

MINISTRY OF ENVIRONMENT AND PARKS
PROVINCE OF BRITISH COLUMBIA

WILLIAMS LAKE
WATER QUALITY ASSESSMENT AND OBJECTIVES
TECHNICAL APPENDIX

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1. INTRODUCTION

Williams Lake is located in British Columbia's Interior Plateau, immediately south east of the City of Williams Lake. The lake drains west to the Fraser River via the Williams Lake River. The San Jose River is the main tributary to Williams Lake. The river begins at the outlet of Lac la Hache and flows for approximately 55 km through the San Jose Valley (Figure 1). There are three major tributaries to the San Jose River: Borland Creek, Jones Creek, and Knife Creek.

Water quality problems have plagued Williams Lake over the past 20 years. As a result, the Planning and Assessment Branch of the Ministry of Environment and Parks requested a complete analysis of water quality within the Williams Lake watershed (Kot, 1985). This report will focus on the water uses and the water quality of Williams Lake, as well as the impact of nutrient loading from the San Jose River on the lake. Lake and river management recommendations designed to improve water quality, identify soil types susceptible to nonpoint nutrient loading from traditional agricultural practices, provisional water quality objectives designed to protect the water quality based on water use, and water quality monitoring are also discussed in this report. Similar analyses of Lac la Hache and the Williams Lake River will be completed in the future using the same terms of reference.

2. HYDROLOGY

The hydrology of Williams Lake has been well surveyed with stream flow gauges situated at the outlets of Williams Lake and Lac la Hache. Stream flow measurements of the Williams Lake River at the outlet of Williams Lake (Water Survey of Canada station 08MC005; Figure 4) have been made daily since July, 1968. The mean monthly hydrograph for the station is shown in Figure 2.

The San Jose River is the main inflow to Williams Lake, draining 95% of the lake's watershed. Groundwater inflow, direct precipitation on the lake's surface, surface runoff from the 5% of the watershed that drains directly to the lake, and evaporation from the lake surface are presumed to be minor influences on the hydraulic loading to Williams Lake.

A stream flow gauge on the San Jose River at the outlet of Lac la Hache (station 08MC006) was operated from 1929 to 1950 and 1973-1974. Only 1973 and 1974 have complete annual records, the remaining years have summer records only. The outflow from Lac la Hache accounted for approximately 70 percent of the hydraulic loading to Williams Lake in 1973 and 1974. The San Jose River and its tributaries account for the remaining hydraulic flow entering Williams Lake.

A stream flow gauge was established in March of 1984 on the San Jose River near Williams Lake (station 08MC040), but the data were not available at the conclusion of this study.

The flushing rate of Williams Lake was calculated from the annual lake outflow measurements. The annual flushing rates from 1974 to 1984 are summarized in Table 1. The average flushing rate based on that period of record was $0.62 \pm 0.37 \text{ yr}^{-1}$ with a maximum and a minimum of 1.37 and 0.24 yr^{-1} , respectively.

3. WATER USE

3.1 WATER LICENCES

The water licences in Williams Lake are summarized in Table 2 and illustrated in Figure 3. Prior to 1987, there were five waterworks licences allowing the withdrawal 17 737 m³/d. Three of the five licences are held by the City of Williams Lake; however, the City relinquished two of its three waterworks licences in 1987, and obtains the majority of its water from four 30 m wells located on Scout Island. The remaining water withdrawal (off Duncan Road) is maintained for emergency use (Lawrence, pers. comm.).

The four wells located on Scout Island draw water from the Central British Columbia aquifer, and are capable of producing up to 38 000 m³/d. The aquifer is effectively sealed from the lake due to an intervening clay/silt layer (Lawrence, pers. comm.). The 1985 average water consumption for the City of Williams Lake was 8 400 m³/d; however, only 106 m³/d was withdrawn from the lake. A storage licence for 2 400 dam³ is also held by the City to prevent lake drawdown due to water withdrawal in the summer.

In addition to the waterworks licences, there are 21 domestic water licences, and three industrial licences (Table 2). Two of the industrial licences are for drinking water supplies on commercial property, while the remaining licence is for lawn watering of a golf course. There are no irrigation water licences extracting water from Williams Lake.

3.2 RECREATION: WILLIAMS LAKE AND AREA

The recreational uses of Williams Lake and Lac la Hache are similar although the volume of use is very different (Lewis, pers. comm.). The recreational uses of the lakes are fishing, water skiing, swimming, sailing, and canoeing. At present there are no studies that quantify the use of each lake; however, the volume of use on Williams Lake is rated low, while Lac la Hache is rated very high by the Regional Office of the Ministry of

Forests and Lands. Lac la Hache and the Chimney Lakes System serve as the main recreational sites for the Williams Lake area.

3.3 FISHERIES

Fisheries information was obtained from Bill Westover, Fisheries Biologist with the Ministry of Environment and Parks Regional Office in Williams Lake. Williams Lake is characterized as having a large coarse fish population with a minor recreational fisheries resource, and no anadromous salmonid spawning or juvenile production. Coarse fish dominate the lake because of its high algal productivity, poor water clarity, and extensive weedy areas. Sports fish include rainbow trout and lake whitefish (Chudyk and Erickson, 1971). In 1985 Kokanee trout were introduced as eyed-eggs to the San Jose River to establish an ice-fishery in Williams Lake.

Resident rainbow trout spawn in the outlet of Williams Lake, the San Jose River, and Borland Creek, a tributary to the San Jose River. Williams Lake has been stocked with rainbow trout in the past, but there are no future plans to add to the native rainbow populations. The rainbow fishery extends for three to four weeks following ice-out, and is estimated at 1 000 angler-days per year. In comparison, the Kokanee ice-fishery on Lac la Hache is estimated at 6 000 angler-days, with an additional rainbow trout fishery of 6-9 000 angler-days in the summer.

4. WASTE DISCHARGES

There are no direct waste discharges that would affect water quality within the Williams Lake or San Jose River drainages. An indirect sewage discharge has occurred in the past from the former Cariboo Student's Residence (now the Cariboo Indian Education Training Centre). However, it is a minor contributor to nutrient loading, amounting to approximately 3% of nutrient loading to the San Jose River (Stitt et al., 1979).

Nonpoint waste discharges from cattle have been a recognized problem for at least the past 10 years. Traditionally, cattle have been overwintered next to the San Jose River and its tributaries. The accumulation of animal waste and the trampling of the river banks is believed to be a major cause of nutrient and suspended solid loading to Williams Lake. During the summer months cattle are moved away from the river and the overwintering pastures are used for growing hay.

A committee of the Ministry of Agriculture and Fisheries, the Ministry of Environment and Parks, and a local Cattlemen's Association representative was established in 1979 to design methods of reducing the nonpoint phosphorus loading from traditional agricultural practices. One important accomplishment of the committee was to move the cattle back from water courses during the winter months. Inspection of the agricultural areas adjacent to the San Jose River, in 1986, showed little stream bank trampling. This result will reduce the stream bank erosion and the suspended solids loading to Williams Lake. Section 5.11 discusses the nonpoint phosphorus loading to the San Jose River.

5. WATER QUALITY

Williams Lake and the San Jose River have been sampled regularly from 1972 to the present. Goddard (1977) and Stitt et al. (1979) summarized the water quality data collected by the Water and Waste Management Branches, respectively. The sample locations and bathymetry of Williams Lake are shown in Figure 4. The morphology of Williams Lake is summarized in Table 3.

5.1 TEMPERATURE

Winter and summer thermal stratification patterns are governed by the continental climate of the Williams Lake area, and a typical thermal pattern for Williams Lake is shown in Figure 5. Summer surface temperatures lie between 20 and 25°C, depending on the ambient temperatures. The summer thermocline begins at a depth of 6 m and extends to 12 m. Beneath 12 m, water temperatures are below 8°C. Fall overturn occurs before the beginning of November and the lake remains isothermal for one or two months. Ice forms on the lake in late November or December and usually remains until early to mid April.

5.2 DISSOLVED OXYGEN

Dissolved oxygen profiles for the winter and summer stratification periods are typical of a eutrophic lake (Figure 6). By mid-August the entire hypolimnion is anoxic, the result of the decomposition of organic matter in the water and sediments. Anoxic conditions remain until fall overturn when aerobic conditions dominate because of the downward mixing of well-oxygenated surface water. Anoxic conditions are less severe during the period of winter stratification as lower water temperature and organic concentrations in the water cause lower hypolimnetic oxygen depletion rates.

During periods of thermal stratification, hypolimnetic oxygen concentrations are critical for adult and juvenile salmonids. Davis (1975) reviewed the oxygen requirements of aquatic organisms, and provided three dissolved oxygen threshold levels. Salmonids (juvenile and adult) are considered by Davis (1975) to be the fresh water species most sensitive to low oxygen levels. Ideal oxygen conditions (level A) provide a high degree of safety for salmonids (>7.75 mg/L). Level B (6.0 mg/L) represents the point where oxygen stress will affect the average member of a species. Level C (4.0 mg/L) represents the point where a large portion of a given species may be affected by low oxygen.

By the end of July, the 6 and 4 mg/L isoclines of dissolved oxygen rise from 18 m to 6 m, a depth which represents 58% of the lake volume. In other words, at the end of July over half of Williams Lake is not suitable for salmonid habitat based on level B or C set by Davis (1975).

Calculation of the hypolimnetic oxygen depletion rates requires regular monitoring of the hypolimnion during aerobic conditions. Using the methods outlined by Wetzel (1975) and the 1974 data collected by Goddard (1977), the oxygen depletion rates are estimated at 0.082 mg/cm²/day. This calculation will slightly underestimate the actual oxygen depletion rate as the hypolimnion was completely anoxic during Goddard's July 25, 1974 sampling trip.

Oxygenation of the hypolimnion to increase the fisheries habitat can be accomplished by two methods. Firstly, a reduction in the lake's algal biomass will reduce the excessive oxygen depletion rates, or secondly, artificial aeration (destratification or hypolimnetic) of the hypolimnion will introduce oxygen to meet the present oxygen depletion rates. Section 5.12 discusses the lake restoration techniques recommended for Williams Lake that are designed to increase fisheries habitat.

5.3 GENERAL WATER CHEMISTRY

The general water chemistry for Williams Lake is summarized in Table 4. The lake is characterized by very hard water (hardness=250 mg/L) due to the very high magnesium and calcium concentrations. Sulphate and chloride are the other minor anions, and averaged 13 and 2.0 mg/L, respectively. The major cations in Williams Lake are magnesium, sodium, and calcium which averaged 42, 34 and 30 mg/L, respectively. The high concentrations of magnesium and sodium are typical for the Williams Lake drainage and are not the result of human activities or pollution sources within the watershed.

Using the procedures outlined by Riehl (1970), the combinations of anions and cations are illustrated in Figure 7 and are summarized in Table 5. Based on these results, Williams Lake can be classified as a magnesium bicarbonate system, where approximately one half of the water's, total dissolved solids and alkalinity is attributed to magnesium bicarbonate. The Williams Lake area is atypical of most areas in British Columbia where calcium bicarbonate dominates the anion-cation balance. The unusual anion-cation balance in Williams Lake is not considered detrimental to the fisheries, recreational, or drinking water suitability of the lake.

The City of Williams Lake obtains its drinking water from wells located on Scout Island near the outlet of Williams Lake. The water quality of the wells located on Scout Island is summarized in Table 6. Although the wells are located near Williams Lake, the water quality has some important differences. The water supply from the wells has a higher pH, higher concentrations of sodium, potassium, iron (total) and sulphate, and lower concentrations of calcium, magnesium, hardness and total alkalinity.

Based on the Ministry of Environment and Parks data, the well water's pH, iron (total), manganese, and occasionally lead are above the guidelines for Canadian drinking water quality (Health and Welfare, 1979). The sodium concentrations are three times above the alert level for very restricted sodium diets, based on the same guidelines. The iron and manganese guidelines are for aesthetics (i.e., not health related), and the pH guideline is

intended to minimize scaling (the formation of carbonate salts) and to maintain the efficiency of chlorine disinfection.

Williams Lake would provide better quality water with respect to pH, sodium, iron, manganese and lead; however, the present eutrophic nature of the lake would cause other problems. Dissolved organics within Williams Lake will colour the water and can combine with chlorine to form halogenated methanes (a group of organic compounds that can be hazardous in water supplies). Also, algae in high concentrations associated with the eutrophic conditions would clog filters, may impart a taste or odour in the water, and under extreme circumstances contain certain cyanobacteria which can be toxic to mammals. Reversal of eutrophication using the techniques outlined in Section 5.12 would eliminate the potential drinking water problems caused by the eutrophic conditions.

5.4 WATER CLARITY

Water clarity, measured by turbidity, suspended solids, or Secchi disc depth, is important aesthetically for recreation and drinking water suitability. Also, very turbid waters are considered unsafe for recreation because of the impaired visibility of potential hazards. Turbidity can be caused by suspended residues entering Williams Lake via the San Jose River (particularly during freshet) or by high concentrations of planktonic algae, the result of eutrophic conditions in the lake.

Turbidity from the San Jose River can influence the water clarity at the east end of the lake, particularly in high flow years. Planktonic algae are the principal cause of poor water clarity in the remainder of the lake during the summer months. Turbidity values at station 0603019 averaged 2.4 ± 1.9 NTU ($n=6$) with a maximum of 6.3 and a minimum of 1.1 (Table 4). The turbidity of Williams Lake exceeded the drinking water standards of an average of 1 NTU and a maximum of 5 NTU (Singleton, 1985).

Secchi disc records at site 0603019 are more frequent ($n=25$), averaging 1.9 ± 1.0 m with a maximum and a minimum of 4 and 0.5 m respectively (Table

4). The Secchi criteria for bathing and swimming areas is 1.2 m (CCREM 1987). It is set to ensure sufficient visibility to estimate depth, to see subsurface hazards easily, and to detect the submerged bodies of swimmers or divers who may be in difficulty. The average of the Secchi results are above the criteria, although some individual results are below the criteria.

Figure 8 illustrates the Secchi measurements for the summer of 1977 and algal biomass (measured as chlorophyll-a) in June and late July. Based on the limited turbidity and Secchi disc measurements, Williams Lake has poor water clarity, principally caused by high concentrations of planktonic algae.

5.5 NUTRIENTS AND CHLOROPHYLL-a

5.5.1 ANNUAL VARIATION

The phosphorus concentration in the water and the phosphorus mass in the lake are two common methods of expressing the annual variability of phosphorus in a lake. Figure 9 represents the mean orthophosphorus concentration from 1973-1985. The actual concentrations are variable from year to year, but the trend shows the epilimnetic and hypolimnetic orthophosphorus concentrations are relatively constant in January, February and March. The high percentage of orthophosphorus (70-80 percent of the total phosphorus) throughout the water column during the winter indicates that there is little to no biological utilization of orthophosphorus in the surface waters, and that chemical precipitation of orthophosphorus with iron does not occur despite aerobic conditions throughout the majority of the water column.

The ice cover is typically gone on Williams Lake by April allowing greater light penetration and biological utilization of orthophosphorus in the surface waters. Surface orthophosphorus concentrations decrease to below $<3 \mu\text{g/L}$ by early July. Hypolimnetic concentrations also decrease in May, but increase following summer stratification, due to phosphorus release from the sediments.

Hypolimnetic orthophosphorus concentrations remain high through the summer while epilimnetic concentrations are below detectable limits. During fall overturn, erosion of the thermocline cause a decrease in hypolimnetic orthophosphorus concentrations and an increase in the epilimnetic concentrations. By November the lake is well mixed, and the surface orthophosphorus concentrations increase to levels typical for the winter months.

The total phosphorus concentrations (Figure 10) show the same cyclical patterns as the orthophosphorus concentrations illustrated in Figure 9. Winter total phosphorus concentrations decrease at spring overturn, the result of biological utilization and sedimentation. Following the spring algal bloom, surface concentrations decrease below 50 $\mu\text{g/L}$ by June and then remain relatively constant through September. Surface concentrations increase in November, the result of mixing with the nutrient-rich hypolimnetic waters. Maximum hypolimnetic total phosphorus concentrations are observed in October, the result of internal phosphorus release from sediments.

5.5.2 NUTRIENTS LIMITING ALGAL GROWTH

In most aquatic systems, nitrogen, phosphorus, or light are usually the factors limiting the growth of microscopic planktonic algae.

In lakes like Williams Lake, where algal growth is not limited by light, nitrogen or phosphorus limitation is usually determined by the total nitrogen to total phosphorus weight ratios in the lake surface water. If the weight ratio in the water is greater than 15:1, then phosphorus is assumed to be limiting; and if the ratio is 5:1 or less, nitrogen is assumed to be limiting (Nordin, 1985).

The mean monthly total nitrogen and total phosphorus weight ratios for the surface waters of Williams Lake are summarized in Figure 11. During the period of ice cover N:P ratios ranged between 11:1 and 13:1. High

phosphorus concentrations relative to nitrogen are responsible for the low ratios during the winter months. Following stratification in May the N:P ratios increased to a level over 30:1. The summer average N:P ratio was approximately 26:1, clearly indicating phosphorus limiting conditions.

5.5.3 SPRING OVERTURN PHOSPHORUS

The productivity and water clarity of a lake during the summer is determined to a large degree by the total phosphorus concentration at spring overturn. Spring overturn is defined as the period prior to summer stratification when the lake is isothermal and mixing. The spring overturn total phosphorus concentrations for Williams Lake from 1974 to 1985 are summarized in Table 7.

In 1974, 1975 and 1976 phosphorus results were collected at station 1100401 (Figure 4). The lake was well stratified at the time of sampling, and the sampling dates are considered to be approximately one month after spring overturn. Spring overturn levels for these three years will be underestimated if the epilimnetic water results are used, and overestimated if hypolimnetic water results are used. Because the hypolimnetic concentrations reflect to some degree the amount of phosphorus sedimented from the epilimnion, the surface and bottom results are averaged to estimate the spring overturn phosphorus levels.

Samples collected from 1977 to 1986 were taken during the period of spring overturn. Consequently, the surface samples taken from stations 0603019 and 0603022 (Figure 4) provide a good estimate of the spring overturn concentrations.

The factors influencing the spring overturn phosphorus concentrations are very important because of the influence of spring overturn on the productivity and the water quality during the summer months. Figure 12 plots the spring overturn phosphorus concentration and the phosphorus loading from the San Jose River. There appears to be a lag time of 1 to 2 years between

phosphorus loading and spring overturn phosphorus concentration. Additional long-term phosphorus loading data from the San Jose River and spring overturn phosphorus conditions for Williams Lake are required to confirm the trend. What is important is the 1-to 2-year lag time between the river-loadings and the spring overturn phosphorus concentrations. The importance of internal phosphorus loading in causing the lag is outlined in Section 5.5.5.

5.5.4 ANNUAL PHOSPHORUS INPUT

Because phosphorus is the nutrient limiting algal growth during the summer months, the sources and annual input are integral components of this report.

