterionella*, *Tabellaria*, *Cyclotella*) and others typical of cool, nutrient-poor waters (*Dinobryon*, *Spondylosium*). The June period was still dominated by diatoms but a number of blue-green algae generally began to appear (*Gomphosphaeria*, *Anaystis*, *Chroococcus*) and the sample at the end of July showed *Gomphosphaeria* to be the most numerous organism. The July samples show that the blue-green algae are the dominant group of organisms. This situation continued through August. The late August early September was typically the period when a peak in population of the blue-green alga *Gleotrichia* occurred (as in 1977-78). This organism is colonial and easily visible to the naked eye as a small (1-2 mm) grey-green sphere in the water column.

The only previous data which are available were collected by the Fish and Wildlife Branch in a lake survey July 1951. The dominant phytoplankton at that time were *Tabellaria*, *Asterionella* and *Fragillaria*.

The phytoplankton biomass data, (Table 20) show no obvious periods of higher biomass. The chlorophyll a concentrations appear to be somewhat higher in spring, (March through June) but only one complete year of data exist so generalizations are difficult. There do appear to be higher concentrations of chlorophyll a at subsurface depths than at the surface, however this is typical of lakes with good water clarity.

There appears to be little difference in biomass between the main basin (station 01) and the south basin (02).

7.3.2 DIATOM STRATIGRAPHY

Another aspect of lake phytoplankton is the changes in lake phytoplankton which are recorded in the sediments. Diatom frustules are relatively resistant to decay and are generally the focus of examination for detecting changes in species composition. Since the cellular remains are deposited in sequence, the sediment gives an excellent record of the chronology of changes in the diatom community. The following information is taken from McKinnell (1978).

Although several studies of diatom assemblages have been undertaken (Pennington 1943; Patrick 1943; Round 1961; Deevey 1942), only recently have attempts been made to correlate these with the historical effects of cultural eutrophication (Stockner and Benson 1967; Bradbury 1977). Relationships between diatom assemblages and limnological parameters have been under examination since the turn of the century. The complexities of these relationships and the difficulties in their interpretation have resulted in few comprehensive and conclusive studies. However, evidence is mounting to support the relationships between diatom assemblages and past limnological conditions.

Diatom floras can be important to the study of lake chronology for several reasons. Firstly, diatoms are often the dominant algal form in many lakes and their numbers alone facilitate sufficient data collection. Secondly, the siliceous frustules enable taxonomic identification and are well preserved in lake sediments under most conditions. The last, most important feature is the ability of diatoms to record relatively distinct ecological events.

In many cases, recent changes in lake histories are associated with human disturbances (Stockner and Benson 1967; Bradbury 1977). Drainage basins, especially lake shores, become sites of intense human settlement and development. The nature and degree of development will ultimately determine the extent to which the aquatic environment is affected. By studying the ecological transition of a lake to its present condition, the effects of nutrient enrichment and disturbance can be documented.

Even though many species occur with relative constancy throughout the sediment core, species proportions have changed, sometimes quite remarkably, with depth. Proportions of the eleven more abundant species in Shawnigan Lake are illustrated in Figure 23. Species' ranked abundance were also determined for each stratum and appear in Table 21. It is evident that species dominance has changed in the upper 6 cm.

Below 6 cm, the dominant species are Cyclotella kutzingiana, C. stelligera, Melosira distans and Fragilaria construens. Above 6 cm., the increased rank of Melosira italica and Tabellaria fenestrata would indicate a general trend toward changing dominance. Several species found abundant in the lower sediments become proportionately less frequent above 6 cm.

To determine the significance of the changes in ranked abundance, a Mann-Whitney test for comparison of rank in unpaired measurements was employed. The measurements referred to are the proportion of respective species within each stratum, above and below 6 cm. Significant differences were observed for ranked abundance within samples between the upper fifth and the lower four-fifths of the core in 6 species. Those species and their significance levels appear in Table 21. The centric diatoms Cyclotella kutzingiana and C. stelligera decreased in rank significantly, while in the upper portions, M. italica increased. The araphidinate species: F. crotonensis, Asterionella formosa and T. fenestrata all showed significant increases above 6 cm.

The appearance of two species, A. formosa and F. crotonensis is possibly the most interesting feature of the recent sediments. Pennington (1943), after finding A. formosa only in the upper 20 cm of sediments in L. Windermere, correlated its sudden appearance with increased human settlement and subsequent nutrient input. The same general trend, was also observed in meso-eutrophic L. Washington, although it was not totally absent from the older sediments (Stockner and Benson 1967).

The pre-disturbance sediments of L. Washington are characterized by C. stelligera, Stephanodiscus astrea, M. italica and F. construens and others not present in Shawnigan Lake. For those species common to both lakes, the same general responses are evident - keeping in mind the differences in the respective trophic states. The exception in Shawnigan L. appears to be the increased ranked abundance of M. italica with nutrient enrichment. Unlike Lake Washington, it was commonly found in several local eutrophic lakes.

In Shawnigan L., F. crotonensis appears only in the most recent sediments. It is very likely that its presence, or at least its increased ranked abundance, results from disturbances within the watershed. In several English lakes, Stockner (1972) recorded F. crotonensis only in those lakes receiving sewage discharge. Cultural eutrophication of L. Washington led to marked increases in numbers of F. crotonensis (Stockner and Benson 1967).

F. pinnata, F. construens, F. brevistriata and C. bodanica maintained relatively constant proportions throughout the core. Several Fragilaria species although common, have not been considered indicative of particular nutrient-enriched environments (Stockner 1972).

As yet, M. distans has not been associated with particular nutrient preferences, although it has been found, as previously described, in certain planktonic associations in unproductive Finnish lakes.

Cyclotella kutzingiana was found to be the dominant species in the older sediments of Shawnigan Lake. It was also dominant in most of the Experimental Lakes Area lakes in northwestern Ontario (Stockner 1971), and was reported by Rawson (1956), and Jarnefelt (1952) to be most abundant in oligotrophic waters.

7.3.3 PERIPHYTON

Sampling of periphyton (attached algae) was carried out to ascertain if littoral productivity was being enhanced by watershed activities or if certain areas of the shoreline were more affected than others in terms of potential algal growth. The results (Table 22) show that periphyton biomass was generally low in all areas sampled. Some difficulty in gathering data was encountered because of sampler and equipment loss (vandalism and weather), however no significant differences were noted between stations.

7.3.4 LONG TERM CHANGES IN WATER QUALITY

The last 75-100 years of Shawnigan Lake history has been radically differenct from the years preceding it. Extensive logging and saw-milling operations were the dominant activities within the watershed in the late 19th and early 20th centuries. In fact, by 1908, the entire shoreline had been deforested. Over the course of 50 years, there were three mills operating on the lake shore. This type of activity within a drainage basin will affect the aquatic system in three major ways.

First, an increased nutrient input will result from leaching of deforested soils. Secondly, the industrial use of the lake water by mills and the subsequent discharge of contaminated water into the lake will be detrimental to the system. And finally, the most influential effect will be the influx of human residents in response to watershed development.

Settlement of the Shawnigan L. basin began in the late 19th century. By 1902, development had progressed to the extent that a small settlement was established at the north end of the lake. Basin development, by accounts of local residents, has proceeded relatively slowly until recently.

Based upon evidence gleaned from the stratigraphic record of diatom flora, it appears that human influence has resulted in significant changes in Shawnigan L. To support this conclusion further, the upper 6 cm. of the lake sediment core would have to correspond to the time span covering the human activities within the basin. The representation of 70 - 80 years by the upper 6 cm would suggest an approximate sedimentation rate of 1 mm per annum. It is important to note that a rate of this magnitude is not inconsistant with rates determined from several, other unproductive lakes (Stockner and Benson 1967; Stockner 1971). This being the case, the correlation between diatom stratigraphy and development history of Shawnigan L. appears to be justified based upon the evidence gathered to date.

7.3.5 ZOOPLANKTON

Examination of the animal component of the plankton is also useful from a number of perspectives. Although the zooplankton itself generally does not cause problems with water quality acceptability or aesthetics, changes in biomass or species composition can provide evidence of changes in the lake ecosystem as a whole, and species composition and biomass can be useful parameters for fisheries considerations. Two incomplete years of data are available for Shawnigan Lake (1977-78 Table 23), and data for 1976 were reported in Nordin (1977).

Very few historical data are available to ascertain if changes have taken place in the zooplankton community. Carl (1940) gives a list of species which were collected in the 1930's: Sida crysallina, Daphnia longispina, Scapholeberis micronata, Bosmina obtusirostris, Polyphemus pediculus, Leptodora kindii, Epischura nevadensis, and Cyclops bicuspidatus. In comparing these species however, it is necessary to note that some taxonomic changes have occurred, e.g., Carl's Daphnia longispina is taxonomically equivalent to Eubosmina lagmanii. Carl notes that Shawnigan Lake had large numbers of collections made from it so his list of species present was probably a reasonable one.

Table 24 lists those species present in the 1930's and in the 1970's sampling, showing the differences between the two communities. One difference is the presence of Bosmina obtusirostris in the 1930's. The present synonym for this species is Eubosmina lagmanii. However, it is unclear which keys Carl used to identify his zooplankton samples or whether the Bosmina he identified is the same species or the Bosmina longirostrus present in 1976-1978. Carl also makes a particular note regarding the absence in Shawnigan Lake (and other lakes in the area) of Epishura and Diaptomus occurring together. The former was present in the 1930's, the latter was not. Diaptomus was present in the 1976 samples although in small numbers.

Other changes in the zooplankton include the absence in the later sampling of Scapholeberis and Polyphemus, and the occurrence in the later sampling of Diaphanosama and Ceriodaphnia.

From the information which is available, it is apparent that no major changes have taken place in the zooplankton community although significant minor changes are apparent. There appears to be a general feeling (Ravera

1980, Brooks 1969) that effects of eutrophication on the zooplankton are relatively difficult to detect and complicated by changes in predation patterns, phytoplankton and other changes.

7.3.4 COLIFORM BACTERIA

They are of particular concern in watersheds where sewage disposal is to the ground and the possibility of contamination of surface water exists. They are especially important when surface waters are used for body contact recreation and drinking water supply.

Water samples were taken from 1976 to 1978 at a variety of sites around the lake and are compiled in Table 25. These data show that the deep water samples, i.e. away from the shore line, were very low in numbers of coliform bacteria with only rare samples showing positive results. The stations which were located in shore areas (1199903, 04, 05) showed significantly higher concentrations. The concentrations did not exceed the criteria for body contact recreation used by the B.C. Ministry of Health (Richards, 1983) which is a maximum for fecal coliform of 200 MPN/100 mL (geometric mean) and a 90th percentile value of 400, but did exceed the guidelines set by the B.C. Ministry of Health (1982) for domestic water supply. The guidelines state that with fecal coliform concentrations of 0-10 MPN/100 mL, disinfection would be required and with concentrations of 10-100 MPN/100 partial treatment (filtration, flocculation, settling, and chlorination) is required.

The most obvious problem is with the numerous water intakes around Shawnigan Lake (see section 5). Few if any of these water systems would have any capability for disinfecting the water used, so potential problems could arise.

The source of this contamination cannot be determined easily. The density of residential housing and the likelihood that some older sewage disposal installations are not functioning efficiently are possible causes. Problems include poor maintenance (lack of pumpout), heavier use than the disposal field was designed for, or poor location of disposal fields, particularly on waterfront lots where high winter lake levels cause disposal fields to be inundated or allow surface waters to be contaminated.

The inflow streams show the highest levels of fecal coliform contamination.

8. DISCUSSION AND RECOMMENDATIONS

8.1 OBJECTIVES FOR WATER QUALITY

Lakes comprise a valuable water resource which can easily become degraded due to human activity. It is unrealistic except in certain unusual cases to completely disallow any human activity in a watershed since provision of recreation and home sites is central to the use and appreciation of such a resource. The key to protection of water quality is to control the location and quantity of activities (particularly home construction) to an extent where there is no effect or a slight effect which may be acceptable to the general population.

This "level of acceptance" is a very subjective concept. However, more objective criteria have evolved which can be used to evaluate water quality change.

If this hypothetical "level of acceptance" is exceeded, then a number of water quality parameters can be identified which are the focus of complaints by the persons who use the water. The water users (noted in section 5) often have very different perspectives as to the specific water quality parameters which are changing. Some parameters are easily identified (e.g., coliform counts, dissolved oxygen) others are more difficult to evaluate or quantify (taste and odour aesthetics). As a consequence the criteria which are used to evaluate suitability of water for various uses are those which can be quantified and more importantly, for which concentrations can be related to particular conditions. The parameters which can be used as indicators of changes in water quality caused by watershed development are discussed below.

There is a group of parameters which reflect the changes brought about by the process of eutrophication. The key parameter is phosphorus since eutrophication in the vast majority of cases (including Shawnigan Lake) is a response to increased phosphorus input to the lake. A variety of other changes are symptoms of eutrophication. These are discussed in further detail below (hypolimnetic oxygen depletion, algal production and biomass, water clarity, total and dissolved organic carbon, colour, increases in manganese, iron and hydrogen sulphide).

8.1.1 EUTROPHICATION PARAMETERS

A range of levels of biological productivity occurs in lakes, and different water uses require different optimal concentrations for the parameters of interest. However, few attempts have been made to establish water quality criteria for these parameters, the most important of which is phosphorus. Since many of the other parameters (water clarity, algal biomass, oxygen depletion etc) can be related directly to phosphorus, phosphorus is examined in detail below. The relationships between phosphorus and the other eutrophication parameters for B.C. lakes are discussed in Appendix 1,

Water quality criteria for total phosphorus are summarized in Appendix 2. Many of the phosphorus criteria for drinking water are far too high (50 - 300 µg/L) to be reasonable for Shawnigan Lake (or most other B.C. locations) unless sophisticated water treatment is available.

The criteria in Appendix 2 for aquatic life make general reference to the oxygen depletion which is an indirect effect of elevated phosphorus. Two separate agencies use 10 µg/L phosphorus as a level to protect against hypolimnetic oxygen depletion. One (Ontario Ministry of Environment 1979) uses mean concentration for the ice free period, and the second (International Joint Commission-IJC) uses spring overturn as their point of reference for measurement. The latter method appears to be more desirable since it can be related more directly to loading and is better for establishing a data base and monitoring lake changes.

Regarding criteria for recreation and aesthetics which is a third water use for Shawnigan, two agencies specify reasonably low concentrations. These could be considered for Shawnigan Lake. Ontario has specified a concentration of 10 µg/L and the International Joint Commission specifies 5-15 µg/L depending on the location (Appendix 2).

Dillon and Rigler (1975) suggested four categories of water use (not directly comparable to Appendix 2 data) which were keyed to mean summer chlorophyll but also correspond to specific spring overturn phosphorus, water clarity and dissolved oxygen concentrations. These are given in slightly modified form in Table 26 for a number of parameters, but only two water uses are considered (fisheries and recreation).

Although criteria are generally specified as concentrations, loading values also can be used, such as Vollenweider's (1968) relationship, as guidelines to protect water qualjty. The disadvantage of using loading as a criterion is the difficulty in obtaining accurate values for lakes. Data collection is time consuming and expensive and requires each lake to be looked at individually and in detail. Because loading can be easily related to spring overturn phosphorus (e.g., Canfield and Bachmann 1981, Reckhow and Simpson 1980, or Vollenweider 1969) it would seem advantageous to use the spring overturn phosphorus value as a criterion rather than the loading value.

The concentrations of total phosphorus in Shawnigan Lake in 1977-78 at spring overturn were approximately 6 µg/L which is less than any of the criteria suggested by other agencies for any water use. However, the unusually high hypolimnetic oxygen depletion would be a strong reason to make efforts to minimize phosphorus input. The only feasible means of preventing oxygen depletion would be by minimizing the phosphorus input and maintaining the concentration as low as possible. A second reason for considering a lower phosphorus concentration for an objective is the generally higher quality of water which is expected by residents of the area. They are accustomed to better quality water in lakes than might be accepted in other parts of North America or Europe where other criteria or objectives have originated.

Another consequence of increasing nutrients can be nuisance aquatic weed growth. Some complaints of weed growth have been noted. The cause for growth of rooted macrophytes is generally not water column nutrients but creation of favourable habitat by restricted water movement or wave action or by accumulation of suitable bottom materials. One genus of plants which does respond to water column nutrients only is Ceratophyllum which does not have roots. Appendix 3 is an unpublished investigation by P.D. Warrington and C.J.P. McKean of Ceratophyllum occurrence and the correlation between concentration of phosphorus and whether the plant is considered a nuisance. It suggests that in general there is a threshold of about 20 µg/L above which Ceratophyllum reaches densities which would be considered a problem. So, in Shawnigan Lake, an increase in Ceratophyllum would seem unlikely until the phosphorus level reaches twice the present level, and as such, nuisance weed growth is a relatively less sensitive indicator of eutrophication.

In summary then, it would appear that a provisional water quality objective for phosphorus appropriate for Shawnigan Lake would be 8 µg/L (spring overturn). This would protect the present uses. Additional discussion on phosphorus concentrations and water quality protection is given in Appendix I. This objective should be met by the average of not less than three samples (surface, mid-depth and near bottom) taken at mid-lake before thermal stratification begins or significant (0.5 µg/L chlorophyll a) algal biomass is present. The best time to sample would be the last two weeks in January.

8.1.2 SUSPENDED SEDIMENTS

Construction activities can result in generation of suspended sediments which enter the lake, generally by erosion. Monitoring of suspended sediments is very difficult and usually requires a large number of samples.

Thus assessing the contribution of watershed activities to lake water quality deterioration is very difficult. However, it would seem appropriate to suggest that the suspended sediment concentration in any inflow to the lake (creeks or ditches or drainage of any kind) be less than 25 mg/L in any grab sample. This critierion has been generally accepted as a level to protect water bodies used for recreation and to protect aquatic life. Another more sensitive use is drinking water. To protect this use turbidity shall not exceed 5 NTU in any grab sample taken within 10 m of a domestic intake, nor shall the mean from at least 10 such samples taken throughout the year exceed 1 NTU.

8.1.3 FECAL COLIFORMS

A notable problem for Shawnigan Lake is the dependence on septic-tanks and tile fields to dispose of sewage. Since the lake serves as a source of the water for a variety of uses, the danger of contamination is a factor which must be considered. The sampling which was done (section 7.3.6) showed that although the water met the standards for body contact, water drawn from the lake required some form of disinfection before use. This is a particular concern for the many small intakes drawing domestic water and is one of most serious problems identified by this study.

It would be an admirable goal to have Shawnigan Lake water without any bacteria indicating health risk for drinking water. However, zero concentration of coliform bacteria would appear to be unreasonable considering the density of population and the inevitable presence of wildlife and birds in the watershed. The present fecal coliform concentration is well within the guideline suggested for domestic water supply; that is, an MPN of less than 10 (for raw water, with disinfection) (the B.C. Ministry of Health, 1982). Since use of surface water for domestic purposes without disinfection is not recommended, no other more stringent standard exists for bulk water supply. The standard of zero MPN at the tap is the only other applicable standard which can be cited.

For Shawnigan Lake the provisional objective which is recommended is a fecal coliform concentration of less than 10/100 mL (90th percentile of samples taken in any 30 day period within 10 m of a domestic intake). This objective will protect domestic water supplies receiving only disinfection.

8.2 PROTECTION OF WATER QUALITY

There are a number of fortunate natural factors which act to minimize the effects of a relatively large watershed population on the water quality of the lake. The lake has a favourable water residence time (flushing rate) of one year which tends to "flush" a significant portion (approximately half) of the phosphorus loading through the outlet. Also, the hydrologic regime is favourable for minimizing effects of nutrients since the major input of nutrients is in the late autumn. The heavy rainfalls which occur in late October or November transport nutrients from decaying vegetation (leaf fall), disturbed soils or erosion (suspended sediments) and nutrients from sewage disposal which would be moved through the soil as the soils become saturated. Since the nutrients enter the lake in late autumn or early winter there is not the response by algae to this input, as there might be in spring or summer, since light is limiting their growth. Because of the high flow-through volume, many nutrients are flushed through the lake and out before they are manifested as algal growth.

Another factor influencing water quality is the location in the watershed of the most significant sources of loading (on a mass/area basis) of nutrients (and suspended sediments and coliforms). These sources are the Village area and to a lesser extent the West Arm area. They are in close proximity to the outlet and so the inputs from these areas affect a relatively restricted area and much of the material is lost through the outlet. If these areas were located in the inlet area or in a central area of the lake, the effect would be very different and the consequences would likely be more severe.

In Shawnigan Lake phosphorus is clearly the nutrient limiting algal growth. It is fortunate that the soils have a very low supply of phosphorus and the soil characteristics are such that phosphorus is well bound - especially from sewage disposal. Any measures which are considered to protect the lake from eutrophication should be directed toward minimizing input of phosphorus to the lake.

8.2.1 LAND USE

A number of activities in the watershed can have an effect on water quality. These include removal of forest cover, disturbance or removal of soil or ground cover and drainage or ditching projects which cause erosion. Any activities which cause increased runoff rate generally lead to increased nutrient loss from the watershed (Gilliom 1980), and watershed development _per se_ causes increased nutrient export (Watson _et al_. 1979, Dillon and Kischner 1975).

Some control of increased water and sediments runoff can be achieved by using settling basins or marsh areas to retain a portion of the material transported.

Logging or extensive tree cutting over a large fraction of the watershed in a short period should be discouraged as it would likely lead to increased supply of nutrients to the lake, according to the literature. Hartman (1981) reported an increase of approximately twofold between pre and post logging stream phosphorus concentrations. Schindler _et al_ (cited in NCASI, 1981) demonstrated increases in export of dissolved and particulate P after tree blowdown in a watershed. Dillon and Rigler (1975) give export coefficients on sedimentary geology for forested watersheds as less than half of watersheds which are at least 15% cleared. The lake sediment stratigraphy for the lake, described earlier, showed that the significant change in the diatom species composition seemed to be a consequence of the extensive watershed logging around the turn of the century. It should also be noted that the age of a forest stand affects the rate of loss of soil nutrients. A vigorously growing youthful forest places greater demand on the soil nutrient pool than a very young or scenescent over mature forest, so the nutrient loss from a forested area varies cyclically over the forest rotation. As the second growth forest reaches maturity in the Shawnigan Lake watershed, the flow of nutrients into the lake can be expected to increase to some extent naturally, in the absence of any human disturbances. disturbances.

Another minor source of phosphorus could be from lawn or garden fertilizers. Since no direct control is possible of this source, the only feasible effort would be to make homeowners (and resort/institutions) aware that over-fertilization leads to loss into the lake and deterioration of water quality. Some form of public information or education may be useful.

8.2.2 SEWAGE DISPOSAL

In dealing with disposal of sewage in the Shawnigan Lake watershed, there are two contaminants which must be considered: nutrients and bacteria. For the latter, Sewage Disposal Regulations exist pursuant to the Health Act to protect the population from the danger of transmission of disease from sewage. These regulations are administered by the Ministry of Health and regional Medical Health Officers. However, for nutrient loss from sewage disposal there are no existing regulations.

In many cases, septic tank/tile field systems may contribute significant amounts of nutrients although the systems are performing entirely satisfactorily in elimninating the hazards of pathogenic organisms. It would seem appropriate in the case of water bodies such as Shawnigan Lake, that possible nutrient contribution be taken into account when new systems are installed. This would include consideration of soil suitabililty for removing nutrients, allowing a greater than minimum required distance from water courses (including drainage ditches and distance above groundwater) and providing for regular maintenance of the system. The latter is necessary since pumping of sludge on a regular basis (3-5 years) is a prerequisite for efficient functioning of a septic system for removal of both pathogens and nutrients.

