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WATER STEWARDSHIP DIVISION
MINISTRY OF ENVIRONMENT

Water Quality Assessment and Objectives for Langford Lake

Technical Assessment

Prepared pursuant to Section 5(e) of the
Environmental Management Act, 2003

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SUMMARY

This document presents a summary of the ambient water quality of Langford Lake, British Columbia, and proposes water quality objectives designed to protect existing and future water uses. The water quality assessment for the lake and an evaluation of the watershed form the basis for the objectives.

Langford Lake is home to numerous permanent residences and also provides significant recreational opportunities (fishing, swimming and boating) and fish and wildlife habitat. There are four water withdrawal licences which are likely for non-consumptive purposes only. Non-point sources of waste from urban runoff, land development and on-site sewage systems are the only major inputs of pollutants to the lake.

The data collected over the past 20 years indicate the lake is eutrophic; however, the overall state of water quality does not appear to be deteriorating at the present time. An aerator was installed in 1984 to address internal nutrient loading issues. Although it is still providing an overall benefit to the lake by reducing internal nutrient loading, it has lost some of its effectiveness over time. Phosphorus and nitrogen concentrations accumulate in the water column throughout the summer and result in conditions that favour blue-green algal blooms in the fall. With the 2006 extension of centralized sewer service in the Langford Lake watershed, there is an opportunity for residents to reduce their impact on the lake's water quality by switching from on-site sewage (i.e., septic) disposal to sewer. Continuing growth and development within the watershed may present challenges in protecting water quality in the future.

Ambient water quality objectives are proposed for temperature, dissolved oxygen, water clarity, total phosphorus, total nitrogen, N:P ratio, chlorophyll *a*, enterococci and fecal coliforms.

Future monitoring recommendations include continuation of annual spring overturn sampling with monitoring of microbiological indicators at bathing beaches to be continued by the Vancouver Island Health Authority.

A glossary of technical terms is included as an appendix to this report.

WATER QUALITY OBJECTIVES FOR LANGFORD LAKE

Site	1100944
Designated Water Uses	Recreation (primary contact), aquatic life
Characteristics	
Temperature	≤ 15° C (summer maximum hypolimnetic temperature)
Dissolved Oxygen	≥ 5 mg•L^{-1} (at any depth throughout the year)
Secchi Depth	≥ 4 m (annual mean)
Total Phosphorus (short-term : 5 – 20 years)	≤ 20 µg•L^{-1} at spring overturn
Total Phosphorus (long-term: > 20 years)	≤ 10 µg•L^{-1} at spring overturn
Total Nitrogen	≤ 500 µg•L^{-1} at spring overturn
N:P Ratio	≥ 20:1
Phytoplankton community	> 50% non-cyannophyte species (cells•mL^{-1})
Chlorophyll *a*	≥ 1.5 µg•L^{-1} ≤ 2.5 µg•L^{-1}
Fecal coliforms	≤ 200 •100 mL^{-1} (geometric mean)

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1.0 INTRODUCTION

1.1 Background

Water quality and quantity have been long-standing issues for Langford Lake. At one time, the lake provided a domestic water source for some lakeshore residents and taste and odour concerns were some of the early signs of declining water quality. Increasing aquatic weed growth, which limited the lake's recreational appeal and blocked the outlet, was another sign that the lake was changing.

Langford Lake originally flowed south to Glen Lake then on to Esquimalt Lagoon. The E&N Railway built an embankment along the western shore which cut off the outflow to the south and in an effort to alleviate some of the flooding risk, a drainage ditch was eventually constructed at the north end. Later, a culvert was built under the railway to allow drainage from the agricultural lands, locally known as Hull's Field, to the south. When these fields became flooded in the fall and winter, water was pumped to Langford Lake in order to minimize the damage.

Although cultural eutrophication was a growing concern in the early 1960's, the key role of phosphorus in limiting primary production in freshwater ecosystems did not become clear until the late 1960's and early 1970's. This resulted in some inaccurate conclusions which may have contributed to the water quality concerns that followed. Around 1960, the Division of Public Health Engineering concluded there was no difference between the water being pumped from Hull's Field and the lake water and therefore, pumping Hull's Field was not contributing to the eutrophication problem. A 1963 Water Rights Branch report (Priestman 1963) identified four possible sources of nutrients: septic systems, detergents, mixing of the water column at overturn and pumping of fertilizer-enriched water from Hull's Field. The report concluded the main inputs were from septic systems and detergents. Also in 1963, the Pollution Control Board passed a resolution to the effect that "algae pollution in Langford Lake is a natural phenomenon and thus out of the

hands of the Control Board". Eventually the contributing role of nutrients to Langford Lake's water quality problems would be more fully understood.

The 1970's saw a significant increase in aquatic weed growth in Langford Lake. This raised concerns regarding the impact on recreational use of the lake and blocking of the outflow, which resulted in lower flushing rates and an increased likelihood of residential flooding problems. Large mats of *Ceratophyllum demersum* and *Elodea canadensis* were covering most areas of the lake that were less than 5 m in depth. The Langford Lake Improvement District submitted an application for a dredging program to remove the macrophytes (aquatic plants) and excess sediments in the lake substrate. In August 1976, a permit was issued to allow testing of a dredge made by local inventor Frank Hillis at the south end of the lake near the boat launch. Although the pilot project was carried out with some degree of success, there is no indication that the full dredging proposal was ever approved.

During the 1980's, some practical efforts were made to address the nutrient concerns in Langford Lake. In 1981, a water quality assessment (McKean & Munteanu 1981) was completed in response to the Langford Lake Improvement District's need for a more detailed analysis of the situation. This report concluded that a high concentration of available phosphorus was the main factor promoting the growth of aquatic plants in Langford Lake and the main sources of phosphorus to the lake were, in decreasing order of importance, the surface flow from Hull's Field, internal loading, septic tanks and aerial loading. This contradicts the conclusions of investigations in the 1960's that the discharge from Hull's Field was having no impact on water quality. In addition to this work, baseline weed surveys were conducted to track the growth and range of the macrophytes. To help address the weed problem, a weed harvester has operated on the lake, on a volunteer basis, since the 1980's.

In an effort to eliminate internal phosphorus loading in Langford Lake, a destratification aerator was installed in 1984 with the following objectives:

- maintain dissolved oxygen levels at greater than 6 $mg \cdot L^{-1}$ throughout the lake;

- eliminate the reducing conditions in the hypolimnion that cause internal phosphorus loading;
- eliminate blue-green algae blooms through lower phosphorus concentrations and pH levels; and
- improve water clarity through reduced chlorophyll *a* concentrations and elimination of blue-green algae.

The aerator has operated every year since and typically runs from April through October.

The 1990's saw a continued increase in the local stewardship of Langford Lake with the establishment of the Langford Lake Area Protection Society in 1995 after the Langford Lake Improvement District was dissolved in 1993. This group is dedicated to protecting Langford Lake and the semi-rural ambiance of the community of Langford. They are currently assisting with water quality monitoring in Langford Lake under the BC Lake Stewardship and Monitoring Program. Most recently, the City of Langford extended centralized sewer service around Langford Lake providing residents the opportunity to switch from on-site sewage disposal. Connection to the centralized sewer system is not mandatory for existing homes but is for all new construction.