Two methods are commonly used to estimate the annual phosphorus input to north temperate lakes. The first method involves the collection of samples and the calculation of loading rates from all possible sources such as atmospheric deposition, streams, septic tanks, agriculture, direct discharges, and internal loading. This method is very time-consuming and costly as it requires large numbers of laboratory analyses. The advantage of this method is that it identifies and quantifies the sources of phosphorus should lake restoration techniques be desirable to reduce the annual phosphorus input.

The second method involves the use of lake models. Models are much simpler as they require fewer data points; however, the commonly used models e.g., Dillon and Rigler (1975) and Reckhow and Simpson (1980) do not allow for internal phosphorus loadings. Other models designed to compensate for the internal release of phosphorus are discussed in Section 5.5.6, but were found to be unsuitable for Williams Lake.

The San Jose River is the only major tributary to Williams Lake, consequently, it represents the largest external phosphorus source to the lake. Although the San Jose River has many tributaries and a large

watershed, the watershed loading to Williams Lake can be calculated by simply measuring stream flow and stream phosphorus concentrations at the inlet to Williams Lake. Sufficient data are available for the San Jose River for the years 1977 to 1985 to calculate the phosphorus loading rate. The loading rate should be calculated for a 12-month period from spring overturn to the next spring overturn.

The total and dissolved phosphorus loading rates from the San Jose River are summarized in Table 8, and illustrated in Figure 12 with the spring overturn phosphorus concentrations. The maximum total and dissolved phosphorus inputs of 10 200 and 6500 kg were recorded in 1976-1977. The minimum loadings for the same fractions of phosphorus were 1690 and 1180 kg respectively in 1979-1980. The average total and dissolved phosphorus loadings for the San Jose River, based on the data presented in Table 8, are 4930 ± 3177 kg (n=11) and 3300 ± 2125 kg (n=11), respectively. The loading of suspended phosphorus can be calculated by subtracting the dissolved phosphorus from the total phosphorus loadings.

The relationship between the annual phosphorus loading from the San Jose River and annual stream flow is shown in Figure 13. The phosphorus loading rates (total and dissolved) increase with increasing flow to approximately 75 000 dam³ of inflow, accounting for 500 to 1800 kg of the total phosphorus input. Above that point, suspended phosphorus input increased to 4000 kg in 1982-1983. Increased stream flow does not appear to influence the suspended phosphorus associated with stream turbidity below 75 000 dam³ of inflow. Above that point, turbidity from stream bank erosion will increase causing elevated suspended phosphorus loading. the decrease in the loading of dissolved phosphorus during high stream flow suggests adsorption of dissolved phosphorus by the suspended particles.

Other phosphorus sources that can be estimated are atmospheric deposition, septic tanks, and the release of phosphorus from the decomposition of aquatic plants. Atmospheric deposition rates in the Okanagan Region are estimated to be approximately 35 kg/km²/yr (Haughton, et al.,

1974). Using this loading rate, the atmospheric loading to Williams Lake is estimated to be 250 kg P/yr.

Septic loadings to Williams Lake have been estimated to be less than 3 percent of the total annual input by Stitt et al. (1979). Considering that development around the lake has not increased significantly since the study, the septic loadings are not expected to have increased.

Aquatic plants are capable of releasing as much as 2000 kg P/km²/yr during decomposition (Smith and Adams, 1986). The majority of the phosphorus released from plants was translocated from the sediments via the root system. Assuming 10% of the lake area is occupied by dense aquatic plants (based on preliminary air photo interpretation) approximately 1400 kg P/yr is released from the weed beds in the fall. This figure is only a rough estimate, but it illustrates the importance of the weeds in phosphorus loading.

5.5.5 INTERNAL PHOSPHORUS LOADING

Internal phosphorus loading refers to the process by which orthophosphorus is released from lake sediments. There appears to be a two-year lag between years of high phosphorus loading from the San Jose River and high spring overturn concentrations (Section 5.5.3). The observed lag appears to be the result of high phosphorus loading during high runoff years being released one or two years later during the anoxic conditions present in the hypolimnion during the summer months.

Quantification of the internal phosphorus loading in Williams Lake can be demonstrated from samples collected on October 3, 1983, by estimating the orthophosphorus mass in the hypolimnion prior to fall overturn. Within the hypolimnion on October 3, 1983, approximately 4 500 kg of orthophosphorus was present. The loading is almost equal to the 4 900 kg of total phosphorus transported to Williams Lake via the San Jose River in an average runoff year.

Sufficient phosphorus is released from the sediments to raise the orthophosphorus concentrations at the lake surface from $<3 \mu\text{g/L}$ in August 1976 to $70 \mu\text{g/L}$ by November of the same year. Assuming all the orthophosphorus is derived from the sediments, there is a net increase of 5 700 kg in orthophosphorus during that time period. The annual variability of internal phosphorus loading is unknown as insufficient data have been collected in other years at fall overturn.

As discussed in Section 5.5.1, the orthophosphorus released from the sediments during the summer months is not precipitated to the sediments during the winter but remains in solution until the following spring overturn. At spring overturn the orthophosphorus is utilized by algae during the summer months causing poor water quality and high hypolimnetic oxygen depletion rates.

In order to interrupt the internal cycling of phosphorus and improve water quality, we must first understand the mechanisms that cause the phosphorus cycle in Williams Lake. Stauffer (1981), Bostrom *et al.* (1982), and Stauffer (1985) give very good reviews of the phosphorus release mechanisms for noncalcareous and calcareous lakes.

Under normal lake conditions, the phosphorus binding capacity of the sediments is determined by the activity of Fe^{3+} ions in the interstitial water. Fe^{3+} ions actively bind with orthophosphorus to form an amorphous iron complex similar in structure to strengite ($\text{FePO}_4 \cdot 8\text{H}_2\text{O}$). For every increase in pH there is a 1000 fold decrease in the Fe^{3+} activity, and for every decrease in pe (a measure of the electron activity and oxidation-reduction potential) the ratio of $\text{Fe}^{2+}:\text{Fe}^{3+}$ increases 10 fold (Lindsay, 1979). The decreased Fe^{3+} activity through elevated pH and lowered pe levels reduces the phosphorus binding capacity of the sediments. The phosphorus binding capacity of the lake sediments is further decreased by the binding of Fe^{2+} ions with reduced sulphur (S^{2-}) to form iron monosulphides. Reduced sulphur is biogenically formed under reducing conditions, and it is an important factor in the inactivation of iron in

sediments. The combination of high pH, low pe, and the formation of iron monosulphides is critical in the phosphorus binding capability of lake sediments, and one or more of these factors can be responsible for the release of orthophosphorus into the hypolimnion described as internal phosphorus loading.

In Williams Lake, the pH near the sediment-water interface during the summer months is approximately 7.6, which is 1.3 pH units lower than the surface waters. The buildup of CO₂ in the hypolimnion is caused by the decomposition of seston and organics in the sediments. Despite the lower pH in the summer hypolimnion, the phosphorus binding capability of the sediments is limited because of the reducing conditions (determined by the presence of hydrogen sulphide) and the build up of iron monosulphides.

During the winter months, the phosphorus binding capacity of the sediments is increased theoretically, through the elimination of reducing conditions by the introduction of oxygen at fall overturn, but this increase appears to be nullified by an increase in pH from 7.6 to 8.3. The result is that phosphorus remains in solution through the winter until spring overturn when it is utilized by the algae forming the nuisance algal blooms and poor water quality. Sedimentation of the algae into the hypolimnion during the summer and the mineralization of the organic phosphorus within the algal cells, combined with the low phosphorus binding capacity of the sediments, causes the cycle to repeat.

5.5.6 LAKE MODELS

The major sources of phosphorus to Williams Lake are internal phosphorus release from the sediments during summer stratification and watershed loading from the San Jose River. Many models have been developed in North America and Europe to estimate the spring overturn phosphorus concentration knowing the phosphorus sedimentation rate, mean depth, hydraulic retention time, and the phosphorus loading rate. The models provide a method of estimating the spring overturn phosphorus concentration using the input

variables. Models also predict the effectiveness of proposed lake restoration techniques before they are implemented. Despite their advantages, the models do not apply to calcareous lakes and lakes with significant internal phosphorus loading (Stauffer, 1985). Stauffer notes that the models were developed using data from oligotrophic to mesotrophic noncalcareous lakes with aerobic hypolimnia. These lakes did not have hydroxyl inactivation of sedimentary amorphous iron which reduces the phosphorus-binding capacity of the sediments, and they did not have anoxic hypolimnia causing internal loading. Both phenomena occur in Williams Lake altering the phosphorus sedimentation rates, a key component of the models.

Models have been developed for lakes with anoxic hypolimnia (Reckhow and Simpson, 1980; and Nurenburg, 1984). Testing of these models on Williams Lake did not adequately estimate the spring overturn phosphorus concentrations (Table 7), or the internal phosphorus load calculated for 1977 and 1983 (Section 5.5.5). As a result, lake models are not applicable to the conditions observed in Williams Lake, and consequently the models were not used.

5.5.7 CHLOROPHYLL-a

Chlorophyll-a is used to estimate algal biomass, which in turn is a measure of a lake's primary productivity. Dillon and Rigler (1974) considered a mean summer chlorophyll-a concentration of 2 $\mu\text{g/L}$ to be the optimal concentration for lakes with important cold water fisheries. They also recommended a chlorophyll-a range of 3 to 5 $\mu\text{g/L}$ for lakes with important recreational values, where some protection is required for the cold water fisheries.

The direct relationship between spring overturn phosphorus and mean summer chlorophyll in phosphorus-limited lakes was first observed by Sakamoto (1966). Several authors (Dillon and Rigler, 1974; Hern et al., 1981; Forsberg and Ryding, 1980) have observed similar relationships in

north-temperate lakes. Nordin and McKean (1984) described the same relationship using data collected from British Columbia lakes (Figure 14).

Chlorophyll-a data were collected from the centre station (0603019) in 1977. One surface sample was collected in late June and two surface samples were collected in early and mid-August. The chlorophyll-a concentration averaged 30 µg/L with a minimum of 6.1 µg/L in June and a maximum of 58 µg/L in early August. The average surface concentration in 1977 is well within the eutrophic zone as determined by Wetzel (1975)(Figure 14).

Based on the spring overturn phosphorus concentrations in Table 7, the mean summer chlorophyll-a concentrations are expected to range between 10 and 20 µg/L in Williams Lake (Figure 14). Stauffer (1985) noted that the mean summer chlorophyll-a concentration for calcareous lakes (like Williams Lake) would be elevated relative to their respective spring overturn phosphorus concentrations. There are two reasons for the difference: 1) the phosphorus-chlorophyll-a relationships were developed from oligotrophic and mesotrophic softwater lakes; 2) calcareous lakes have higher internal phosphorus loading rates because of hydroxyl ion reduction through the complexing of phosphorus with iron (Bostrom et al. 1982).

Because Williams Lake is a hard water, eutrophic lake, the chlorophyll-a levels are expected to be higher than shown by the relationship in Figure 14. Precise prediction of chlorophyll-a using spring overturn phosphorus can only follow several years of data collection.

5.6 TROPHIC STATUS

The trophic status of Williams Lake can be designated using a variety of criteria. The most common criteria are spring overturn phosphorus, chlorophyll-a, Secchi disc depth, and hypolimnetic oxygen depletion rates. Based on all the above criteria, Williams Lake is considered eutrophic (Table 9).

5.7 PHYTOPLANKTON AND ZOOPLANKTON

There are no known zooplankton samples collected from Williams Lake, but nine phytoplankton samples have been taken during the summer months (Stitt et al. 1979). The blue-green alga Aphanizomenon flos-aquae was the dominant species (typical for eutrophic lakes) in all nine phytoplankton samples collected from 1974 to 1977. Palmer (1962) notes that A. flos-aquae at high densities can cause taste and odour problems as well as clog filters. There are no algae density data reported by Stitt et al. (1979); however, based on the authors' personal observations, the concentration of algae during the summer can be sufficient to impart a taste or odour and clog filters.

5.8 AQUATIC MACROPHYTES

The aquatic macrophytes throughout Williams Lake were identified and mapped in 1974 by Goddard (1977). A more complete and more current species list (Table 10) was available from Warrington (1980). The macrophyte Myriophyllum exalbescens forms dense mats at the inlet and outlet of the lake. The outlet has extensive shallow areas with slow moving water which is ideal habitat for this aquatic plant. Although the eutrophic nature of Williams Lake does not determine which rooted macrophytes grow in the lake (i.e., the plants are responding to habitat preference), the high nutrient concentrations in the water are expected to enhance the density of macrophyte growth. A survey to map macrophyte species distributions and densities should be completed to provide a bench mark for future comparison.

Because aquatic plants can obtain as much as 25% of the nutrients required for growth from the water (Smith and Adams, 1986), significantly reducing the lake phosphorus concentrations would theoretically reduce the density of aquatic macrophytes at the outlet of the lake. Section 5.12 outlines methods of reducing the phosphorus concentrations in Williams Lake.

5.9 LAKE SEDIMENTS

Sediments from the bottom of the lake can be used to explain sediment-water interactions, while sediment cores can be used to explain the history of a lake. To date, several surface samples and a two-metre core have been taken from the deepest point in the lake.

The surface sediments are characterized as being very watery and blackish brown. In the core, the blackish-brown colouration is homogeneous to a depth of 170 cm below the sediment-water interface where there is a minor shift to a brownish black sediment (Zirnhelt, 1984). The black colouration is caused by iron monosulphides which are an insoluble black compound formed in the hypolimnia of lakes under reducing conditions (Lindsay, 1979).

Chemical analyses of the Williams Lake sediments and other lakes within the Cariboo Aspen-Lodgpole Pine Biogeoclimatic Zone are summarized in Table 11. Phosphorus and the metals in Table 11 do not indicate excessive loading when compared to the other lakes in the Cariboo area. Of particular interest are the Kjeldahl nitrogen and loss on ignition (which is an indirect measure of organic carbon). Eutrophic lakes with flushing rates less than once per year (like Williams Lake) typically have much higher Kjeldahl nitrogen and 'loss on ignition' results. The low results in Williams Lake are due to high inorganic sedimentation rates derived either from stream erosion within the San Jose River or from mineral carbonate formations within the lake (e.g., calcium carbonate). Some calcium carbonate formation is likely as the inorganic carbon concentrations within the lake are near saturation (Section 5.3), and the calcium concentrations are twice the values in sediments from other lakes in the area. Based on the core data collected the calcium is associated with the inorganic carbon, presumably as CaCO_3 .

Lakes that undergo calcium carbonate precipitation are called 'marl' lakes. These lakes are characterized as being less productive than non-marl

lakes as calcium carbonate is effective in co-precipitating dissolved and suspended phosphorus, rendering it unavailable for algal growth. Marl typically precipitates into the hypolimnion where a portion will become dissolved because of lower temperatures and pH. Portions of the suspended and dissolved phosphorus are liberated in the hypolimnion, but the majority of the nutrients will not be available for algal production until fall overturn when the hypolimnion and epilimnion mix.

True marl lakes will have as much as four times the calcium content in their sediments as Williams Lake. Consequently, although it appears that calcium carbonate precipitation does occur in Williams Lake, the lake can not be considered a true marl lake.

Magnesium concentrations in sediments are much lower than the calcium values despite higher concentrations in lake water. This is caused by magnesium carbonate's increased solubility relative to calcium carbonate.

Zirnhelt (1984) analysed a 2 m core which is thought to extend past the cultural horizon of 1850. The trophic status of the lake prior to cultural development is thought to have been mesotrophic to slightly eutrophic which is the present state of Lac la Hache.

5.10 COLIFORM BACTERIA

Significant fecal coliform contamination of the San Jose River has been documented by Stitt et al. (1979), with concentrations being generally low at the outlet of Lac la Hache (<10 MPN/100 mL for 82% of samples) and increasing downstream to the mouth where concentrations averaged 94 MPN/100 mL. The major inputs appear to be during February from the two major tributaries, Borland Creek and Knife Creek. The increases coincided with the major inputs of nutrients from nonpoint sources during runoff conditions in the spring and are therefore attributed to a cattle manure source.

Total and fecal coliform samples are monitored in Williams Lake on a weekly basis from June through August, primarily at Scout Island Beach (Figure 4). Rotary and Fethers Beach are also monitored, but less frequently. To date, the fecal coliform density has not exceeded a running log mean of 4/100 mL, and no single sample has exceeded 21/100 mL in the past four years (Vath, 1987).

While the river water entering the lake is contaminated by agricultural runoff, the only sample taken from the centre of the lake (August, 1978) was <2 MPN/100 mL. It is expected that bacteria will die and settle to the bottom once in the lake. This was also found to be the case in Hendrix Lake which received a direct discharge of sewage from a treatment plant (Zirnhelt, unpublished data).

5.11 NONPOINT PHOSPHORUS LOADING

5.11.1 ANALYSIS OF WATER QUALITY

The phosphorus loading to Williams Lake via the San Jose River is a function of the phosphorus concentration in the river and the volume of water discharged by the river. Section 5.5.4 discusses the annual phosphorus load from the San Jose River and its impact on the water quality of the lake. This section discusses the phosphorus concentration in the San Jose River and its tributaries, to identify areas of nonpoint phosphorus loading.

Within the San Jose watershed there are 25 stations with up to 99 samples per station (Figure 15). Phosphorus analyses have been carried out from 1977 to 1985. By calculating changes in concentration down the river the areas of significant nonpoint phosphorus loading can be evaluated. The nonpoint phosphorus loadings are calculated for the San Jose River as a whole and then for the following individual sections of the river: a) the San Jose River between Lac la Hache and Knife Creek, b) the San Jose River between Knife and Borland Creeks, c) Knife Creek, and d) Borland Creek.

Figure 16 illustrates the dissolved and total phosphorus concentrations measured in 1978 on the San Jose River downstream from Borland Creek and the monthly stream flow for the same period. The figure clearly shows high dissolved phosphorus concentrations prior to the peak stream flow. The difference between total and dissolved is the suspended phosphorus concentration which is minimal in February and March, but is significant from April to August. Stream bank erosion due to higher stream flows and summer rains is responsible for the higher suspended phosphorus concentrations.

This pattern is typical of other years; however, the concentration of suspended phosphorus from stream bank erosion is dependent on the amount of watershed runoff (Section 5.5.4). The peak in dissolved phosphorus prior to higher stream flows is thought to be due to localized snow melt in the lowlands around the San Jose River and its tributaries. These areas are used to overwinter cattle.

The phosphorus loadings from nonpoint sources are expressed as a change in concentration per km of river and a change in concentration per km² of watershed (Table 12). The sampling stations used for the loading calculations and their associated stream lengths and watershed sizes are illustrated in Figure 17.