8.3 RECOMMENDATIONS FOR PROTECTION OF WATER QUALITY

1. The authorities responsible for zoning and subdivision in the Shawnigan Lake watershed (Ministry of Highways, Regional District) should take appropriate measures to minimize the generation and transmission of nutrients, particularly phosphorus which could enter the lake. These measures might include control of land clearing, drainage, ditching, erosion or other consequences of development in the watershed. This could be done by specifying the time of year construction could take place or by having drainage (especially storm drainage) controlled by retention ponds or by directing the water and materials to areas which would minimize their impact.

2. Measures should be considered which would minimize the contribution of phosphorus from both new sewage disposal facilities and existing systems. This would include taking into account the recommendations of Wiens and Nagpal (1983) who identified those soils most suitable for sewage disposal and those soil characteristics which would maximize the field efficiency.

3. Higher standards for outside disposal systems should be considered (deeper soil, increased set-backs) in approval of new septic systems or alternatives. These should reflect the need to reduce the loss of nutrients as well as pathogen removal.

4. Other measures should be considered to reduce nutrient loadings. These might include mandatory septic tank maintenance (pump-outs) or different methods of on-site disposal such as construction of a septic field so that a group of houses on unsuitable soil might dispose of their combined effluent to an area of soil which was more suitable. Anotner possibility would be to consider establishing a management authority to ensure that septic systems are correctly installed, maintained and upgraded as needed. An example of this type of administrative system is described in Nova Scotia (1983).

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TABLE 1

LAKE MORPHOMETRY

Lake:	Surface area 537 ha
:	maximum depth 50 m
:	mean depth 12 m
:	% littoral 25 (area <6 m)
:	mean surface elevation 115 m
:	lake volume 64 057 dam^3
:	mean water residence time 1.0 yr

Watershed	- area 69.4 km^2
	- max elevation 485 m
	- mean elevation 230 m

TABLE 2
WATER QUALITY MONITORING LOCATIONS

1.	Lake	1199901	Deepest point of main basin, depth 45 m
		1199902	Deep point in south basin midway between Memory Island and west shore of lake, depth 19 m.
		1199903	West Arm, mid channel directly in line with A-frame home, depth 3 m.
		1199904	North end of lake, 25 metres west of Shawnigan railway station.
		1199905	Near lake outlet, 100 metres south of outlet.
2.	Streams	1199906	Shawnigan Creek inflow
		1199907	"Round House Creek"
		1199908	Unnamed Creek - first creek north of Round House Creek
		1199909	McGee Creek
		1199910	Unnamed Creek - second creek north of McGee Creek.
		1199911	West Arm inflow
		1199912	Shawnigan Creek outflow
		1199913	Landfill Creek, site 1
		1199914	Landfill Creek, site 2
		1199915	Village Creek
		1199916	East Shawnigan Creek

for locations see Figure 1. Last digits in EQUIS number correspond to number in Figure 1

TABLE 3

LICENCED WATER WITHDRAWALS - SHAWNIGAN LAKE

TYPE OF USE	NO OF LICENCES	TOTAL QUANTITY	COMMENTS
Shawnigan Lake			
Domestic	124	66 376 m^3/d	most are for 2.27 m^3/d
	1	3 201 m^3/d	Victoria Aqua ski Co-operative
Industrial	3	40.9 m^3/d	Restaurant & resort
	1	68.2 m^3/d	Sherwood Waterworks Ltd
	1	272.8 m^3/d	Shawnigan Lake school
Waterworks	5	876 m^3/d	Dougan, Victor, R.
	7	1 130 m^3/d	Sherwood Waterworks Ltd
Irrigation	3	7 277.5 m^3	
Storage	1	1 234 dam^3	Mill Bay Waterworks District
Shawnigan Creek			
Domestic			
Upstream lake	2	9.1 m^3/d	
Downstream lake	5	11.4 m^3/d	

TABLE 4

SHAWNIGAN LAKE OUTFLOW VOLUMES (dam^3)

	1975-76	1976-77	1977-78	1978-79	1979-80	1980-81	1981-82	1982-83	1 in 10 Low Flow
	1975	1976	1977	1978	1979	1980	1981	1982	
Mar.	12 700	13 340	14 960	4 525	13 300	12 540	5 790	10 200	5 150
Apr.	4 514	6 970	2 855	3 355	2 850	3 550	4 930	2 510	2 310
May	1 550	1 810	813	1 120	1 120	1 070	2 765	1 300	846
June	367	545	372	325	133	970	1 530	78	115
July	93	230	95	35	30	750	675	103	25.3
Aug.	78	93	26	1	1	260	160	155	0.3
Sept.	71	46	0	6.2	4	100	8.6	43	3.3
Oct.	4 110	34	30	11	24	150	1 440	60	7.9
Nov.	23 040	73	5 685	57	139	5 190	8 735	780	54
Dec.	23 175	195	18 450	3 270	27 700	24 390	22 800	13 500	2 100
	1976	1977	1978	1979	1980	1981	1981	1983	
Jan.	22 614	4 480	11 500	2 880	13 750	11 935	19 800	19 540	2 890
Feb.	15 900	6 400	7 845	12 400	13 925	15 220	18 770	17 370	6 380
Total	108 212	34 216	62 631	27 985	73 006	76 125	87 400	65 640	19 880
TOTAL OUTFLOW FROM OCTOBER THROUGH FEBRUARY									
	88 839	11 182	43 510	18 618	55 538	56 885	71 545	52 280	11 430

TABLE 5

WATER EXCHANGE RATES (yr^{-1})

	1975-76	1976-77	1877-78	1978-79	1979-80	1980-81	1981-82	1982-83	1 in 10 low flo
	1975	1976	1977	1978	1979	1980	1981	1982	
Mar.	0.20	0.21	0.24	0.07	0.21	0.20	0.09	0.16	0.09
Apr.	0.07	0.11	0.05	0.05	0.05	0.06	0.08	0.04	0.04
May	0.02	0.03	0.01	0.02	0.02	0.02	0.04	0.02	0.01
June	<0.01	0.01	0.01	0.01	<0.01	0.02	0.02	<0.01	<0.01
July	<0.01	<0.01	<0.01	<0.01	<0.01	0.01	0.01	<0.01	<0.01
Aug.	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
Sept.	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
Oct.	0.07	<0.01	<0.01	<0.01	<0.01	<0.01	0.02	<0.01	<0.01
Nov.	0.37	<0.01	0.09	<0.01	<0.01	0.08	0.14	0.01	<0.01
Dec.	0.37	<0.01	0.29	0.05	0.44	0.39	0.36	0.21	0.03
	1976	1977	1978	1979	1980	1981	1982	1983	
Jan.	0.36	0.07	0.18	0.05	0.22	0.19	0.31	0.32	0.05
Feb.	0.25	0.10	0.12	0.20	0.22	0.24	0.30	0.29	0.11
WATER EXCHANGE RATES FOR THE 12 MONTHS PRIOR TO SPRING OVERTURN									
	1.72	0.54	0.99	0.44	1.16	1.2	1.39	1.09	0.33
WATER EXCHANGE RATES FOR THE 5 MONTH PERIOD PRIOR TO SPRING OVERTURN (OCT. THROUGH FEB.)									
	1.4	0.18	0.69	0.30	0.88	0.90	1.13	0.87	0.19

TABLE 6

LAKE WATER QUALITY DATA SUMMARY

	Main Basin 1199901 (6-76 to 5-82)				South Basin 1199902 (6-76 to 9-80)				West Arm 1199903 (6-76 to 9-78)			
Characteristic	mean	min	max	n=	mean	min	max	n=	mean	min	max	n=
pH (lab)	7.2	6.5	7.9	91	7.1	6.5	7.6	70	7.3	6.8	7.5	24
Residue total 105°C	40.0	36	50	62	42.9	34	54	46	40.9	34	48	14
Residue 550°C	20.1	11	28	13	22.7	18	28	6	-	-	-	-
Residue filterable 105°C	38.6	32	50	60	39.6	30	46	46	40	32	48	16
Residue NF 550°C	1.3	<1	3.0	13	2.6	<1	8.0	36	-	-	-	-
Residue volatile	1.1	<1	2	6	1.6	<1	3.0	3	-	-	-	-
Specific Conductance (Field)	43.0	35	50	34	43.1	38	52	27	-	-	-	-
Specific Conductance (lab)	51.5	44	59	91	52.1	44	66	70	53.7	46	60	24
Turbidity (NTU)	0.7	0.3	1.6	24	1.5	0.4	7.4	14	0.82	0.6	1.1	5
Extinction depth (Secchi) m	6.0	1.2	8.4	29	5.9	3.7	8.3	25	-			
color TAC (TAC units)	5.8	1.0	11.0	79	6.7	2	37	56	5.7	3	11	18
Total alkalinity	16.2	14.7	17.9	27	16.3	13.5	20.2	17	16.7	15.5	18.1	4
Carbon organic	3.2	<1.0	7.0	77	3.5	1.0	6.0	61	3.4	1.0	6.0	18
Chloride	4.4	3.7	4.6	4	-	-	-	-	-	-	-	-
Hardness	17.6	14.9	19.1	9	17.8	15.2	19.1	6	-	-	-	-
Nitrogen-ammonia	0.011	<0.005	0.029	93	0.019	<0.005	0.227	68	0.009	<0.005	0.017	23
Nitrogen - Nitrate/nitrite	0.047	<0.02	0.140	97	0.038	<0.020	0.130	72	0.035	<0.02	0.110	19
Nitrate	0.043	<0.02	0.130	60	0.041	<0.020	0.130	49	0.035	<0.02	0.110	19
Nitrogen organic	0.154	0.02	0.400	79	0.170	0.020	0.360	59	0.162	0.07	0.420	20
Chemical oxygen demand	14.6	12.3	16.8	6	13.7	12.3	16.4	3	-	-	-	-
phosphorus ortho	<0.003	<0.003	<0.003	97	<0.003	<0.003	<0.003		<0.003	<0.003	<0.003	
phosphorus total dissolved	0.004	<0.003	0.008	75	0.004	<0.003	0.008	55	0.004	<0.003	0.006	19
phosphorus total	0.006	<0.003	0.010	91	0.006	<0.003	0.015	70	0.005	<0.003	0.009	24
carbon inorganic	3.3	<1	6.0	61	3.2	<1	7.0	50	3.1	<1	5.0	16
Calcium total	6.0	5.7	6.3	12	6.2	5.8	6.7	8	6.4	-	-	1
Magnesium total	1.0	0.9	1.1	12	1.1	1.0	1.2	8	1.1	-	-	-
Sodium total	2.9	2.7	3.1	4	-	-	-	-	-	-	-	-

all values are mg/L unless noted otherwise

TABLE 7

WATER QUALITY DATA SUMMARY: LAKE SHALLOW STATIONS AND OUTFLOW

Characteristic	Near Railway Station 1199904 (05-77 to 01-79)				Near Outlet 1199905 (11-77 to 01-79)				Shawnigan Creek Outflow 1199912 (5-77 to 1-79)			
	mean	min	max	n=	mean	min	max	n=	mean	min	max	n=
pH (relative units)	7.3	6.8	7.6	14	7.3	7.0	7.6	12	7.2	6.7	7.7	28
Residue total 105°C	42.7	38	48	9	41.5	38	46	8	40.8	30.0	54.0	24
Residue filterable 105°C	39.2	34	44	10	38.8	34	42	8	37.9	28.0	44.0	23
Residue non-filterable 105°C	2.6	2.0	4.0	7	2.7	2.0	4.0	6	1.8	<1	90	18
Specific Conductance (μS/cm)	53.4	49	57	10	53.9	51	56	12	53.9	49.0	57.0	28
Turbidity (NTU)	0.8	0.5	1.6	5	0.7	0.5	1.1	5	0.8	0.4	1.8	17
Color (TAC)	5.5	4.0	9.0	8	6.1	4.0	9.0	7	6.0	2.0	10	24
Alkalinity - total	17.3	16.8	17.8	2	16.2	-	-	1	16.4	15.6	17.2	2
Organic Carbon	2.8	<1.0	5.0	6	2.6	<1.0	4.0	7	3.2	2.0	6.0	11
Nitrogen-ammonia	0.008	<0.005	0.021	14	0.008	<0.005	0.014	12	0.011	<0.005	0.038	28
Nitrogen - Nitrate	0.039	<0.020	0.110	27	0.037	<0.020	0.1	23	0.044	<0.020	0.110	28
Nitrogen organic	0.141	0.030	0.230	14	0.173	0.060	0.260	12	0.159	0.060	0.320	28
Phosphorus ortho	all <0.003			97	all <0.003				all but one <0.003			28
Phosphorus total dissolved	0.003	<0.003	0.005	13	0.004	<0.003	0.005	12	0.004	<0.003	0.009	28
Phosphorus total	0.005	<0.003	0.010	14	0.005	<0.003	0.007	12	0.007	<0.003	0.021	28
Inorganic carbon	2.8	2.0	3.0	6	3.3	2.0	5.0	7	2.9	1.0	5.0	11
Calcium total	6.3	-	-	1	5.9	-	-	1	6.0	5.3	6.9	15
Magnesium total	1.1	-	-	1	1.0	-	-	1	1.0	1.0	1.1	15
Sodium total	-	-	-	-	-	-	-	-	3.0	2.9	3.2	2

all values mg/L unless noted

TABLE 8

WATER QUALITY DATA SUMMARY: INFLOW STREAMS I

Characteristic	Shawnigan Creek Inflow 1199906				Roundhouse Creek 1199907				Creek North of Roundhouse 1199908			
	mean	min	max	n=	mean	min	max	n=	mean	min	max	n=
pH (relative units)	7.1	6.7	7.5	31	6.8	6.4	7.2	20	6.8	6.7	7.0	11
Residue total 105°C	44.4	30	82	29	42.1	30	70	19	37.8	30	42	11
Residue filterable 105°C	39.6	28	80	28	40.3	28	66	18	35.8	28	40	11
Residue non-filterable 105°C	1.5	<1	4	28	1.3	<1	4	18	1	<1	1	11
Specific Conductance (μS/cm)	50.8	37	111	31	49.2	40	95	20	42.3	37	48	11
Turbidity (NTU)	0.65	0.4	1.6	19	0.47	0.2	0.9	14	0.5	0.2	1.1	11
Color (TAC)	10.7	5	22	29	8.9	5	23	17	8.8	6	12	11
Alkalinity total	8.4	-	-	1	10.8	-	-	1	10.8	-	-	1
Organic carbon	3	2	5	11	3	2	5	9	2.6	2	3	8
Chloride	3.9	3.8	4	2	2.5	2.3	2.9	3	2.3	-	-	1
Nitrogen-ammonia	0.008	<0.005	0.015	31	0.009	<0.005	0.229	20	0.008	<0.006	0.011	11
Nitrogen - Nitrate	0.225	0.040	0.640	31	0.405	0.080	1.4	37	0.152	<0.020	0.280	11
Nitrogen organic	0.108	0.04	0.260	31	0.139	<0.01	0.380	20	0.074	0.01	0.14	11
Phosphorus ortho	0.003	<0.003	0.005	31	<0.003	<0.003	<0.003	18	all except one		<0.003	11
Phosphorus total dissolved	0.004	<0.003	0.007	27	0.004	<0.003	0.005	18	0.003	<0.003	0.004	11
Phosphorus total	0.005	<0.003	0.007	31	0.004	<0.003	0.006	20	0.003	<0.003	0.004	11
Inorganic carbon	1.6	<1	3	11	2.1	1	3	9	2.5	1	4	8
Calcium total	4.2	3.2	5.3	14	5.5	4.3	6.6	13	4.8	3.7	5.6	11
Magnesium total	0.9	0.7	1.1	14	0.9	0.7	1	13	0.8	0.7	1	11
Sodium total	3.3	3	3.6	2	2.5	1.9	3.0	2	2			1

All values mg/L unless noted.

1199906 - Sampled from May 1977 to March 1979.

1199907 - November 1977 to March 1979.

1199908 - December 1978 to March 1979.

TABLE 9

DATA SUMMARY: INFLOW STREAMS II

Characteristic	McGee Creek 1199909				N. McGee Creek 1199910				West Arm Inflow 1199911			
	mean	min	max	n=	mean	min	max	n=	mean	min	max	n=
pH (relative units)	7.5	7.0	7.9	30	6.6	6.4	6.8	5	7.3	6.7	7.8	29
Residue total 105°C	61.8	42	88	29	40.8	38	49	5	81.7	40	122	26
Residue filterable 105°C	59.9	40	86	28	38.8	36	42	5	79.9	38	120	25
Residue non-filterable 105°C	1.5	<1	3	28	1	1	1	5	2.4	<1	5	14
Specific Conductance (μS/cm)	80.2	57	107	30	50.4	46	57	5	107	48	163	29
Turbidity (NTU)	0.7	0.3	2.1	17	0.8	0.5	1.3	5	1.7	0.7	8.8	20
Color (TAC)	14.0	6	33	26	11.6	7	22	5	20.6	4	44	25
Alkalinity total	26.9	22.3	31.5	2	12.2	-	-	1	41.5	-	-	1
Organic carbon	3.4	1	6	11	2.5	2	3	2	4.3	<1	12	10
Chloride	2.5	2	3	3	3.7	-	-	1	3.3	2.8	3.8	2
Nitrogen-ammonia	0.010	<0.005	0.024	30	0.008	0.006	0.014	5	0.013	0.005	0.026	29
Nitrogen - Nitrate	0.253	0.02	0.83	57	0.192	0.04	0.52	5	0.475	0.05	1.84	29
Nitrogen organic	0.122	0.010	0.310	30	0.08	0.07	0.09	5	0.186	0.03	0.52	29
Phosphorus ortho	all but 2 values <0.003			27	4 of 5 values <0.003				19 of 28 <0.003		0.004	29
Phosphorus total dissolved	0.004	<0.003	0.008	25	0.004	0.003	0.004	5	0.006	<0.003	0.008	25
Phosphorus total	0.005	0.003	0.012	29	0.005	0.004	0.006	5	0.010	<0.003	0.037	28
Inorganic carbon	6.5	4	9	10	4.5	4	5	2	10.5	7	15	10
Calcium total	10.4	6.8	13.3	15	5.3	4.7	6.4	5	14.5	7.9	20.3	15
Magnesium total	2.0	1.1	2.7	15	1.0	0.9	1.1	5	2.0	1.0	2.9	14
Sodium total	2.4	2.1	2.7	2	2.9	-	-	1	3	2.3	3.7	2

All values mg/L unless noted.

McGee Creek: May 1977 to March 1979.

N. McGee Creek: February 1979 to March 1979.

West Arm Inflow: October 1977 to March 1979.

TABLE 10
DATA SUMMARY: INFLOW STREAMS III

Characteristic	Landfill Inflow #1 1199913 (11-77 to 12-78)				Landfill Inflow #2 1199914 (11-77 to 03-79)				Village Inflow 1199915 (11-77 to 03-79)			
	mean	min	max	n=	mean	min	max	n=	mean	min	max	n=
pH (relative units)	7.0	6.7	7.4	10	6.8	6.5	7.1	10	7.4	7.1	8.0	26
Residue total 105°C	153	136	174	8	141.4	114.0	156	10	200.8	164	240	23
Residue filterable 105°C	146	132	172	7	137.6	106.0	154	10	190.5	140	212	23
Residue non-filterable 105°C	3.2	1	7	5	2	1	4	4	10.3	2	42	18
Specific Conductance (μS/cm)	218.2	192	266	10	203.8	175	225	10	292.9	207	231	26
Turbidity (NTU)	1.25	0.9	1.6	2	2.6	1.5	6.6	6	2.4	0.6	5.9	18
Color (TAC)	10.6	7	16	5	27.3	19	35	7	7.2	1	14	23
Alkalinity total	-	-	-	-	45.4	-	-	1	79.9	60.3	99.4	2
Organic carbon	3.5	3	4	2	5	4	6	2	2.1	<1	5	9
Chloride	21.3	20.5	22	2	21.6	17.8	24.5	3	27.4	20.1	31.7	3
Nitrogen-ammonia	0.011	0.006	0.018	10	0.021	0.009	0.04	10	0.118	<0.005	0.780	26
Nitrogen - Nitrate	1.14	0.06	2.41	17	0.821	<0.020	1.7	10	1.811	0.660	3	26
Nitrogen organic	0.272	0.12	0.81	10	0.331	0.200	1	10	0.319	0.140	1	26
Phosphorus ortho	0.005	<0.003	0.008	10	0.006	<0.003	0.016	9	0.018	0.006	0.060	26
Phosphorus total dissolved	0.009	0.007	0.014	10	0.013	0.007	0.028	9	0.066	0.009	0.230	26
Phosphorus total	0.019	0.008	0.039	9	0.019	0.011	0.037	9	0.040	0.015	0.111	26
Inorganic carbon	22.5	20.0	25.0	2	20.5	15.0	26.0	2	25	18	29	9
Calcium total	27.7	-	-	1	22.1	20	23.6	5	38.3	28.8	42.1	13
Magnesium total	6.0	-	-	1	4.4	3.9	4.7	5	5.5	4.0	6.1	13
Sodium total	-	-	-		11.6	-	-	1	11.3	-	-	2

All values mg/L unless noted.
Landfill Inflow #1: November 1977 → December 1978.
Landfill Inflow #2: November 1977 → March 1979.
Village Inflow : November 1977 → March 1979.

TABLE 11
TOTAL PHOSPHORUS LOADING TO SHAWNIGAN LAKE 1977-1979
(kg)

		OUTFLOW	SHAWNIGAN CREEK INFLOW	MCGEE CREEK	AREA A	AREA B	AREA C	VILLAGE* CREEK
1977	OCT	0.07	0.44	0.13	0	0.2	0.8	0.30
	NOV	48.4	13.8	3.3	1.7	5.6	18.3	0.27
	DEC	188	35.9	10.8	5.4	14.4	37.7	0.72
1978	JAN	241	26.6	8.0	4.2	7.5	20.8	0.17
	FEB	39.2	14.8	3.3	2.2	5.2	15.5	0.24
	MAR	40.7	8.0	2.4	1.6	2.8	9.6	0.33
	APR	26.8	6.3	1.9	1.6	0.9	16.6	0.22
	MAY	6.7	2.5	1.0	0.3	0.7	-	0.47
	JUN	1.9	0.9	0.18	-	0.35	-	0.24
	JUL	-	0.19	0.06	-	0.08	-	0.82
	AUG	-	0.15	0.06	-	0.08	-	0.83
	SEP	-	0.26	0.06	-	0.07	-	0.66
	OCT	-	0.22	0.04	-	0.07	-	0.66
	NOV	-	0.30	0.09	-	0.09	-	0.66
	DEC	22.9	6.8	2.0	0.8	3.8	0.04	0.44
1979	JAN	20.1	3.9	2.4	0.8	4.5	11.8	0.42
	FEB	88.3	31.5	9.5	3.7	6.3	28.4	0.44
	MAR	96.2	31.2	11.2	4.4	6.2	18.1	0.27
	APR	20.5	6.3	2.4	0.9	1.8	18.7	0.21
	MAY	6.0	1.5	0.9	-	0.6	-	0.24

	OUTFLOW	SHAWNIGAN CREEK INFLOW	MCGEE CREEK	AREA A	AREA B	AREA C	VILLAGE* CREEK	
Oct/77→June 78	592.8	109.2	31.0	17.0	37.7	119.3	3.0	Σ=317**
July/78→May/79	254.0	82.3	28.7	10.6	23.6	77.0	5.7	Σ=227

all values in kg.

Areal loading from sub-basins (kg/km^2)

	SHAWNIGAN CREEK INFLOW	MCGEE CREEK	AREA A	AREA B	AREA C
Oct/77→June 78	3.4	3.2	2.7	5.9	12.4
July 78→May 79	2.6	3.0	1.7	3.7	8.0

* Village Creek had a ground water source, draining the Village of Shawnigan Lake. Nutrient loadings as a function of area for this creek as the area supplying water to the creek is not known.