1.2 Water Quality Objectives

Water quality objectives are prepared for specific bodies of fresh, estuarine and coastal marine surface waters of British Columbia as part of the Ministry of Environment's mandate to manage water quality. Objectives are prepared only for those waterbodies and water quality characteristics that may be affected by human activity now or in the future.

Water quality objectives are based on scientific guidelines that are safe limits of the physical, chemical or biological characteristics of water, biota (plant and animal life) or sediment which protect water use. Objectives are established in British Columbia for waterbodies on a site-specific basis. They are derived from the guidelines by considering

local water quality, water uses, water movement, waste discharges and socio-economic factors.

Water quality objectives are set to protect the most sensitive designated water use at a specific location. A designated water use is one that is protected in a given location and is one of the following:

- Raw drinking water, public water supply and food processing;
- Aquatic life and wildlife;
- Recreation and aesthetics;
- Agriculture (livestock watering and irrigation)
- Industrial water supplies.

By protecting the most sensitive water use, all designated uses for a given waterbody are also protected.

Water quality objectives have no legal standing at this time and are not directly enforced. However, they do provide policy direction for resource managers for the protection of water uses in specific waterbodies. Objectives guide the evaluation of water quality, the issuing of permits, licences and orders, and the management of fisheries and the province's land base. They also provide a reference against which the state of water quality in a particular water body can be checked, and help determine whether basin-wide water quality studies should be initiated. Water quality objectives are also a standard for assessing the Ministry's performance in protecting water uses.

Water quality objectives are established to protect all uses which may take place in a waterbody. Monitoring is undertaken to determine if all the designated water uses are being protected. The monitoring usually takes place at a critical time, when the water quality objectives may not be met, that is generally determined as part of the water quality objective setting exercise. It is assumed that if all designated water uses are protected at the critical time, then they also will be protected at other times when the threat is less. For practical reasons, the monitoring usually takes place during a five week period, which allows the specialists to measure the worst, as well as the average condition

in the water. For some waterbodies, the monitoring period and frequency may vary, depending upon the nature of the problem, severity of threats to designated water uses, and the way the objectives are expressed (e.g., mean and/or maximum values).

2.0 SITE DESCRIPTION

2.1 Watershed Description and Hydrology

Langford Lake is a small lake (60 ha) located approximately 15 km west from Victoria on southern Vancouver Island (Figure 1). The watershed is relatively small (3.3 km^2) and the average water residence time has been calculated at 3.6 years (McKean & Munteanu 1981).

The surficial geology of the area is mainly a mixture of well-drained course to medium texture materials with some mountainous land of thinly-mantled bedrock. The inflow area at the southeast end of the lake and the outflow area to the northwest are characterized by very poorly drained organic soils. Approximately 40% of the watershed covers Skirt Mountain to the north; this is an area of shallow, rocky soils which are typically well-drained (McKean & Munteanu 1981).

The watershed lies within the Coastal Douglas Fir Biogeoclimatic Zone and consists of several biomes: rocky outcrops of salal and lichens, well-drained stable areas of Douglas fir, Garry oak, arbutus and red cedar, and low wetlands of willow, skunk cabbage, *Spirea* bushes, sedges and grasses (McKean & Munteanu 1981).

There is one main inflow to the lake at the southeast corner of the lake (which was the original outflow) and the outflow to the Goldstream River is at the northwest corner of the lake (Figure 1). There are also several smaller seasonal inflows around the lake. The flow of water is somewhat restricted in Langford Lake by vegetative growth in and around the outlet of the lake, although periodic vegetation removal has been done. Lake levels are highest during the winter months and typically around 63 m above sea level. Water levels drop throughout the summer and generally reach a low of around 62.5 m at the beginning of October. The fall freshet normally starts around this time and the water levels quickly rise over the next several weeks to winter levels.

2.2 Lake Morphometry

Langford Lake is shallow with a mean depth of 6.4 m, a maximum depth of 16 m and a volume of approximately 3,800 dam^3. There is a deep basin in the southeast end of the lake where the aerator is located. A bathymetric map of Langford Lake is provided in Figure 2. Approximately half of Langford Lake's 60 ha surface area represents littoral areas less than 3 m in depth. The perimeter of the lake is 4.8 km.

3.0 WATER USES

There are four current water licences authorized to withdraw water from Langford Lake. Two are authorized for irrigation at approximately 300 $m^3 \cdot year^{-1}$ and the other two are domestic withdrawals authorized for approximately 830 $m^3 \cdot year^{-1}$. It is unlikely that the domestic withdrawals are actually used for domestic (i.e., drinking) purposes and are probably used for watering lawns and other non-consumptive uses. Langford Lake still provides significant recreational values for both lakeshore residents and visitors. With approximately half of the shoreline developed and occupied with full time residences, aesthetics are a very important aspect of the lake for shoreline residents. There are four established swimming areas on Langford Lake including two public beaches which are well used in the summer months. Boating and canoeing are also popular activities although the use of outboard motors and personal water craft is not permitted.

Langford Lake provides a popular recreational fishery and is stocked annually with rainbow (*Oncorhynchus mykiss*) and cutthroat trout (*O. clarki clarki*). Between 1996 and 2005 Langford Lake has been stocked with more than 66,000 rainbow trout (the majority of which were of catchable size) and over 25,000 cutthroat trout (yearlings and fingerlings). The estimated value of these is fish at the time of release is approximately $80,000 (based on $1 per catchable trout and $0.60 per yearling or fingerling). Smallmouth bass (*Micropterus dolomieui*), pumpkinseed (*Lepomis gibbosus*) and yellow perch (*Perca flavescens*) are also present and provide angling opportunities.

4.0 LAND USE

Residential development is concentrated along the southeast and northeast shorelines with significant riparian vegetation along the northwest shoreline. There is a gravel pit at the northeast end of the lake which is partially encompassed within the watershed boundaries. This area is currently undergoing residential development which may result in removal of some of the native vegetation in that area. The Hull's Field area at the southeast end of the lake is being developed to enhance its recreational value. Included in the development will be a series of detention ponds to treat stormwater before it flows into the lake. There are also developments planned for the southwest end of the lake, which includes 105 ha of land between Langford Lake and Glen Lake, and on Skirt Mountain to the north of the lake.

There are land-based recreational opportunities in this watershed as well with hiking trails and boardwalks along the western and northern shore of the lake which are popular with local residents.

5.0 STUDY DETAILS

Four sites were regularly sampled in this assessment. The locations of the sample sites are illustrated in Figure 3 and a summary of site location details is provided in Table 1. The majority of past water quality data has been collected in the spring at EMS site 1100944 in the southeast deep basin at the aerator float. Sampling was conducted at this site only when the aerator was not operating (November through March). When the aerator was operating, sampling was conducted at the original deep site (EMS site 1100953), which is located approximately 50 m to the southeast from site 1100944, to minimize any influence of the aerator. Two additional sites were sampled to the northwest of the deep sites: E256395 and E256396. Samples were also taken from the main inflow to the lake (E234410) and the outflow (E234413). All samples were collected following Ministry of Environment approved methods (B.C. RISC 1997a).