The dissolved and total phosphorus loadings based on stream length averaged $1.19 \pm 1.19^*$ and 1.62 ± 1.30 $\mu\text{g P/km}$ for the entire San Jose River. The San Jose River between Lac la Hache and Knife Creek, and between Knife Creek and Borland Creek, had lower loading rates (Table 12). Knife and Borland Creeks had loading rates approximately 200 and 600 percent higher than the San Jose River as a whole.

In contrast, the average dissolved and total phosphorus loading rates ($n=13$) from Dragon Creek (McKean, 1981) were 18 ± 4 and 81.4 ± 101 $\mu\text{g P/km}$. The higher loading rates in Dragon Creek are caused by higher concentrations of cattle adjacent to the water course.

*Standard deviation

The dissolved phosphorus loading rate based on area for the San Jose River (n=38) is $0.059 \pm 0.058 \mu\text{g P/km}^2$. The San Jose River between Lac la Hache and Knife Creek had a similar loading rate, while the lower section of the San Jose River had a 120 percent increase. Knife and Borland Creeks had the highest loadings, averaging a 500 and 950 percent increase over the loading rates for the entire river. The total phosphorus loading followed a similar pattern.

It is important to note that the phosphorus loading rates (based on watershed area) for the reaches of the San Jose River are less than loading rates for the individual tributaries. This may be due to two reasons: 1) reduced agricultural density or 2) better agricultural practices.

Detailed sampling of Borland Creek was completed in 1985 to identify areas of high nonpoint phosphorus loading. Dissolved phosphorus concentrations in Borland Creek above the highway at 150 Mile House (station 0600183) averaged $47 \pm 16 \mu\text{g/L}$ (n=14). No increase in dissolved phosphorus occurred in Borland Creek between 150 Mile House (0600183) and above the confluence of Valley Creek (0600322), or between Valley Creek (0600330) and the confluence of 5 Mile Creek (0600105).

This is a sharp contrast to the data presented by Stitt et al. (1979). Although the data are not directly comparable (due to hydrological differences), dissolved phosphorus loadings prior to 1980 from a feedlot operation located next to Borland Creek above Valley Creek were very high. Since 1980 the feedlot has reduced its nonpoint loading with the result of no detectable impact in 1985.

Samples collected from Valley (0600319) and 5 Mile Creeks (no site number) showed elevated dissolved phosphorus concentrations ($118 \pm 33 \mu\text{g/L}$ (n=7) and $372 \pm 191 \mu\text{g/L}$ (n=8), respectively).

The results of the 1985 study indicate that phosphorus loading occurs in the headwaters of Borland Creek (above 150 Mile House), and very high loadings occur in Valley and 5 Mile Creeks.

5.11.2 ANALYSIS OF SOILS AND LANDFORMS

Nonpoint pollution from pastures and rangelands supporting livestock can be attributed to several factors. These include (i) those related to stock density, length of grazing period, manure loading rate, manure spreading uniformity by livestock etc.; (ii) management practices such as continuous or rotational grazing etc.; (iii) proximity of feeding, watering and resting areas to streams and waterways; (iv) surface runoff; and (v) surface erosion potential of the area in question. The rangelands in the proximity of the San Jose River, between Lac la Hache and Williams Lake, have been evaluated for their nonpoint pollution potential. Since the cultural and management factors (i) and (ii) are unknown in the study area, the problem is analysed using factors (iii), (iv), and (v).

The potential for producing surface runoff from a soil/land unit is determined using Cook's method (Table 13). It is assumed that land areas in question supported a good to excellent (90% of drainage area) vegetal cover and that surface storage of water is low. The potential of a soil unit to produce sediments, on the other hand, is based upon the erosion potential of bare soils of standard slope subjected to similar rainfall (Table 14). The soil and watershed characteristics, necessary for determining runoff and erosion potentials for each soil unit, are obtained from 1:50 000 soils and landforms maps and accompanying soil survey reports. The major soils in the study area and their characteristics are shown in Figure 18 and Table 15. The soils and landforms maps were also used to determine proximity of a land unit to a waterbody.

The surface runoff and erosion potentials, and the proximity of land units to receiving waters, are shown in Table 16. The most critical areas

from the viewpoint of contributing nonpoint pollution are those located adjacent to the receiving waters.

The phosphorus loading outlined in the previous section appears to be related to the soil and landforms within the watersheds (Figure 18). The soils and landforms adjacent to the San Jose River, between Lac la Hache and Knife Creek, are generally characterized as having a normal potential for producing runoff and a moderate erosion potential. The landforms adjacent to the San Jose downstream from Knife Creek have a normal to high potential for producing runoff and a moderate to high erosion potential. The landforms located on the lower portions of Knife and Borland Creeks have normal to high runoff potential and high erosion potential in areas of steep topography (less severe erosion potential in flat areas).

Some recommendations are noted below to alleviate nonpoint pollution problems from critical areas in the San Jose watershed.*

1. Livestock having free access to streams and congregating around feeding, watering, and resting areas located close to receiving waters cause water quality problems. (i) Fencing livestock away from streams and (ii) establishing feeding, watering, and resting areas away from waterways are, therefore, recommended.
2. Over-wintering cattle in high densities could damage vegetative cover and soil structure by trampling and compacting soil of the dormant sod. This results in increased surface runoff and pollutant (including sediment) yield and is detrimental to the water quality of nearby receiving waters.

* These recommendations are for the consideration of the Ministry of Environment and Parks, Ministry of Agriculture and Fisheries, the Cariboo Cattlemen's Association, and the land owners. A site specific evaluation is required by these groups to consider the cost and effectiveness of the recommendations.

The potential for producing surface runoff from most soil/land units adjacent to receiving waters is variable. Normal to high runoff potential is noted for units 4, 15, 16, 17, 21, 22, 23, 24, 25, 32, 42. These units, and in addition units 19 and 20, are prone to yielding moderate to high erosion losses in adverse conditions (Table 16). Maintenance of healthy vegetative cover by (i) limiting cattle activities to well-drained areas (Tables 15 and 16) and (ii) rotating cattle feeding and watering grounds will alleviate the problem to a large extent. Where damage to vegetation has already occurred, the area can be rehabilitated by (i) routing the surface runoff from up above away from the damaged area, and/or by (ii) routing the surface water generated by the damaged area through detention/settling ponds and areas of good vegetation cover.

3. Cattle activity should be limited to a minimum level in areas (i) prone to spring flooding (e.g., soil/land unit 9) and (ii) of imperfectly or poorly drained soils (units 27 and 28) with poor renovation ability.

5.12 LAKE RESTORATION TECHNIQUES

The San Jose River is the major external source of phosphorus to Williams Lake and significant reductions in nonpoint loading would improve the water quality of the lake. Overwintering of cattle away from the San Jose River has occurred since 1980, and has reduced stream bank trampling and direct phosphorus loading from manure. Additional improvements which could be made, based on soil landforms, are outlined in Section 5.11. Because of the internal phosphorus cycling problem in Williams Lake, reduction in nonpoint loading will not cause significant improvements in water quality. Some type of lake treatment will be required to reduce the impact of internal phosphorus loading. The options are 1) aeration, 2) iron addition and 3) liming.

It is difficult to determine without some laboratory experiments which technique will provide the best and most cost-effective results. All tech-

niques have been successfully used in other lake systems, and their benefits are outlined below. Should restoration be implemented, the drinking water suitability and aesthetics of Williams Lake will improve.

1. Aeration: This technique is designed to introduce oxygen into the hypolimnia of lakes during summer and winter stratification. The benefits are increased fisheries and zooplankton habitat, and elimination of the reducing conditions that cause internal phosphorus loading.

Installation and operating costs are estimated at \$100 000 and \$10 000/year respectively (Ashley pers. comm.).

2. Iron Addition: The addition of sufficient iron, in the form of iron chloride, would increase the phosphorus binding capacity of the sediments. Preliminary work by Nordin (pers. comm.) on Wood Lake indicates a dosage rate of 450 litres of pickling liquor per hectare would be required every five years. Pickling liquor is an iron chloride/hydrochloric acid mixture which is a byproduct of the iron galvanizing industry in the Vancouver area. It is obtainable at no charge when available, but the transportation and application costs would be approximately \$40/ha or \$30 000 per application.

3. Liming: The addition of calcium hydroxide causes the pH to temporarily increase to approximately 9.4. At these high pH levels a substantial percentage (7-10%) of the bicarbonate will be transformed to carbonate. For lakes with high bicarbonate concentrations (350 mg/L), carbonate will exceed its saturation point and precipitate as calcium carbonate. The precipitate physically strips dissolved and suspended phosphorus, as well as magnesium and sodium, from the surface waters of the lake. The result would be improved drinking water (i.e., softer water and lower sodium concentrations) and reduced algal productivity during the summer months.

Liming on Frisken Lake (near Kamloops) in 1983 and 1984 successfully removed 80-90% of the chlorophyll-a, and between 60-97% of the dissolved inorganic phosphorus (Murphy et al. 1985). The costs of liming a lake are approximately \$1500 per hectare per year, or \$1 000 000 per year for Williams Lake.

Based on the recent success of the destratification-aeration project on Langford Lake (Nordin and McKean, 1987), a destratification aerator in combination with periodic iron additions would be the best strategy to improve the water quality in Williams Lake. It must be emphasized that this assessment and associated costs are preliminary estimates only. Further research and experimentation will be required before the techniques can be implemented.

6. PROVISIONAL WATER QUALITY OBJECTIVES

6.1 DESIGNATED WATER USES

Water quality objectives are desirable to ensure that the present and future uses of Williams Lake are protected. This report has shown that the lake is important for:

- aquatic life (recreational freshwater fishery)
- primary-contact recreation
- drinking water supply

It is proposed that these be adopted as the designated water uses to be protected in Williams Lake. The provisional water quality objectives outlined below are set to protect the most sensitive use. If the objectives are met for the most sensitive use, then all of the other uses of the lake will be protected.

6.2 FECAL CONTAMINATION

The most sensitive uses are for drinking water supply and primary-contact recreation such as swimming and water skiing.

The objective for drinking water supplies is designed to ensure that no water treatment apart from disinfection is required. The provisional objective for drinking water supply is as follows: the fecal coliform density shall not exceed 10 per 100 mL in 90 percent of lake water samples taken in any consecutive 30-day period. The objective is based on the Ministry of Health's guidelines for the treatment of raw water supplies (B.C. Ministry of Health, 1982).

The objective for primary-contact recreation is based on the recommendations by Richards (1983). The provisional objective is as follows: the fecal coliform density shall not exceed a running log mean of 200/100 mL,

calculated from at least five weekly samples taken during the recreation season, nor shall more than 10 percent of samples during a 30-day period exceed 400/100 mL.

Fecal coliform densities have been measured in the lake by the Ministry of Health (Section 5.10) but further monitoring is necessary to determine if these objectives are being met. The objective for drinking water supply should apply to grab samples taken near or in drinking water intakes in the lake. The objective for recreation applies to bathing beaches.

6.3 ALGAL GROWTH, NUTRIENTS, AND HYPOLIMNETIC OXYGEN CONCENTRATION

Nuisance algal growth is the result of excessive phosphorus in Williams Lake. Excessive algae can cause taste and odours in drinking water, clog filters, cause aesthetic problems for recreation, reduce water clarity, and create high hypolimnetic oxygen-depletion rates. A lack of oxygen results in a loss of fisheries and zooplankton habitat.

As mentioned earlier, the biomass of algae and the hypolimnetic oxygen depletion rates are controlled by the availability of phosphorus. Although the present recreational and fisheries values of Williams Lake are minimal, the provisional water quality objectives for chlorophyll-a are set to protect future recreational and fisheries values of the lake. The mean summer chlorophyll-a guideline of 3 to 5 µg/L is recommended by Dillon and Rigler (1975) for lakes with important recreational values and where some protection is required for the cold water fisheries. A chlorophyll-a level of 5 µg/L or less would provide excellent quality water which could be used as the drinking water supply for the City of Williams Lake. To achieve a mean summer chlorophyll-a concentration below 5 µg/L, the spring overturn total phosphorus concentration should not exceed 20 µg/L. Consequently, the provisional water quality objective for the lake is a spring overturn total

phosphorus concentration of 20 µg/L. The objective applies to the samples collected at spring overturn at the surface, mid-lake, and near the sediment-water interface at sites 0603019 and 0603022.

A water quality objective for chlorophyll-a of an average of 5 µg/L or less is proposed for the May to August period. It applies to the average of at least four monthly samples, each sample consisting of a discrete sample collected from 0.5 m. Algal blooms can occur in the fall as the result of the erosion of the summer thermocline and the mixing of the epilimnetic and hypolimnetic waters. Because this bloom is not dependent on the spring overturn phosphorus concentration, the chlorophyll-a objective does not apply after September 1.

An objective is also set for the dissolved oxygen content of the hypolimnion to provide some protection of the fish and zooplankton habitat. The dissolved oxygen should not drop below 4.0 mg/L (level C, Davis 1975) to provide a moderate level of protection to juvenile and adult salmonids at any point >5 m above the hypolimnetic sediment-water interface. This objective takes into account that the dissolved oxygen content of lakes at the sediment-water interface may drop below 4 mg/L because of the oxygen demand of the sediments. An objective which applies 5 m above the sediment-water interface will ensure that over 90% of the hypolimnetic volume of Williams Lake will be suitable for fisheries production.

6.4 TURBIDITY

The water use most sensitive to turbidity is drinking water supply with disinfection only (i.e., no removal of turbidity or suspended residues).

The proposed provisional water quality objective for turbidity is a maximum level of 5 NTU and an average level of ≤1 NTU. The objectives are based on the B.C. Ministry of Health (1982) Drinking Water Standards. The maximum level shall apply to any grab sample taken near or in a domestic intake, and the average level shall apply to at least 5 weekly samples taken in a period of 30 days at any point in the lake.

In addition to the turbidity objective, a water clarity objective is proposed. The Canadian Water Quality Guidelines (CCREM, 1987) recommend a water clarity for water contact recreation of 1.2 m as measured with a Secchi disc. The Secchi disc objective of 1.2 m applies to point any in the lake from May through August. To check the objective we recommend weekly readings at site 0603019 between 10 a.m. and 2 p.m.

7. WATER QUALITY MONITORING

The purpose of the monitoring program outlined below is to check compliance with the water quality objectives outlined in Section 6, as well as to provide a complete biological and water quality data base so that future lake management programs can be formulated.

7.1 FECAL COLIFORM BACTERIA

The fecal coliform monitoring related to drinking water supplies should be five weekly samples over 30 days, anytime of the year. The number of sites and their location are left to the discretion of the sampler, but should be associated with waterworks or domestic water licences. The samples can be collected near an intake, at a tap, or at an outside location that comes directly from the lake or stream (before chlorination or filtering). The waterline should be thoroughly flushed before filling the sample bottle.

Monitoring for primary-contact water recreation should be 5 weekly samples over 30 days, during the summer months (July or August). Samples should be taken 1 m from the shore at beaches.

7.2 PHOSPHORUS, CHLOROPHYLL-a AND HYPOLIMNETIC OXYGEN

The interdependence of these three measurements necessitates a similar monitoring program. The objective of the monitoring is to calculate the spring overturn and mean summer phosphorus concentrations, the mean summer chlorophyll-a concentrations, and the hypolimnetic oxygen depletion rates.

Spring overturn is defined as the period of mixing after ice-out and before summer stratification (usually mid April). The nutrient analyses to be completed at that time are: ammonia-nitrogen, nitrate-nitrogen, orthophosphorus, and total phosphorus. Samples should be collected at the deepest part of the lake, site 0603022 (0, 10 and 20 m) and 0603019 (0, 7 and 15 m).

Chlorophyll-a, phytoplankton, and total phosphorus samples should be collected monthly from May to August at site 0603019. Chlorophyll-a, and phytoplankton samples should be discrete samples taken at 0.5 m.

Temperature and dissolved oxygen profiles should be taken at site 0603019 during spring overturn and at the same time as the chlorophyll-a samples, and once in February during ice cover.

Total and orthophosphorus samples should be collected at the surface and 1 m above the sediment-water interface. Samples should be collected at the same time as the chlorophyll-a and dissolved oxygen.

7.3 TURBIDITY

Turbidity samples should be collected 0.5m at site 0603019 during spring overturn, and then monthly using the same schedule as chlorophyll-a. In total, five turbidity samples will be collected in a summer.

Secchi disc readings are also recommended using the same schedule as turbidity. In addition, a local resident should be encouraged to take weekly Secchi disc measurements between 10 and 2 p.m. at site 0603019.

7.4 AQUATIC MACROPHYTES

Increased recreational use of Williams Lake is inevitable. The impairment of water recreation by aquatic macrophytes will generate complaints about the macrophyte growth. A survey to map existing macrophyte species and densities will provide a data base for checking public complaints and provide a bench mark to ascertain if macrophytes are becoming more of a nuisance.

7.5 NONPOINT PHOSPHORUS LOADING

Due to the importance of nonpoint phosphorus loading from the San Jose watershed some monitoring is required. Monitoring of the San Jose River for dissolved and total phosphorus at the outlets of Lac la Hache (0600021) and Williams Lake (0600013), and at the inflow to Williams Lake (0600316), will provide useful long-term data adequately describing changes in nonpoint phosphorus loading. Maintaining stream flow gauges on the San Jose River and at the outlet of Williams Lake will also be important in order to calculate nonpoint phosphorus loading and the lakes's flushing rate.