** Does not include the outflow loading values.

TABLE 12
TOTAL DISSOLVED PHOSPHORUS LOADING TO SHAWNIGAN LAKE 1977-1979
(kg)

		OUTFLOW	SHAWNIGAN CREEK INFLOW	MCGEE CREEK	AREA A	AREA B	AREA C	VILLAGE* CREEK
1977	OCT	0.05	0.30	0.07	0	0.09	0.62	0.20
	NOV	27.7	11.1	2.5	1.7	3.3	11.6	0.18
	DEC	75.5	17.9	8.0	5.4	9.9	29.6	0.22
1978	JAN	103	21.3	6.4	4.2	6.4	17.6	0.13
	FEB	31.4	11.1	3.3	2.2	3.7	13.3	0.16
	MAR	23	8.0	2.4	1.2	2.4	7.8	0.28
	APR	10	4.7	1.9	1.3	0.9	3.3	0.16
	MAY	3.3	1.7	0.4	0.25	0.5	-	0.11
	JUN	1.0	0.7	0.13	-	0.2	-	0.13
	JUL	-	0.1	0.03	-	0.05	-	0.18
	AUG	-	0.1	0.03	-	0.06	-	0.50
	SEP	-	0.15	0.03	-	0.05	-	0.33
	OCT	-	0.15	0.03	-	0.05	-	0.33
	NOV	-	0.3	0.09	-	0.09	-	0.42
	DEC	13.1	6.8	2.4	0.8	1.4	0.03	0.19
1979	JAN	14.4	3.9	1.6	0.8	1.8	7.8	0.31
	FEB	58.8	21.0	7.1	3.7	4.2	20.5	0.27
	MAR	64.1	31.2	9.4	4.4	3.7	18.7	0.18
	APR	11.7	6.3	1.5	0.76	0.8	2.6	0.16
	MAY	3	1.5	0.5	-	0.5	-	0.12

Oct./79→June/78 274.95	77	25.1	16.3	43.6	83.8	1.6	Σ = 247**
July/78→May/79 165.10	71.3	22.7	10.5	12.7	49.6	3.0	Σ = 169

Areal loading from sub-basins (kg/km^2)

Oct.77→June/78	2.4	2.6	2.5	6.8	8.7
July/78→May/79	2.2	2.4	1.6	2.0	5.2

* See comment on Table 11.
** Does not include the outflow loading values.

TABLE 13
TOTAL NITROGEN LOADING TO SHAWNIGAN LAKE 1977-1979
(kg)

		OUTFLOW	SHAWNIGAN CREEK INFLOW	MCGEE CREEK	AREA A	AREA B	AREA C	VILLAGE CREEK*
1977	OCT	2.7	59	17.5	0	37	13.3	37.7
	NOV	1597	2023	249	333	371	623	35.5
	DEC	4529	143	404	790	754	3301	22.2
1978	JAN	3103	112	383	239	319	1598	17.7
	FEB	1569	590	121	88.7	162	886	18.9
	MAR	1041	439	77.8	120	123	215	20.0
	APR	805	284	85.3	57	47.4	473	13.3
	MAY	168	92	12.7	15.9	14.7	-	13.3
	JUN	68	41	10.2	-	5.6	-	13.3
	JUL	-	10	2.5	-	1.6	-	12.8
	AUG	-	4	2.5	-	1.0	-	15.5
	SEP	-	5	2.5	-	0.7	-	15.5
	OCT	-	9	6.1	-	0.9	-	15.5
	NOV	-	58	22.2	-	5.2	-	21.1
	DEC	817	783	417	303	405	0.9	27.7
1979	JAN	632	617	334	170	225	590	33.3
	FEB	2706	1838	708	394	714	2205	38.9
	MAR	2950	1374	562	187	425	2248	25.5
	APR	672	228	78	27.8	50.5	303	25.5
	MAY	170	90	10	-	14.0	-	13.3

Oct./77→June/78								
	12 882	3793	2133	1643.6	1833.7	7109	191.9	Σ = 16 704**
July 78/→May/79								
	7 947	5006	2142	1081.8	1842.9	5346.9	244.6	Σ = 15 665
Areal loading from sub-basins (kg/km²)								
Oct./77→June/78		118.5	222.2	256.8	286.5	740.5	-	
July/78→May/79		156.4	223.1	169.0	287.8	556.9	-	

* See comment on Table 11.

** Does not include outflow loading values.

TABLE 14
NITRATE NITROGEN LOADING TO SHAWNIGAN LAKE 1977-1979
(kg/)

		OUTFLOW	SHAWNIGAN CREEK INFLOW	MCGEE CREEK	AREA A	AREA B	AREA C	VILLAGE CREEK*
1977	OCT	0.13	34	13.3	0	23.7	6.7	33.3
	NOV	69.2	1385	150	250	277	357	31.1
	DEC	1510	988	243	287	440	2425	17.7
1978	JAN	1265	692	143	127	223	1294	16.1
	FEB	471	259	55	36.9	56.9	664	7.2
	MAR	407	279	47.9	71.7	71.7	59.8	16.1
	APR	134	110	18.9	17.4	31.5	9.5	8.6
	MAY	11	29	3.8	5	5.0	-	10.7
	JUN	3.2	19	5.3	-	1.8	-	11.1
	JUL	-	6.3	1.3	-	0.6	-	11.1
	AUG	-	1.5	1.3	-	0.4	-	11.1
	SEP	-	3.0	1.3	-	0.4	-	11.1
	OCT	-	6.3	4.4	-	1.1	-	11.1
	NOV	-	47.4	17.9	-	3.9	-	13.8
	DEC	98	621	263	202	324	0.6	18.9
1979	JAN	143	460	133	114	170	118	25.5
	FEB	706	1470	300	236	536	1417	26.1
	MAR	898	810	131	100	200	1498	22.2
	APR	175	100	34	20.2	22.8	38.0	22.2
	MAY	15	20	4.0	-	4.5	-	10.0
Oct./77→June/78		3870.5	3795	680.2	795	1130.6	4816	151.9	Σ = 11 368**
July/78→May/79		2035.0	3545.5	891.2	672.2	1263.7	3071.6	183.1	Σ = 9 627
Areal loading for sub-basins (kg/km^2)								
Oct./77→June/78			118.6	70.8	124.2	176.6	501.7	-
July/78→May/79			110.8	92.8	97.4	197.5	320	-

* See comment on Table 11.

** Does not include outflow loading values.

TABLE 15
SUSPENDED SOLIDS INPUT TO SHAWNIGAN LAKE 1977-1979

		OUTFLOW	SHAWNIGAN LAKE INFLOW	MCGEE CREEK	AREA A	AREA B	AREA C	VILLAGE CREEK *	
1977	OCT	13	148	22.2	-	44	22.2	388	
	NOV	6 920	5 543	831	550	1665	831	333	
	DEC	37 745	17 960	2695	1797	3594	5390	266	
1978	JAN	22 992	10 650	1598	1065	2131	1598	277	
	FEB	15 690	7 385	1107	738	1477	1107	166	
	MAR	9 054	3 985	598	398	398	598	111	
	APR	6 710	1 580	1420	1263	631	473	199	
	MAY	2 242	420	127	83	209	-	166	
	JUN	650	148	88	-	29	-	44	
	JUL	-	37	9	-	29	-	466	
	AUG	-	37	9	-	29	-	166	
	SEP	-	37	9	-	15	-	111	
	OCT	-	37	9	-	7	-	111	
	NOV	-	74	9	-	15	-	22	
	DEC	3 268	1 350	200	270	540	2.5	88	
1979	JAN	2 874	1 315	393	262	1050	786	10.8	
	FEB	11 765	10 500	1575	1050	1050	6300	44.4	
	MAR	12 830	6 245	1874	1250	1250	7494	44.4	
	APR	2 920	1 265	380	252	253	379	11	
	MAY	800	400	100	80	100	-	11	
Oct./77→June/78		102 016	47 819	8486	5894	10 178	10 019	1950	Σ = 84 346**
July/78→May/79		34 457	21 297	4467	3164	4 338	14 961	1129	Σ = 49 357

Areal loading from sub-basins (kg/km^2)

	SHAWNIGAN LAKE INFLOW	MCGEE CREEK	AREA A	AREA B	AREA C	VILLAGE CREEK
Oct/77→June/78	1 494	884	921	1 590	1 044	-
July/78→May/79	566	465	494	678	1 558	-

* See comment on Table 11.
** Does not include outflow loading values.

TABLE 16

ESTIMATED PHOSPHORUS LOADING TO SHAWNIGAN LAKE USING VARIOUS MODELS

YEAR	τ	ρ	q_s	TP mgL^{-1}	L_1	L_2	L_3
1976	1.7	0.6	20 1				
1977	0.5	2.0	6.4	0.006	0.179	0.116	0.269
1978	1.0	1.0	11.7	0.006	0.107	0.153	0.150
1979	0.4	2.5	5.2	0.008	0.310	0.143	0.438
1980	1.2	0.8	13.6	0.009	0.150	0.179	0.189
1981	1.2	0.8	14.2	-			
1982	1.4	0.7	16.3	0.007	0.100	0.218	0.114
1983				0.007			
Average:		1.2	12.5				

TP - total phosphorus in mgL^{-1}

L - loading in $g\ m^{-2}yr^{-1}$

Z - mean depth (12 m)

σ - phosphorus sedimentation coefficient (0.49)*

p - flushing rate (yr^{-1})

τ - hydraulic detention time (yr)

q_s - water overflow rate ($m\ yr^{-1}$) (outflow rate ÷ Surface Area)

L1. $TP = \frac{L}{Z(\sigma + \rho)}$ (Vollenweider, 1969)

L2. $TP = \frac{L}{11.6 + 1.2\, q_s}$ (Reckow and Simpson, 1980)

L3. $TP = \frac{0.603L}{Z(.257 + \rho)}$ (Canfield and Bachmann, 1981)

Surface Area $5.37 \times 10^6\ m^2$

* estimated from Larsen and Mercier (1976) Table 4 (mean value)

TABLE 17

HYPOLIMNETIC OXYGEN DEPLETION RATES FOR SHAWNIGAN LAKE

	$mg/m^2/day$
1976	248
1977	216
1978	350
1980	290
mean	276

TABLE 17A

CALCULATED HYPOLIMNETIC OXYGEN DEPLETION RATES ($mg/m^2/day$) AT DIFFERENT PHOSPHORUS CONCENTRATIONS

Concentration (mg/L)	Cornett and Rigler (1979)		Welch and Perkins (1979)
0.006	203*	221**	237
0.008	214	232	260
0.010	226	243	284

* mean weighted hypolimnetic temperature 6.5°C

** mean weighted hypolimnetic temperature 7.0°C

TABLE 18

SHAWNIGAN LAKE WATER CLARITY

DATE	1199901 Secchi (m)	1199901 1% (m)*	1199902 Secchi (m)	1199902 1% (m)
760629			3.6	-
760909	6.1	-	6.1	-
761020	6.1	-	6.1	-
761125	4.6	-	4.0	-
770509	6.3	-	-	-
770511	6.0	-	6.1	-
770525	5.6	-	5.2	-
770613	5.0	-	4.9	-
770802	6.0	13	6.0	11.5
770816	7.8	-	6.9	-
770929	6.3	13.8	5.0	-
771102	5.5	11	-	-
771103	-	-	4.9	12
771205	4.7	13.5	-	-
780220	7.0	-	-	-
780322	6.1	10	5.2	12.5
780417	5.6	11	6.3	10
780508	6.1	10	-	-
780509	-	-	6.4	11
780529	5.6	8.5	6.0	9.0
780614	8.4	9.0	8.4	9.0
780705	7.0	11.0	6.0	11
780725	6.7	-	6.1	-
780828	8.2	-	8.3	-
781116	-	12.2	-	10
790205	-	-	4.8	10
790308	-	9.3	-	8
mean	6.2	11.0	5.8	10.4

* 1% light penetration value represents the depth at which 1% of surface illumination is present.

TABLE 19

DOMINANT NET PLANKTON, SHAWNIGAN LAKE 1977

Date	Dominant net plankton
May 16	Asterionella formosa
	Dionobryon divergens
	Tabellaria fenestrata
	Cyclotella kutzingiana
	Anacystis cyanea
	Spondylosium planum
	Cyclotella bodanica
June 7	Asterionella formosa
	Gomphosphaenia lacustris
	Anacystis cyanea
	Dinobryon divergens
	Cyclotella kutzingiana ,
	Tabellaria fenestrata
June 14	Asterionella formosa
	Gomphosphaeria lacustris
	Anacystis cyanea
	Dinobryon bavaricum
	Tabellaria fenestrata
	Chroococcus turgidus
June 27	Gomphosphaeria lacustris
	Asterionella formosa
	Anacystis cyonea
	Chroococcus turgidus
	Dinobryon bavaricum
July 18	Gomphosphaeria lacustris
	Anabaena flos-aquae
	Anacysis cyanea
August 3	G. lacustris
	Anacystis cyanea
	Anacystis incenta
	Asterionella formosa
	Anabaena flos-aquae
	Rhabdoderma Gorskii
August 17	Anacystis cyanea
	Anabaena flos-aquae
	Rhabdoderma gorskii
	Sphaerocystis schroeteri
	Anabaena limnetica
	Gleotrichia echinulata

TABLE 20

ALGAL BIOMASS (CHLOROPHYLL *a*) FOR SHAWNIGAN LAKE 1977-1979

DATE	DEPTH	STATION 1199901	STATION 1199902	STATION 1199903	STATION 1199904	STATION 1199905
77 08 03	1 m	2.9	2.5			
77 08 17	1 m	2.1	2.1			
78 03 06	1 m	2.7	-			
78 03 22	1 m	2.2	2.3			
78 04 17	1 m	3.1	3.3			
78 05 08	1 m	2.3	1.9			
78 05 29	1 m	2.1	-			
78 06 14	0 m	2.3	2.2			
	1 m	4.2	2.2			
	2 m	6.9	5.2			
78 07 12	0 m	1.9	2.2			
	1 m	2.3	2.3			
	2 m	2.1	5.9			
	3 m	7.8	-			
	4 m	3.3	-			
78 08 08	0 m	0.7	-			
	3 m	2.2	-			
	6 m	2.9	-			
	9 m	2.9	-			
	12 m	4.1	-			
78 08 24	1 m	-	-	1.1	0.8	0.5
78 09 28	1 m	1.4	1.3	1.0	0.6	0.7
78 11 16	1 m	2.4	2.0	2.4	2.1	2.3
79 02 05	1 m	-	2.3	2.0		
mean		2.9	2.9			

Values in μg/L

TABLE 21
SEDIMENT CORE: SIGNIFICANCE OF CHANGES IN SPECIES RANKED ABUNDANCE

SPECIES	AVERAGE RANK Above 6 cm	AVERAGE RANK Below 6 cm	SIGNIFICANCE
Tabellaria fenestrata	3.7	16.5	0.01*
Asterionella formosa	11.0	absent	0.01*
Fragilaria crotonensis	9.2	rare	0.01*
Fragilaria construens	4.0	3.6	.GT.0.5
Fragilaria brevistriata	8.0	6.1	.GT.0.5
Fragilaria pinnata	7.0	6.2	.GT.0.5
Cyclotella bodanica	5.3	5.0	.GT.0.5
Cyclotella Kutzingiana	6.3	1.2	0.01*
Cyclotella stelligera	8.7	4.0	0.01*
Melosira italica	3.0	12.0	0.01*
Melosira distans	2.0	2.0	0.10

* Significantly changed in rank above 6 cm.

The core was obtained from the south basin of Shawnigan Lake in 1978.

For details of the methodology see McKinnell (1978).

The rank represents ranked abundance by proportion determined for each stratum.

TABLE 22
PERIPHYTON BIOMASS 1978

STATION		JULY 4	JULY 24	AUGUST 10
1199903		0.06 (n=1)	0.043 (n=3)	-
1199904		<0.03 (n=1)	0.03 (n=3)	-
1199905		0.04 (n=1)	0.053 (n=3)	-
	(mill)	0.03 (n=1)	0.04 (n=3)	0.055 (n=2)
	(south island)	-	0.03 (n=3)	0.055 (n=2)
	(inlet)	0.13 (n=1)	0.04 (n=3)	-

all values in $\mu g/cm^2$
plates installed June 13, 1978

TABLE 23

SHAWNIGAN LAKE ZOOPLANKTON 1977-78

SITE 1199901: Main Basin	11 May	16 May	25 May	14 June	27 June	18 July	16 Aug	29 Sept	3 Nov	22 Mar	17 Apr	8 May	29 May	25 July	28 Aug
Leptodora kindtii															
Immature cladocerans	24					8	24	32				64			20
Diaphanosoma leuchtenbergianum				4											12
Chydorus sp.	16														
Bosmina longirostris	56	16	8					24		36		240		12	16
Daphnia rosea	136	136	152	112	176	104	320	656		28		112	56	40	356
D. longirgmus	152	916	240	4	40	44	20	8		52		592	312	52	52
Nauplii & Copepodites	328	8	32	84	416	664	376	88		132		224	216	1376	832
Cyclops bicuspidatus	648	408	448	236	504	356	332	1632		472		1056	696	392	1300
Epischura nevadensis	32		32	12	24	96	132	8		16		16	24	20	
Kellicottia longispina	640	640	400	184	208	64	116	72		128		1808	1392	140	268
Keratella cochlearis	1104	2152	760	40	72	68	52	32		52		304	152	52	148
K. quadrata	120	240	96					8		52		176	16		12
Polyarthra vulgaris					64	152	160	112					24	48	28
TOTAL*	3256	4016	2168	676	1504	1556	1532	2672		968		4592	2888	2132	3044
SETTLED VOL. (mL)	0.8	0.8	1.0	.65	0.6	0.45	1.25		1.2	0.7		1.7	1.1	0.5	0.75

TABLE 23 - CONTINUED
SHAWNIGAN LAKE ZOOPLANKTON 1977-78

SITE 1199902: South Basin	11 May	16 May	25 May	14 June	27 June	18 July	16 Aug	29 Sept	3 Nov	22 Mar	17 Apr	8 May	29 May	25 July	28 Aug
Leptodora kindtii									4					4	4
Immature cladocerans	12		16		8		28							28	60
Diaphanosoma leuchtenbergianum															16
Chydorus sp.															
Bosmina longirostris	92								24	216	628	56		28	64
Daphnia rosea	136		168	244	60		136		324	20	48	8	116	28	624
D. longirgmus	184		240	8	32		16		20	132	168	104	536	16	4
Nauplii & Copepodites	48		16	64	228		480		12	128	252	24	108	856	648
Cyclops bicuspidatus	672		416	28	144		232		472	628	1056	728	628	576	1404
Epischura nevadensis	16		40	16	60		60				20	8	12	60	4
Kellicottia longispina	568		640	240	76		12		32	124	948	1016	608	76	320
Keratella cochlearis	452		632	152	28				16	72	344	216	276	16	164
K. quadrata	40			8			12		4	112	636	208	8	8	8
Polyarthra vulgaris					44		112							24	12
TOTAL*	2120		2168	760	680		1088		908	1432	4100	2368	2292	1720	3332
SETTLED VOL. (mL)	0.7	0.6	0.7	0.6	0.55	0.25	1.05		1.2	0.65	1.0	1.6	1.1	0.4	1.0

*numbers represent organisms/sample (11.75 cm net, 15 m (station 02) 20 m (station 01) vertical tow)

TABLE 24
SHAWNIGAN LAKE CRUSTACEAN ZOOPLANKTON SPECIES COLLECTED IN THE 1930's AND 1970's

	Carl (1940)	This Study
Cladocera		
Daphnia rosea (Carl's longispina)	√	√
D. longiremus		√
Bosmina longirostris		√
Sida crystallina	√	√ (1976-rare)
Chydorus sphaericus		√ (rare)
Leptodora kindtii	√	√
Diaphanosoma leuchtenbergionum		√
Ceriodaphnia reticulata		√
Scapholeberis mucronata	√	
Bosmina obtusirostris	√	
Polyphemus pediculus	√	
Copepoda		
Cyclops bicuspidatus	√	√
Epischura nevadensis	√	√
Diaptomus oregoneusis		√

TABLE 25
TOTAL AND FECAL COLIFORM CONCENTRATIONS 1976-1979

SITE	1199901		1199902		1199903		1199904		1199905	
	TOTAL	FECAL	TOTAL	FECAL	TOTAL	FECAL	TOTAL	FECAL	TOTAL	FECAL
DATE										
76 06 29	<2	-	2	-	-	-	-	-	-	-
76 10 20	<2	<2	-	-	6	4	-	-	-	-
76 11 25	23	5	-	-	11	<2	-	-	-	-
77 02 13	-	-	-	-	23	12	-	-	-	-
77 03 10	-	-	-	-	33	17	350	170	17	8
77 05 09	<2	<2	<2	<2	-	-	-	-	-	-
77 05 11	<2	<2	<2	<2	<2	<2	-	-	-	-
77 05 16	<2	<2	<2	<2	-	-	-	-	13	<2
77 06 13	<2	<2	-	-	23	12	-	-	-	-
77 07 05	2	<2	2	<2	8	8	-	-	-	-
77 08 11	<2	<2	-	-	-	-	-	-	-	-
77 08 16	-	-	2	<2	17	5	-	-	-	-
77 09 29	2	<2	5	<2	-	-	-	-	-	-
77 11 02	49	8	2	2	79	11	17	5	23	8
77 12 05	2	2	-	-	13	2	-	-	-	-
78 04 17	<2	<2	<2	<2	7	5	5	-	-	-
78 05 08	<2	<2	-	-	<2	<2	<2	<2	2	2
78 05 29	<2	<2	-	-	-	-	-	-	-	-
78 06 14	2	2	<2	<2	9	7	2	2	4	2
78 07 05	11	<2	<2	<2	8	2	<2	<2	2	<2
78 08 28	<2	<2	<2	<2	49	17	2	<2	46	31
78 09 20	<2	<2	-	-	-	-	-	-	-	-
78 09 28	-	<2	2	<2	5	2	17	17	5	5
78 11 16 #1	5	2	2	<2	8	<2	7	<2	2	2
78 11 16 #2	-	-	2	<2	-	-	-	-	-	-
79 02 05	-	-	<2	<2	<2	<2	8	<2	4	<2

all values are MPN/100 mL.

TABLE 25 - CONTINUED

SHAWNIGAN CREEK - OUTLET (at bridge)

DATE		TOTAL	FECAL
25/11/76		49	33
10/03/77		94	31
10/05/77		49	33
17/05/77		23	2
24/05/77		33	33
15/06/77		70	70
17/04/78		5	2
MCGEE CREEK (at bridge)			
10/05/77		46	2
17/05/77		140	17
24/05/77		49	<2
15/06/77		110	110
SHAWNIGAN CREEK - INFLOW (at bridge)			
20/10/76		220	130
10/03/76		33	17
10/05/77		62	17
17/05/77		39	8
24/05/77		47	7
15/06/77		79	79
MISCELLANEOUS LAKE SITES (see Black et.al. 1977)			
10/03/77	#1	2	<2
	#2	33	8
	#4	13	5
	#5	17	8
	#6	22	11

TABLE 26

WATER QUALITY CRITERIA FOR PHOSPHORUS, CHLOROPHYLL a, WATER CLARITY, AND HYPOLIMNETIC DISSOLVED OXYGEN AS RELATED TO PARTICULAR WATER USES.

ADOPTED FROM DILLON AND RIGLER (1975)

	Spring Overturn Phosphorus µg/L	Mean Summer Chlorophyll a µg/L	Water Clarity Secchi disc m	Minimum Hypolimnetic Oxygen Concentration µg/L
1.(a) Both body contact water recreation and cold water fishery (trout)	10	2	>5	>5
2. water recreation important but preservation of cold water fishery not imperative.	18	5	2-5	
3. body contact recreation of little importance but warm water fisheries (bass, perch, pike) important	30	10	1-2	
4. suitable only for warm-water fisheries, considerable danger of winter-kill except in deep lakes.	56	25	<1.5	0

Note: The relationship between the eutrophication parameters is discussed in Appendix I.