Water chemistry was sampled at all four lake basin sites (Table 1 and Figure 3). Samples were collected in November 2004 and January 2005 while the water column was mixed, then monthly from March to November 2005, which included the entire period the water column was thermally stratified (May to October). Follow-up sampling took place in February 2006, again while the water column was well mixed. Grab samples were taken at three depths in the water column (0.1 m, 6 m and 12 m) for the deep stations (1100944 and 1100953) and at the surface and bottom of the water column for the other basin sites (E257395 and E257396). Once the aerator began operation, sampling at three depths was done at 1100953 and only at the surface and bottom of 1100944.

Surface samples were collected by hand and water column samples were collected using a Van Dorn bottle. Spring overturn samples were analyzed for the following parameters:

- Physical: pH, true colour, conductivity, total suspended solids, total solids
- General inorganics: alkalinity
- Carbon: total inorganic carbon, total organic carbon
- Nitrogen: total nitrogen (N), total Kjeldahl N, total organic N, ammonia N, nitrate + nitrite

- Phosphorus: total phosphorus (P), ortho-P, dissolved P.

Total metals were also analyzed during the 2005 and 2006 spring overturn monitoring at site 1100944.

Monthly water chemistry sampling focussed on nutrients and for the deep sites (either 1100944 or 1100953, depending on whether the aerator was operating) analyses were done for the following parameters:

- Carbon: total inorganic carbon, total organic carbon
- Nitrogen: total Kjeldahl N, total N, total organic N, ammonia N, nitrate + nitrite
- Phosphorus: ortho-P, total P.

The other three sites were sampled monthly at the surface and bottom and analyzed for total P and total N. The purpose of this was to determine the primary nutrient concentrations in the water column away from the aerator. The inflow (E234410) and outflow (E234413) were sampled periodically for total Kjeldahl N, total N, nitrate + nitrite, ortho-P and total P during the fall freshet to get an indication of how much nitrogen and phosphorus was being delivered from the main inlet and removed from the system via the outlet. Water chemistry analyses were conducted by Maxxam Analytics Inc. in Burnaby, British Columbia.

Temperature and dissolved oxygen (DO) concentrations were measured on each sampling day at all sites with a YSI 550A Handheld DO and Temperature System at one metre intervals from surface to bottom. Water clarity was measured on each sampling day using a Secchi disc.

Phytoplankton and chlorophyll *a* samples were collected by taking one litre grab samples at a depth of 0.5 m at the deep site (either 1100944 or 1100953, depending on operation of the aerator). Chlorophyll *a* samples were processed at the laboratory (Maxxam Analytics Inc.). Zooplankton samples were collected to determine community composition and densities using a 10 m vertical tow in a Wisconsin-style net with a net

opening diameter of 30 cm and a mesh size of 80 μm. Phytoplankton and zooplankton taxonomy was done by Fraser Environmental Services in Surrey, British Columbia.

Microbiological data was provided by the Vancouver Island Health Authority. The swimming beaches located at Leigh Road and Goldstream Avenue (Figure 3) are regularly monitored throughout the summer months and analyzed for fecal coliform counts. Geometric means were calculated using data from a minimum of 30 consecutive days and combining results from both sites.

6.0 WATER QUALITY

6.1 Limnological Characteristics

6.1.1 Lake Temperature Stratification

Water temperature was measured at each lake site from November 2004 to November 2005 and time/depth temperature profiles for each site are illustrated in Figures 4 through 7. The water temperature ranged from a minimum of approximately 4° C in the winter months to a maximum surface water temperature of 24° C in August at all sites. Sites 1100944 and 1100953 showed similar patterns of stratification, which is not surprising because of the close proximity to each other. A strong thermocline was established at approximately 7 m to 8 m in depth in May and the lake remained stratified at these sites until late October.

Site E257395 showed a similar stratification pattern and water temperatures as the deep sites with mixing of the water column taking place at the end of November.

Site E257396 was the shallowest site measured and the warmest. A slightly different pattern of thermal stratification was observed with a well-defined thermocline established by June at similar depths to the other sites. The water column destratified by early September and was completely mixed by the end of September with a temperature of 18° C throughout.

In 1984, an aerator designed to destratify the water column was installed in Langford Lake at site 1100944. The objective was to prevent internal phosphorus loading from the sediments by increasing dissolved oxygen concentration in the hypolimnion through destratification of the water column. Temperature profiles from 1984 and 1985 show that the unit was effective at destratifying the water column, however, it increased overall water temperatures throughout (Nordin & McKean 1988). The original intent was to retrofit the unit with a hypolimnetic aerator when funding was available which would provide a cool, oxygenated refugia for salmonids. Although a more efficient diffuser was

installed on the aerator in 1985, the planned retrofit was never completed and the aerator has operated in the same manner ever since.

The temperature profiles presented here are very similar to those for pre-aeration conditions (Nordin & McKean 1988) demonstrating that the unit, over time, has lost some efficiency and is no longer effectively destratifying the water column. However, this does not mean that aerator is not providing an overall benefit to the lake and this is discussed further in Section 6.1.4.

The proposed water quality objective for temperature is a summer maximum hypolimnetic (from 6 m in depth to the bottom of the lake) temperature of 15°C. This objective is designed to maintain cold water habitat for fish during the summer months (May through August) and was met in 2005.

6.1.2 Dissolved Oxygen

Dissolved oxygen (DO) concentrations were measured throughout the water column at each lake site from November 2004 to November 2005. Dissolved oxygen concentrations are presented in Figures 4 to 7.

At sites 1100944 and 1100953, the water column was well oxygenated (>5 $mg \cdot L^{-1}$) until May when decreased DO concentrations in the deep water were noted. In June, low DO (<1 $mg \cdot L^{-1}$) concentrations were observed in the bottom two metres at these sites and in July and August, the whole hypolimnion was showing concentrations below 1 $mg \cdot L^{-1}$. In September and October, the thermocline began to break down and DO concentrations were slightly higher, but remained below 1 $mg \cdot L^{-1}$ from a depth of 11 m to the lake bottom. By November, the water column was mixed and generally well oxygenated throughout.

Sites E257395 and E257396 showed a similar patterns to sites 1100944 and 1100953 with low DO concentrations below the thermocline during the summer months. E257396

had a completely oxygenated water column by September because the water column was mixed earlier than the other sites.

These observations indicate that, despite operation of the aerator from May through October, DO concentrations are low in the deepest and coolest part of the lake. This could be potentially stressful on trout if they are forced into the oxygenated, yet warmer, surface waters.

The original goal for DO concentrations in Langford Lake following aeration was 6 $mg \bullet L^{-1}$ (Nordin & McKean 1988). The current approved provincial water quality guideline for dissolved oxygen in the water column is 5 $mg \bullet L^{-1}$ (instantaneous minimum for the protection of aquatic life) (B.C. MOE 1997). This guideline was generally not met throughout the hypolimnion in Langford Lake from June to October, 2005.