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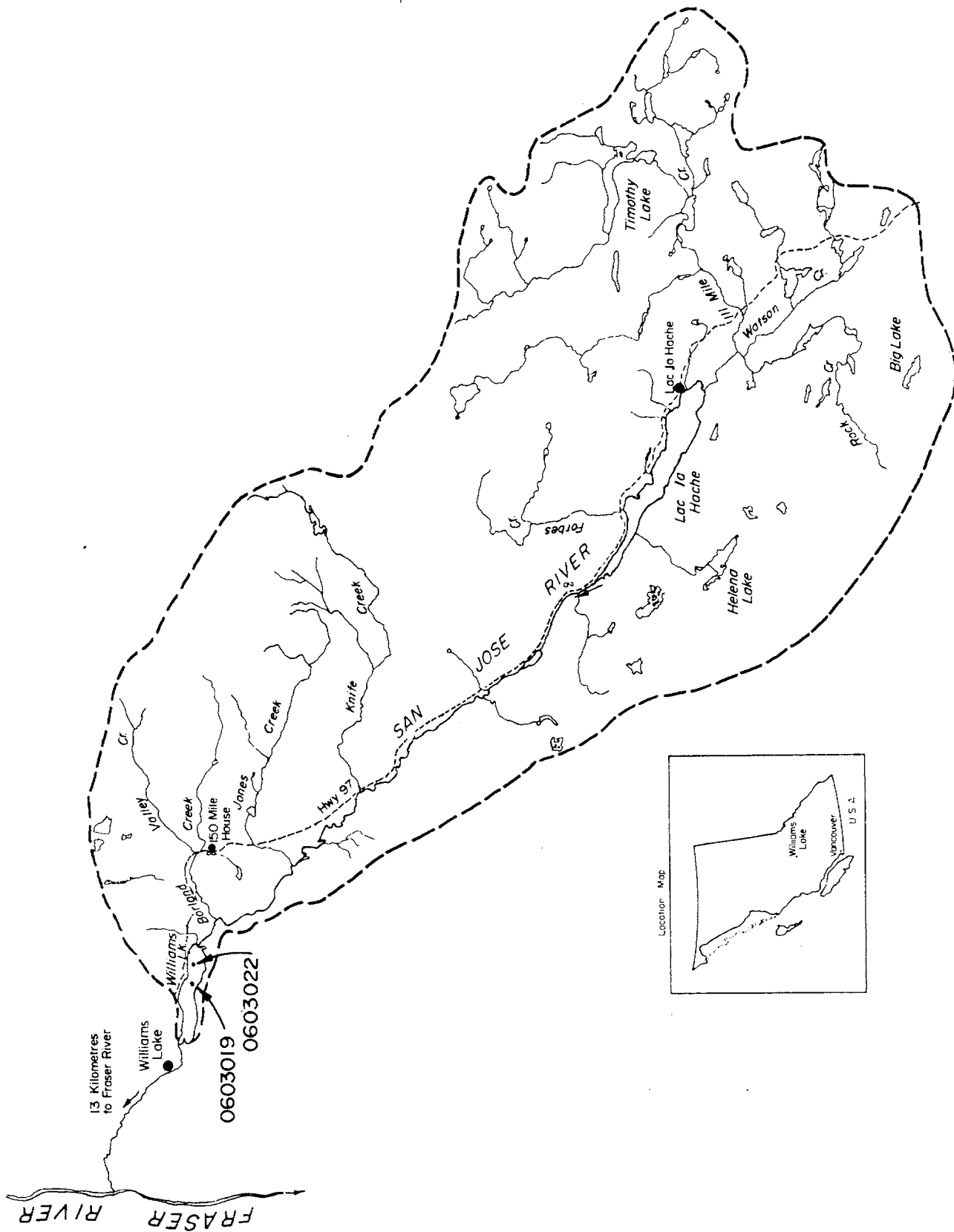


FIGURE 1 - Williams Lake Watershed, scale 1cm = 4 km

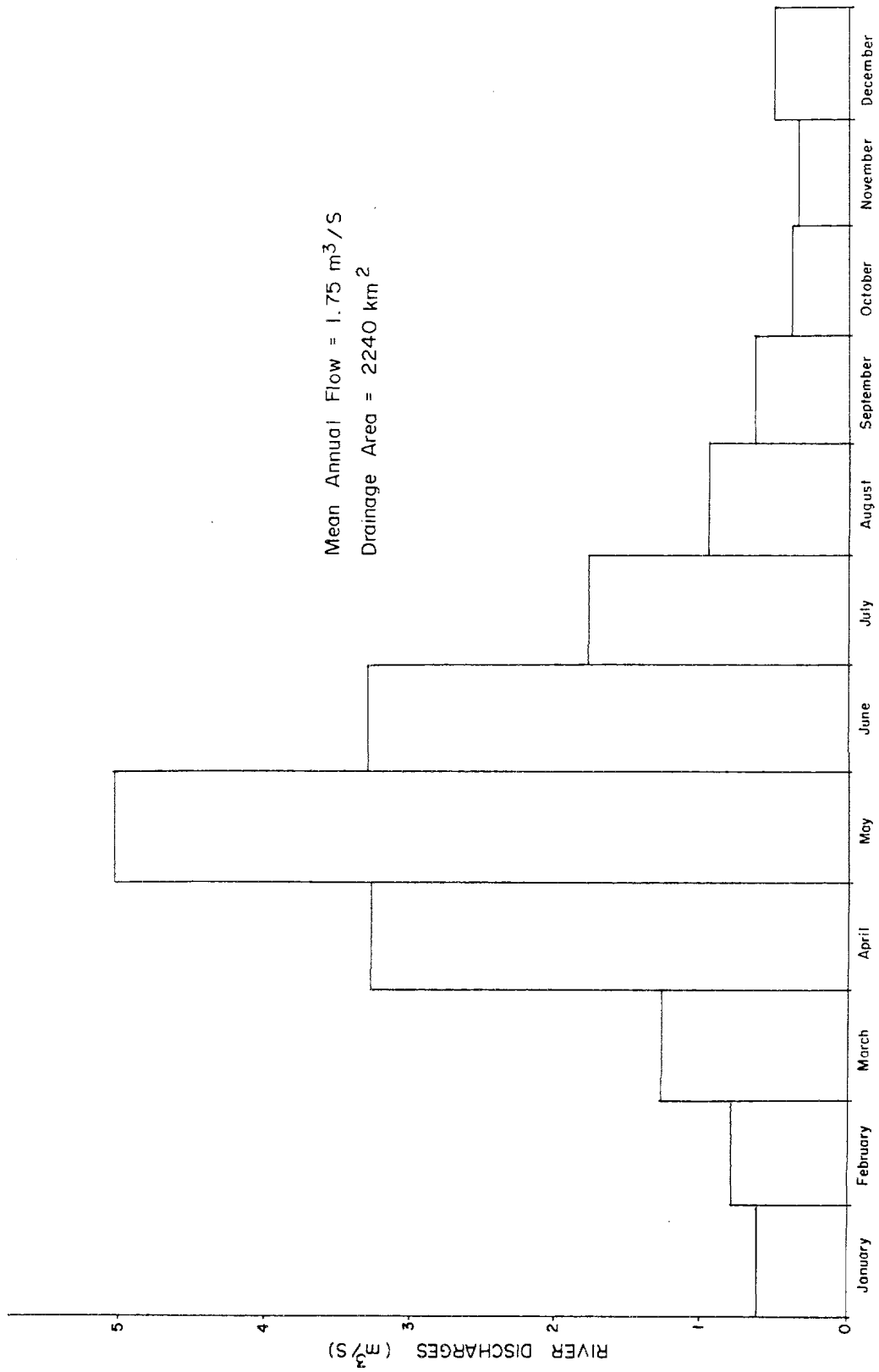


Figure 2: Mean monthly flow : Williams Lake River (08 MC 005) 1969 - 1982

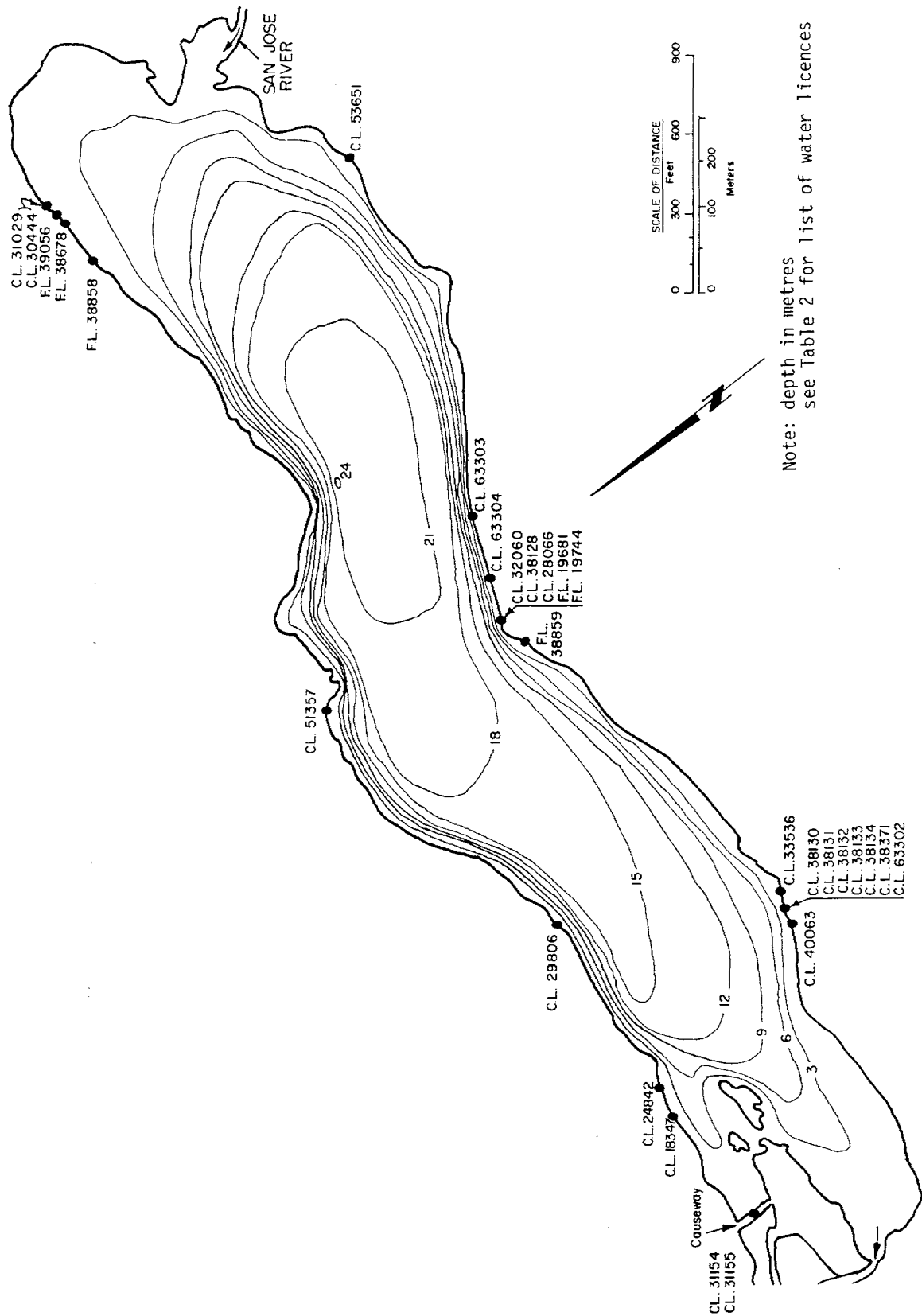


Figure 3: Water Licence points of diversion for Williams Lake.

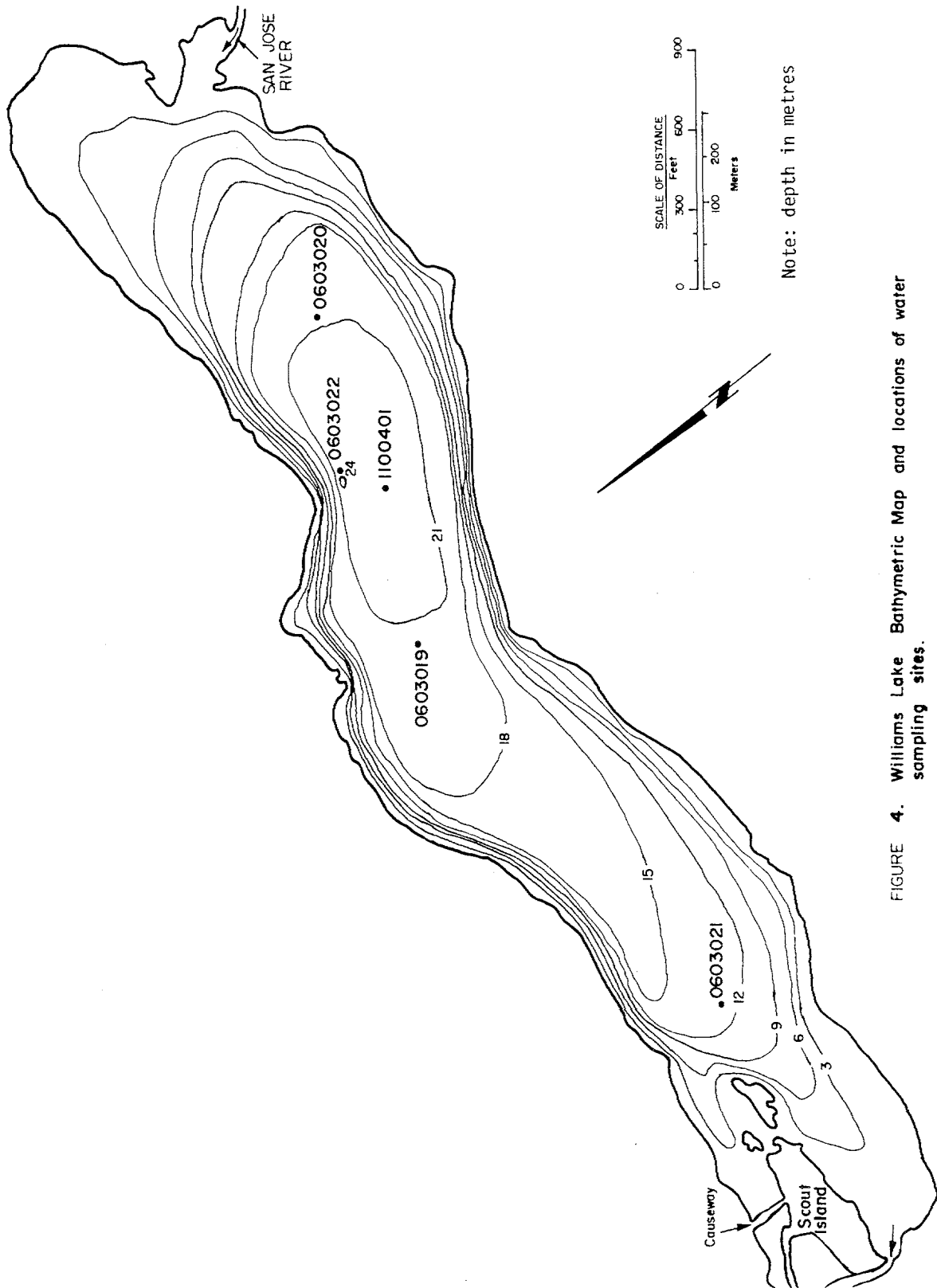


FIGURE 4. Williams Lake Bathymetric Map and locations of water sampling sites.

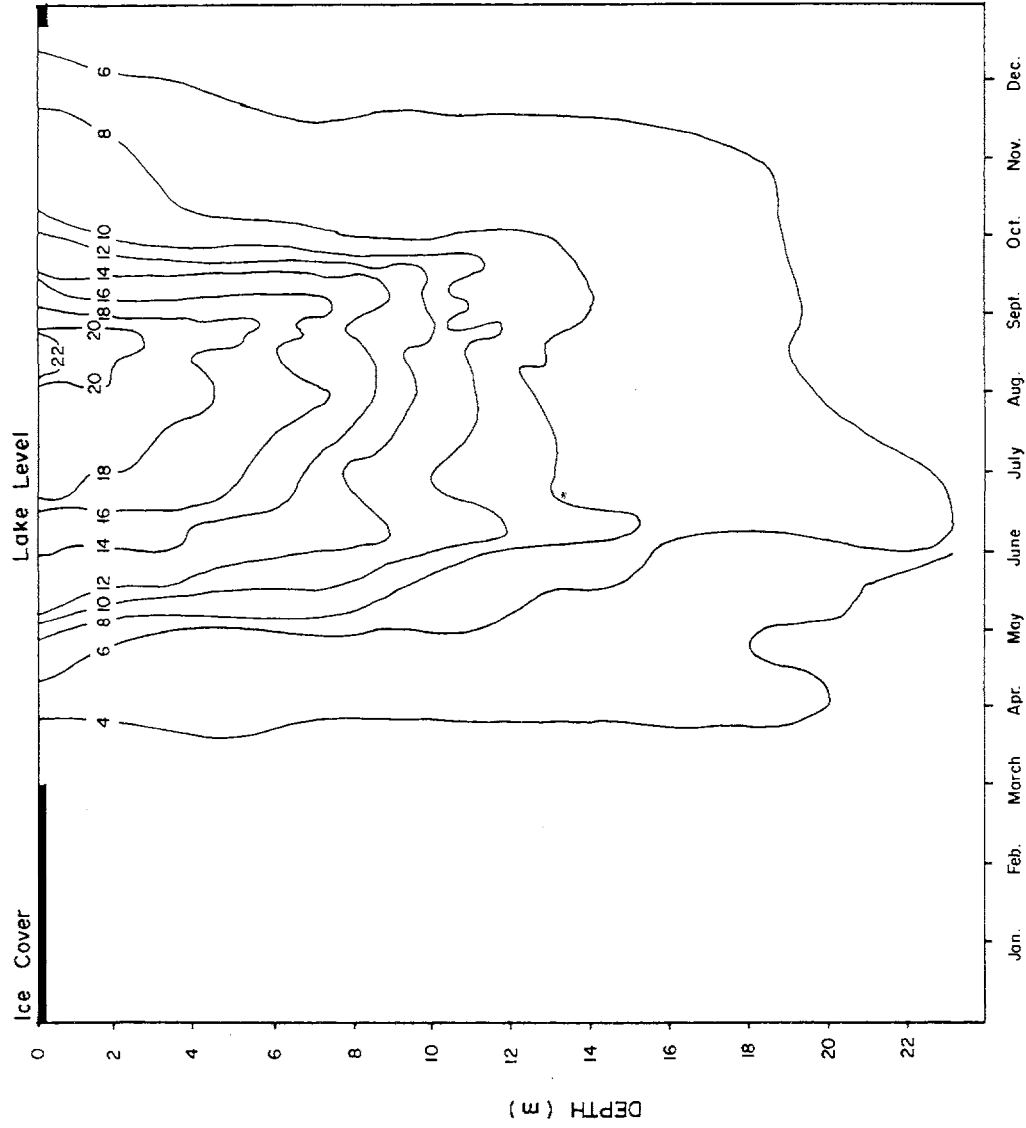


Figure 5. Time - Depth - Temperature profiles in Williams Lake ($^{\circ}\text{C}$) at station 0603019 in 1977

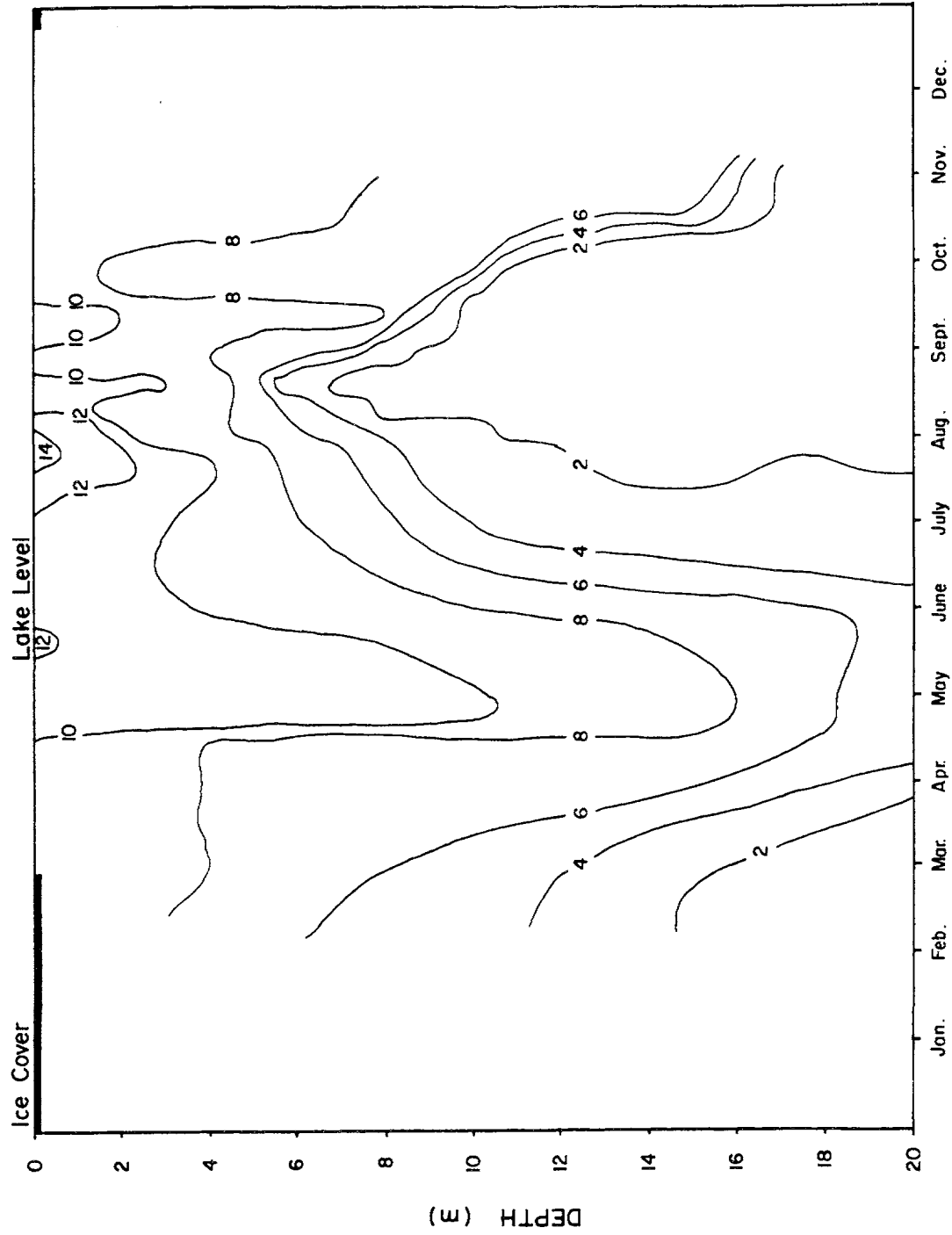


Figure 6. Time - Depth - Dissolved oxygen concentrations (mg/L) in Williams Lake at station 0603019 in 1977.