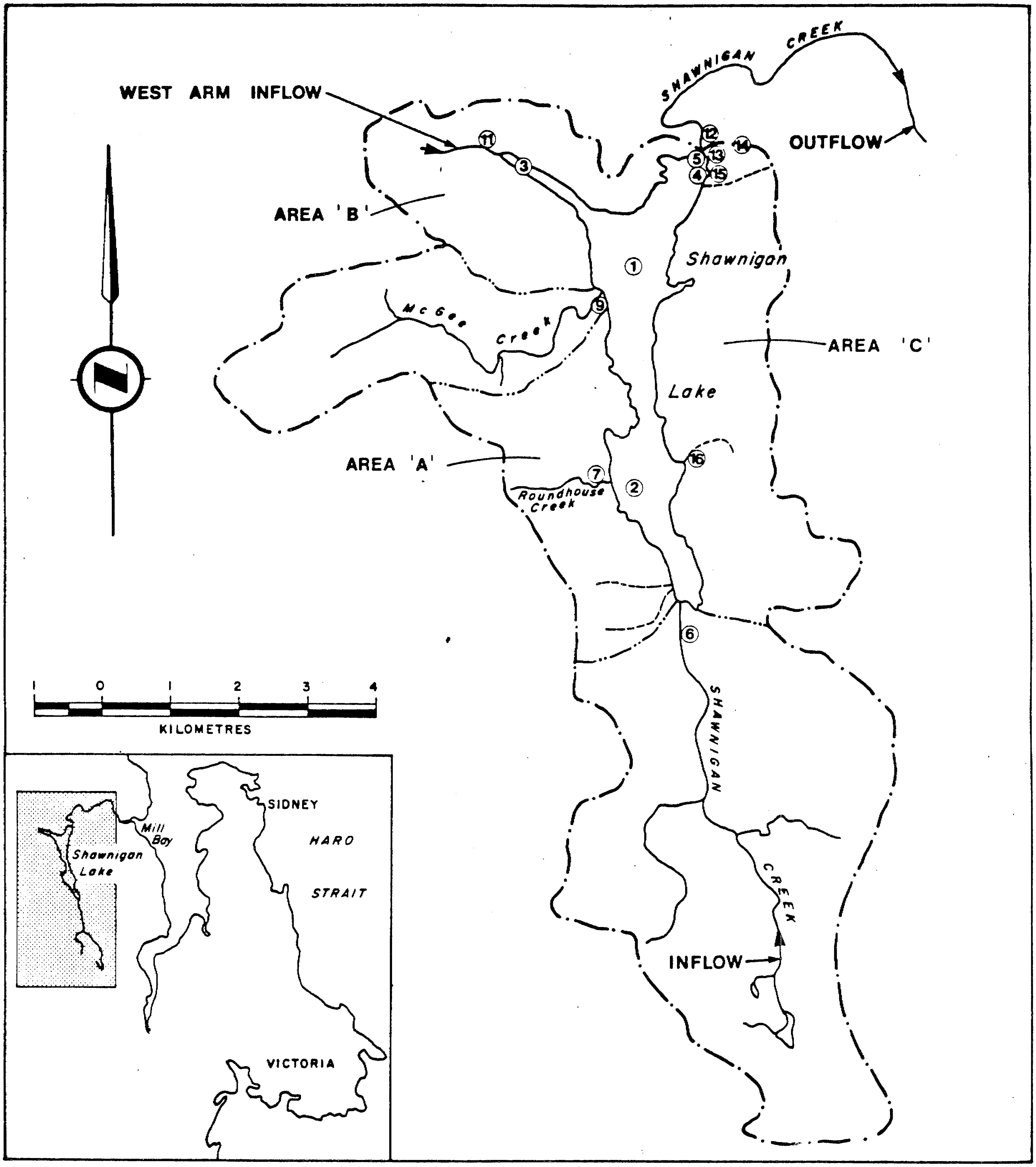


Figure 1 Water Quality Monitoring Sites
(all site numbers are prefixed 11999..)

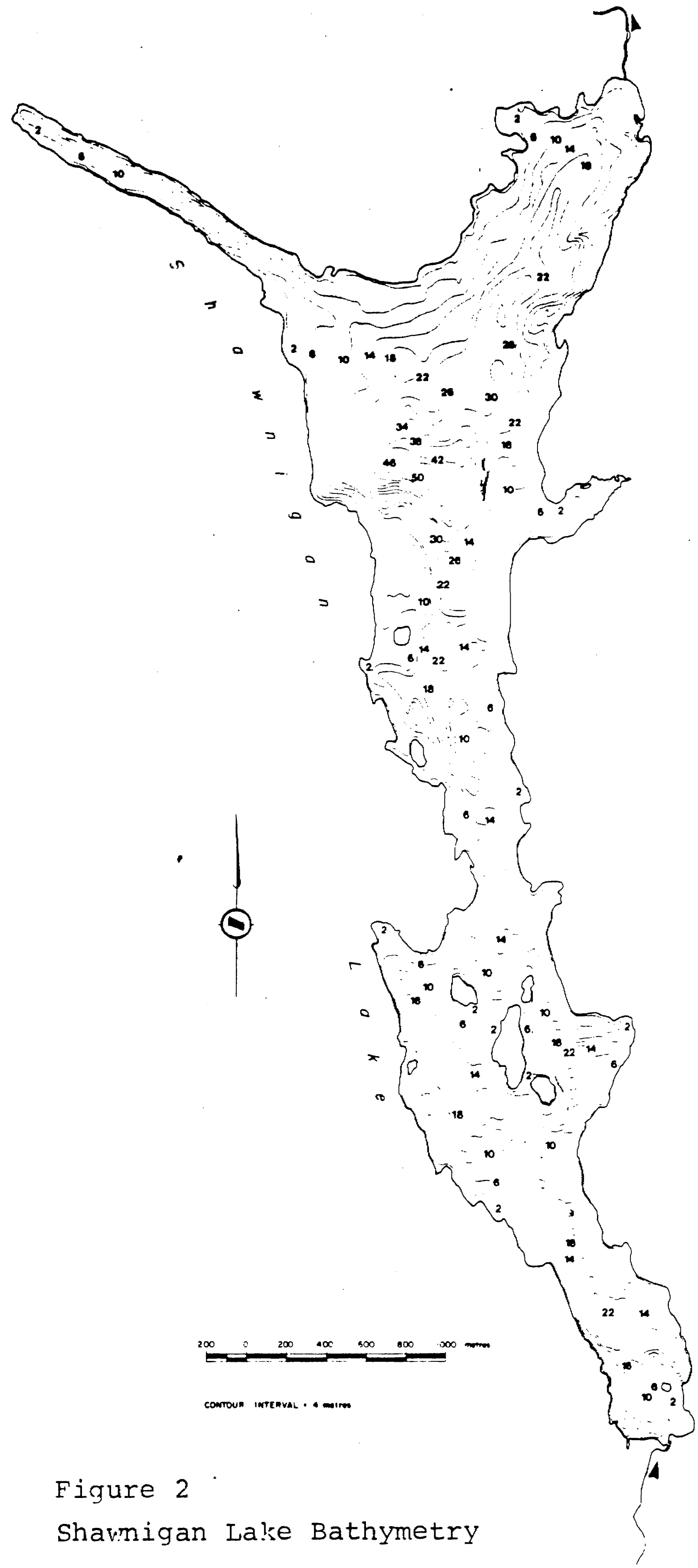


Figure 2
Shawnigan Lake Bathymetry

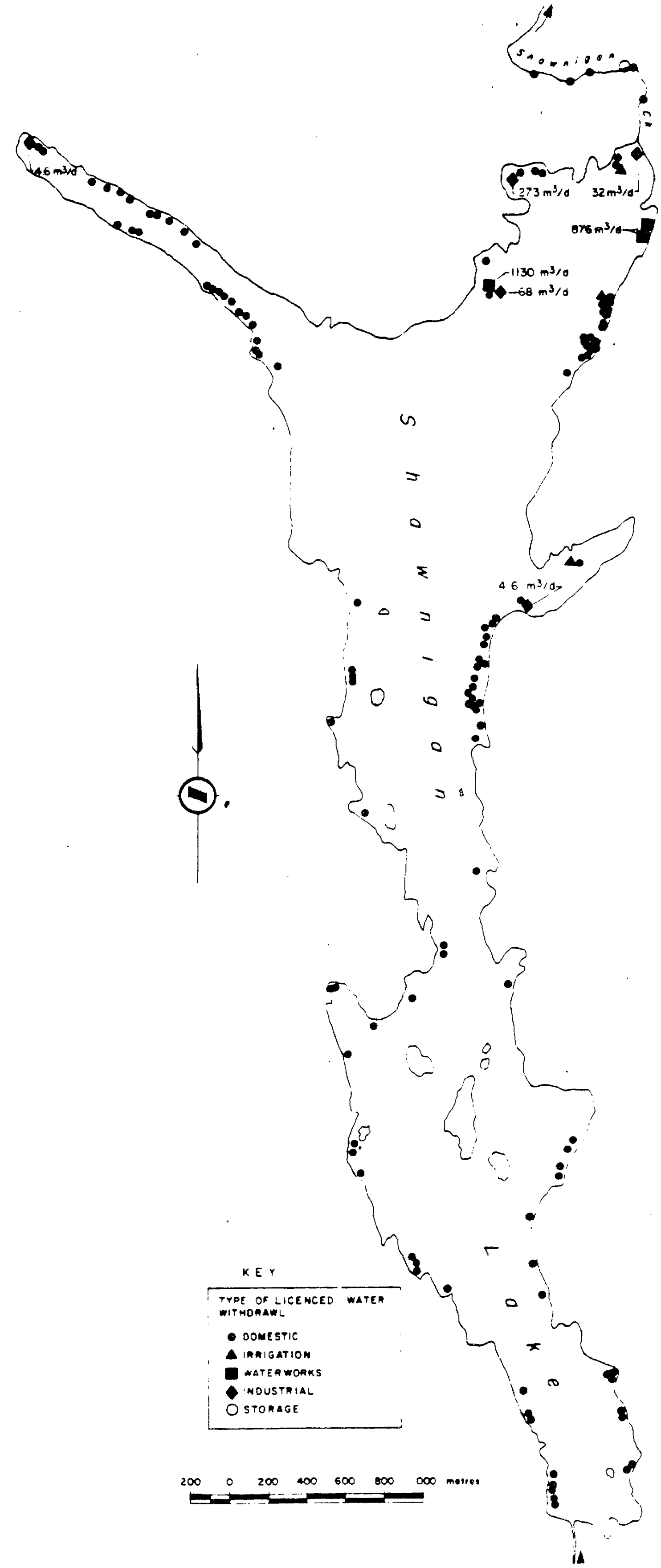


Figure 3 Water Withdrawals from Shawnigan Lake

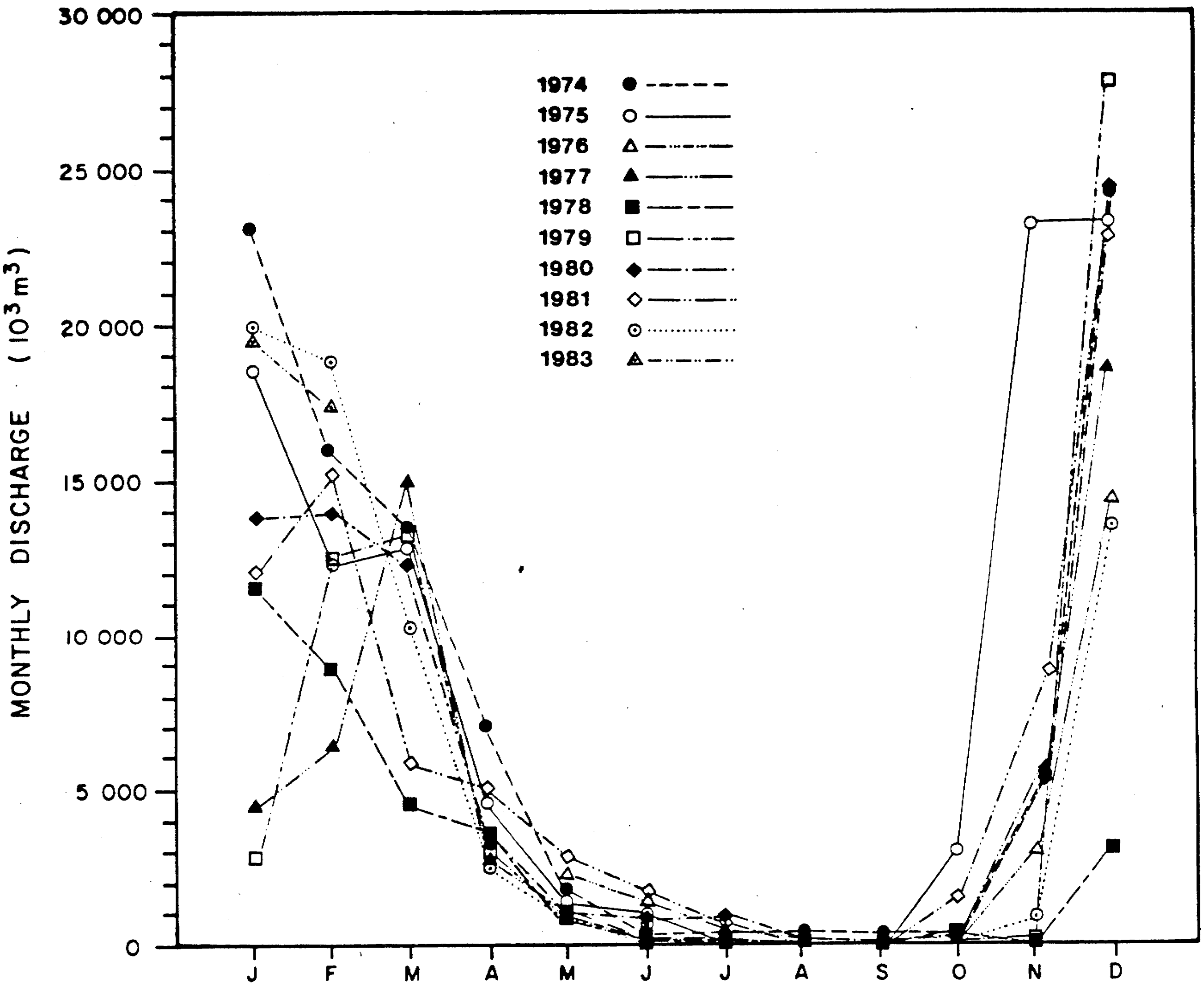


Figure 4 Shawnigan Creek - outflow hydrograph

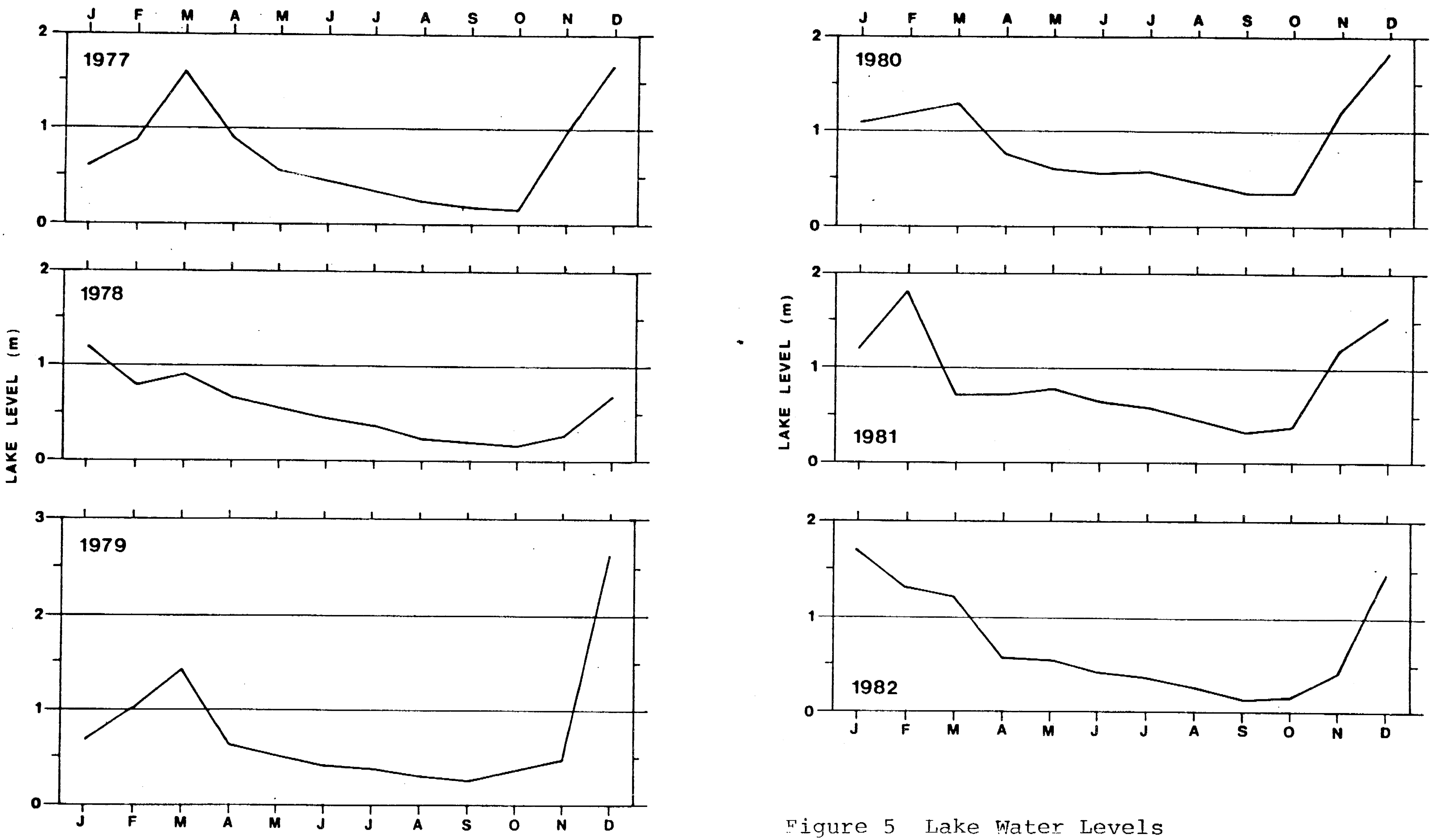


Figure 5 Lake Water Levels

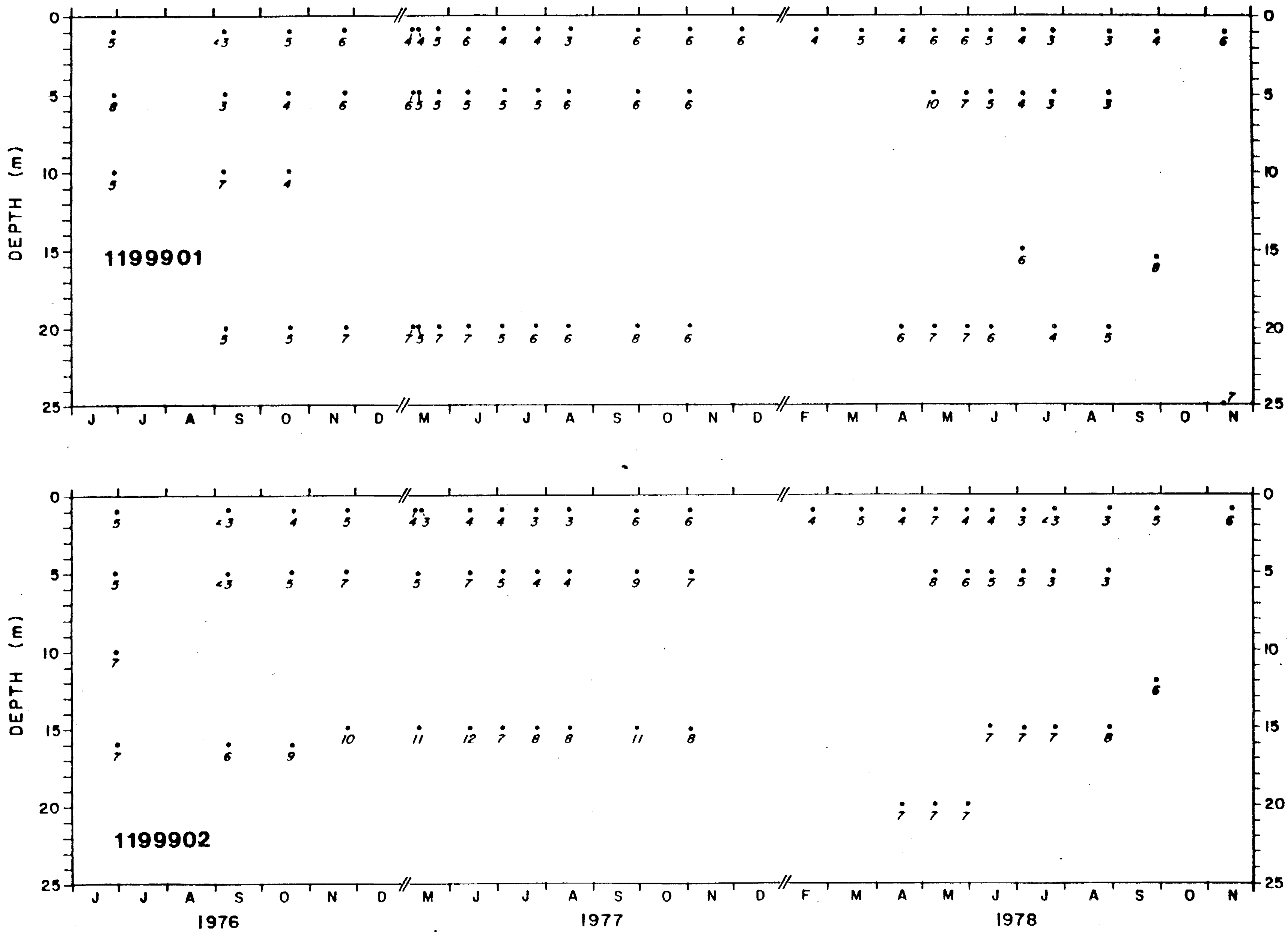


Figure 6 Total Phosphorus time/depth main and south basins 1976-1978
(all values in ug/L)