The proposed long-term water quality objective for dissolved oxygen in Langford Lake is ≥ 5 $mg \bullet L^{-1}$ at any depth. This level represents the minimum concentration to minimize stress in salmonids and will also protect other species present which are more tolerant of lower DO concentrations (e.g. smallmouth bass). Although it is very unlikely that this objective can be met under the current conditions in Langford Lake, it is included as a goal for consideration in future lake management and restoration efforts.

6.1.3 Water Clarity

Water clarity was measured using a standard 20 cm Secchi disc, lowered in the water column until it was no longer visible from the surface. This is a standard, yet simple, measure of water clarity that can be used to indicate changes in water quality, as clarity decreases with increasing colour, suspended sediments or algal abundance.

Secchi depth readings are presented in Figure 8 and show a slight increasing trend. The mean post-aeration Secchi depth, from 1986 to 2006 (4.3 m, SD = 1.5 m) was greater than the pre-aeration Secchi depth measured in 1983 (3.8 m, SD = 0.5 m). One-way

analysis of variance was used to compare the mean Secchi depths for the pre-aeration period, the initial aeration period (1984 – 1985) and the post-aeration period and they were not found to be significantly different ($p = 0.19$).

The Canadian Environmental Quality Guideline for water clarity is 1.2 m to ensure there is negligible risk to the health and safety of recreational users and the results reported here easily meet that guideline. **The proposed water quality objective for Secchi depth in Langford Lake is an annual mean of greater than 4 m**. The calculation of the annual mean Secchi depth will be based on a minimum of four measurements taken quarterly. This objective is to be used as an indicator of water quality and will ensure the current level of water clarity in Langford Lake is maintained.

6.1.4 Limnological Interpretation

Although the Langford Lake aerator was initially effective during 1984 and 1985 (and presumably for some years after that) the thermal stratification patterns and dissolved oxygen levels presented here are similar to those observed pre-aeration (1983). Because the current productivity of the lake is similar to that of the mid-1980's (based on total phosphorus (Section 6.2) and chlorophyll *a* concentrations (Section 6.3)) we can assume that the aerator is no longer operating as designed. In August 2005, surface temperatures reached a maximum of 24° C. Although the proposed temperature objective was met (<15° C in the hypolimnion), the proposed dissolved oxygen objective was not met in the deepest parts of the lake. This could be potentially stressful on salmoninds.

The aerator, however, is still likely having a positive effect on the lake. For example, the anoxic conditions in the bottom waters are not as extensive as they were prior to aeration and this may be limiting the internal loading of nutrients during periods of thermal stratification (see Section 6.2.4). Water clarity appears to have increased slightly but the increase was not statistically significant. Trends can be difficult to determine as water clarity is dependent on a number of conditions at the time of sampling; phytoplankton concentrations and temporal hydrology can affect water clarity. Results can also be

influenced by weather and water conditions at the time of sampling as well as the individual recording the measurement.

6.2 Water Chemistry

6.2.1 General Parameters

All general parameters showed low levels and no significant trends were identified over the period of record. A summary of water quality data is provided in Appendix 1. Site-specific results are provided in Appendix 2. The water quality of Langford Lake can be described as neutral (mean pH = 7.5, SD = 0.3) with low dissolved solids (mean specific conductivity = 188 $\mu S \cdot cm^{-1}$, SD = 19 $\mu S \cdot cm^{-1}$) and average turbidity values well below the guidelines for the protection of aquatic life (5 Nephelometric Turbidity Units (NTU)) (mean turbidity = 1.2 NTU, SD = 0.5 NTU). The mean alkalinity was 61.7 $mg \cdot L^{-1}$ (SD = 2.9) indicating a low sensitivity to acidic inputs.

6.2.2 Nutrients

Nutrients can be defined as any material assimilated by organisms for growth and maintenance. In freshwater ecosystems the primary nutrients for algal growth are phosphorus (P) and nitrogen (N) although there are a number of others required in lesser amounts (e.g. carbon, silica, sodium, manganese, potassium, calcium, magnesium, iron, cobalt, copper, zinc and molybdenum). Nutrients required for metabolic processes are available through natural pathways and processes and, along with light, limit primary production in aquatic environments (NRC 2000). Nutrients dissolved in water are absorbed by primary producers (algae and aquatic plants) and provide primary consumers (e.g., zooplankton) with a nutrient source. The primary consumers, in turn, provide a source of nutrients for organisms higher in the food web and in this way nutrients are transferred to other organisms, both aquatic and terrestrial.

Nitrogen and phosphorus are natural resources that, before human development, were in limited supply. Water entering lakes is generally richer in N than P because N is naturally more abundant and because P compounds tend to be more insoluble (Grimm 1987). The availability of both has increased dramatically in the past several decades because of the increased use of fertilizers, burning of fossil fuels, urban development, land clearing and deforestation. The influx of nutrients has disrupted the natural N and P cycles, greatly increasing the bioavailability of these elements. The environmental consequences of this increase include accelerated eutrophication of waterbodies, contamination of groundwater, fish kills due to ammonia toxicity, increased toxic algal blooms, contamination of water supplies and increased economic burdens as a result of the need for monitoring, treatment and remediation of contaminated water (Environment Canada 2001).

Although the various forms of N and P in the aquatic environment can be measured, total N and total P are the most widely used variables for predicting eutrophication responses. They tend to overestimate the actual amount of bioavailable N and P, but are considered the most reliable indicators because of the dynamic nature of nutrients in the aquatic environments in terms of cycling (US EPA 2000).

6.2.2.1 Phosphorus

Phosphorus plays a major role in nearly all phases of algal metabolism, particularly in photosynthesis. It is required in the synthesis of nucleotides, phospholipids, sugar phosphates and other compounds and is bonded in a number of essential low molecular weight enzymes and vitamins (Wetzel 1983).

Phosphorus is least abundant in relation to other required nutrients, and therefore usually limits primary productivity in lakes (Brönmark & Hansson 1998), however to raise algal biomass, additions of both P and N are usually required (Kallf 2002). The most available fraction of P for direct uptake by aquatic primary producers is the dissolved inorganic form, orthophosphate (PO_4^-), which is also referred to as soluble reactive phosphorus (Nordin 1985). More than 90% of P occurs as biologically unavailable

organic phosphates and cellular constituents within the biota (Wetzel 1983) while PO_4^- is usually less than 5% of all P. Other fractions include organic P and particulate P which are transformed to more available forms at rates dependent on microbial action, environmental conditions, origin of material, kinetics and water residence time. In lakes with short water residence there is a higher rate of flushing which limits the amount of organic P that is transformed to inorganic P (Nordin 1985).

Phosphorus is measured as total P (TP), PO_4^- and total dissolved P (TDP). Total P consists of PO_4^-, total dissolved P and particulate P (>0.45 μm in diameter), which can be both organic and mineral in origin. The TDP fraction consists of PO_4^- and a fraction of dissolved organic P from the breakdown of biotic material and polyphosphates. In most lakes (except those with heavy suspended sediment loads), total P is the best estimate of biologically available P (Nordin 1985).