SYMBOLS

Ca - Calcium
Mg - Magnesium
Na - Sodium
K - Potassium
CO₃ - Carbonate
HCO₃ - Bicarbonate
SO₄ - Sulphate
Cl - Chloride

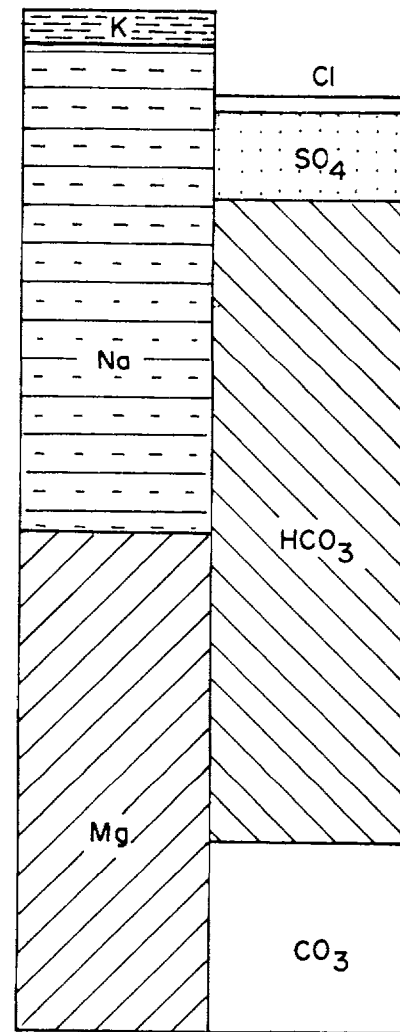
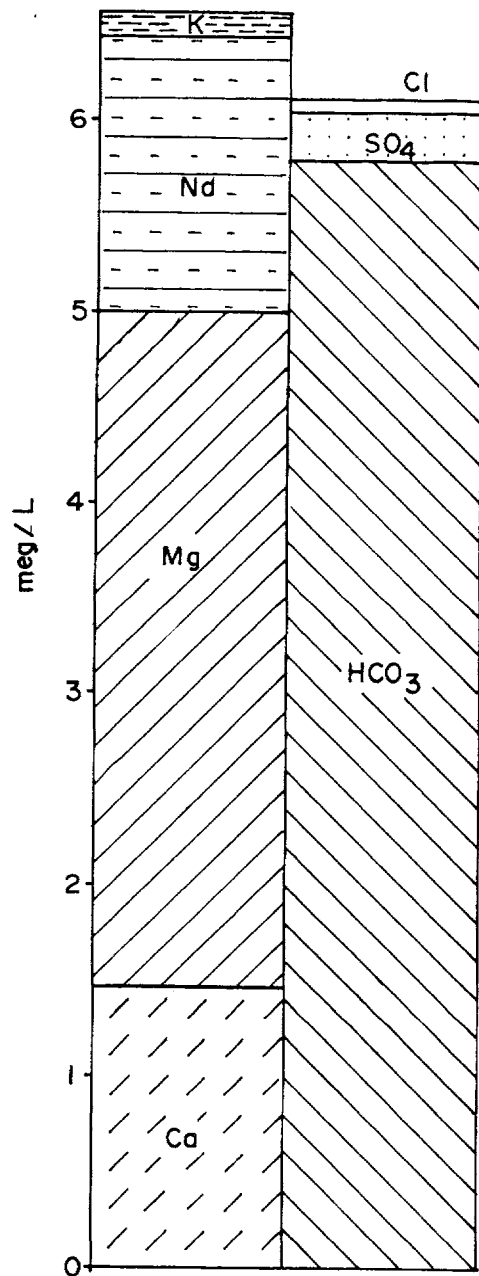


Figure 7. Anion - Cation balance for Williams Lake and the water supply for the City of Williams Lake.

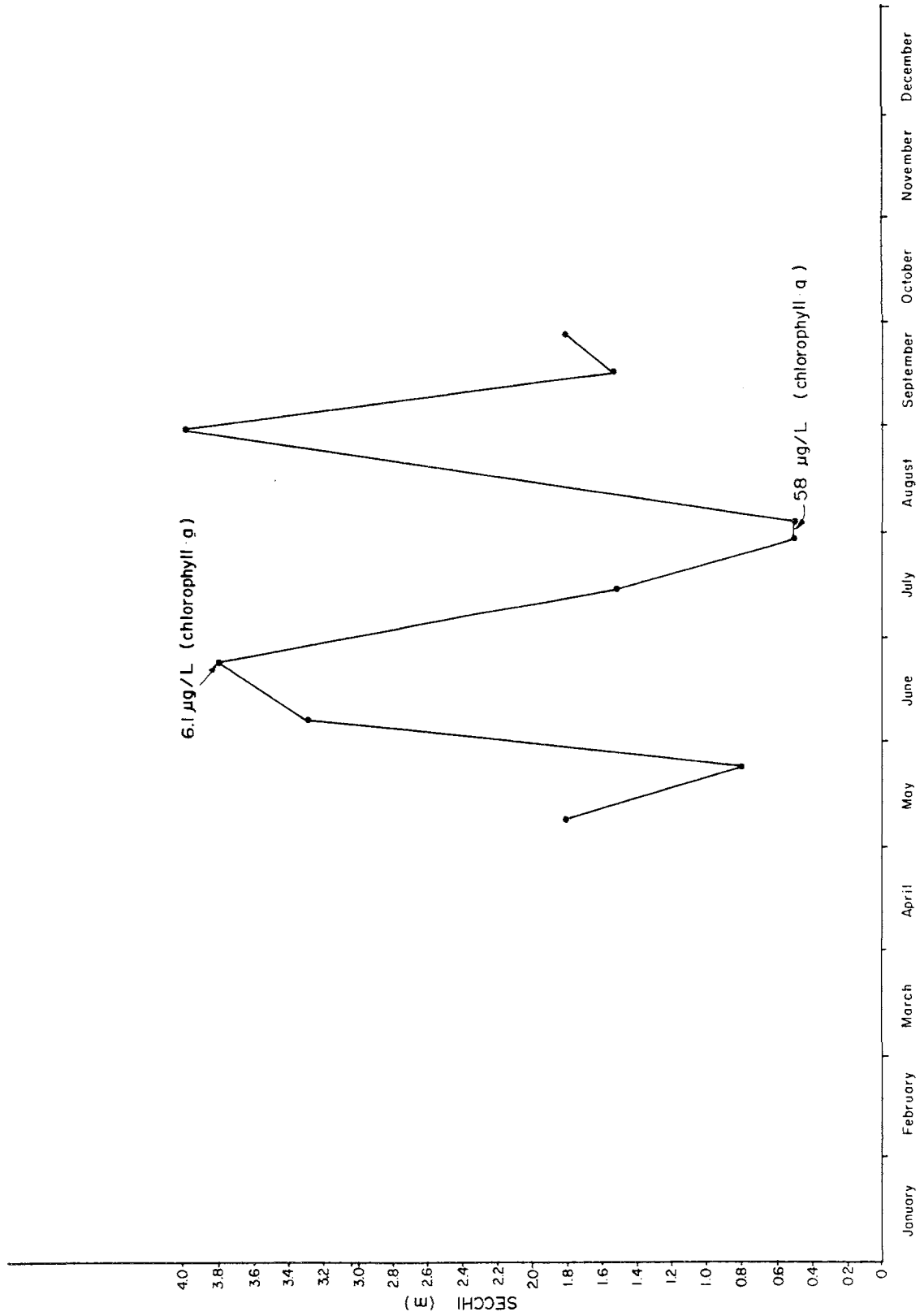


Figure 8. Secchi disc and algal biomass measurements for Williams Lake in 1977 at site 0603019.

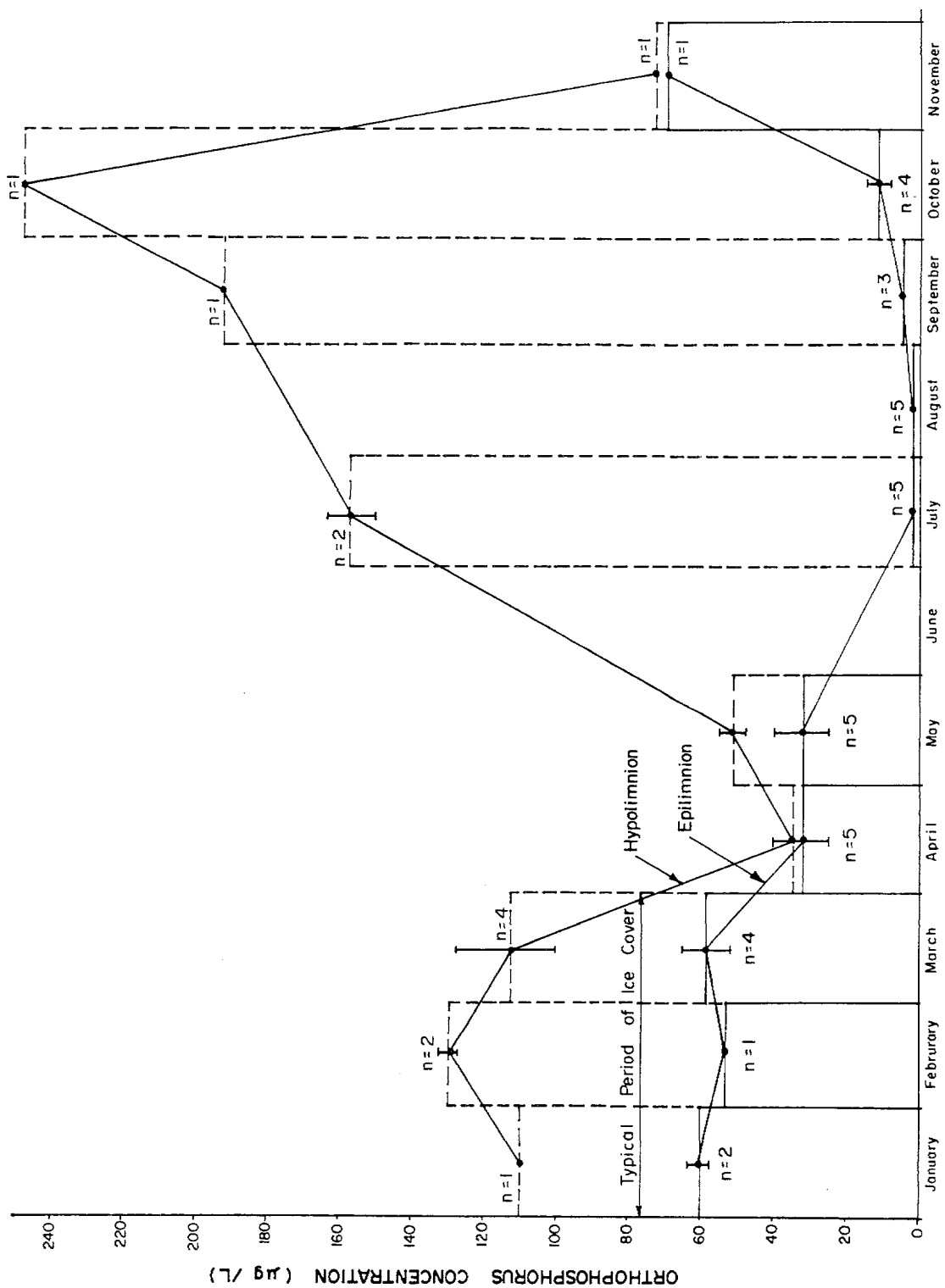


Figure 9: Mean monthly Orthophosphorus concentrations for the Epilimnion and Hypolimnion of Williams Lake 1973 - 1985

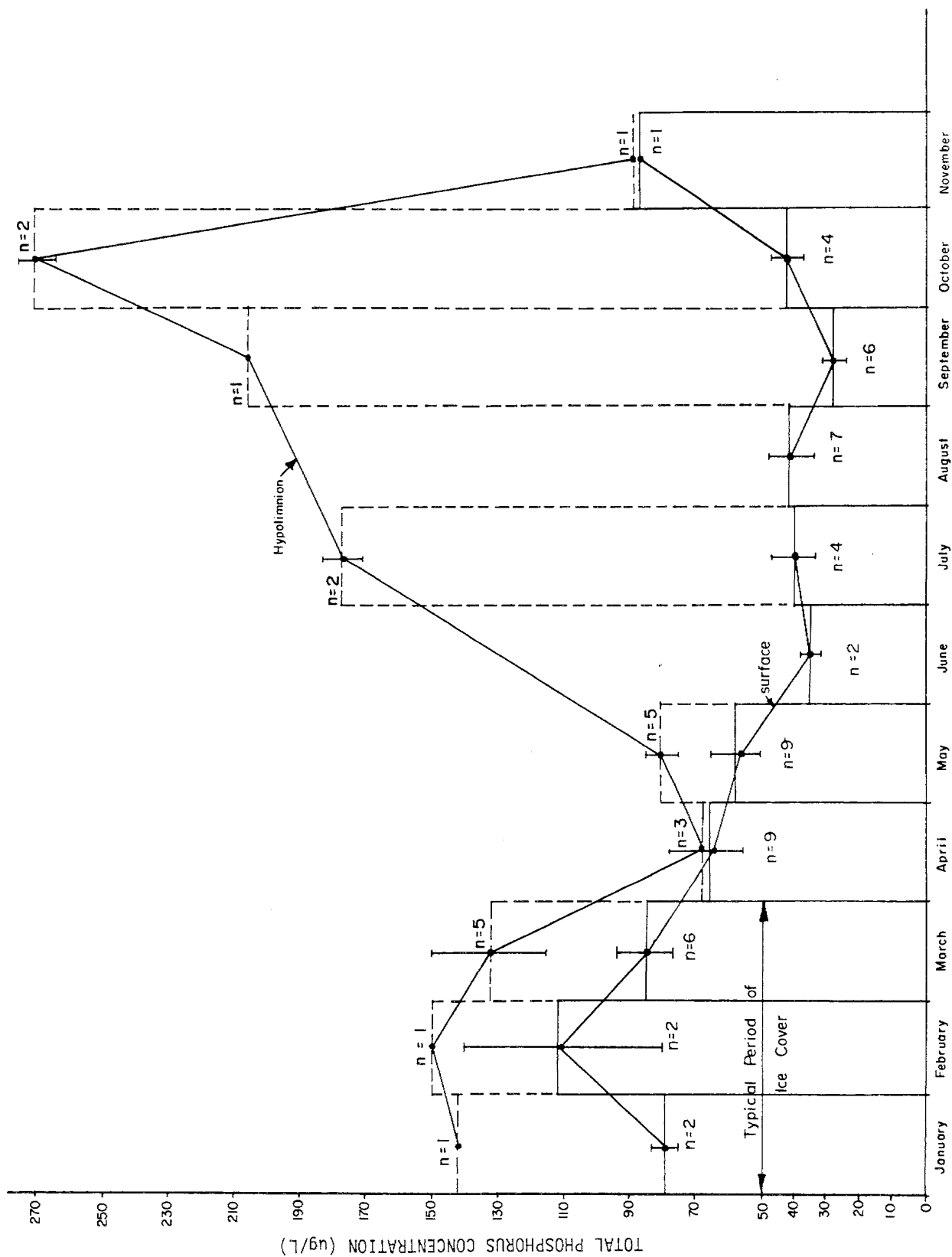


Figure 10. Mean monthly total phosphorus concentrations for the Epilimnion and Hypolimnion of Williams Lake 1973 - 1985