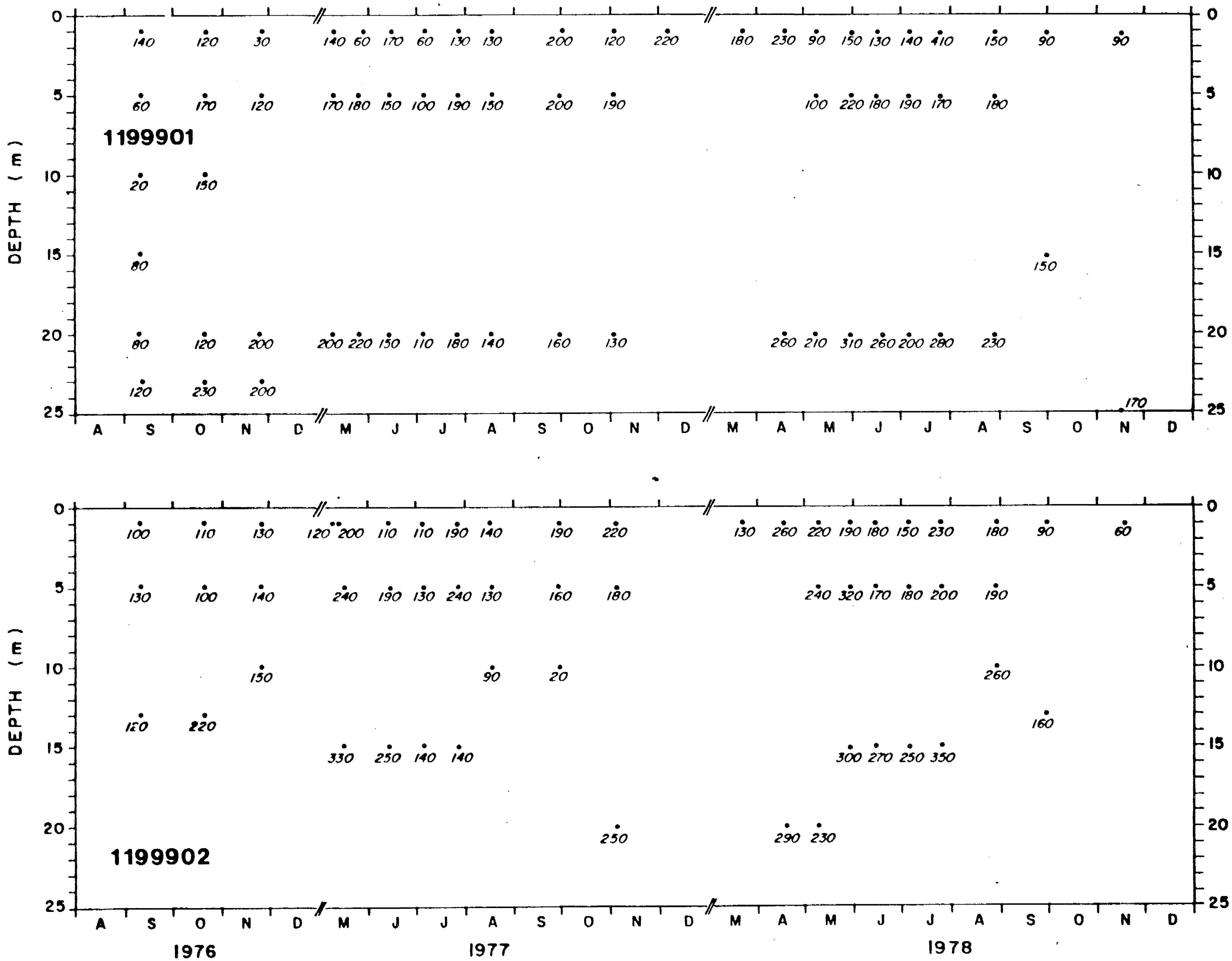


Figure 7 Total Nitrogren time/depth main and south basins 1976-1978
(all values in ug/L)

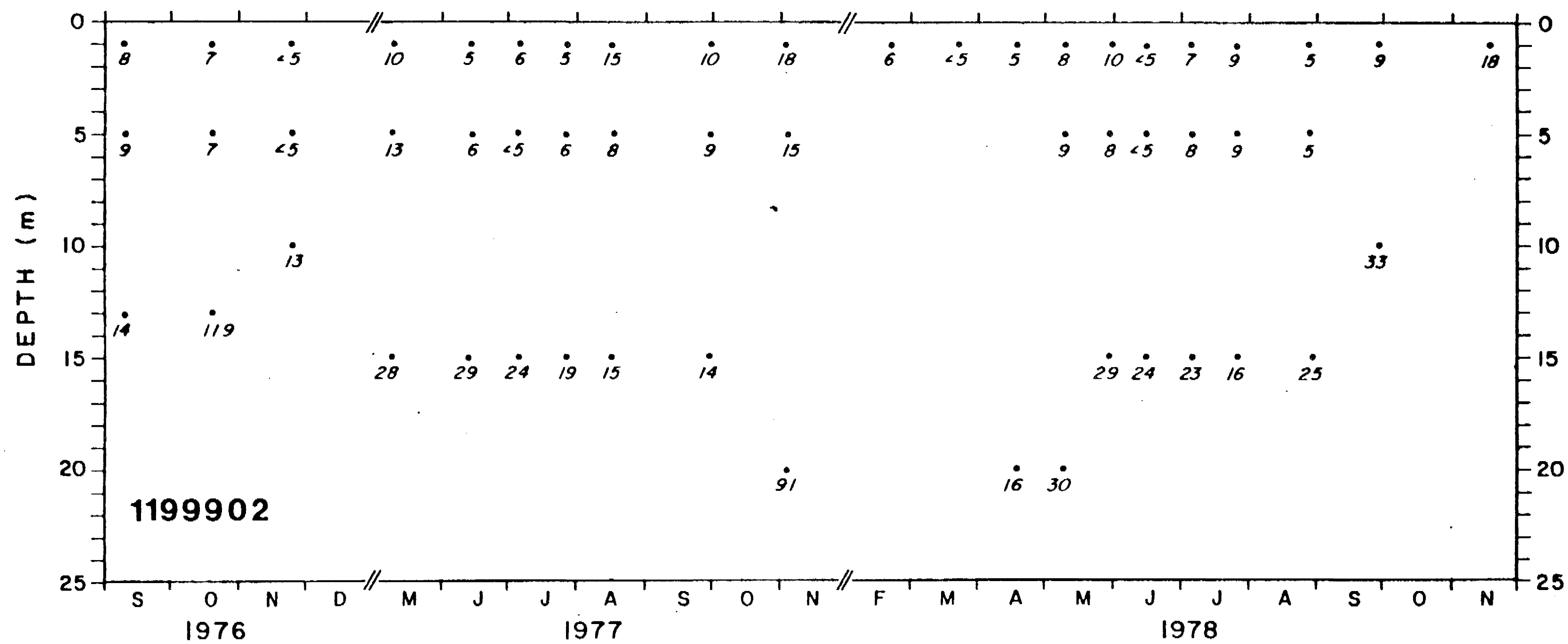


Figure 8 Ammonia time/depth south basin 1976-1978
(all values in ug/L)

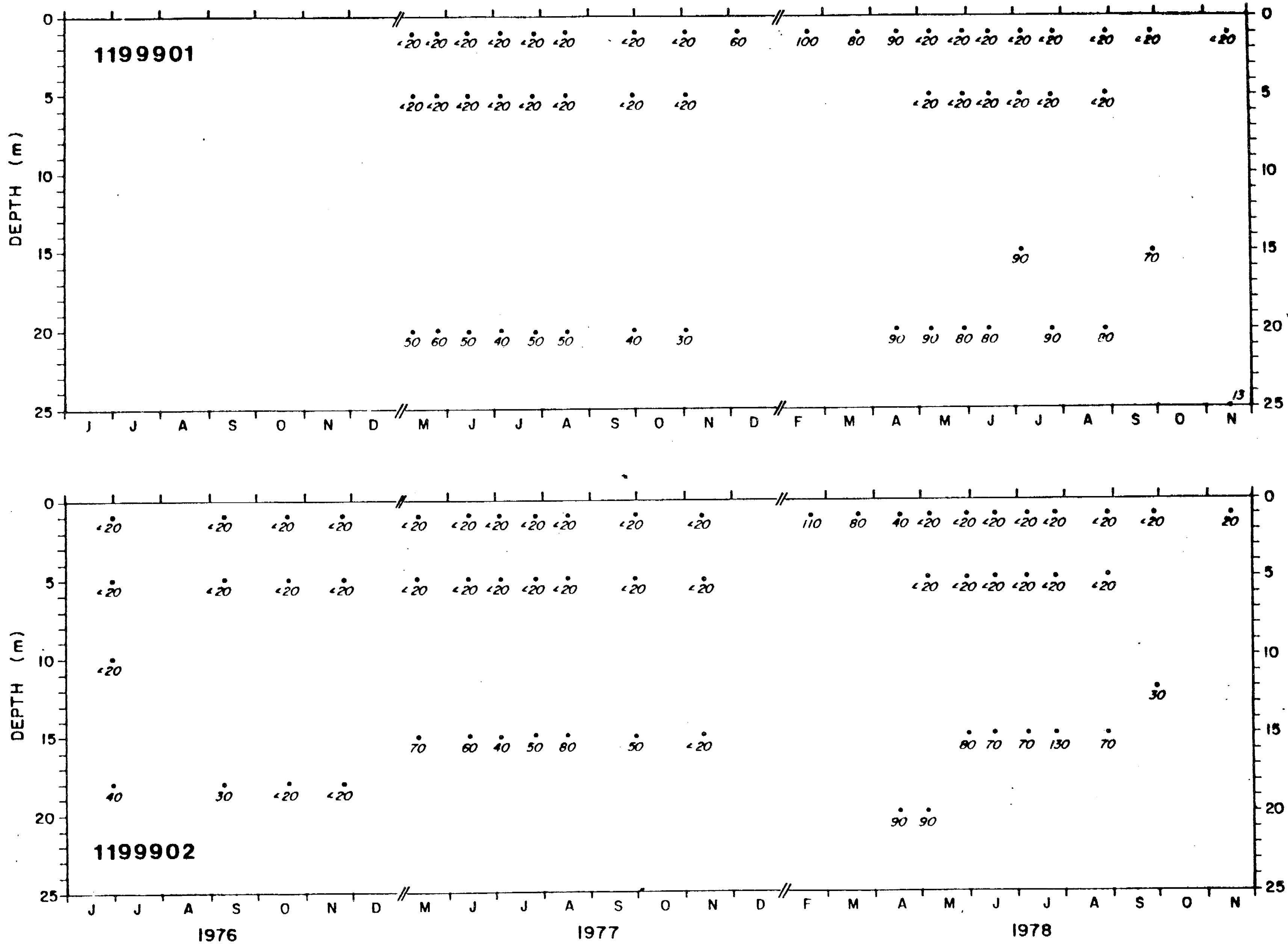


Figure 9 Nitrate time/depth main and south basins 1976-1978
(all values in ug/L)