Phosphorus is non-volatile and atmospheric transport of P is limited to movement as a dust or aerosol. Surface waters receive P mainly through surface flows rather than groundwater because soils tend to bind P unless the soil is saturated and anoxic. In soils, PO_4^- in solution reacts quickly with ions to become unavailable to plants and decreases the potential for leaching of P to lakes. Transport of P in soils to lakes in significant levels is more likely to occur as a result of erosion rather than leaching. Once delivered to a lake, P is transported through the water column, mainly as particulates, and quickly removed from the water column (Allen 1995). Phosphorus is usually retained fairly efficiently by a combination of biological assimilation and the deposition of sediments and biota to the bottom sediments where binding with aluminium and ferric hydroxides is particularly strong. In anoxic conditions the ferric ions are reduced and the binding is weakened allowing the phosphate to diffuse more freely to the water column (Correll 1998). Exchange of P across the sediment/water interface is regulated by redox interactions, which are dependent on oxygen supply, mineral solubility and sorptive mechanisms (Stumm & Morgan 1996), the metabolic activities of bacteria and fungi, and the turbulence from physical and biotic activities (Wetzel 2001). The primary mechanism by which organic P is converted to inorganic phosphate in the sediment is

through the bacterial mediated breakdown of organic matter. Such mechanisms also create the reducing conditions required for the release of P into the water (Wetzel 2001).

The exchange of P between the sediments and the overlying water is a major component of the P cycle in natural waters; the ability of lake sediments to serve as a sink for P and the release of P from sediments back to the water column depends on a number of chemical, physical and metabolic factors. Traditionally, aerobic oligotrophic lake sediments have been considered a sink for P while sediments in eutrophic waters (where phosphate release is high when sediments turn anaerobic) are considered a source of P for the water column (Environment Canada 2003). There is little correlation between the P concentration of sediments and that of the overlying water (Wetzel 2001). Factors affecting P concentrations in sediments include: the ability of the sediments to retain P (e.g. iron (Fe) content, total organic carbon, particle size); the conditions of the overlying water (aerobic vs. anaerobic); and the biota within the sediments which can alter the exchange equilibria and thus affect the P transport back to the water (Environment Canada 2003).

Phosphorus turnover rates vary seasonally and can be very rapid during the summer when demand for plant growth is high and external inputs are lower. During winter and spring, external P loading and sedimentation is high while internal loading is low. During the summer months, the rate of P release from the sediments is greater and sedimentation from the water column (organic P from algae as well as Fe complexes) increases through the season. There can also be some vertical redistribution of P released from the sediments across the thermocline (Environment Canada 2003).

6.2.2.1.1 Results

Historical P results are summarized in Appendix 1. For this study, total phosphorus (TP) results are summarized temporally and spatially in Table 2. Site-specific results are provided in Appendix 2. The samples collected on January 24, 2005 indicate the lake is still vertically mixed at that time of year and provided the best estimate of TP

concentration at spring overturn for the lake with a mean concentration for all sites of 40 $\mu g \cdot L^{-1}$ (SD = 2 $\mu g \cdot L^{-1}$). This is well above the upper guideline limit for the protection of aquatic life (5 – 15 $\mu g \cdot L^{-1}$) and recreational guidelines (<10 $\mu g \cdot L^{-1}$). The effects of thermal stratification are evident by May 2005 with elevated concentrations in the hypolimnion at all four sites. All sites showed high hypolimnetic TP concentrations (i.e. > 100 $\mu g \cdot L^{-1}$) coinciding with periods of low hypolimnetic DO concentrations; however, the greatest concentrations were seen at the deepest site (1100953). The hypolimnetic P concentrations were much lower at site 1100944 which is closest to the aerator. In February 2006, the phosphorus concentrations are comparable among sites and indicate that the water column is once again vertically mixed.

The mean TP concentrations for all sites are illustrated in Figure 9. Mean concentrations were calculated, by sampling date, for both the top of the water column (epilimnion) and the bottom (hypolimnion) and plotted over time. This graph illustrates the differences between epilimnetic and hypolimnetic TP throughout the season. Total P concentrations are fairly consistent throughout the water column until thermal stratification occurs after which time the epilimnetic TP decreases through biological assimilation (e.g. uptake by phytoplankton and macrophytes) and the hypolimnetic TP increases through internal loading processes. As the thermal stratification of the water column breaks down in the fall, the water is mixed and dissolved oxygen levels increase and the TP concentrations decrease in the deeper waters. Sedimentation of algae and the binding of P with Fe complexes also reduce the overall TP concentration which eventually reaches equilibrium throughout the water column.

The most complete data for Langford Lake is from site 1100944 in the southeast half. Average TP concentrations, as close to spring overturn as possible, are listed in Table 3. With historical data, it is sometimes difficult to identify any trends in water quality because the data are not always collected under consistent conditions and that seems to be the case here. Ideally, the lake would have been sampled between late January, to ensure complete mixing had occurred, and mid February, before any significant biological activity has taken place.

The earliest available results are from May 14, 1973 and show an average concentration of 28 $\mu g \cdot L^{-1}$. At this time of the year we could expect to see some thermal stratification although on this date the TP concentrations were fairly uniform throughout the water column. If we assume there was an increased amount of biological activity (plant and algae growth) in the lake by this time, it is reasonable to conclude that the TP concentrations would have been much higher if they had been measured in January or February when light and productivity were lower.

Samples collected from 1983 to 1986 (with the exception of 1984) were collected at the preferred time (i.e. late January to mid February) and show higher mean concentrations ranging from 29 $\mu g \cdot L^{-1}$ to 46 $\mu g \cdot L^{-1}$. The 1984 samples were lower at 19 $\mu g \cdot L^{-1}$ but were collected in April and this may be the result of increased biological uptake.

In 1992, 1993 and 1995, samples were collected all within the same two week timeframe (February 27 – March 15) and showed mean concentrations of approximately 10 $\mu g \cdot L^{-1}$. The lower concentrations may have been caused in part by increased biological uptake. The aerator may have also contributed to this decrease by limiting the internal P loading in the previous year.

Samples collected in early March of 1997 and 1998 showed higher mean TP concentrations than the previous five years (32 $\mu g \cdot L^{-1}$ and 35 $\mu g \cdot L^{-1}$, respectively). In 2000 and 2001, sampling was conducted around the third week of March and mean TP concentrations measured (22 $\mu g \cdot L^{-1}$ and 26 $\mu g \cdot L^{-1}$, respectively) were lower than 1997 and 1998, but higher than those measured in the mid-1990's. In 2002 and 2003, samples were collected in mid-March and mean concentrations were again lower (12 $\mu g \cdot L^{-1}$ and 9 $\mu g \cdot L^{-1}$, respectively) and within the water quality guidelines for the protection of freshwater aquatic life. The most recent spring overturn samples were taken in late January 2004 and 2005 and early February 2006. The results showed elevated mean TP concentrations of 32 $\mu g \cdot L^{-1}$, 41 $\mu g \cdot L^{-1}$ and 33 $\mu g \cdot L^{-1}$, respectively.

The data presented in Table 3 and illustrated in Figure 10 do not show any trend over time in TP concentration at spring overturn and this is likely the result of several factors. The data were not collected under consistent conditions – some were collected well into spring when the biological activity of the lake was increased which would result in lower TP concentrations. The aerator likely has had little direct effect on spring overturn TP concentrations as operation does not begin until after the sampling is done. Other than two periods showing lower TP concentrations (1992 – 1995 and 2002 – 2003), the measured springtime TP concentrations have been typically greater than the guideline levels for the protection of freshwater aquatic life.