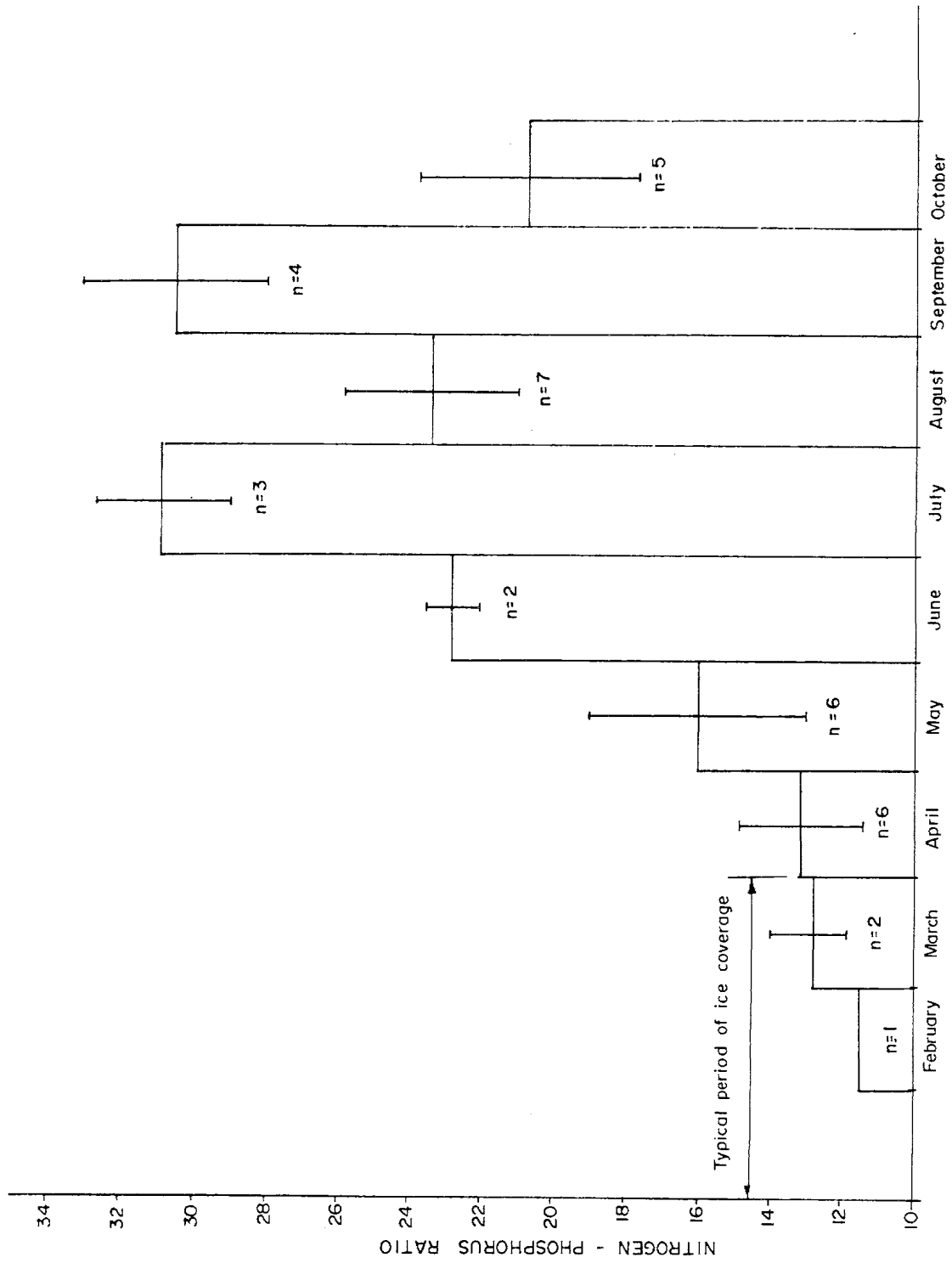


Figure II. Mean monthly nitrogen - phosphorus weight ratio for the surface water of Williams Lake. (1983 - 1985)

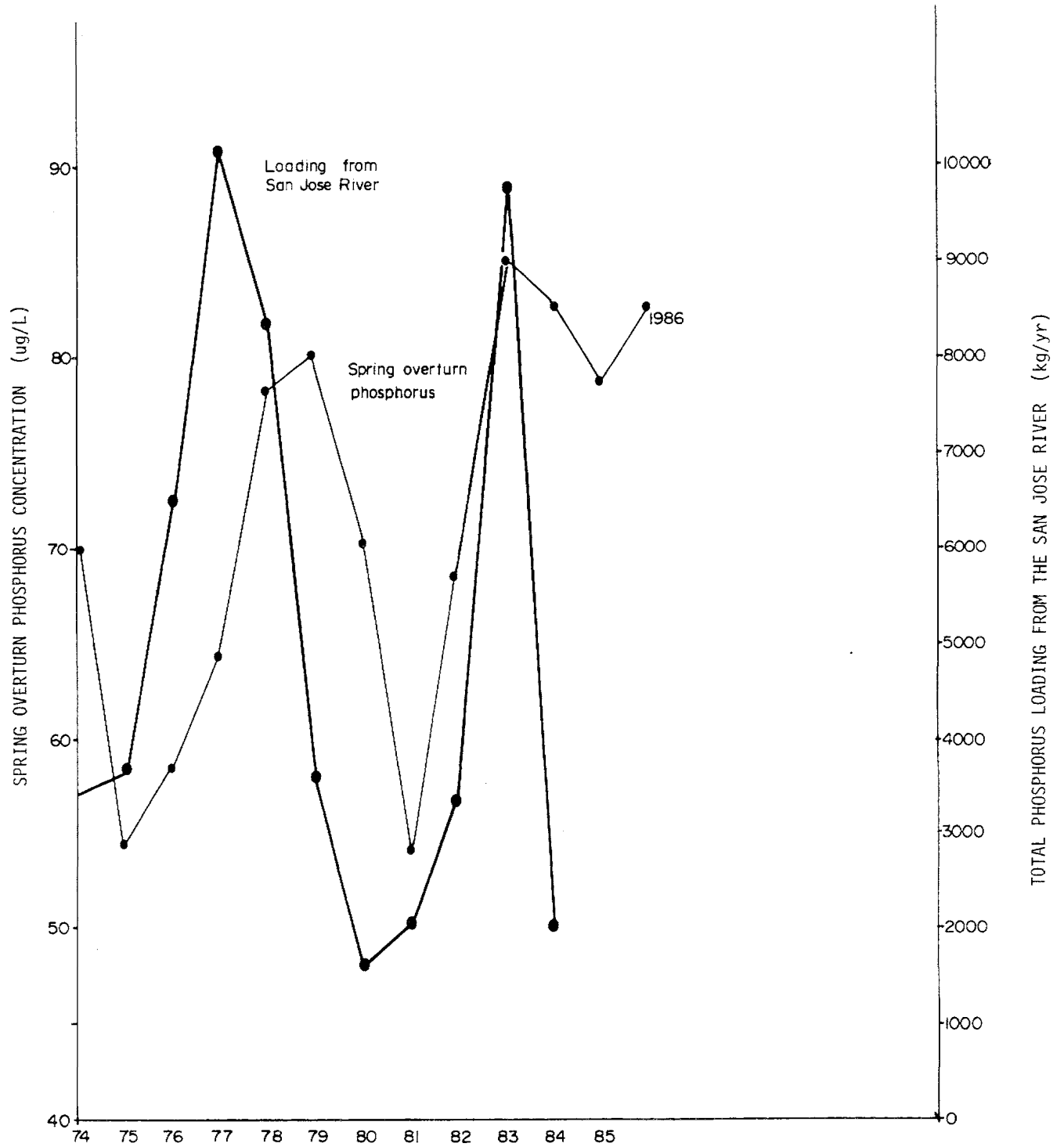


Figure 12. Spring overturn total phosphorus concentrations and nutrient loading via the San Jose River from 1974 to 1986.

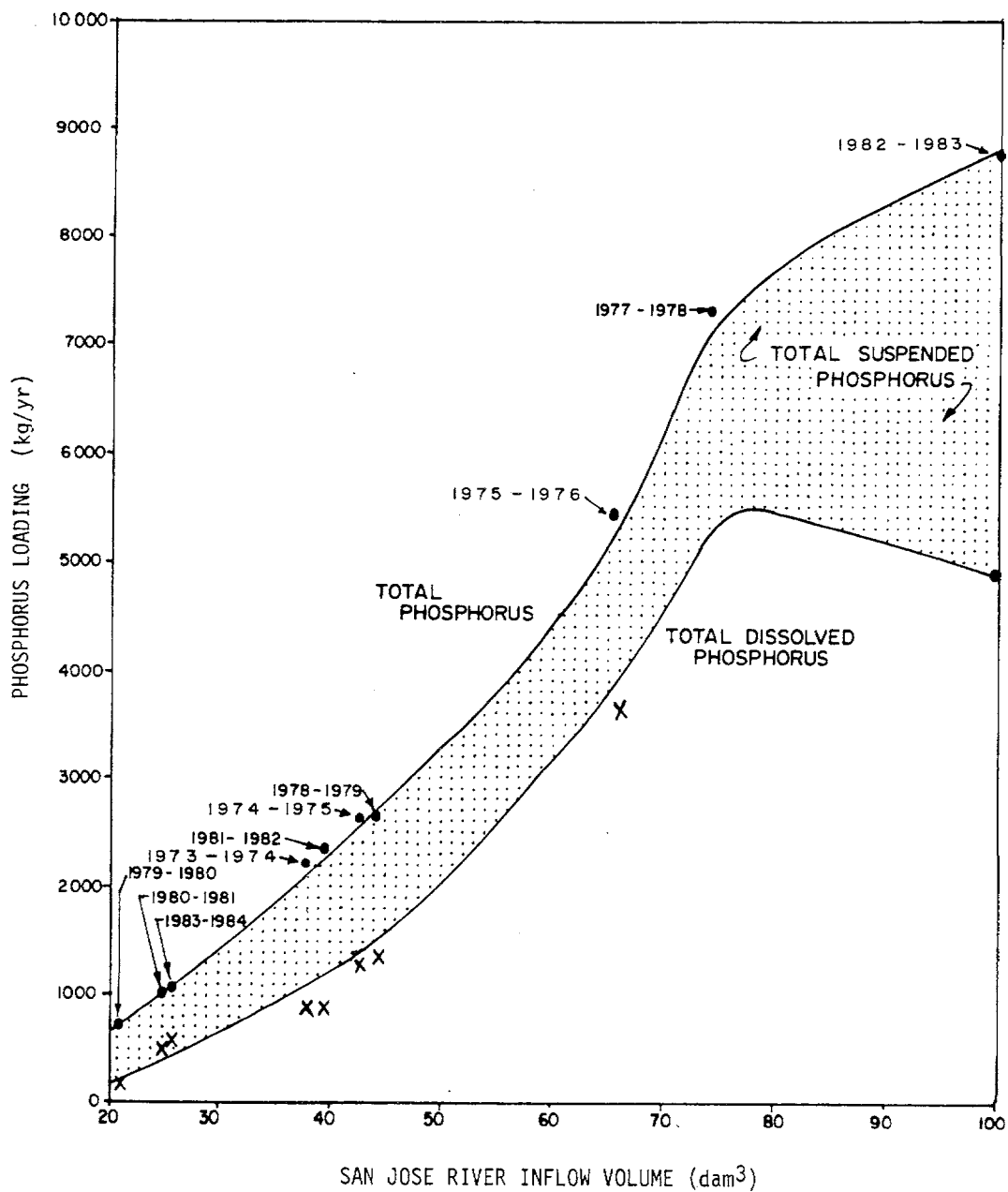


Figure 13. Annual phosphorus loading rates for the San Jose River as a function of inflow volume.

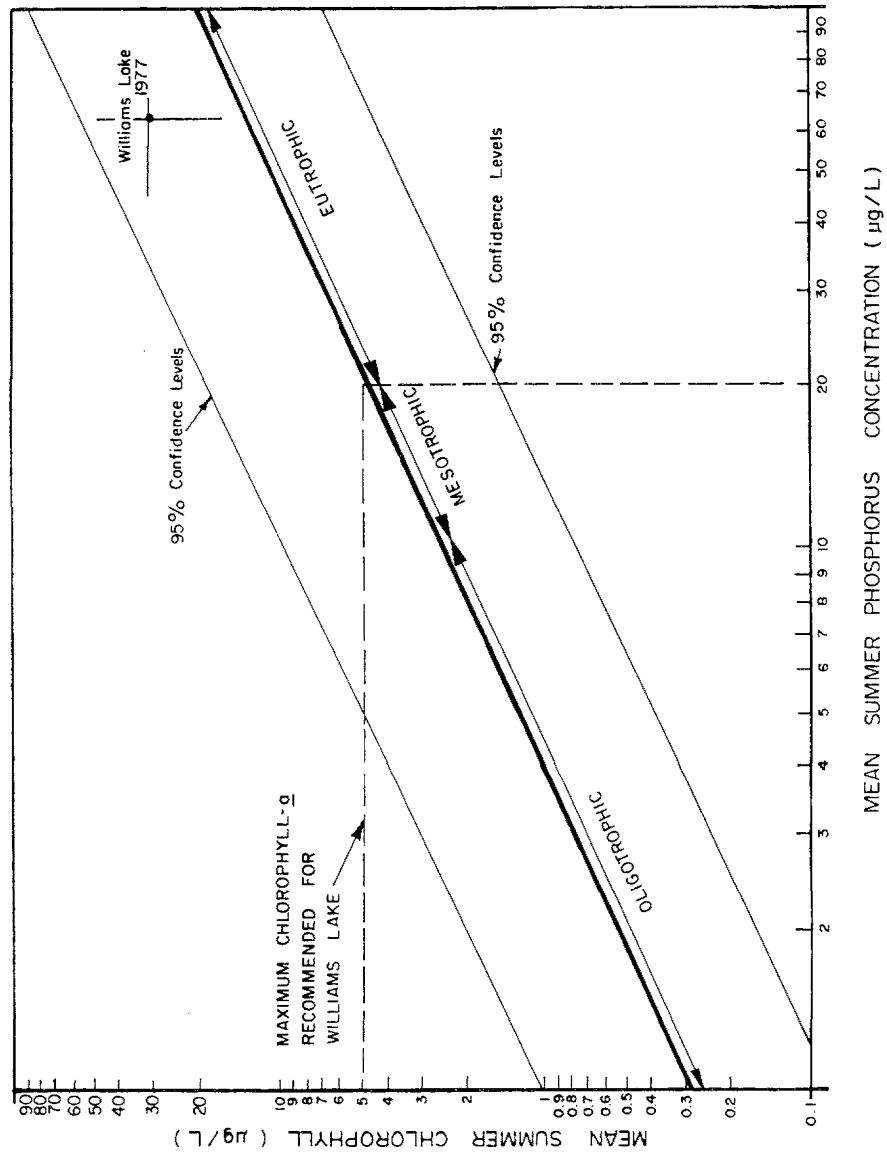


Figure 14. Mean summer chlorophyll-a concentration as a function of mean summer phosphorus concentration.

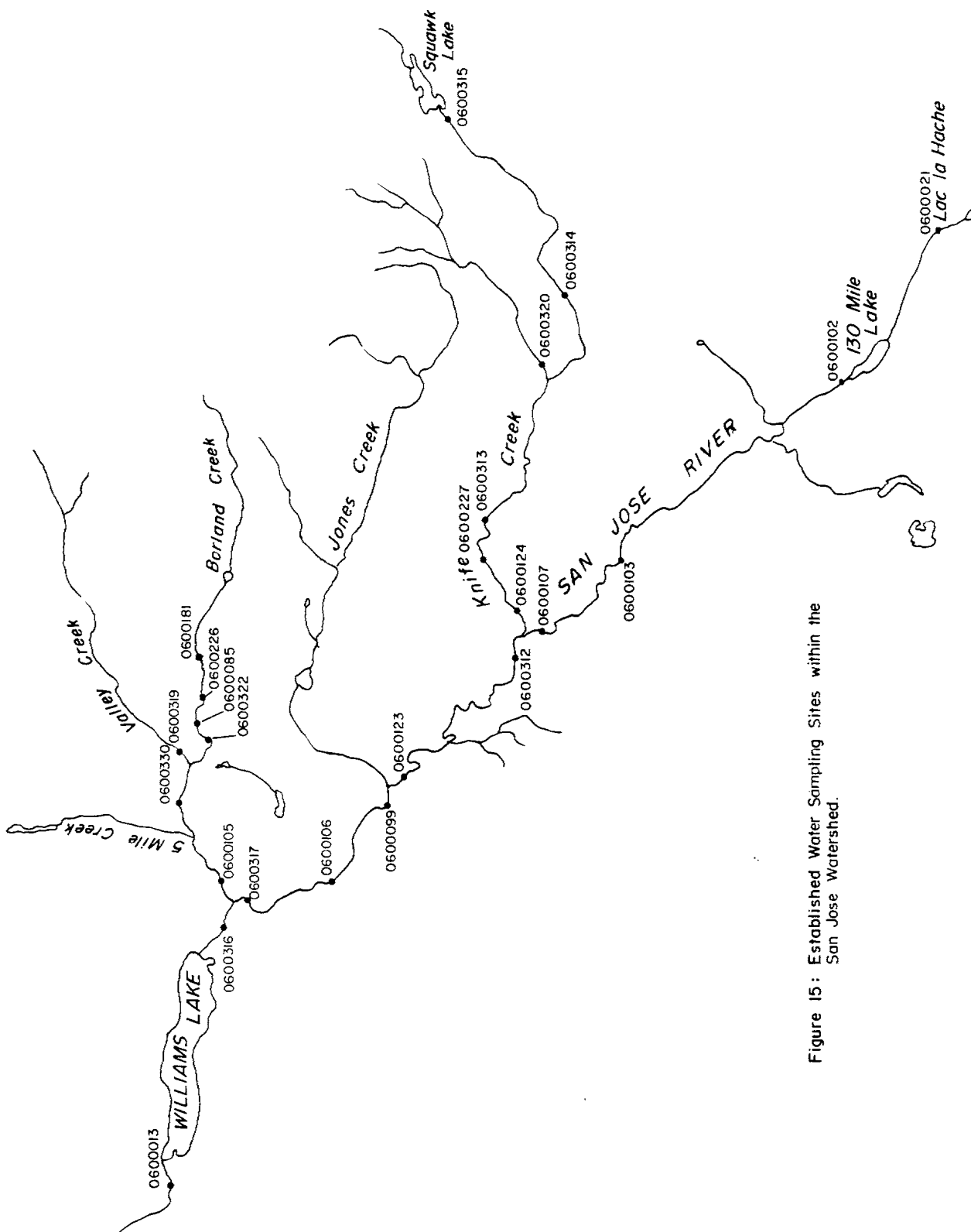
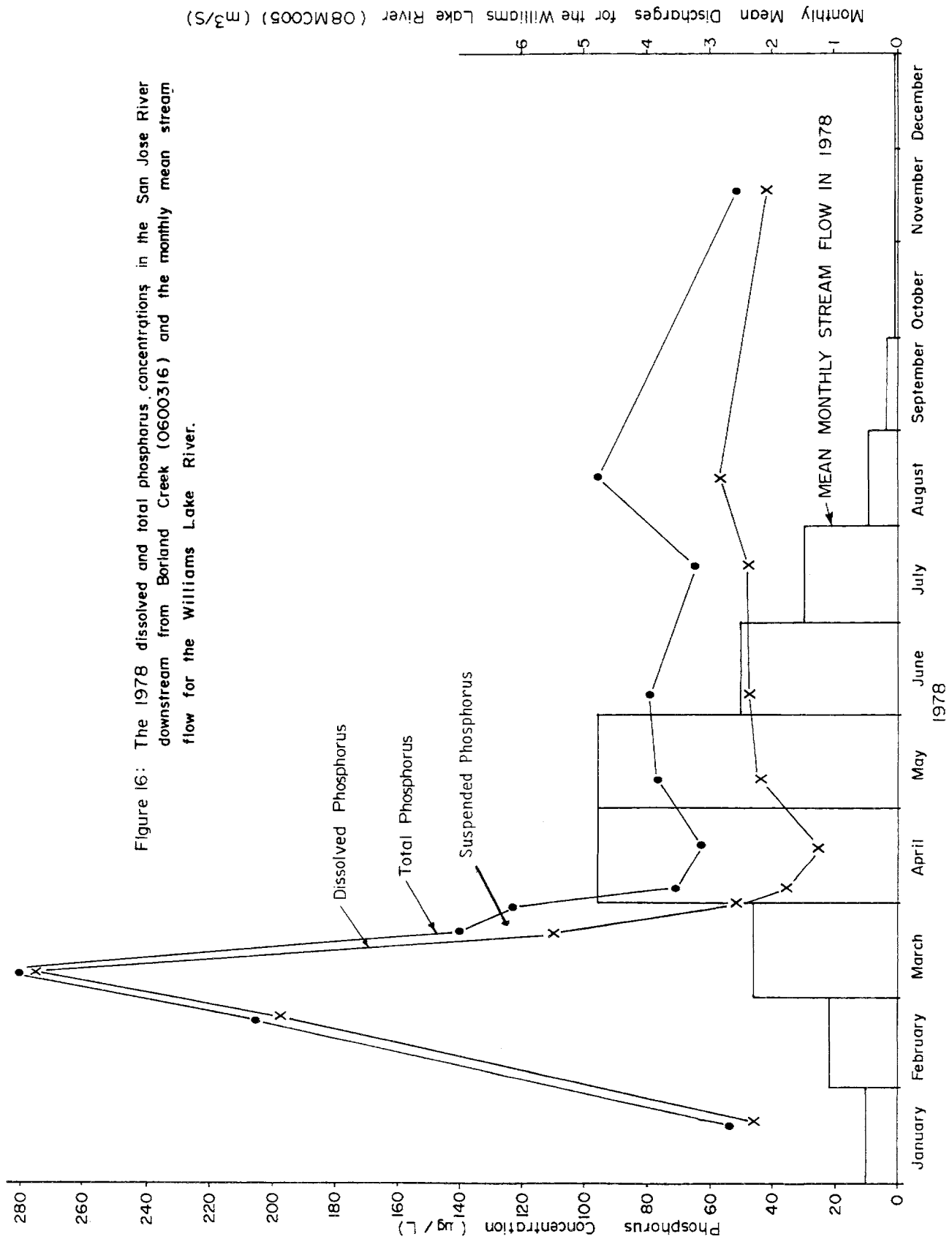