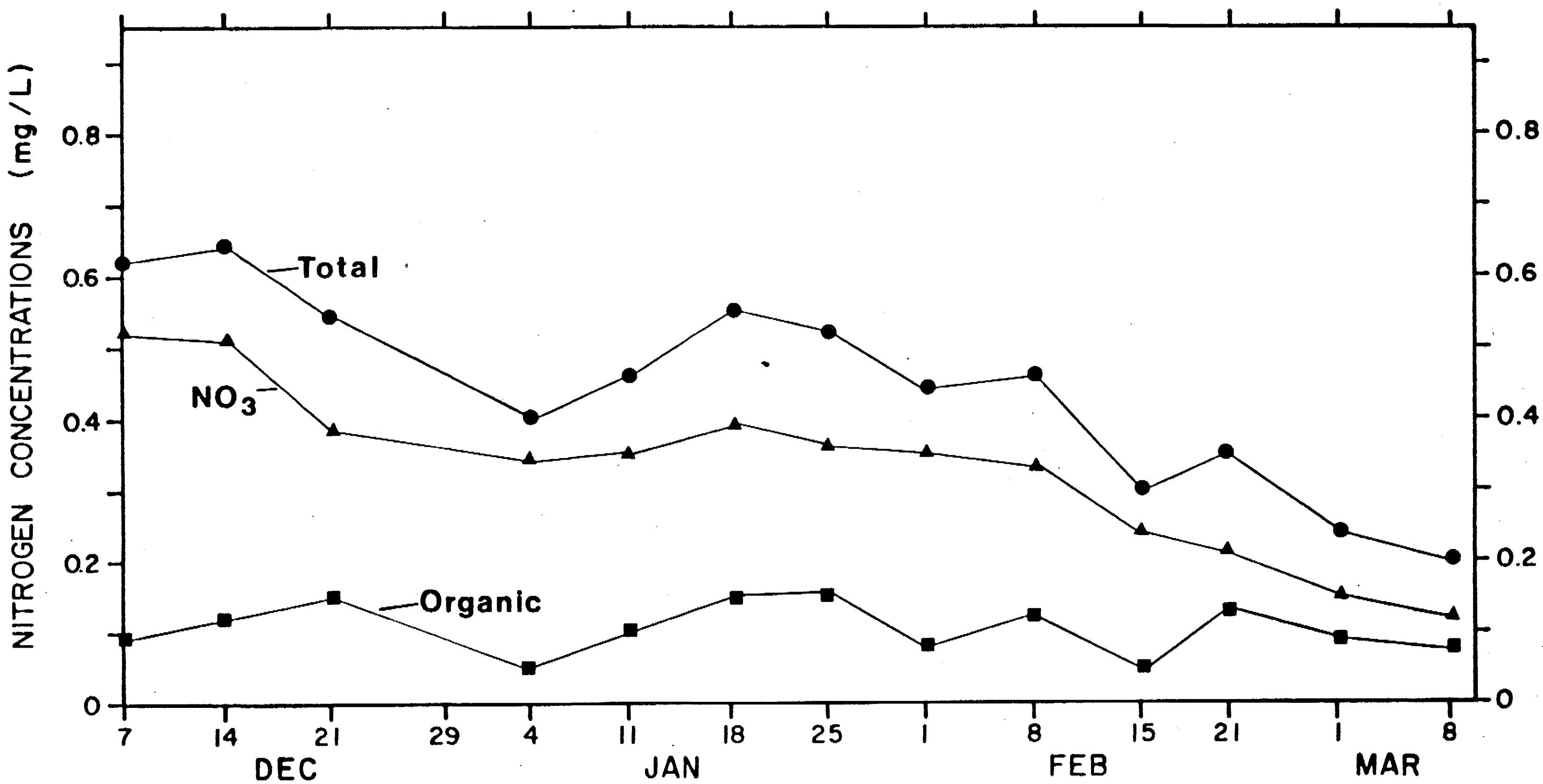


Figure 10 Nitrogen in Shawnigan Creek inflow, 1978-79

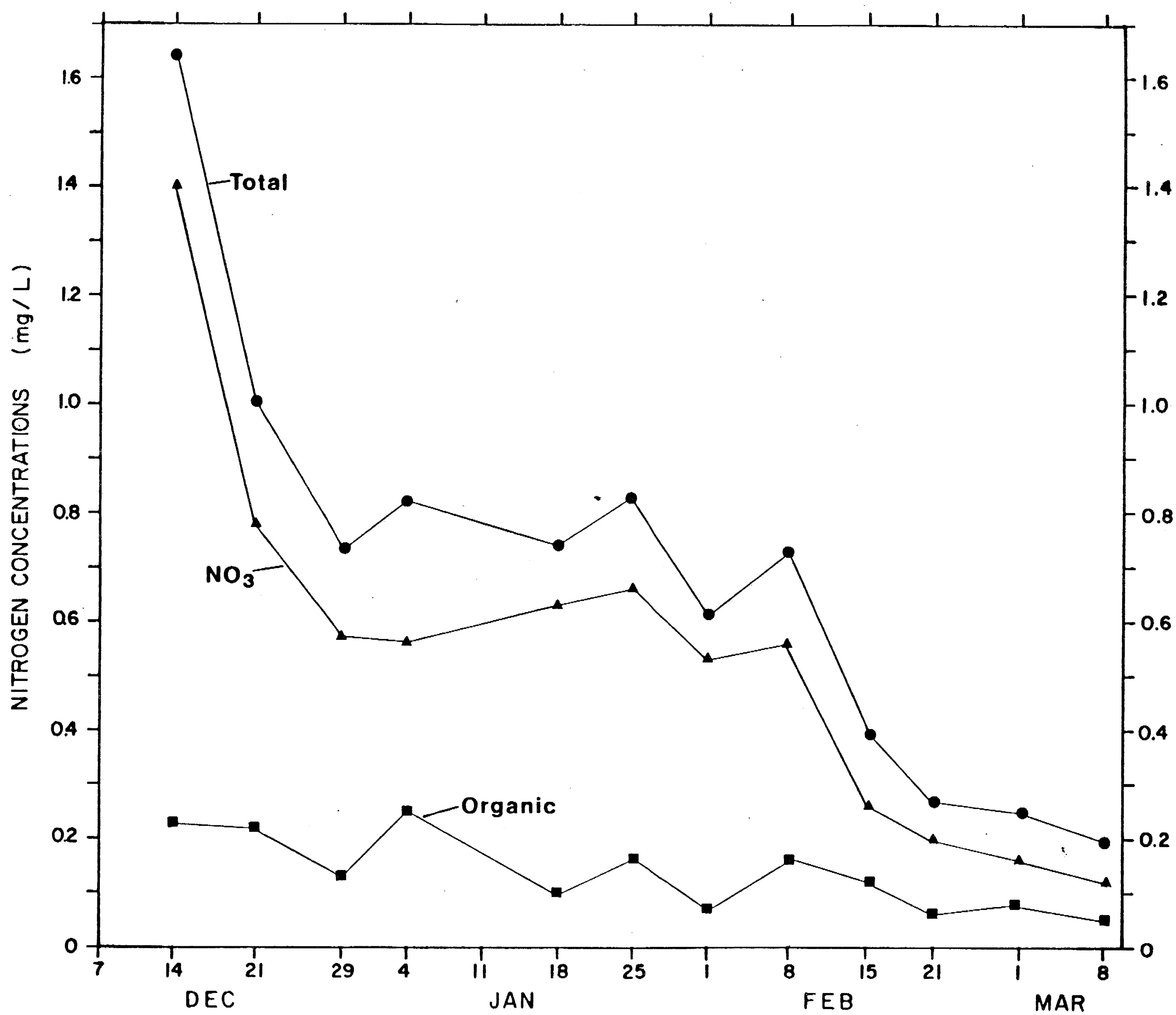


Figure 11 Nitrogen in Roundhouse Creek, 1978-79

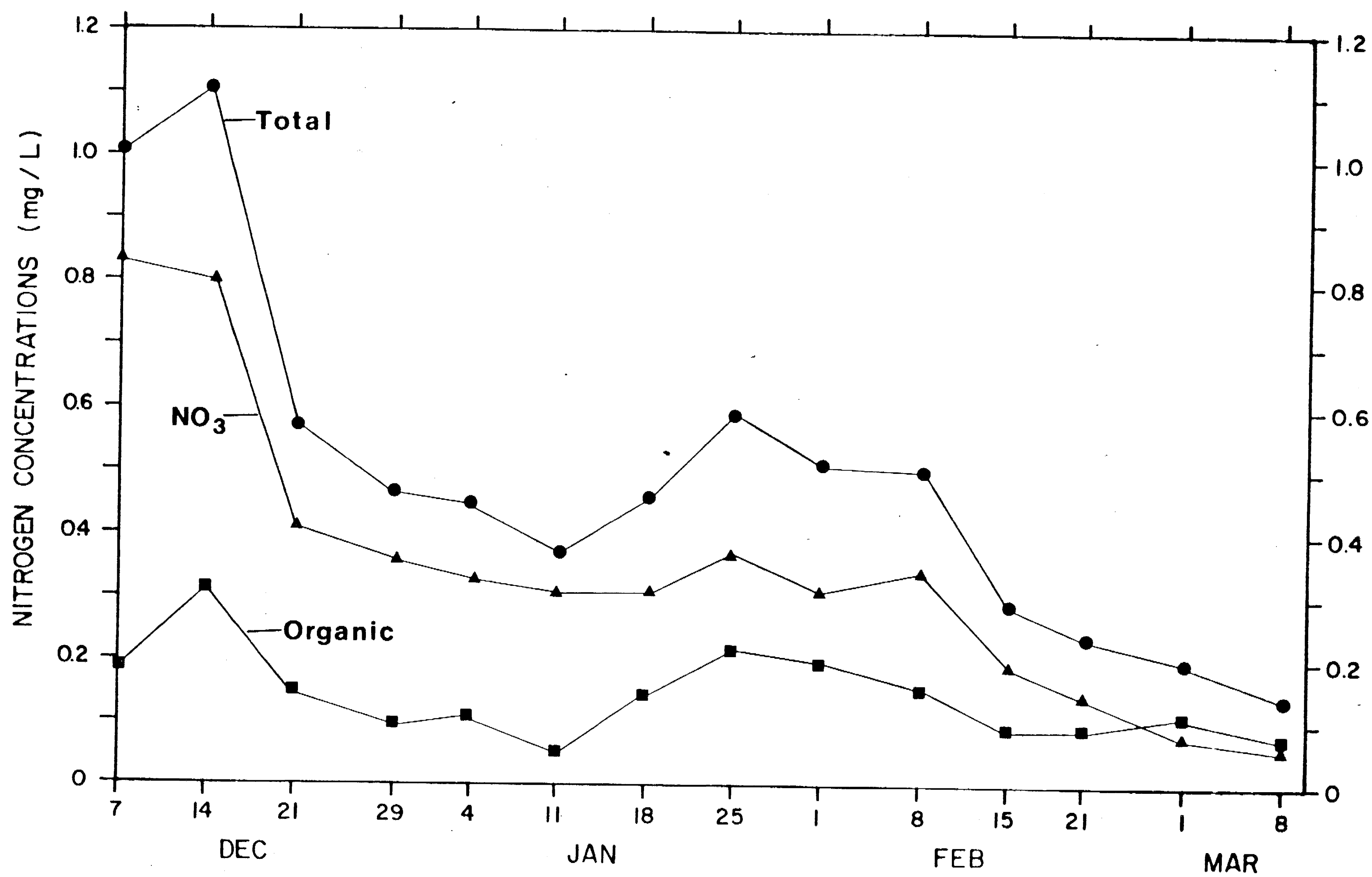


Figure 12 Nitrogen in McGee Creek, 1978-79

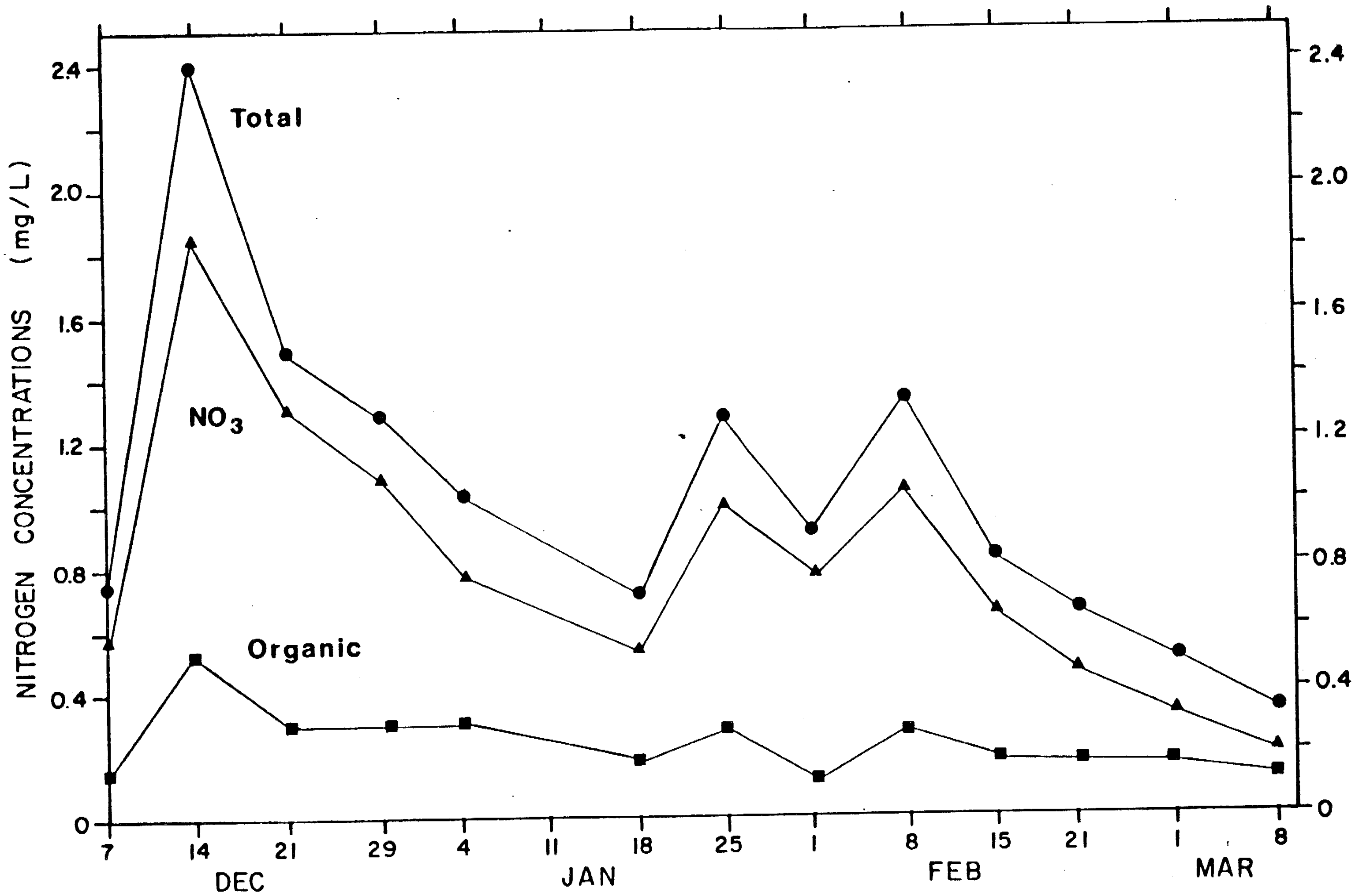


Figure 13 Nitrogen in West Arm Inflow, 1978-79

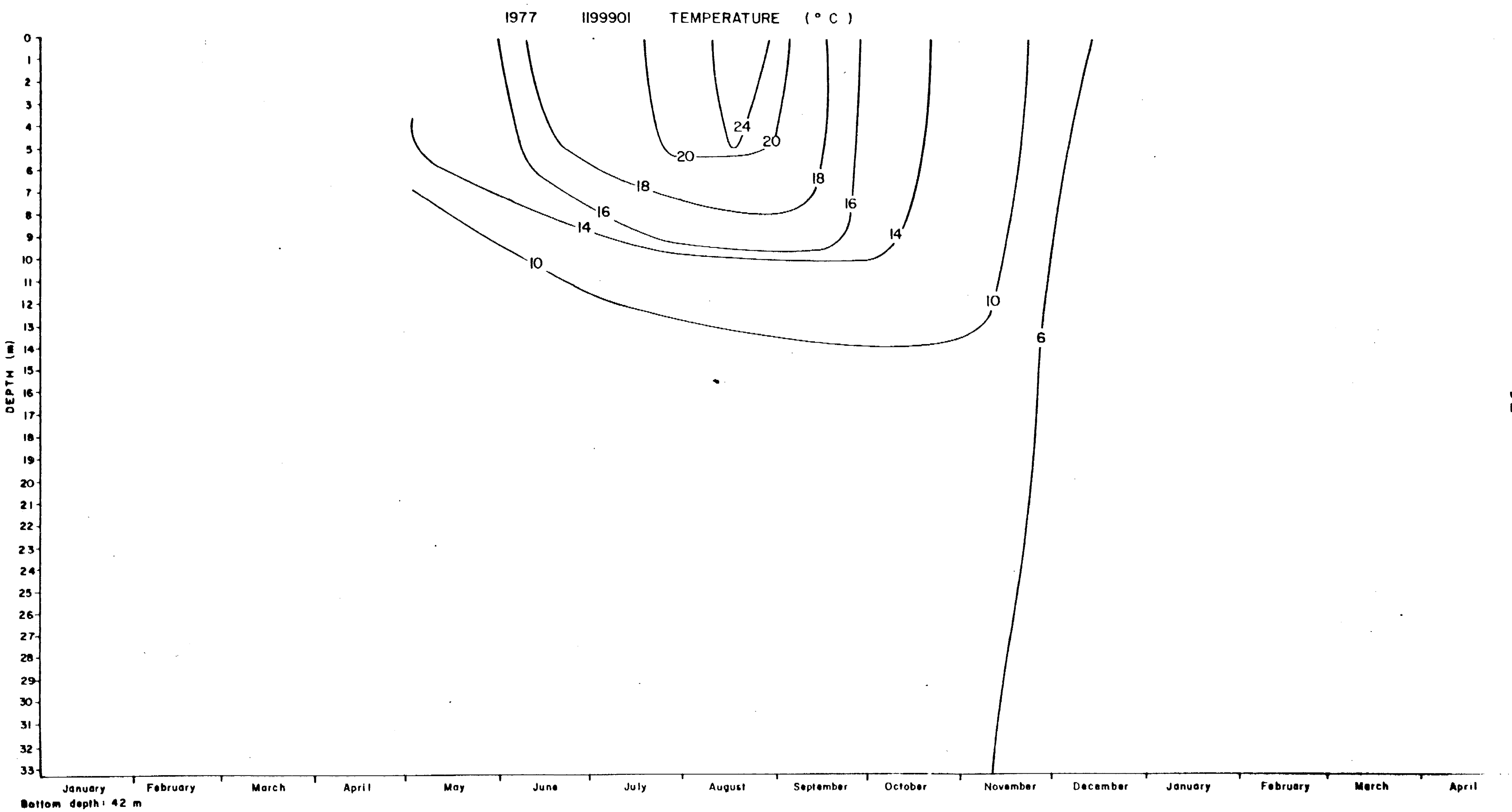


Figure 14 Time/Depth figure for temperature, main basin 1977

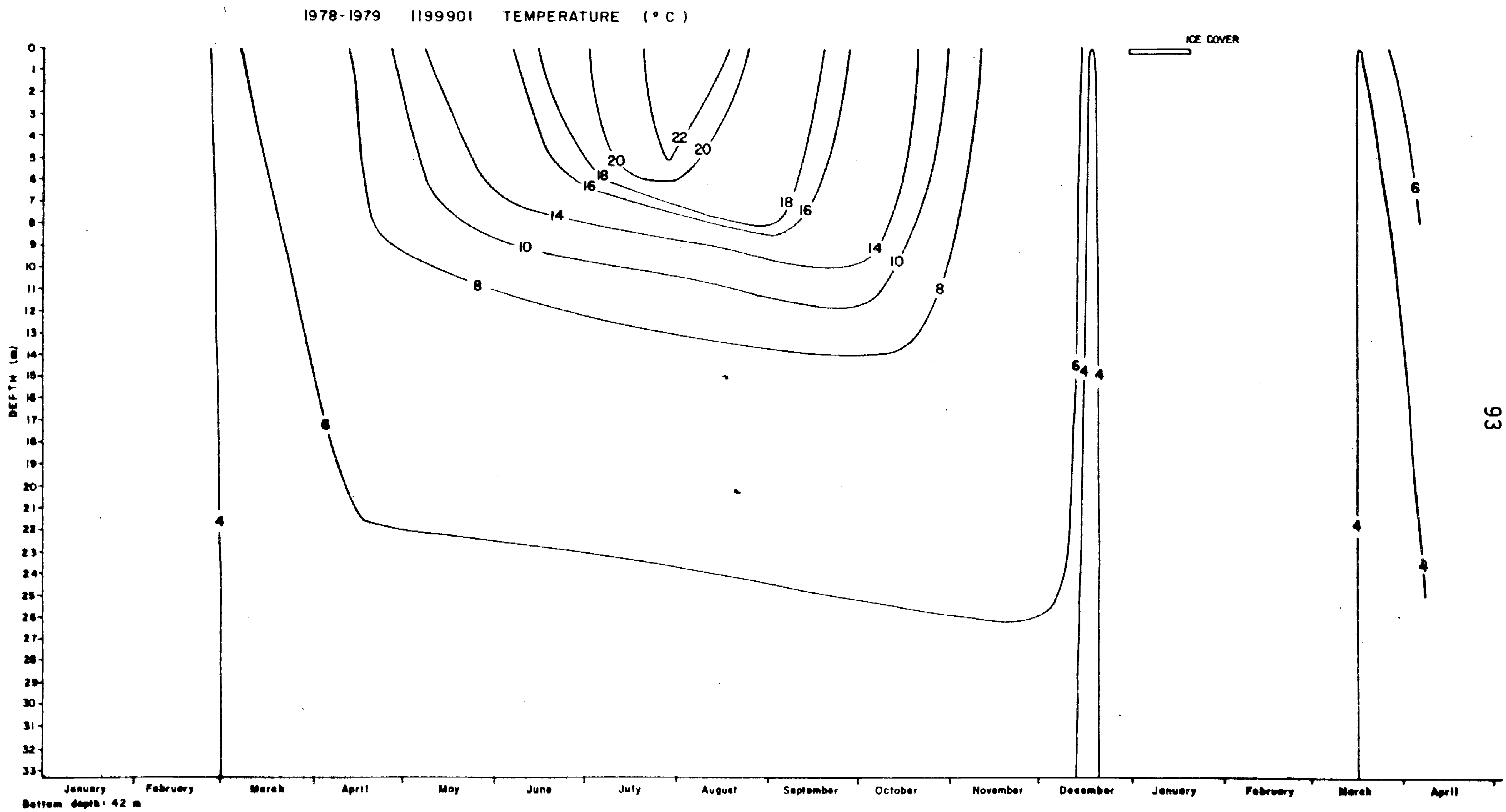


Figure 15 Time/Depth figure for temperature, main basin 1978

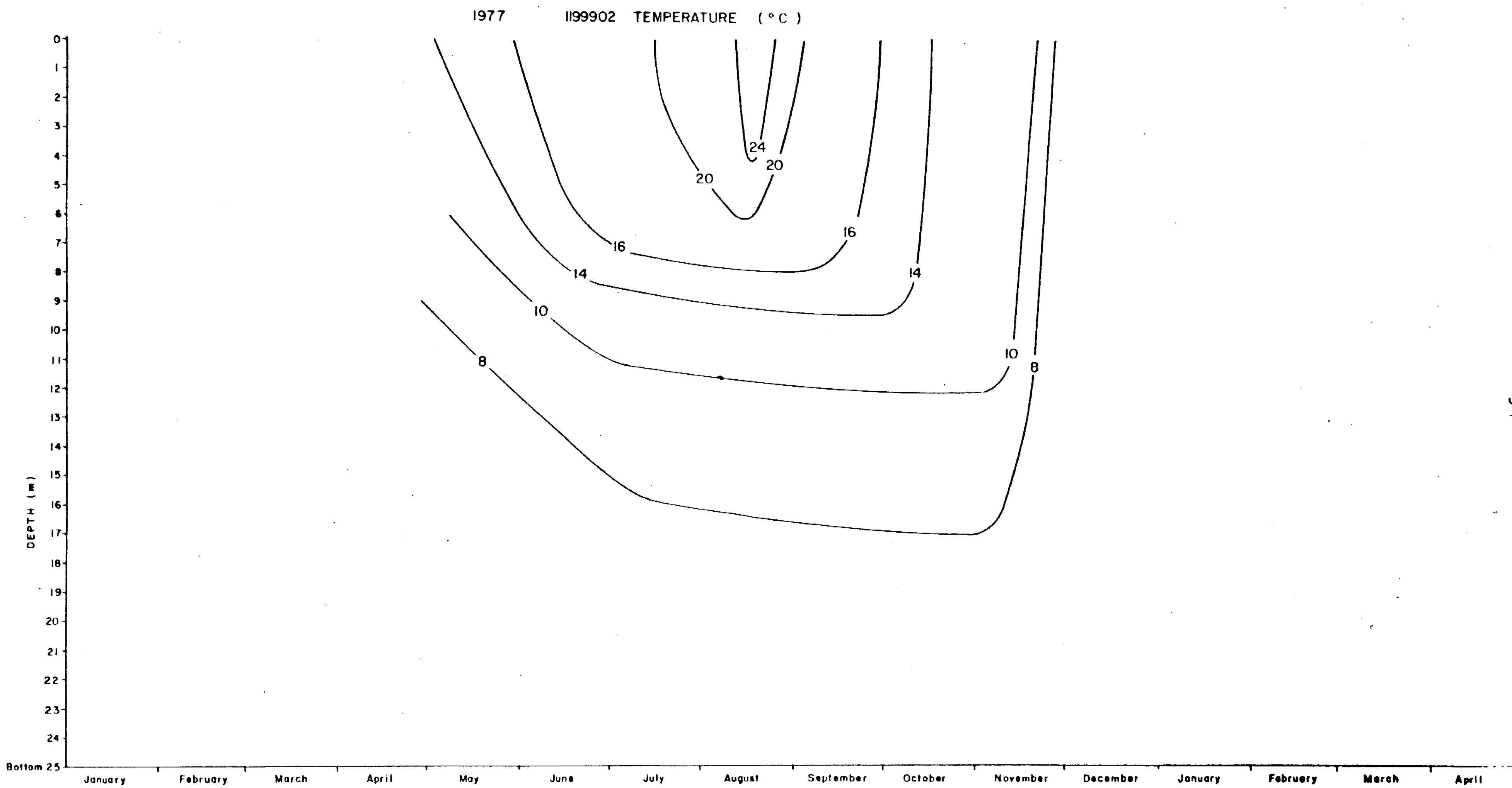


Figure 16 Time/Depth figure for temperature, south basin 1977

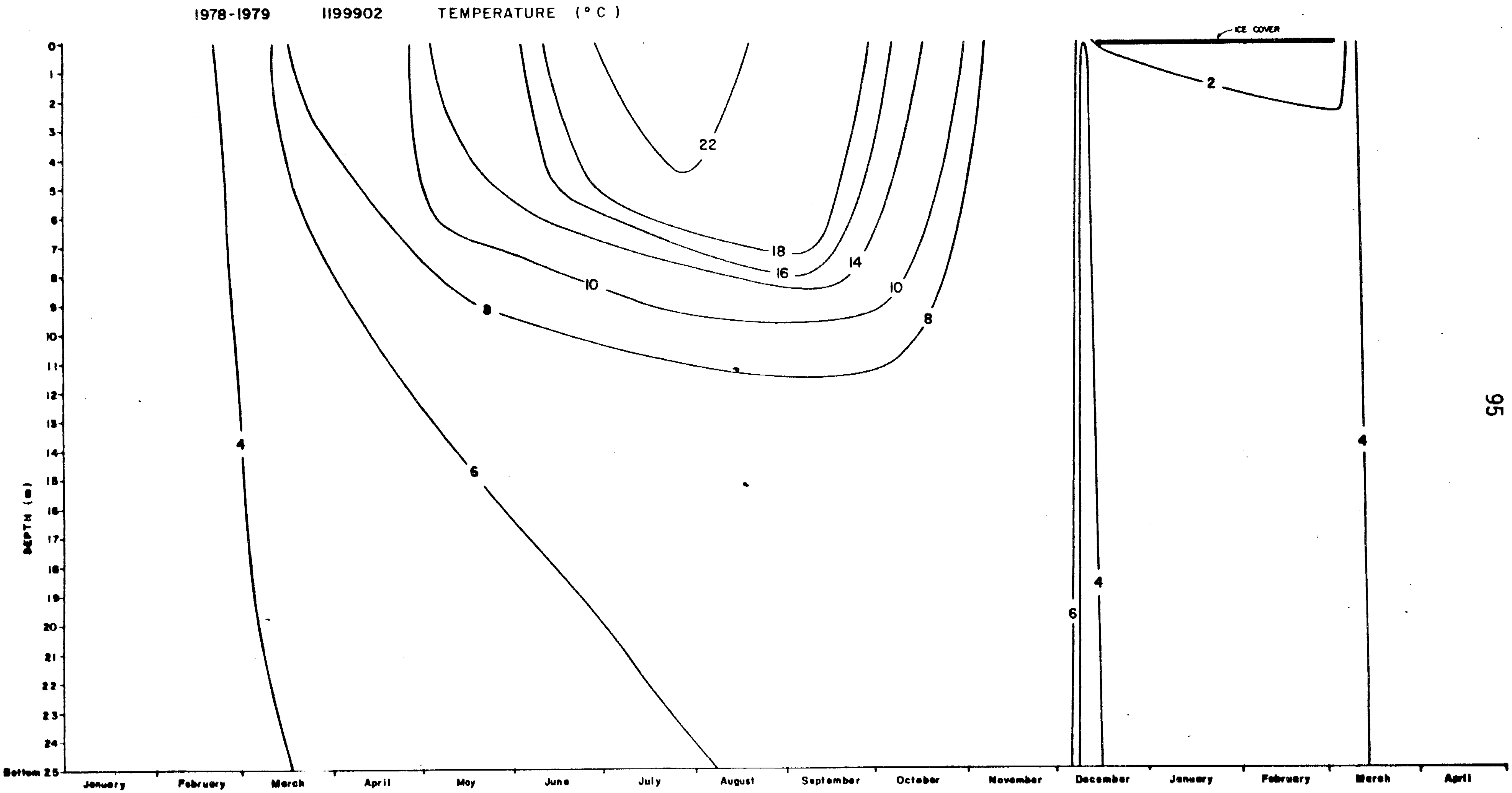


Figure 17 Time/Depth figure for temperature, south basin 1978

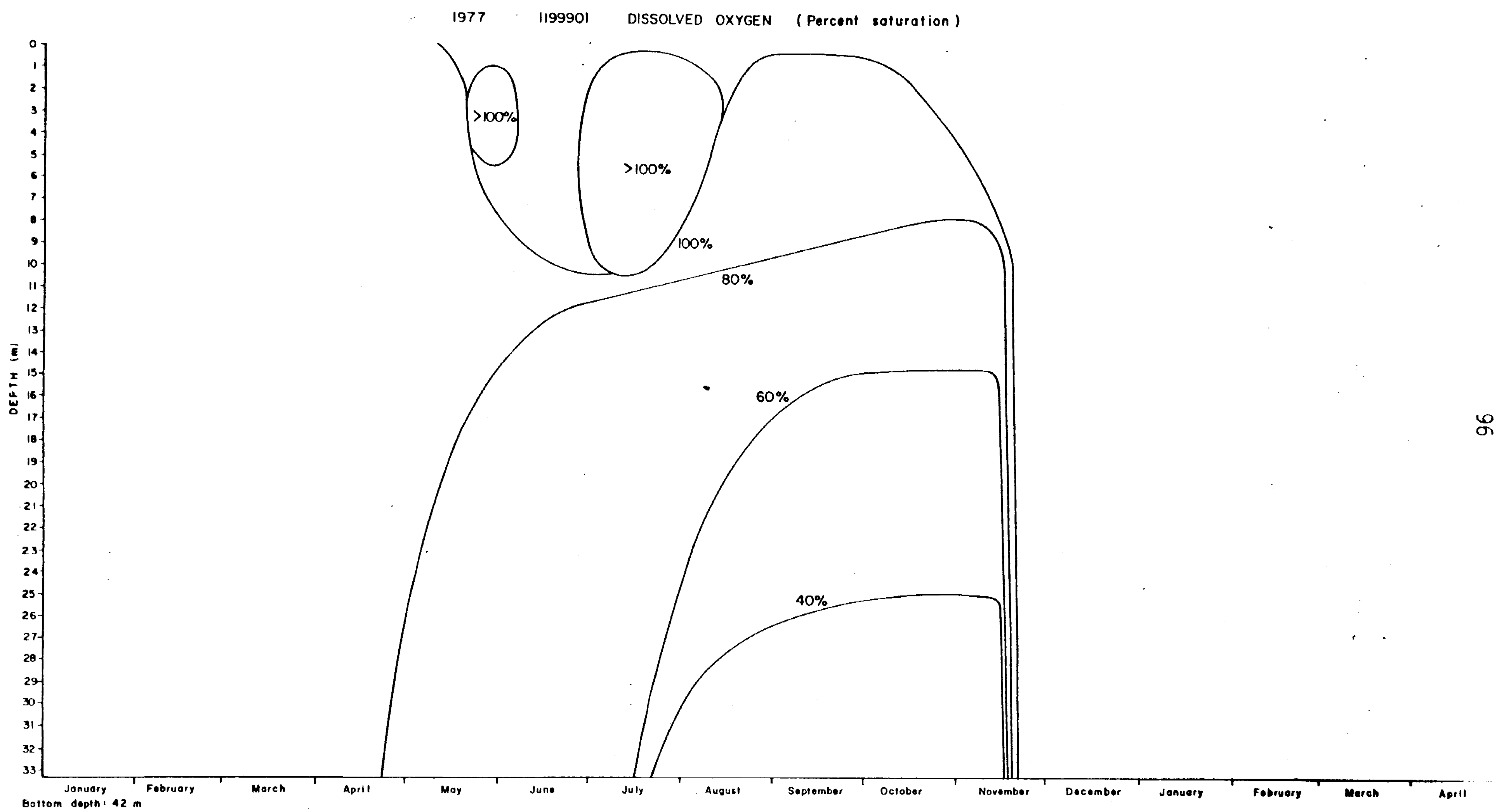


Figure 18 Time/Depth figure for dissolved oxygen, main basin 1977

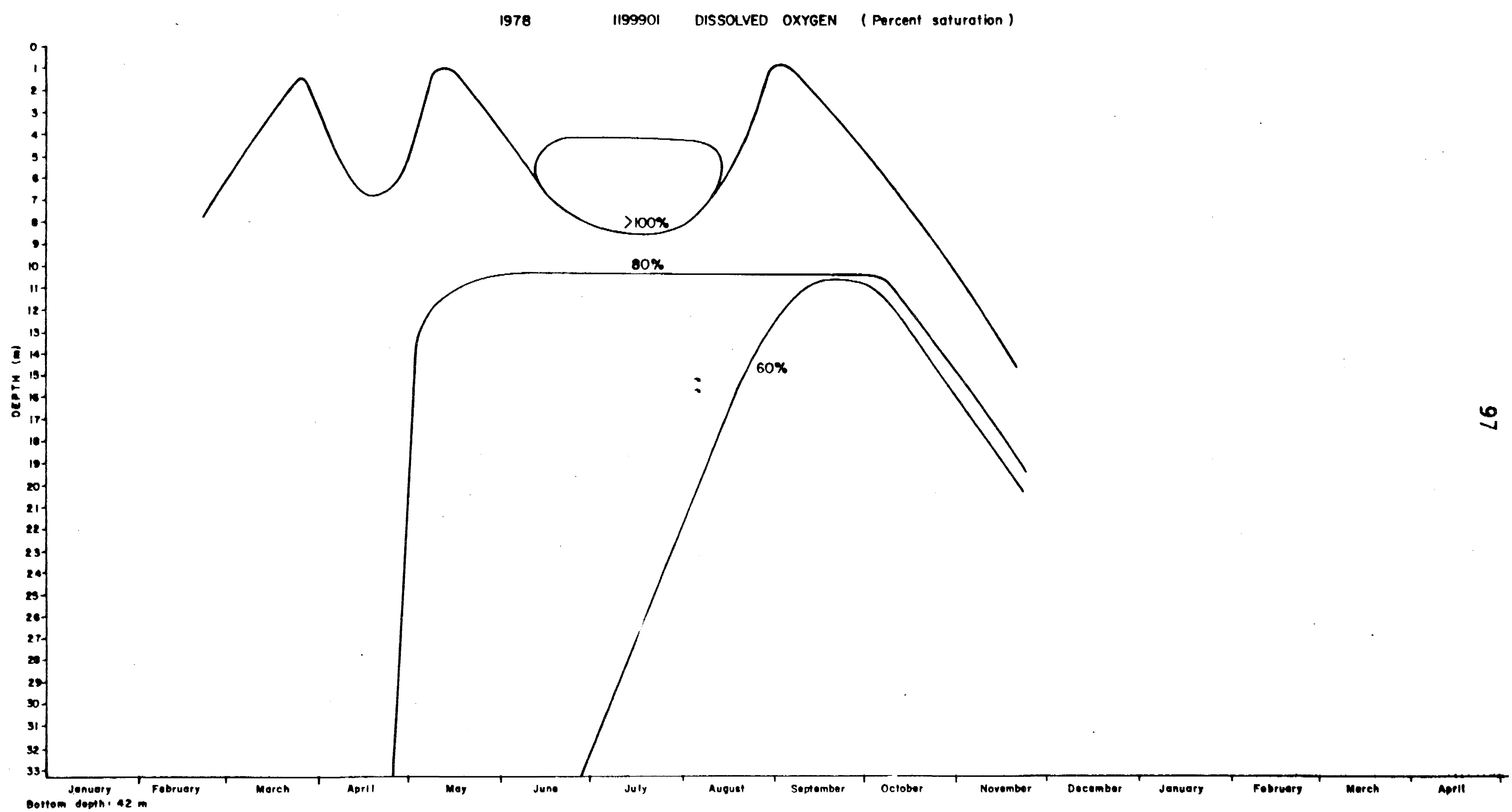


Figure 19 Time/Depth figure for dissolved oxygen, main basin 1978

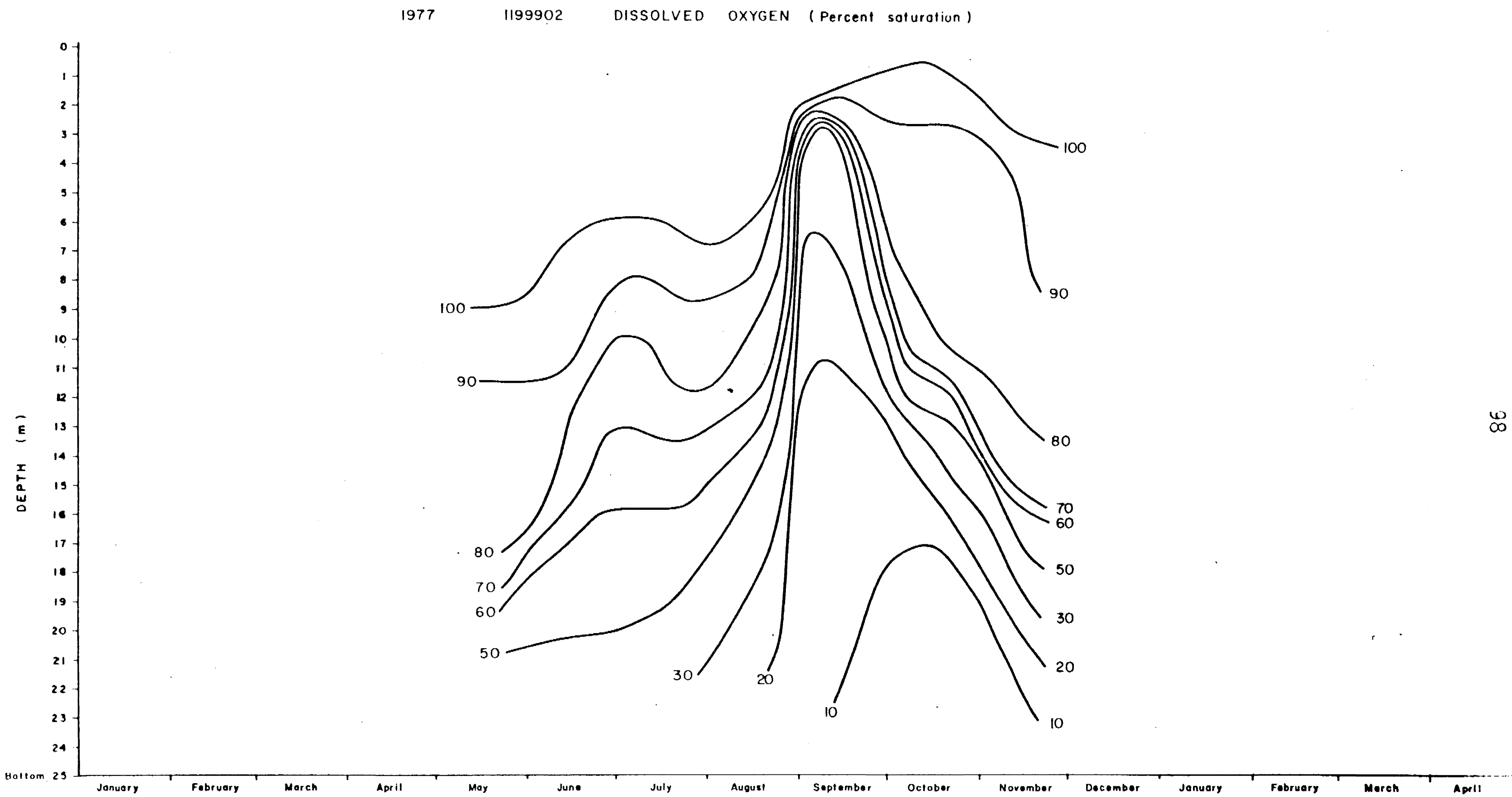


Figure 20 Time/Depth figure for dissolved oxygen, south basin 1977

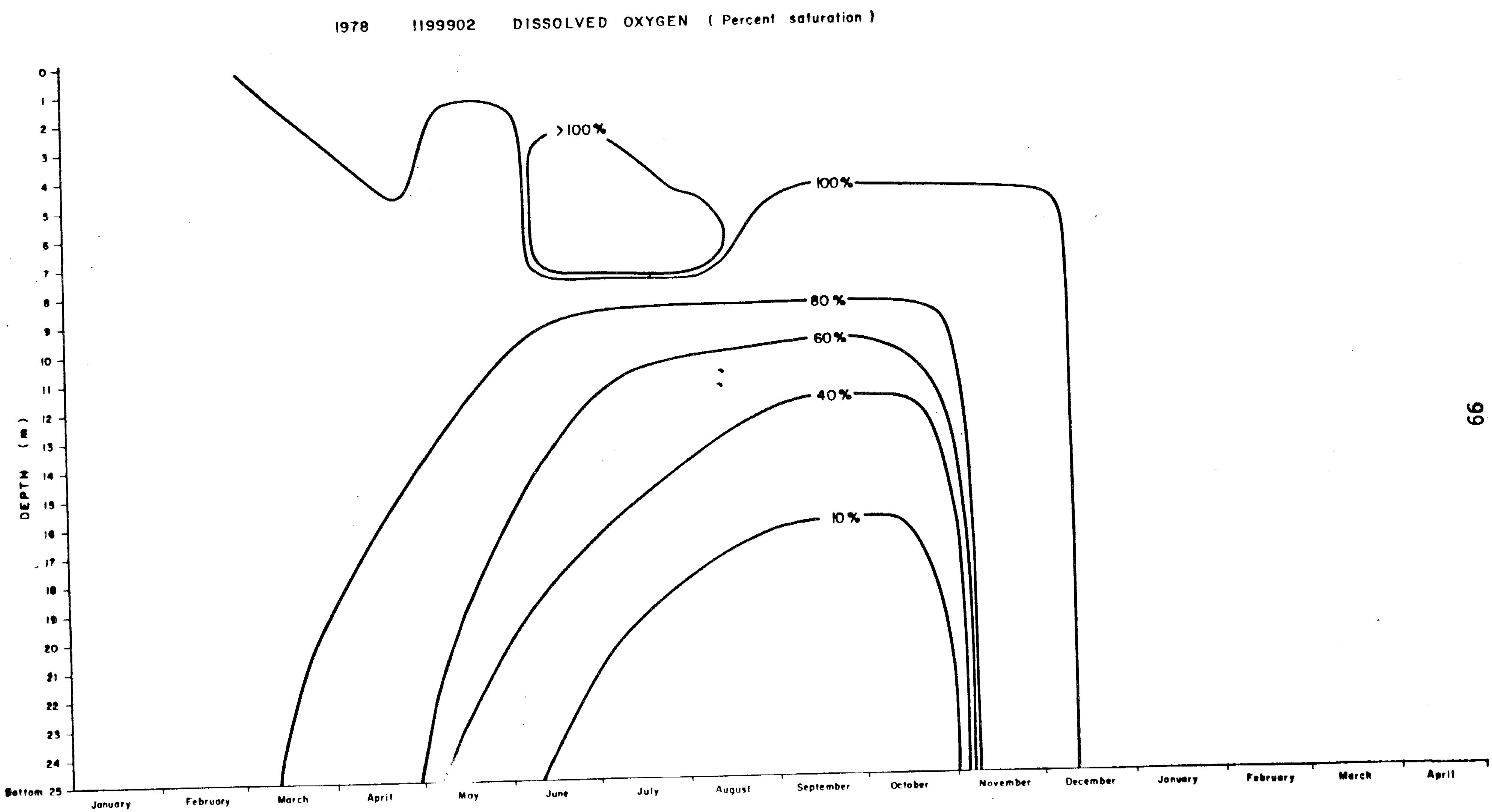


Figure 21 Time/Depth figure for dissolved oxygen, south basin 1978

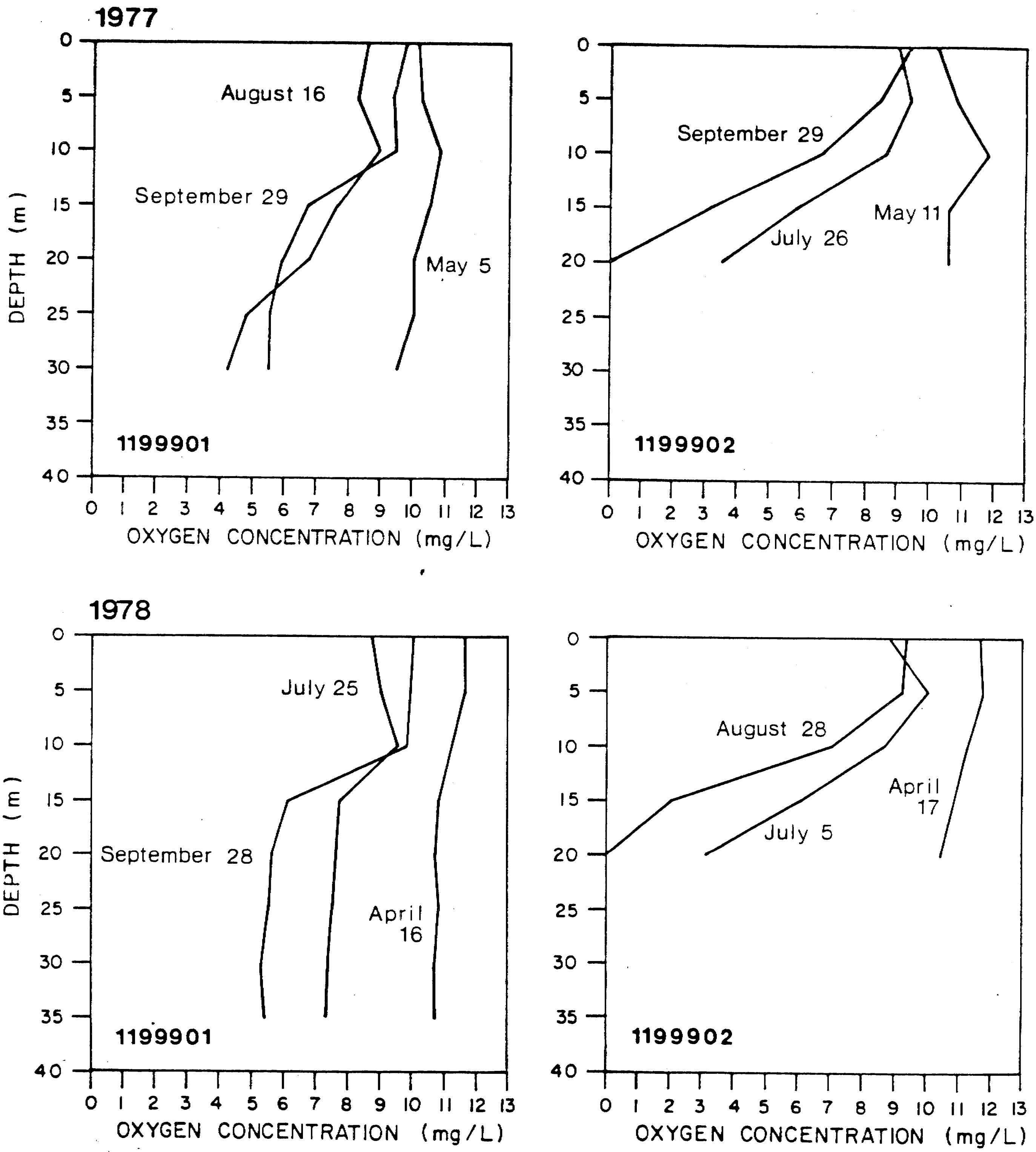


Figure 22   Dissolved Oxygen Profiles

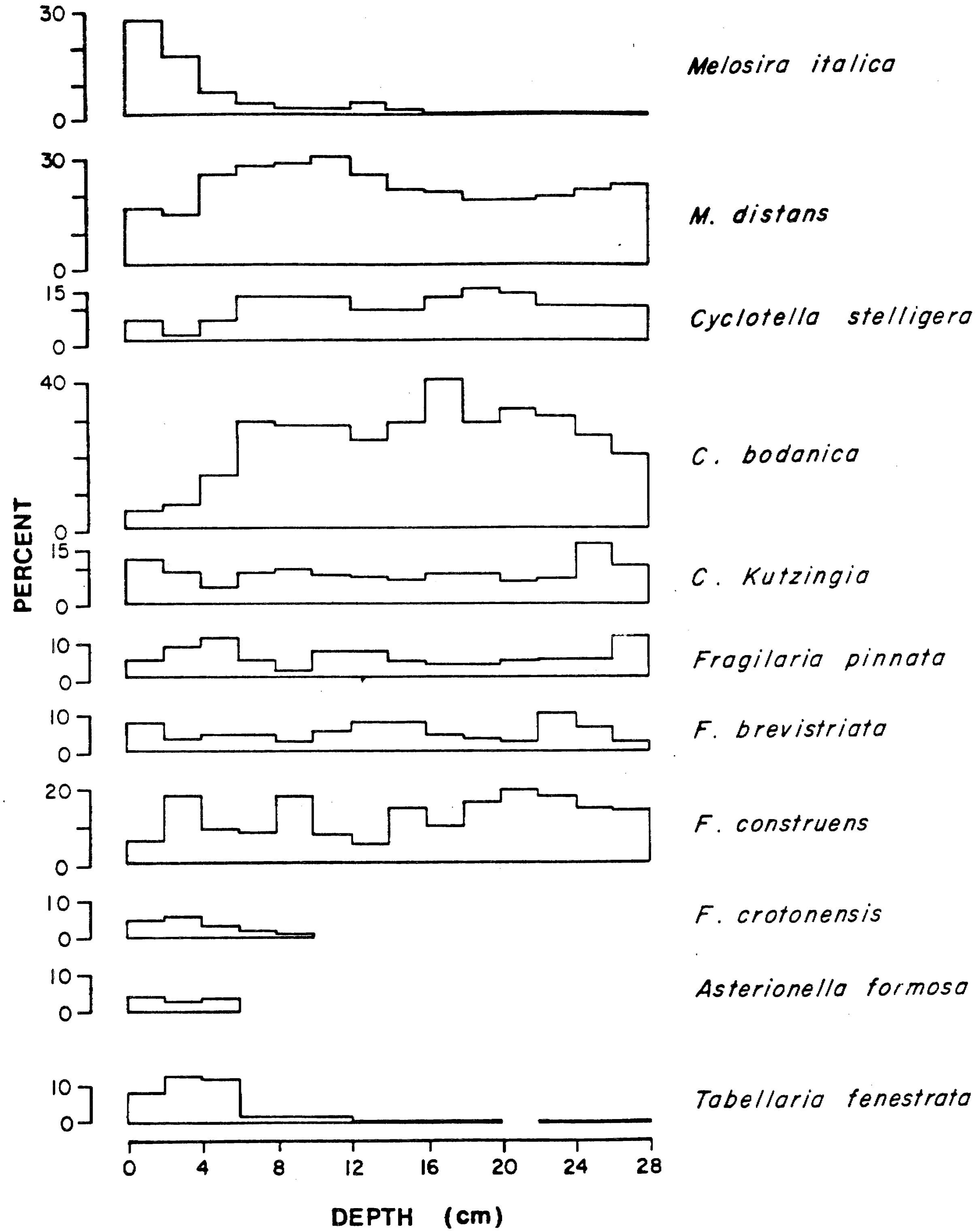


Figure 23 Diatom Species Proportion with Depth in ..ake Sediments

APPENDIX 1 Eutrophication

As noted earlier in the report, a number of the factors related to the eutrophication process are interrelated. Generally the key factor is phosphorus and since the relatonship between phosphorus and chlorophyll was described by Sakamoto (1966), a number of studies have confirmed the universality of the relationship between phosphorus and chlorophyll (Dillon and Rigler 1975, Rast and Lee 1978) and between chlorophyll and Secchi disc depth - i.e. water clarity (Janus and Vollenweider 1981, Rast and Lee 1978).

These relationships provide an excellent opportunity to project possible future changes in those water quality parameters related to eutrophication and set upper limits on certain parameters (objectives) to protect particular water uses.

Appendix Figures 1 and 2 show the relationships between phosphorus and chlorophyll and chlorophyll and water clarity for a number of B.C. lakes. The lines represent a regression line for best fit, and 50 and 95% confidence limits as well as the line representing the overall regression line for a large group of lakes (OECD lakes, see Janus and Vollenweider 1981). The relationships are essentially the same as have been reported from other geographical areas.

What is important however is to differentiate between the general relationship and positions of individual lakes in the figure. Because of a number of factors (sampling frequency, inherent water colour or turbidity, trophic transfer efficiency, mixing depth) lakes may occupy a position on the figure above or below the line. What also is necessary is that the lake's algal production be limited by phosphorus rather than nitrogen. For projecting a future situation, for instance to predict how much of an increase in chlorophyll a would occur with an increase in phosphorus of 2 or 5 µg/L a line is drawn through the point representing the lake and parallel to the general regression line. The chlorophyll corresponding to the projected phosphorus would occur on that line. See McKean and Nordin (1982) for an example.