Simple linear regression was used to determine if the date of sampling influenced spring overturn average TP concentrations (Figure 11). Only post-aeration TP concentrations were used in this comparison and although the relationship was not statistically significant ($p = 0.002$), 58% of the variation in TP concentrations could be explained by the timing of sampling. This suggests that spring overturn sampling should be conducted in January or February to obtain a more accurate measure of nutrient levels prior to the onset of significant biological activity. Additionally, those values illustrated in Figure 10 which are within the guidelines may not be representative of conditions typical for Langford Lake.

Orthophosphate was measured at one of the two deep stations (1100944 or 1100953) throughout the study, depending on operation of the aerator (Table 4). Orthophosphate was relatively low in the epilimnion and metalimnion except when the water column was completely mixed in the winter months. Any available orthophosphate would have likely been assimilated fairly quickly by algae and other plants keeping the concentrations down in the photic zone. The effects of thermal stratification on the mixing ability of the upper and lower areas of the water column is demonstrated by the high orthophosphate and TP concentrations measured in the hypolimnion in June through November. This also demonstrates the reduced efficiency of the aerator which was designed to destratify the water column and increase dissolved oxygen concentrations deep in the water column to prevent internal P loading. When the water column mixes in the fall, the P liberated from

the bottom sediments can alter lake conditions and may increase the risk of fall algal blooms, particularly cyanobacteria.

The main inflow (site E234410) to the lake has been sampled periodically and the TP results are listed in Table 5. Total P concentrations decreased between November 1999 and November 2005 from 115 $\mu g \cdot L^{-1}$ to 37 $\mu g \cdot L^{-1}$. The lower levels noted in 2005 are likely due to the elimination of agricultural activities in Hull's Field. A further decrease to 6 $\mu g \cdot L^{-1}$ was noted in January 2006. Although it is difficult to draw any conclusions from the limited amount of data, there does appear to be an overall decrease from the concentrations reported previously (McKean & Munteanu 1981) of 500 $\mu g \cdot L^{-1}$ in 1979 and 1980.

The proposed short-term (5 – 20 years) water quality objective for total phosphorus in Langford Lake is an average concentration of ≤ 20 $\mu g \cdot L^{-1}$ at spring overturn. The objective will be based on the average of three measurements taken from the water column (surface, mid-column and bottom) at the deep site (1100944) and should be collected by mid-February before thermal stratification. This objective recognizes that because of past land-use practices within the watershed, the B.C. guideline of 5 $\mu g \cdot L^{-1}$ – 15 $\mu g \cdot L^{-1}$ for the protection of freshwater aquatic life and <10 $\mu g \cdot L^{-1}$ for the protection of recreational primary contact will not likely be met in most years. If serious remediation efforts to reduce phosphorus concentrations in the lake are implemented in the future, a long-term (>20 years) objective of <10 $\mu g \cdot L^{-1}$ should be achievable.

6.2.2.2 Nitrogen

Nitrogen, along with phosphorus, carbon and hydrogen, is one of the major constituents of the cellular protoplasm of organisms and plays a key role in the productivity of freshwaters (Wetzel, 1983). Nitrogen occurs mainly in the amino acids and proteins of organisms (Brönmark and Hansson, 1998) and is also involved in the functions of nucleotides, nucleic acids, chlorophyll and coenzymes. Nitrogen limitation in lakes is

less common than P limitation because of the potential for fixation of atmospheric N by cyanobacteria (Nordin, 1985).

The aquatic nitrogen cycle is a balance of N inputs and N losses from an aquatic environment. Nitrogen inputs to aquatic systems include atmospheric (particulate fallout and precipitation), N fixation in both the water and the sediments, and inputs from both groundwater and surface water. Nitrogen losses occur through outflows from the basin, nitrogen gas (N_2) losses to the atmosphere, and sedimentation of inorganic and organic N-containing compounds (Wetzel 1983).

Nitrogen occurs in the environment in a number of forms, the most important in primary production are the inorganic forms ammonia and nitrate (Nordin 1985). Ammonia is the most reduced inorganic form found in water and exists in two states in equilibrium, depending on environmental conditions: ammonia (NH_3) and ammonium (NH_4^+) ion (Nordin & Pommen 1986). Ammonia is the most favourable form of N for cell uptake (Brönmark & Hansson 1998), however the preference for either of ammonia or nitrate (NO_3^-) will be dependent on algal species (Nordin 1985). Ammonia is usually low in aerobic waters because of the utilization by plants in the photic zone (Wallace 2002) and nitrification to NO_3^-. Inorganic N can also be absorbed to sediments and released when conditions in the water change. Other forms of N include nitrite (NO_2^-), which is generally not present in large quantities in undisturbed lakes; dissolved organic N (DON) usually in the form of polypeptides and complex organics; and particulate organic N present as phytoplankton, zooplankton and detritus. Dissolved organic N may be biologically available as amino acids and is quickly utilized by bacteria.

Ammonia is generated by heterotrophic bacteria as the primary nitrogenous end product of decomposition of proteins and other nitrogenous organic compounds and is present primarily as NH_4^+ which is readily assimilated by plants in the trophogenic zone. Eventually, all organic nitrogen in domestic wastewater is converted to ammonia through a process called mineralization or ammonification. This process can occur under aerobic or anaerobic conditions, although in septic tanks anaerobic is most common (Wallace

2002). In water, NH_3 rapidly hydrolizes to NH_4^+ at which point it is available for uptake by phytoplankton. In soils and water, NH_4^+ is biologically converted to NO_3^- through nitrification at the interface between oxic and anoxic water or sediments (Kalff 2002). Ammonium readily attaches to soil particles and is therefore relatively immobile. Once converted to NO_3^-, via nitrification, it is relatively mobile in the soil and readily leaches to ground and surface water (Wallace 2002). Nitrate is the most common form of inorganic nitrogen entering freshwaters from the drainage basin in surface waters, groundwater and precipitation.

Nitrification is usually defined as the biological oxidation of NH_4^+ to NO_3^- with NO_2^- as an intermediate product (Wallace 2002); it is an aerobic process facilitated by bacteria of the genuses *Nitrosomonas* and *Nitrobacter*. In reaction 1 below, ammonium is oxidized to nitrite in an aerobic process requiring at least 1.5 $mg \cdot L^{-1}$ of dissolved oxygen. In reaction 2, nitrite is oxidized to nitrate. Hydrogen ions are produced in the process which could increase the acidity of the environment and affect the function of the *Nitrobacter* bacteria. There must be sufficient alkalinity present as a buffer to prevent the pH from dropping (Wallace 2002).

$$(1) \quad NH_4^+ + 1.5\ O_2 \xrightarrow{\textit{Nitrosomonas}} NO_2\text{-} + 2\ H^+ + H_2O$$

$$(2) \quad NO_2^- + 0.5\ O_2 \xrightarrow{\textit{Nitrobacter}} NO_3^-$$

$$NH_4^+ + 2\ O_2 \longrightarrow NO_3^- + 2\ H^+ + H_2O$$

Nitrate is assimilated into organic nitrogenous compounds within organisms and is bound and cycled in photosynthetic and microbial organisms. During the normal metabolism of these organisms and at death, the nitrogen is liberated as ammonia (Wetzel 1983).