Figure 15: Established Water Sampling Sites within the San Jose Watershed.

Figure 16: The 1978 dissolved and total phosphorus concentrations in the San Jose River downstream from Borland Creek (0600316) and the monthly mean stream flow for the Williams Lake River.



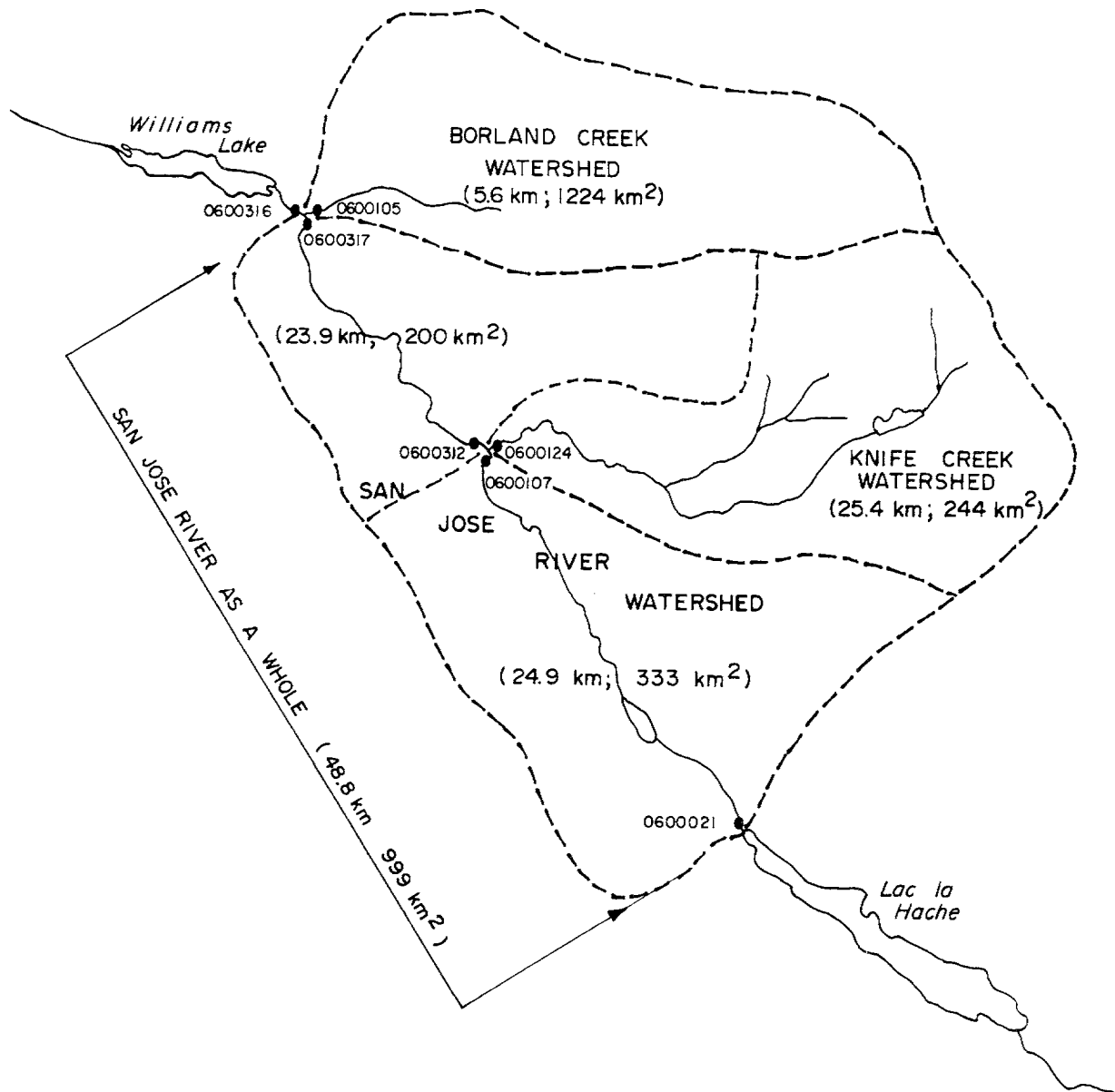


Figure 17. Illustration of Sub-watersheds within the San Jose River watershed.

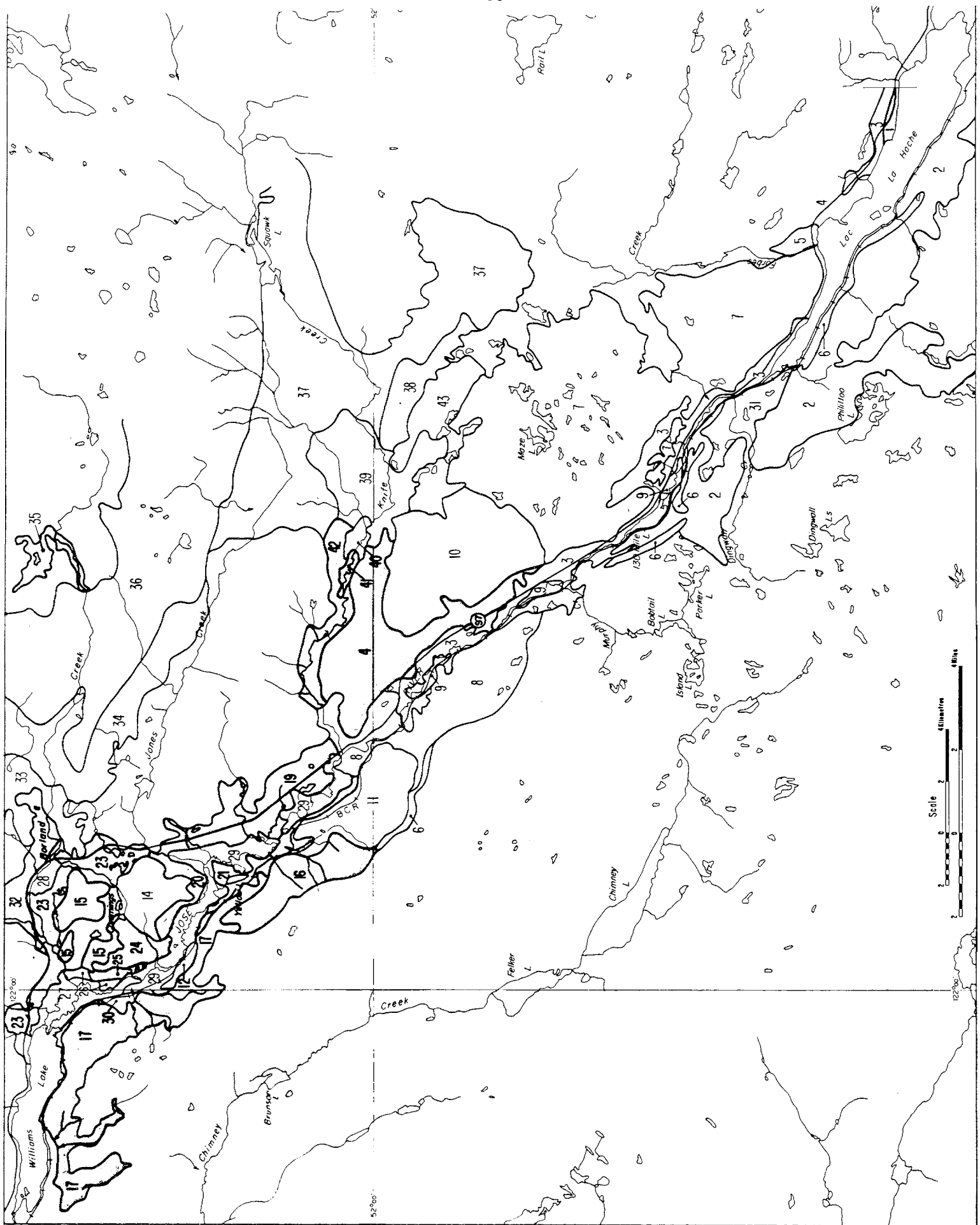


Figure 18. Erosion and runoff potential for the soil and landforms of the San Jose River Watershed. (shaded areas have normal to high runoff or erosion potential, see Table 16).

TABLE 1

LAKE OUTFLOW, WATER RETENTION TIME, AND
FLUSHING RATE FOR WILLIAMS LAKE FROM 1973-1984

HYDROLOGICAL PERIOD*	LAKE OUTFLOW (dam ³)	WATER RETENTION TIME (yr)	FLUSHING RATE (yr ⁻¹)
1973 - 1974	38 131	2.31	0.43
1974 - 1975	42 835	2.06	0.49
1975 - 1976	65 194	1.35	0.74
1976 - 1977	120 226	0.73	1.37
1977 - 1978	74 219	1.19	0.84
1978 - 1979	44 162	2.00	0.50
1979 - 1980	21 038	4.20	0.24
1980 - 1981	25 035	3.52	0.28
1981 - 1982	39 364	2.24	0.45
1982 - 1983	101 084	0.87	1.15
1983 - 1984	<u>25 862</u>	3.41	0.29
	54 300		

* Hydrological period is defined as the 12 months prior to spring overturn.

Based on a lake volume of 88 000 dam³, the average flushing rate and water retention time is 0.62 yr⁻¹ and 1.62 years respectively for the above period of record.

TABLE 2
WATER LICENCES FOR WILLIAMS LAKE

Priority Date	Licence Number	Point of Diversion	Licence Volume	Use*	Licence Description	Licencee
1947.11.05	C 18397	A	454 m ³ /d	WWK	WWK undertaking of licence	City of Williams Lake
1958.11.23	C 24842	B	1135 m ³ /d	WWK	Land in Village of Williams Lake & part of L 71, 7045, 8815, 8816 & 8817 under C of PC & N	City of Williams Lake
1962.03.05	F 19681	XX	2.3 m ³ /d	DOM	L 4 of L 8833 Cariboo Dist Plan 6365	Private
1962.03.05	C 28066	XX	2.3 m ³ /d	DOM	L 3 of L 8833 Cariboo Dist Plan 11933	Private
1963.09.17	F 38858	UU	2.3 m ³ /d	DOM	Part of L 15 of L 8829 Cariboo Dist Plan 7762 lying S of the Rd	Private
1963.10.21	C 31029	A3	68 m ³ /d	WWK	Land within the Bdy	Lexington Improvement Dist.
1964.03.03	C 30444	A3	2.3 m ³ /d	DOM	Part of L 2 of L 8829 Cariboo Dist Plan 7762 lying S of Rd	Private
1964.03.12	F 38678	B3	2.3 m ³ /d	DOM	L 3 of L 8829 Cariboo Dist Plan 7762 lying S of Rd	Private
1964.03.12	F 39056	C3	2.3 m ³ /d	IND	L 3 & 4 of L 8829 Cariboo Dist Plan 7762 lying S of Rd	Private
1964.04.23	C 29806	D3	2.3 m ³ /d	DOM	L 1 of L 8819 Cariboo Dist Plan 6641	Private
1964.04.24	C 31155	AB	15.890 m ³ /d	WWK	WWK undertaking of licence	City of Williams Lake
1964.04.27	F 38859	E3	2.3 m ³ /d	DOM	Blk A of L 8833 Cariboo Dist	Private

* WWK = Waterworks
DOM = Domestic
IND = Industrial
STD = Storage

TABLE 2 (Continued)
WATER LICENCES FOR WILLIAMS LAKE

Priority	Licence Number	Point of Diversion*	Licence Volume	Use	Licence Description	Licencee
1965.02.12	C 30855	ZB	7.3 m ³ /d	IND	Part of L 72 Cariboo held under PGE Lease No 923	Weldwood of Can Ltd-Merrill & Wagner Div
1966.06.21	F 19744	XX	2.3 m ³ /d	DOM	L 2 of L 8833 Cariboo Dist held under PGE Lease No 923	Private
1966.03.31	CX 31154	C	2 087 dam ³	STO	Stor for CL 18397, 24842 & 31155	City of Williams Lake
1966.07.21	C 32060	XX	2.3 m ³ /d	DOM	L 1 of L 8833 Cariboo Dist Plan 11933	Private
1968.01.30	C 33536	E4	6.8 m ³ /d	DOM	L 5 plan 7633 & L 9 Plan 10885 of L 8834 Cariboo Dist	Private
1971.04.19	C 38128	XX	2.3 m ³ /d	DOM	L 4 & 5 of L 8833 Cariboo Dist Plan 11933	Private
1971.06.14	C 38130	U4	2.3 m ³ /d	DOM	L 12 of L 8834 Cariboo Dist plan 10885	Private
1971.06.14	C 38131	U4	2.3 m ³ /d	DOM	L 2 of L 8834 Cariboo Dist Plan 7633	Private
1971.06.14	C 38132	U4	2.3 m ³ /d	DOM	L 11 of L 8834 Cariboo Dist Plan 10885	Private
1971.06.14	C 38133	U4	2.3 m ³ /d	DOM	L 5 of L 7734 Cariboo Dist Plan 12401	Private
1971.06.14	C 38134	U4	2.3 m ³ /d	DOM	L 3 of L 8834 Cariboo Dist Plan 7633	Private
1971.06.14	C 38371	U4	2.3 m ³ /d	DOM	L 10 8834 Cariboo Dist Plan 12401	Private
1971.08.23	C 40063	V4	15.9 m ³ /d	IND	L 1 of Plan 7633 & L 14 Plan 10885 Cariboo Dist	Private

TABLE 2 (Continued)
WATER LICENCES FOR WILLIAMS LAKE

Priority	Licence Number	Point of Diversion*	Licence Volume	Use	Licence Description	Licencee
1974.05.27	C 53651	P3	190 m ³ /d	WWK	Undertaking of the licensee as set out in PC & N 190 Cariboo Dist	Fetters Developments Ltd
1975.01.31	C 51357	Q3	2.3 m ³ /d	DOM	L 2 of L 8819 Cariboo Dist Plan 21526	Private
1984.08.23		U3	4.6 m ³ /d	DOM	L 1 of L 8833 Cariboo Dist Plan	Private
1985.06.04		V3	2.3 m ³ /d	DOM	L 6 & 7 of L 8833 Cariboo Dist Plan 11933	Private
1985.07.09		U4	2.3 m ³ /d	DOM	L 7 of L 8834 Cariboo Dist Plan 12401	Private

TABLE 3
MORPHOLOGICAL PARAMETERS OF WILLIAMS LAKE

Altitude (m)	:	565
Lake Area (ha)	:	720
Watershed Size (km ²)	:	2 240
Mean Depth (m)	:	12
Maximum Depth (m)	:	24
Latitude	:	52 07 21 N
Longitude	:	120 08 31 W

TABLE 4

WATER CHEMISTRY SUMMARY FOR WILLIAMS LAKE
1972 TO 1984 (SITE 0603019)

Parameter	Average	Max.	Min.	Standard Deviation	N
Alkalinity, total	303	311	298	6.1	6
Bicarbonate Carbon, dissolved*	353	447	290	50	7
Calcium, dissolved	29.8	31.7	27.0	1.9	5
Chloride, dissolved	2.0	2.3	1.6	0.4	3
Hardness, dissolved	252	260	240	11.0	3
Iron, total	**	0.2	<0.1	-	3
Lead, total	**	0.001	<0.003	-	3
Manganese, total	0.025	0.03	0.02	-	3
Magnesium, dissolved	42.2	45	37	3.8	4
pH	8.4	9.0	7.7	0.3	52
Potassium, dissolved	4.5	4.7	4.4	0.2	3
Secchi	1.9	4.0	0.5	1.0	25
Sodium, dissolved	33.5				1
Specific conductance	562	610	497	28.3	52
Sulphate, dissolved	13	13	13		2
Total Dissolved Solids	367	374	356	8.1	4
Turbidity	2.4	6.3	1.1	1.9	6

(all results in mg/L except pH, Specific conductance: $\mu\text{S}/\text{cm}$, Secchi:m, and turbidity:NTU).

* calculated from inorganic carbon results.