For Shawnigan Lake, the suggested objective for total phosphorus based on the most apparent sensitive eutrophication parameters (hypolimnetic oxygen depletion, body contact recreation and aesthetics - see below) is 8 µg/L (0.008 mg/L). For a phosphorus concentration of 8 µg/L the corresponding chlorophyll or water clarity can be obtained by using the relationships shown in the Figures. The concentration of chlorophyll which corresponds to 8 µg/L P is 3.8 µg/L. The water clarity which corresponds to 3.8 µg/L chlorophyll is 4.9 m. The present mean chlorophyll and water clarity are 2.9 µg/L and 6.0 m respectively.

Another way of projecting changes is to use a standard Vollenweider curve to project changes in loading on a lake. This however only places a lake in a general trophic zone and shows its general movement but does not show the amount of change in chlorophyll or water clarity. A Vollenweider curve is shown as Appendix Figure 3 giving lake positions for loadings corresponding to spring overturn values of 0.006, 0.008 and 0.010 mg/L.

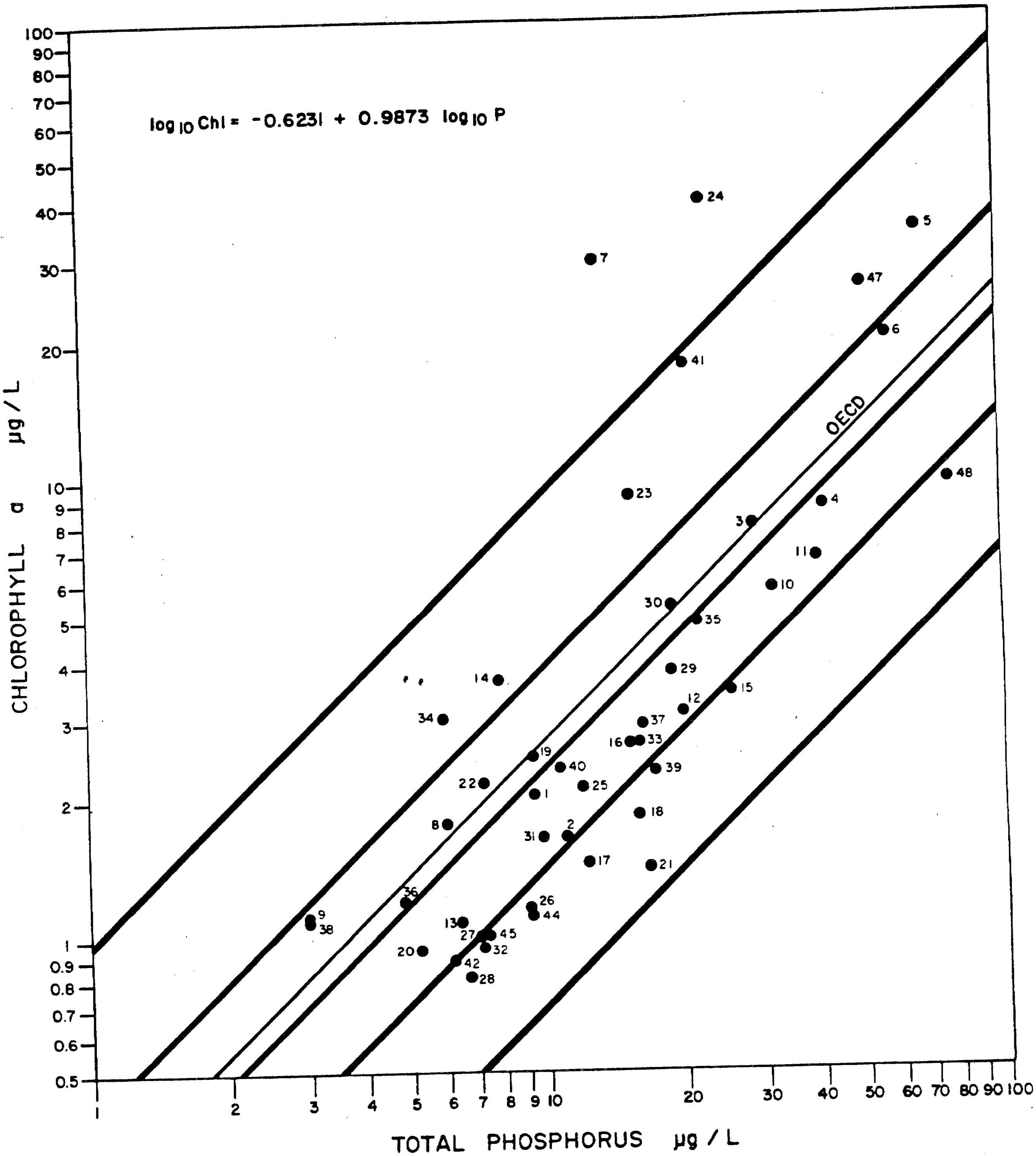
A eutrophication factor which is important to Shawnigan Lake is hypolimnetic oxygen depletion. Some of the depletion in Shawnigan lake may be due to factors oth than pelagic input of organic carbon to the hypolimnion (wood waste or watershed groundwater inputs with organic i.e., sewage content). Thus, using a correlation between some index of biological production (phosphorus, standing crop) and oxygen depletion may not be completely appropriate. However, some estimation of increases in oxygen depletion with increasing phosphorus concentrations are discussed below.

Relationships to estimate hypolimnetic oxygen depletion have been discussed by Walker (1979) Welch and Perkins (1979) and Cornett and Rigler

(1979). Welch and Perkins found a good correlation between phosphorus loading (corrected for flushing rate) and oxygen depletion rate. For Shawnigan Lake, this model would give an oxygen depletion rate of approximately 237 mg O_2/m^2/day at a phosphorus concentration of 6 μg/L which is lower than the measured oxygen depletion for the lake (section 7.2.2). The relationship developed by Cornett and Rigler suggests that the present hypolimnetic oxygen depletion should be approximately 203-221 mg O_2/m^2/day. In comparison to the measured oxygen depletion rates (Table 17) the calculated rates are much lower (Table 17A).

With an oxygen depletion rate higher than would be expected, the risk of adverse conditions occurring with increasing eutrophication is a serious concern. The oxygen concentration in the south basin is already near or at zero in late summer, so increased oxygen depletion is most undesirable. From the oxygen depletion/phosphorus relationships (Table 17A), it would appear that oxygen depletion rates would increase 5-10% for each increase of 2 μg/L of P. Thus 5-10% more oxygen would be consumed in the same period at 8 μg/L than at 6 μg/L. An increase of this magnitude <u>might</u> be acceptable However, an increase in the lake to 10 μg/L P could result in an oxygen depletion rate of 10-20% higher than the already unusually high depletion rate and present a much more serious risk of water quality deterioration, particularly initiation of internal phosphorus loading from lake sediments.

These data reinforce the assumption that dissolved oxygen is a very sensitive aspect of the lakes general water quality. They justify the provisional objective for phosphorus concentration of 8 μg/L, to protect the lake from further deterioration.