Denitrification (reaction 3 below) occurs when O_2 within the system has been depleted; bacteria are able to use the oxygen in NO_3^- as an alternate electron acceptor for metabolic purposes:

$$(3) \quad 6(CH_2O) + 4\ NO_3^- \longrightarrow 6\ CO_2 + 2\ N_2 + 6\ H_2O$$

This reaction is irreversible and occurs in the presence of available organic carbon and anoxic or anaerobic conditions (Wallace 2002). Nitrification and denitrification are closely coupled, but denitrification is the process responsible for the loss of fixed nitrogen (primarily as N_2) to the atmosphere (Kalff 2002). There are approximately 17 genera of bacteria capable of denitrification (Wallace 2002).

6.2.2.2.1 Results

The forms of nitrogen monitored in this study include total nitrogen (TN), ammonia (dissolved), nitrate + nitrite (dissolved), total organic N and total Kjeldahl N. Total N was measured at all four sites throughout the study period and the results are summarized in Table 6 and illustrated in Figure 12. Historical results are summarized in Appendix 1 and site-specific results are presented in Appendix 2.

The effects of aeration on nitrogen concentrations are apparent in the results of Table 6. Site 1100944 showed elevated hypolimnetic TN concentrations during stratification; however, site 1100953 (which is within 50 m of 1100944 and the aerator) showed much higher concentrations peaking at 2,300 $\mu g \cdot L^{-1}$ in November 2005. Hypolimnetic TN concentrations at site E25795 were similar to site 1100944. Except for July and August 2005, lower hypolimnetic TN concentrations were observed at site E257396 than site 1199044. This may be the result of earlier destratification and quicker flushing as this site is the shallowest and closest to the lake outlet. The overall difference between epilimnetic and hypolimnetic concentrations is illustrated in Figure 12 which shows the accumulation of nitrogen compounds in the deeper waters as the season progresses.

Historical average total nitrogen concentrations at spring overturn are illustrated in Figure 13. There is no apparent trend over time with values ranging from a high of 774 $\mu g \cdot L^{-1}$ in 1984 to a low of 320 $\mu g \cdot L^{-1}$ in 2003.

The other N parameters (ammonia, nitrate + nitrite, total organic N and total Kjeldahl N) were measured at either site 1100944 or site 1100953, depending on whether the aerator was operating (Table 7). Anoxic conditions commonly lead to the accumulation of

ammonia and orthophosphate in the hypolimnion (Beutel 2001). This is evident by the results listed in Table 7, which show increasing average ammonia concentrations as the season progresses. At the same time there is a corresponding decrease in nitrate + nitrite concentrations. This is caused by the lack of oxygen which prevents nitrification from taking place. With decreasing dissolved oxygen concentrations in the hypolimnion, ammonia accumulates and nitrate concentrations decline as the cycling of N is interrupted. This is illustrated in Figure 14, which shows hypolimnetic ammonia, nitrate+nitrite and dissolved oxygen through the year. It is this accumulation of ammonia that contributes the most to increasing TN concentrations seen through the growing season.

Nitrite and un-ionized ammonia (NH_3) are toxic to fish and B.C. has approved water quality guidelines for each parameter for the protection of aquatic life (Nordin & Pommen 1986). The results from this study show that neither parameter is of concern at this time in Langford Lake. For example, the guideline for the maximum concentration for total ammonia nitrogen based on conditions in November 2005 (i.e., pH estimated at 7.5 and water temperature approximately 10º C) would be 12.7 $mg \bullet L^{-1}$. Our results showed an average concentration throughout the water column of approximately 0.6 $mg \bullet L^{-1}$. The guideline for the maximum nitrite concentration (based on an average chloride concentration of 9.3 $mg \bullet L^{-1}$) would be 0.3 $mg \bullet L^{-1}$. Our highest average result for nitrate+nitrite was 0.311 $mg \bullet L^{-1}$ in April 2005. Because nitrite (NO_2^-) is an intermediate product in the nitrification process which converts NH_4^+ to NO_3^-, we can assume that the majority of this is NO_3^-, not NO_2^-, and therefore well below the guideline level. We conclude then that excess ammonia and nitrite concentrations are not a concern in Langford Lake at the present time.

Total nitrogen was measured periodically in the main inflow from Hull's Field and these results are listed in Table 5. The highest TN concentrations measured were in 1999 at 2,800 $\mu g \bullet L^{-1}$ and the most recent autumn concentrations (1,800 $\mu g \bullet L^{-1}$ in November 2005) were lower. This reduction is likely due to the elimination of agricultural activities in Hull's Field and blockage of inflow due to the current development of the area.

The proposed long-term water quality objective for total nitrogen in Langford Lake is an average concentration of ≤500 μg·L^{-1} at spring overturn. The objective is based on the average of three measurements taken from the water column (surface, mid-column and bottom) at the deep site (1100944) and should be collected by mid-February before thermal stratification. This objective approximates the average concentration measured at spring overturn over time and reflects the current eutrophic conditions of the lake. Although this objective may be difficult to achieve at the present time, it does provide a reasonable goal for management efforts.

6.2.2.3 Nitrogen:Phosphorus Ratio

The nitrogen:phosphorus (N:P) ratio is a useful indicator of lake trophic status and whether primary production is limited by phosphorus or nitrogen concentrations. Algae require N and P in particular ratios and comparing these values to what is actually available in the water can be a valuable diagnostic tool (Nordin 1985). The N:P ratio is derived from the Redfield Ratio (Redfield 1958) which describes the general requirements of carbon, nitrogen and phosphorus for freshwater primary producers. The ratio is 106 C: 16 N: 1 P (by atoms) or 40 C: 7 N: 1 P (by weight). Since our analytical results, and much of that in the literature, are reported by weight, we will use the latter ratio in this discussion.

Most lakes are P-limited and N:P ratios tend to decrease with increasing eutrophication, either natural or anthropogenically mediated. Therefore, decreasing N:P ratios would indicate deteriorating water quality. As a general guideline, lakes with N:P ratios greater than 7:1 are P-limited and those less than 7:1 are N-limited although there is considerable variation around this “ideal” ratio.

Nordin (1985) proposed the following range of total N: total P ratios to describe nutrient limitation:

- <5:1 is indicative of N limitation;
- 5:1 – 15:1 indicates no limitation or co-limitation, and;
- >15:1 indicated P limitation.

Slightly different levels were reported by the OECD (1982) as follows:

- <10:1 = N limitation;
- 10:1 – 17:1 = no limitation or co-limitation, and;
- >17:1 = P limitation.

The observed variation in the limiting nutrient ratios is due to a number of factors. One contributing factor could be the luxury uptake of both N and P, whereby primary producers are able to take up and store more of the nutrients than are immediately required for optimal growth (Forsberg 1975). Inter-specific differences in nutrient requirements among different phytoplankton species used, as well as environmental conditions such as water residence time (Meakin Consultants (in draft) 2002) could also account for variation. Finally, the varying experimental conditions and uncertainties in the bioavailable fractions of the nutrients among the reported experiments (OECD 1982) will also contribute to variation in reported results.