** most results below detection levels, average not calculated.

TABLE 5

IONIC ASSOCIATIONS AND IONIC CONTRIBUTIONS TO ALKALINITY

Association	Relative Contribution to Total Dissolved Residues	Contribution to Alkalinity
$\text{Mg}(\text{HCO}_3)_2$	52%	145 mg/L
$\text{Ca}(\text{HCO}_3)_2$	23%	75 mg/L
NaHCO_3	13%	72 mg/L
Na_2SO_4	7%	N/A
NaCl	3%	N/A
Others	2%	N/A

TABLE 6

WATER CHEMISTRY FOR THE CITY OF WILLIAMS LAKE'S DRINKING WATER SUPPLY
(OBSERVATION WELL WR 85A-69, SITE 1401073)

Parameter	Average	Max.	Min.	S.D.	N
Alkalinity, phenolphthalein (mg/L)	32.9	39.5	29.8	4.5	4
, total (mg/L)	220	233	187	21.8	4
Aluminum, dissolved (mg/L)	*	0.02	<0.02	-	3
, total (mg/L)	0.04	0.06	0.02	0.02	3
Arsenic, dissolved (mg/L)	<0.250	-	-	-	4
, total (mg/L)	<0.250	-	-	-	4
Barium, dissolved (mg/L)	0.03	0.06	<0.1	0.03	3
Cadmium, dissolved (mg/L)	<0.01	-	-	-	3
, total (mg/L)	<0.01	-	-	-	3
Calcium, dissolved (mg/L)	2.0	2.5	1.6	0.4	4
Chloride, dissolved (mg/L)	3.4	3.6	3.2	0.2	4
Chromium, dissolved (mg/L)	<0.01	-	-	-	3
, total (mg/L)	0.01	0.01	0.01	-	3
Cobalt, dissolved (mg/L)	<0.100	-	-	-	3
, total (mg/L)	<0.100	-	-	-	3
Copper, dissolved (mg/L)	*	0.01	<0.01	-	3
, total (mg/L)	0.13	0.26	0.02	0.12	3
Iron, dissolved (mg/L)	0.10	0.27	<0.01	0.14	3
, total (mg/L)	33.5	38.1	27.3	2.3	3
Lead, dissolved (mg/L)	<0.1	-	-	-	3
, total (mg/L)	*	0.250	<0.1	-	3
Magnesium, dissolved (mg/L)	27.6	30.2	24.6	2.4	4
, total (mg/L)	29.9	32.7	28.4	2.4	3
Manganese, dissolved (mg/L)	*	0.01	<0.01	-	3
, total (mg/L)	0.10	0.20	0.05	0.07	4
Molybdenum, dissolved (mg/L)	*	0.02	<0.01	-	3
Molybdenum, total (mg/L)	0.01	0.02	<0.01	0.01	3
Nickel, dissolved (mg/L)	<0.05	-	-	-	3
, total (mg/L)	<0.05	-	-	-	3
pH, laboratory	9.4	9.7	9.3	0.2	4
Potassium, dissolved (mg/L)	6.8	6.9	6.7	0.1	4
Residue, total (mg/L)	260	276	246	12.5	4
Sodium, dissolved (mg/L)	58.9	61.5	57.6	1.8	4
Specific Conductivity (μ S/cm)	469	483	440	20.3	4
Vanadium, dissolved (mg/L)	<0.010	-	-	-	3
, total (mg/L)	<0.010	-	-	-	3
Zinc, dissolved (mg/L)	<0.01	-	-	-	3
, total (mg/L)	0.08	0.15	0.03	0.06	3

*Average not calculated: only one result above detection levels.

TABLE 6 (CONTINUED)

WATER CHEMISTRY FOR THE CITY OF WILLIAMS LAKE'S DRINKING WATER SUPPLY

Wells 1, 2, 3 and 4 on Scout Island - March 31, 1987

Parameter	Well #1	Well #2	Well #3	Well #4
pH (units)	7.94	7.89	7.78	7.88
Dissolved Solids	610.	472.	614.	630.
Hardness (as CaCO ₃)	473.	349.	500.	508.
Manganese	0.14	0.10	0.24	0.09
Total Iron	0.20	0.18	0.05	0.60
Sulfate, (as SO ₄)	162.	64.3	112.	114.
Calcium (as Ca)	46.7	38.8	52.0	46.5
Nitrite	<0.002	<0.002	<0.002	<0.002
Fluoride	0.11	0.15	0.12	0.15

NOTE: All results expressed in mg per litre unless otherwise specified.
 < = less than

Analyzed by Eco-Tech Laboratories Ltd.,
 10041 E. Trans Canada Hwy., R.R. #2
 Kamloops, B. C.
 V2C 2J3

TABLE 7

SPRING OVERTURN PHOSPHORUS CONCENTRATIONS FOR WILLIAMS LAKE
FROM 1974 TO 1985

YEAR	SPRING OVERTURN PHOSPHORUS CONCENTRATION ($\mu\text{g/L}$)	COMMENT
1974	70	1 month after turnover: surface and bottom average
1975	54	6 weeks after turnover: surface and bottom average
1976	58	2 weeks after turnover: surface and bottom average
1977	64	good composite sample
1978	77	good composite sample
1979	80	good composite sample
1980	70	good composite sample
1981	50	good composite sample
1982	70	good composite sample
1983	85	good composite sample
1984	83	good composite sample
1985	74	good composite sample
1986	80	good composite sample
1987	75	good discrete sample at 0 & 5 m

TABLE 8

**ANNUAL PHOSPHORUS LOADING RATES TO WILLIAMS LAKE VIA
THE SAN JOSE RIVER**

HYDROLOGICAL PERIOD*	INFLOW VOLUME (dam ³)	TOTAL PHOSPHORUS LOADING (kg)	DISSOLVED PHOSPHORUS LOADING (kg)
1973 - 1974	38 131	3 200	1 900
1974 - 1975	42 835	3 600	2 350
1975 - 1976	65 194	6 500	4 600
1976 - 1977	120 226	10 200	6 500
1977 - 1978	74 219	8 300	6 510
1978 - 1979	44 162	3 640	3 220
1979 - 1980	21 038	1 690	1 180
1980 - 1981	25 035	2 005	1 585
1981 - 1982	39 364	3 360	1 895
1982 - 1983	101 084	9 720	5 905
1983 - 1984	25 862	2 010	1 590

* Hydrological period is defined as the 12 months prior to spring overturn.

TABLE 9

**TYPICAL RANGES OF PHYTOPLANKTON AND NUTRIENTS FOR
DIFFERENT TROPHIC LEVELS**

Trophic Status	Chlorophyll <u>a</u> Growing Season Mean (µg/L)	Total P at Spring Overturn (µg/L)	Secchi Disc (m)	Hypolimnetic Oxygen Depletion Rates (mg O ₂ /cm ² /day)
Oligotrophic	<2	<10	>6	<0.025
Mesotrophic	2-5	10-20	2-6	-
Eutrophic	>5	>20	<2	>0.055
Williams Lake	30	50-80	1.9	0.082

From Nordin (1985).

TABLE 10

AQUATIC MACROPHYTES COLLECTED FROM WILLIAMS LAKE
(From Warrington, 1980)

Potamogeton vaginatus	Potamogeton pectinatus	Potamogeton zosteriformis
Potamogeton praelongus	Potamogeton Richardsonii	Potamogeton natans
Potamogeton nodosus	Potamogeton friesii	Potamogeton berchtoldii
Myriophyllum exalbescens	Sparganium eurycarpum	Sparganium emersum
Lemna minor	Lemna trisulca	Spirodela polyrhiza
Utricularia vulgaris	Ranunculus gmelinii	Ranunculus aquatilis
Alisma plantago-aquatica	Najas flexilis	Ceratophyllum demersum
Hippuris vulgaris	Polygonum amphibium	Equisetum fluviatile
Zannichellia palustris	Sium suave	Veronica americana
Mimulus guttatus	Typha latifolia	Eleocharis palustris
Scirpus lacustris	Mimulus	Chara
	Sagittaria	

TABLE 11

SEDIMENT ANALYSIS FOR WILLIAMS LAKE AND OTHER LAKES
IN THE CARIBOO-ASPEN-LODGEPOLE-PINE BIOGEOCLIMATIC ZONE

	July 23, 1985 Williams Lake Surface Sediment	Lakes in Cariboo Aspen-Lodgepole Pine Zone (n=34) Surface Sediment	Williams Lake 2 m CORE (n=34)
Aluminum	6 880	9 900	9 080
Calcium	90 500	46 800	76 610
Carbon, Total	93 000	181 100	74 550
Iron	18 000	22 100	24 210
Loss on Ignition (%)	20.5	38.4	14.7
Magnesium	13 000	17 000	13 000
Manganese	1 130	713	965
Nitrogen, Kjeldahl	7 750	15 000	4 545
Phosphorus	1 1240	41 340	919
Zinc	59	79	59

All results except Loss on Ignition are µg/g dry.

TABLE 12

PHOSPHORUS LOADING RATES FOR THE SAN JOSE RIVER
AND ITS TRIBUTARIES

	SAN JOSE RIVER	SAN JOSE RIVER BETWEEN LAC LA HACHE AND KNIFE CREEK	SAN JOSE RIVER BETWEEN KNIFE AND BORLAND CREEKS	KNIFE CREEK	BORLAND CREEK
	n = 38	n = 38	n = 34	n = 35	n = 35
TDP ($\mu\text{g}/\text{km}$)	1.19 \pm 1.19	0.82 \pm 1.28	1.14 \pm 1.59	2.26 \pm 34.48	7.70 \pm 20.98
TDP ($\mu\text{g}/\text{km}^2$)	0.059 \pm 0.058	0.061 \pm 0.096	0.131 \pm 0.183	0.357 \pm 0.292	0.627 \pm 0.655
TP ($\mu\text{g}/\text{km}$)	1.62 \pm 1.30	0.96 \pm 1.40	1.56 \pm 1.55	2.91 \pm 2.16	6.50 \pm 23.5
TP ($\mu\text{g}/\text{km}^2$)	0.079 \pm 0.063	0.072 \pm 0.107	0.180 \pm 0.188	0.459 \pm 0.347	0.717 \pm 0.703

TABLE 13
SURFACE RUNOFF PRODUCING POTENTIAL (COOK'S METHOD)

Watershed Character- istics	RUNOFF POTENTIAL			
	(100) Extreme	(75) High	(50) Normal	(25) Low
Relief	(40) Steep, rugged terrain, with average slopes generally above 30%	(30) Hilly, with average slopes of 10 to 30%	(20) Rolling, with average slopes of 5 to 10%	(10) Relatively flat land, with average slopes of 0 to 5%
Soil infiltration	(20) No effective soil cover, either rock or thin soil mantle of negligible infil- tration capacity	(15) Slow to take up water; clay or other soil of low infiltration capacity as gumbo	(10) Normal: deep loam with infiltration about equal to that of typical prairie soil	(5) High: deep sand or other soil that takes up water readily and rapidly
Vegetal cover	(2) No effective plant cover; bare or very sparse cover	(15) Poor to fair; clean-cultivated crops or poor natural cover; less than 10% of drain- age area under good cover	(10) Fair to good; about 50% of drainage area in good grassland, woodland, or equivalent cover; not more than 50% of area in clean- cultivated crops	(5) Good to excellent; about 90% of drainage area in good grassland, woodland, or equivalent cover
Surface storage	(20) Negligible; surface depressions few and shallow; drainageways steep and small; no ponds or marshes	(15) Low; well-defined system of small drainageways; no ponds or marshes	(10) Normal; consider- able surface depression stor- age; lakes, ponds and marshes less than 2% of drain- age area	(5) High; surface- depression storage high; drainage system not sharply defined

TABLE 14

SURFACE EROSION POTENTIAL AS A FUNCTION OF
SOIL TEXTURE AND SOIL SLOPE

Soil Texture	Slope		
	<10 %	10% to 20%	>20%
s, ls	Low	Low	Moderate
cl, sl l, sicl	Moderate	Moderate	High
si, sil	High	High	High

P = sand or sandy

ls = loamy sand

sk = sandy loam

l = loam

si = silt

sil = silt loam

sicl = silty clay loam

cl = clay loam

TABLE 15
SOIL DESCRIPTIONS FOR THE SOIL AND LANDFORMS
WITHIN THE SAN JOSE RIVER WATERSHED

Symbol(s) ⁺	Name	Parent Material* Texture	Most Common Soil	Drainage**
Bf ₂	Buffalo	sil (Lacustrine)	Orthic Dark Gray	w
Cx ₁ , Cx ₁	Canim	Cobbly sl (Glacial Till)	Orthic Dark Brown	w
Cy ₁ , Cy ₁	Chimney	gl (Morainal blanket)	Orthic Dark Gray	w
El ₁	Elliot	sicl (Fluvial)	Humic Gleysol & Cumilic Regrosol	i
Hs ₁ , Hs ₁	Hawks	gs (Fluvioglacial)	Orthic Entric Brunisol	r
Hl ₁ , Hl ₂	Helena	gsl-1 (Morainal blanket)	Orthic Gray Luvisol & Gleyed Gray Luvisol	w to i
Mk	McGuckin	loamy lacustrine and fluvial	Orthic Black	w
Su	Sugarcane	loam fluvial	Gleyed Regosol	i to p
Ta ₁	Tatton	strong l - sl (Colluvial)	Orthic Gray Luvisol	mw
Tri ₁ , Tr ₃	Trurans	gsl - ls (Glacial outwash)	Dystic Brumisol	r
Tu ₁	Tubbs	gsl - ls (Morainal blanket)	Eutric Brumisol	w
Te ₁ , Te ₃	Tyee	gl - cl (Morainal blanket)	Orthic Gray Luvisol	mw
Wl ₁ , Wl ₂ ,	Williams	gl - cl	Orthic Gray Luvisol	mw
WL, Wl ₁ ,	Lake	(Morainal blanket)		
RI ₂	Rail	Organic	Typic Mesisol	vp

* g=gravel or gravelly, s=sand or sandy, l=loam or loamy, si=silt or silty, c=clay.

** i=imperfect, mw=moderately well, p=poor, r=rapid, w=well, vp=very poor.

+ as indicated on soils and landforms maps.

TABLE 16

SURFACE RUNOFF GENERATION POTENTIAL, EROSIONAL POTENTIAL, AND
PROXIMITY TO A WATERBODY FOR THE SOILS AND LANDFORMS
WITHIN THE SAN JOSE RIVER WATERSHED

LAND/SOIL UNIT		PROXIMITY TO WATERBODY	RUNOFF PRODUCING POTENTIAL	EROSION POTENTIAL
*No.	Symbol**			
1	Bf ₂ /c-d	adjacent	normal	high
2	Hl ₁ /c***	"	normal	moderate
3	Cx ₁ -Hl ₂ /c-e	"	normal	moderate
4	Te ₁ /c-e	"	normal-high+	moderate
5	Tr ₃ /d-e	"	normal	low
6	Ta ₁ /E-F	"	normal	moderate+
7	Tu ₁ -Tr ₄ /c-d	"	low to normal	low
8	Tu ₁ ⁶ -Tr ₁ ⁴ /gcMbF ^G =df	"	normal	low-moderate+
9	El ₁ /b+++	"	normal	moderate
10	Te ₃ /c=d	setback	normal	moderate
11	Wl ₁ /mMb=CD	"	normal	moderate
12	Wl ₁ /mMb=cb	"	normal	moderate
13	Wl ₁ /mMb=bC##	"	normal	moderate
14	Wl ₂ /mMb=CD	"	normal	moderate
15	Wl ₂ /mMb=DE	"	normal-high+	moderate
16	Wl ₂ ⁶ -Ta ⁴ /mMbC=cf	setback	normal-high+	moderate
17	Wl ₂ ⁷ -Hs ₁ ³ /msMbF ^G =df	setback	normal-high+	moderate
18	Wl ₂ ⁷ -Hs ₁ ³ /lsMbF ^G =df##	adjacent	normal-high+	moderate
19	Cy ₁ /mMb=Df	adjacent	normal	moderate-high+

TABLE 16 (Continued)

LAND/SOIL UNIT		PROXIMITY TO WATERBODY	RUNOFF PRODUCING POTENTIAL	EROSION POTENTIAL
*No.	Symbol**			
20	Cy ₁ /mMb=Ef	setback	normal	moderate-high ⁺
21	Cy ₁ -Wl ₁ /mMb=DF	adjacent	normal-high ⁺	moderate-high ⁺
22	Cy ₁ ⁸ -Wl ₁ ² /mMb=Dg##	setback	normal-high ⁺	moderate-high ⁺
23	Cy ₁ ⁸ -Wl ₁ ² /lMb=Dg	adjacent	normal-high ⁺	moderate-high ⁺
24	Cy ₁ ⁷ -Te ₃ ³ /mMb=CE	setback	normal-high ⁺	moderate
25	Cy ₁ ⁶ -Te ₄ ⁴ /mMb=Ef	setback	normal-high ⁺	moderate-high ⁺
26	Mk/mLt=bc	adjacent	normal	low-moderate
27	Su/mFp=AB ⁺⁺	setback	normal	low-moderate
28	Su/mFp=BA ⁺⁺		normal	low-moderate
29	Su ⁶ -Cy ₁ ⁴ /mFpMb=bE	adjacent	normal	moderate
30	CY ₁ /lMb=cd	setback	normal	moderate
31	Hl ₂ -XZ ₁ -Rl ₂ /c-d	setback	normal	moderate
32	Cy ₁ -Wl ₂ /mMb=cf	setback	normal-high	moderate-high ⁺
33	Wl ₁ ⁶ -Hs ₁ ⁴ /mCmbF ^G =CD	adjacent	normal	low-moderate
34	Te ₁ /mMb = CD	adjacent	normal	low-moderate
35	Sz/Ob =b	adjacent	low-high ⁺⁺⁺	low#
36	Te ₁ /mMb=bc	adjacent	normal	low-moderate
37	Hl ₁ ⁶ -St ₁ ⁴ /gcmMb=Cd	adjacent	normal	low-moderate

TABLE 16 (Continued)

LAND/SOIL UNIT		PROXIMITY TO WATERBODY	RUNOFF PRODUCING POTENTIAL	EROSION POTENTIAL
*No.	Symbol**			
38	Hl ₁ -Tu ₁ /c-e	adjacent	normal	low-moderate
39	Hl ₁ /gcMb=CD	adjacent	normal	low-moderate
40	Su/mFp=BC	adjacent	normal	low-moderate
41	Rl/Ob=b	adjacent	low-high+++	low#
42	Wl ₂ ⁶ -Cy ₁ ⁴ /mMb=df	adjacent	normal-high	low-high ⁺
43	Hl ₂ -Rl ₁	setback	normal	low-moderate

* as shown in the map attached with this report (Figure 18)

**as shown in soils and landforms maps

***some areas of wet imperfectly drained soils

+ steep topography

+++often flooded in spring

++ imperfectly to poorly drained soils

organic soils

not shown in the map; units 18 and 22 are essentially the same as units 17 and 23 respectively.