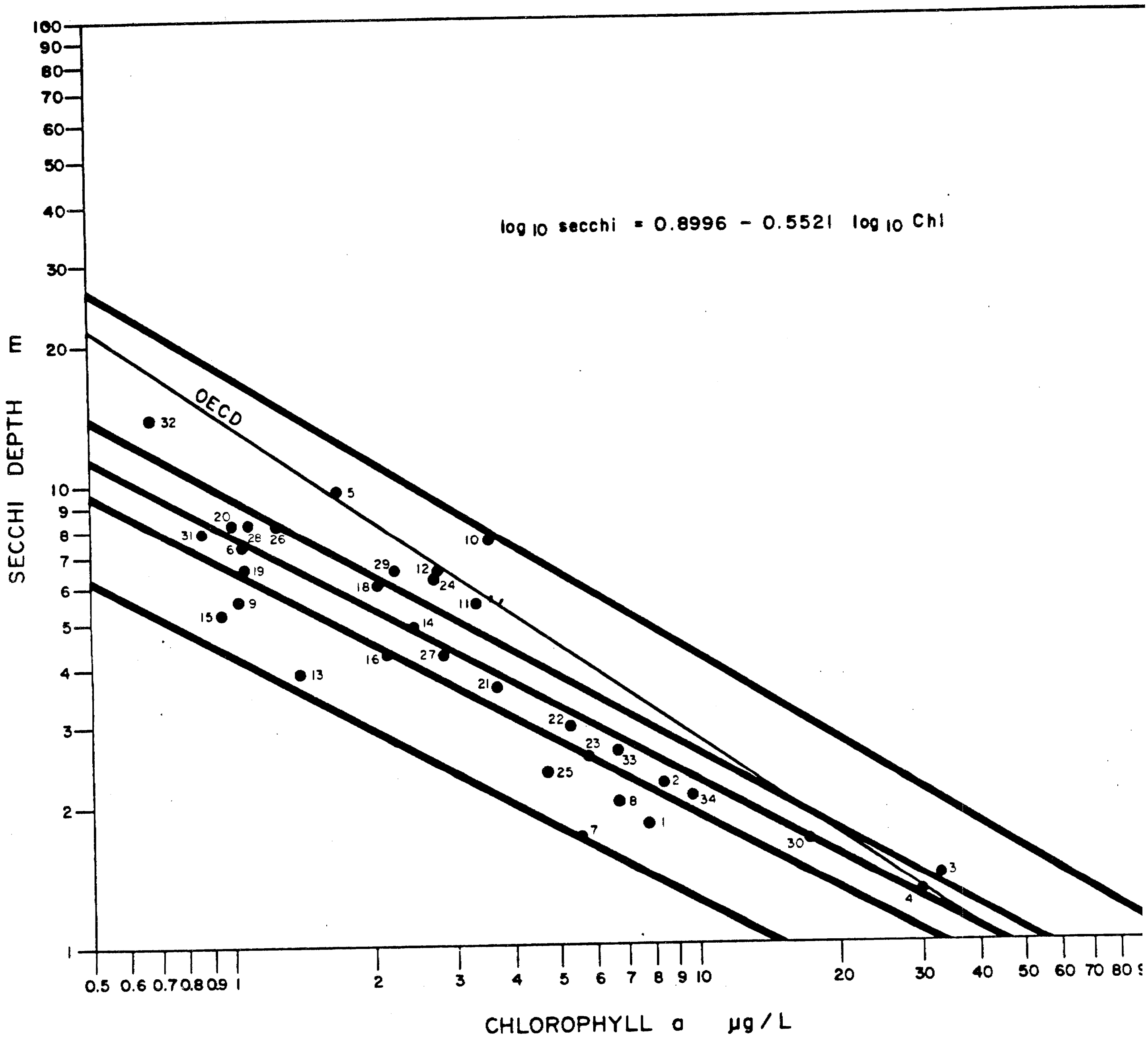


Appendix Figure 1 Chlorophyll/Phosphorus in B.C. Lakes

Appendix Figure 1

Numbers represent the following lakes:

1. Bednesti
2. Berman
3. Burns 1974
4. Burns 1975
5. Charlie 1974
6. Charlie 1975
7. Chief 1976
8. Christina 1978
9. Crooked 1977
10. Decker 1974
11. Decker 1975
12. Eena 1978
13. Kalamalka
14. Kawkawa 1976
15. Kootenay 1975-76
16. Kootenay 1977-78
17. Lac La Hache 1978
18. Lac Le Jeune 1977
19. Lakelse 1974
20. Mabel 1976 and 1978
21. Murch
22. McKinley
23. Nulsko
24. Nulki
25. Okanagan north basin
26. Okanagan central
27. Okanagan south
28. Opatcho
29. Osoyoos north basin 1975-78
30. Osoyoos south 1975-78
31. Pand
32. Purden
33. Saxton
34. Shawnigan
35. Shuswap - Tappen
36. Shuswap
37. Skaha
38. Slocan
39. Stump
40. Summit
41. Tabor
42. Three Valley
43. Trout
44. Verdant
45. Vivian
46. West
47. Williams
48. Wood

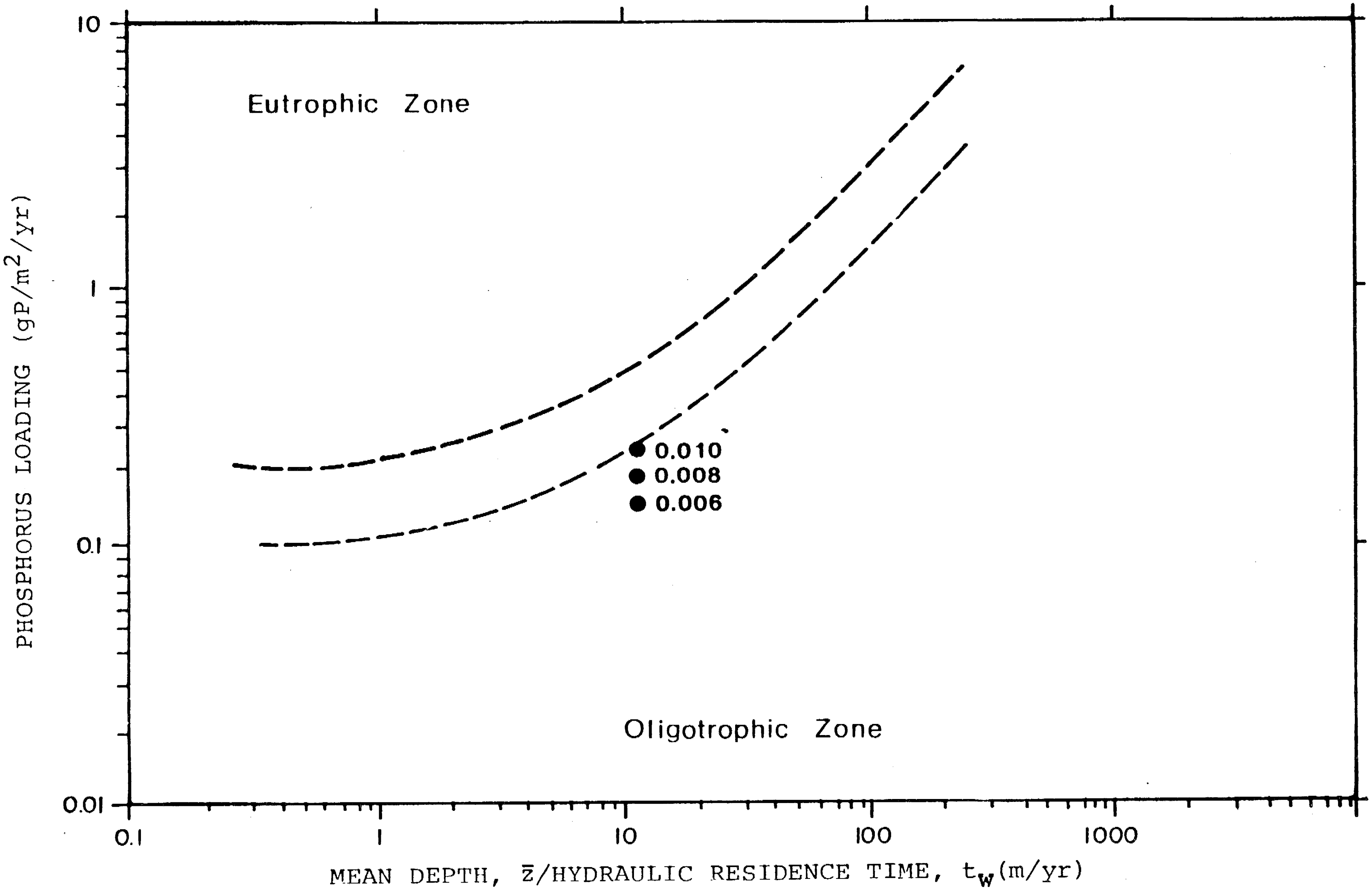


Appendix Figure 2 Secchi/Chlo: hyll in B.C. Lakes

Appendix Figure 2

Numbers represent the following lakes:

1. Burns 1974
2. Burns 1975
3. Charlie
4. Chief
5. Christina
6. Crooked
7. Decker 1974
8. Decker 1975
9. Kalamalka
10. Kawkawa
11. Kootenay 1975-76
12. Kootenay 1977-78
13. Lac La Hache
14. Lakelse
15. Mabel
16. McKinley
17. Nukko
18. Okanagan north
19. Okanagan central
20. Okanagan south
21. Osoyoos north
22. Osoyoos south
23. St. Mary
24. Shawnigan
25. Shuswap - Tappen
26. Shuswap
27. Skaha
28. Slocan
29. Stump
30. Tabour
31. Three Valley
32. Trout
33. West
34. Williams



Appendix Figure 3 Vollenweider Curve

APPENDIX 2. Water Quality Criteria for Phosphorus (adapted from Singleton and Clark, 1983, Clark 1978)

PHOSPHORUS TOTAL

REVISION DATE: Jan. 1983

CRITERIA		REFERENCE	DATE	PAGE(S)
(a) DRINKING WATER				
Objective for drinking water = < 0.2 mg/L (as PO_4)	<0.2 mg/L	B.C. Health	1969	
Recommended standard for drinking water 0.2 mg/L (as PO_4); (i.e. 0.065 as P)	0.065 as P	B.C. Health	1969	
Suggested level for drinking water = 0.2 mg/L (i.e. 0.065 as P)	0.065 as P	GWQOS	1972	
Maximum phosphorus concentration (total inorganic and organic) as PO_4 = 0.15 mg/L (0.05 mg/L as P)	0.015 mg/L as PO_4 0.05 mg/L as P	Sask.	1975	7
Maximum phosphorus concentration (total inorganic and organic) as PO_4 = 0.015 mg/L	0.015 mg/L as PO_4	Alta.	1977	6
Proposed guideline: 90th percentile for total phosphorus in potable water is 0.31 mg/L	0.31 mg/L	Eng.	1982	81,82
(b) AQUATIC LIFE				
Maximum phosphorus concentration (total inorganic and organic) as PO_4 = 0.015 mg/l (0.05 mg/L as P)	0.015 mg/L as PO_4 0.05 mg/L as P	Sask.	1975	7
Maximum phosphorus concentration (total inorganic and organic) as PO_4 = 0.015 mg/L	0.015 mg/L	Alta.	1977	6
A total phosphorus concentration for the ice-free period of <10 μg/L to prevent oxygen depletion by plant life	10 μg/L	Ont.	1979	139-141
Present objective - total phosphorus concentration should be limited to the extent necessary to prevent nuisance growth of algae, weeds and slimes that are or may become injurious to any beneficial water use		IJC	1980	4

APPENDIX 2. Water Quality Criteria for Phosphorus - (Continued)

PHOSPHORUS TOTAL (Continued)

REVISION DATE: Jan. 1983

CRITERIA		REFERENCE	DATE	PAGE(S)
AQUATIC LIFE - (Continued)				
Recommended objective - the basin wide mean total phosphorus concentration in the Central Basin of Lake Erie in the spring should not exceed 10 μg/L to restore year-round aerobic conditions in the bottom waters of the basin		IJC	1980	5
(c) WILDLIFE				
Maximum phosphorus concentration (total inorganic and organic) as PO_4 = 0.15 mg/L (0.05 mg/L as P)	0.015 mg/L	Sask.	1975	7
Maximum phosphorus concentration (total inorganic and organic) as PO_4 = 0.015 mg/L	0.015 mg/L	Alta.	1977	6
(d) AGRICULTURE				
(i) Irrigation				
Maximum phosphorus concentration (total inorganic and organic) as PO_4 = 0.15 mg/L (0.05 mg/L as P)	0.015 mg/L	Sask.	1975	7
Maximum phosphorus concentration (total inorganic and organic) as PO_4 = 0.015 mg/L	0.015 mg/L	Alta.	1977	6
(ii) Livestock				
Maximum phosphorus concentration (total inorganic and organic) as PO_4 = 0.15 mg/L (0.05 mg/L as P)	0.015 mg/L	Sask.	1975	7
Maximum phosphorus concentration (total inorganic and organic) as PO_4 = 0.015 mg/L	0.015 mg/L	Alta.	1977	6

APPENDIX 2. Water Quality Criteria for Phosphorus - (Continued)

PHOSPHORUS TOTAL (Continued)

REVISION DATE: Jan. 1983

CRITERIA		REFERENCE	DATE	PAGE(S)
(e) RECREATION AND AESTHETICS				
Maximum phosphorus concentration (total inorganic and organic) as PO_4 = 0.015 mg/L (0.05 mg/L as P)	0.015 mg/L	Sask.	1975	7
Maximum phosphorus concentration (total inorganic and organic) as PO_4 = 0.015 mg/L	0.015 mg/L	Alta.	1977	6
A high level of protection against aesthetic deterioration will be provided by a total phosphorus concentration for the ice-free period of 10 µg/L or less. This should apply to all lakes naturally below this level	10 µg/L	Ont.	1979	139-141
Excessive plant growth in rivers and streams should be eliminated at a total phosphorus concentration below 30 µg/L	30 µg/L	Ont.	1979	139-141
To avoid nuisance concentrations of algae in lakes, average total phosphorus concentration for the ice-free period should not exceed 20 µg/L	20 µg/L	Ont.	1979	139-141
Present objective - total phosphorus concentration should be limited to the extent necessary to prevent nuisance growth of algae, weeds and slimes that are or may become injurious to any beneficial water use		IJC	1980	4
Recommended total phosphorus objectives range from 5-15 µg/L depending on the location	5-15 µg/L	IJC[2]	1980	4,5
(f) INDUSTRY				
Maximum phosphorus concentration (total inorganic and organic) as PO_4 = 0.015 mg/L (0.05 mg/L as P)	0.015 mg/L 0.05 mg/L as P)	Sask.	1975	7

APPENDIX 2. Water Quality Criteria for Phosphorus - (Continued)

PHOSPHORUS TOTAL (Continued)

REVISION DATE: Jan. 1983

CRITERIA		REFERENCE	DATE	PAGE(S)
INDUSTRY (Continued)				
Maximum phosphorus concentration (total inorganic and organic) as PO_4 = 0.015 mg/L	0.015 mg/L	Alta.	1977	6
(g) MISCELLANEOUS				
Permissible nutrient (N & P) additions to lakes in terms of Vollenweider loading formulae		GWQOS	1972	
Algorithm for protection of lakes from euthophication		GWQOS	1972	
Objective for raw water, as PO_4 = <0.2 mg/L (i.e. <0.065 as P)	<0.065 as P	GWQOS	1972	
Acceptable for raw water, as PO_4 = 0.2 mg/L (i.e. 0.065 as P)	0.065 as P	GWQOS	1972	

Alta. (1977) = Alberta Surface Water Quality Objectives, Water Quality Branch, Standards and Approvals Division, January 1977.

B.C. Health (1969) = Recommended Water Quality Standards, Dept. of Health Services and Hospital Insurance, Province of B.C., Victoria 1969.

Eng. (1982) = Anglican Water Authority. 1982. Water Quality Criteria, Cambridge, England.

GWQOS (1972) = Guidelines for water quality, objectives and standards, Technical Bulletin No. 67, Inland Waters Branch, Dept. of the Environment, Ottawa, 1972

IJC (1980) = Report of the Aquatic Ecosystems Objectives Committee, Great Lakes Science Advisory Board, IJC. 1980.

Ont. (1979) = Rationale for the Establishment of Ontario's Provincial Water Quality Objective, Ministry of Environment. Sept. 1979.

Sask. (1975) = Saskatchewan Water Quality Objectives, Environment Saskatchewan, Water Pollution Control Branch, Jan. 1975.

APPENDIX 3. Correlations Among Phosphorus, Trophic Status and the 'Weedy' Growth of Ceratophyllum in B.C. lakes.

Notes by P.D. Warrington and C.J.P. McKean

The aquatic macrophyte genus Ceratophyllum does not posess roots; the radicle fails to develop during seed germination. Early in the season the plants may be anchored in the sediment by the lower portion of the stem but later in the growing season most of the biomass is floating near the surface as a mat. The plants may overwinter vegetatively in deep water but also form vegetative overwintering turions and many hard, spring seeds (achenes) which settle to the bottom of ponds and lakes. Ceratophyllum is often a dominant invasive species in newly formed lakes and temporary ponds, in sewage lagoons and in most eutrophic habitats on rich in dissolved nutrients and high in organic matter.

Two species are found in British Columbia. C. demersum which is a common and widespread species in Canada and much of the world and C. echinatum which is primarily a north-eastern seaboard species of North America. In B.C. C. demersum is ubiquitous but uncommon in oligotrophic water, while C. echinatum is found primarily in the south-west coastal portion of B.C. in more mesotrophic waters. C. echinatum rarely surfaces and prefers deeper water than the commonly surfacing C. demersum.

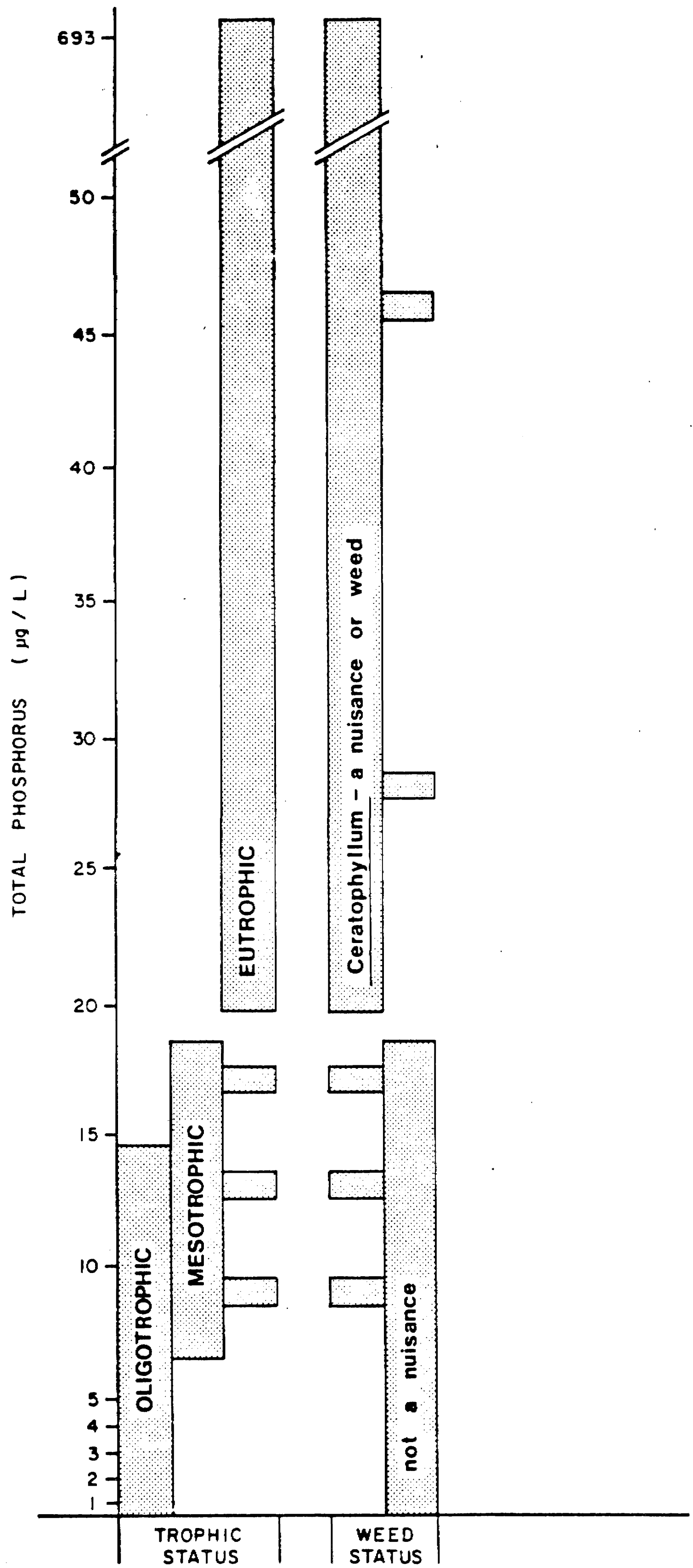
Since Ceratophyllum has no roots and often grows completely detached from from the bottom, particularly C. demersum, it must absorb all its nutrients through its leaves and stem from the water column and not through the sediments. Thus Ceratophyllum should be a good indicator of dissolved nutrients, especially phosphorus. To explore this relationship a listing of all the lakes in B.C. where Ceratophyllum was known to occur was extracted from an aquatic plant distribution data base maintained by the authors[1].

(1) This is a computerized data set covering some 1200 bodies of water in B.C. for which aquatic plant data are known. In addition to plant species information, lake chemistry and lake morphometric data are also included. The data set is maintained by Water Management Branch, Ministry of Environment, Province of British Columbia.

For each lake subjective estimates of the trophic status and of whether or not Ceratophyllum was a nuisancewere made. Next, from the same data base, the mean phosphorus levels were recorded where available. Only complete records, where trophic status, Ceratophyllum 'weed' status and phosphorus were known, were used for the following analysis.

The data were ranked by phosphorus level and a clear relationship became evident among subjective 'weediness' assessments, subjective trophic status and objective phosphorus levels. The results were the same whether the two species of Ceratophyllum were analyzed separately or combined. Only the combined analysis is shown in Figure 4. There is no nuisance growth of Ceratophyllum below about 20 μg/L of mean total phosphorus unless very high levels of organic nitrogen and other nutrients are also present. These are rare occurrences. One always observes a 'weedy' growth of Ceratophyllum above 20 μg/L of mean total phosphorus unless the lake is light-limited by high colour or turbidity values or limited by some other essential nutrient. These are also rare occurrences. There is no nuisance growth of Ceratophyllum in oligotrophic lakes which have an upper limit of about 15 μg/L mean total phosphorus or in mesotrophic lakes with an upper mean total phosphorus level of about 19 μg/L.

There are several ways in which this information can be useful. The presence of a nuisance population of Ceratophyllum will usually indicate a lower limit of about 20 μg/L to the mean total phosphorus in the lake. If phosphorus levels are known for lakes which do not already have Ceratophyllum one can predict whether or not Ceratophyllum would become a problem if introduced. When managing recreationally important lakes one can state the maximum phosphorus level which may be permitted before Ceratophyllum will interfere with recreation and take steps to control phosphorus loading before a problem develops. This could take the form of setting minimum lot sizes, of switching to sewers from septic tanks, of increasing the lakes flushing rate, of controlling septic tanks set-backs, field sizes, field



Appendix Figure 4 Lake trophic status and the nuisance growth of Ceratophyllum characteristics or seasonal use patterns or a number of other ways of balancing the inflow and outgo of phosphorus from the lake. When rehabilitating lakes which already have a weed problem with Ceratophyllum one now knows how far the phosphorus levels must be reduced in order to eliminate the weed problem. This will in turn help determine which control methods might be applicable in a given situation.