Watershed characteristics also play a role in the N:P ratio of a lake; generally oligotrophic lakes receive runoff from undisturbed areas with high N:P ratios and eutrophic lakes receive runoff with low N:P ratios originating from feedlots, stormwater drainage, sewage and other inputs (Meakin Consultants (in draft) 2002). Nutrient inputs to lakes aren't strictly the result of human land-use disturbances; in many areas, soil-borne P can also result in naturally high P levels in lakes and streams.

At one time, it was thought that N:P ratios could be used in managing the risk of excessive cyanobacterial blooms with TN:TP ratios >29 typically exhibiting low proportions of blue-green algae (Smith 1983). Since that time, researchers have shown that it is the relative amount of each nutrient, rather than the N:P ratio, that influences the biomass (and toxicity) of cyanobacteria (Downing et al. 2001; Giani et al. 2005). In Meakin Consultants' review of N:P ratios (in draft, 2002) they conclude that there is no distinct N:P ratio at which to expect changes in phytoplankton species composition in surface waters, especially in regards to predicting timing of potentially harmful cyanobacterial blooms. Instead, they suggest that N:P ratios be used as one of several

tools available to ecosystem managers in an integrated lake management strategy. For example, N:P ratios could be used to help assess the general trophic status of a lake because high N:P ratios are associated with oligotrophic conditions (low algal biomass dominated by green algae) and low N:P ratios are associated with eutrophic conditions (higher algal biomass dominated by cyanobacteria), providing an approximation of potential nutrient limitation in a system.

In this study, N:P ratios were calculated by dividing the average spring overturn total N concentration by the average spring overturn total P concentration. Where total N data were not available, it was calculated by summing the Kjeldahl nitrogen (ammonia and organic N) and nitrate + nitrite values. The N:P ratios over time for 1100944 are illustrated in Figure 15. The N:P ratio for this site has been variable over the years which is a reflection of variation in nutrient concentration, both N and P. The results show a low value of 12 in 1986 and a high value of 50 in 1995. The most recent results show lower N:P ratios, however they are still indicating that Langford Lake is predominantly P-limited.

Figure 16 illustrates the average epilimnetic and hypolimnetic N:P ratios (from all sites) and the changes that occur over the course of the growing season. The epilimnetic N:P ratio increases through the course of the year suggesting P-limitation during summer and early fall months. The hypolimnetic N:P ratio decreases, which is a reflection of the increasing P loading as a result of anoxia. In the fall as thermal destratification occurs, the epilimnetic N:P decreases, because of disproportionate increases in TN and TP with a greater increase seen in TP. This results in conditions which may favour the formation of blue-green algae in the photic zone.

The proposed short-term water quality objective for N:P ratios in Langford Lake is ≥20:1. This objective reflects the current conditions of the lake. Calculation of N:P ratios will be based on average total nitrogen and total phosphorus concentrations measured at spring overturn. If the objectives for total N (500 $\mu g \cdot L^{-1}$) and total P (25 $\mu g \cdot L^{-1}$) are met, the N:P ratio objective will also be attained ensuring the lake continues

to be P-limited. This objective can be revised in the future to reflect any improvements in total N and total P concentrations.

6.2.2.4 Organic Carbon

The primary concern with organic carbon is with respect to drinking water sources and the production of disinfection by-products (e.g., trihalomethanes) as a result of chlorination. Because Langford Lake is not a drinking water source, this is not an issue.

There are, however, some biological concerns related to carbon concentrations in lakes. For example, mercury availability and bioaccumulation increase as dissolved organic carbon (DOC) increases in drainage lakes partly because mercury brought to an aquatic system from the surrounding catchment area is attached to humic substances. As well, reductions in DOC inputs have been shown to decrease primary productivity in lakes, which can have important effects on invertebrates and fish communities (Moore 1998 and citations within).

Dissolved organic carbon has not consistently been measured in Langford Lake, but total organic carbon has and this data is provided in Appendices 1 and 2. The mean TOC concentration measured was 2.9 $mg \cdot L^{-1}$ (SD = 2.6, n = 18) in the 1980's; 3.4 $mg \cdot L^{-1}$ (SD = 0.2, n = 4) in the 1990's, and 3.7 $mg \cdot L^{-1}$ (SD = 1.3, n = 29) in the 2000's. Results from samples collected on September 21, 2005 were unusually high (15 $mg \cdot L^{-1}$ – 16 $mg \cdot L^{-1}$); because the reason for this is not obvious and they appear to be outliers, these results were not included in this analysis. One-way Analysis of Variance was used to determine if there was a difference between these means and they were not found to be significantly different ($p = 0.2$).

The average TOC concentrations measured in 2005-2006 are illustrated in Figure 17 (except for the September 21, 2005 results discussed above). The typical range for TOC in B.C. waters is 1 $mg \cdot L^{-1}$ to 30 $mg \cdot L^{-1}$ (Cavanaugh et al. 1998) and generally less than 5 $mg \cdot L^{-1}$ except for lakes and rivers with high natural sources (Moore 1998). The levels

measured in Langford Lake fit within this range. Therefore, no objective for total organic carbon for Langford Lake is proposed at this time.

6.2.3 Metals

Total metals have been measured periodically in Langford Lake since the 1980's, usually at spring overturn prior to stratification of the water column. All samples were collected from site 1100944 and the results are summarized in Appendix 1. Most results were below analytical detection limits; however low level analyses conducted on samples collected in 2004 confirmed the following metal concentrations were below the water quality guidelines for the protection of aquatic life: aluminum (Al), antimony (Sb), arsenic (As), barium (Ba), beryllium (Be), cadmium (Cd), chromium (Cr), cobalt (Co), copper (Cu), iron (Fe), lead (Pb), manganese (Mn), molybdenum (Mo), nickel (Ni), selenium (Se), silver (Ag), thallium (Tl), titanium (Ti) and zinc (Zn).

The following metals also showed levels below analytical detection limits, however there are no aquatic life guidelines available: bismuth (Bi), potassium (K), tellurium (Te), tin (Sn), vanadium (V), and zirconium (Zr).

Finally, the following metals showed measurable results, but no water guidelines are available for the protection of aquatic life: magnesium (Mg), silicon (Si), sodium (Na), strontium (Sr) and sulphur (S).

The mean concentration for total calcium (21.7 $mg \cdot L^{-1}$, SD = 1.4 $mg \cdot L^{-1}$) indicates that Langford Lake has a low sensitivity to acidic inputs which agrees with previous assessments (Swain 1987).

There are no water quality objectives proposed for metal concentrations in Langford Lake at this time.

6.2.4 Water Chemistry Interpretation

The water chemistry parameters of concern in Langford Lake are nitrogen and phosphorus in their various forms. Spring overturn TP values typically exceed both the

guidelines to protect recreational primary contact (10 $\mu g \cdot L^{-1}$) and aquatic life (5 $\mu g \cdot L^{-1}$ – 15 $\mu g \cdot L^{-1}$). While the aquatic life protection guidelines have been met occasionally in recent years, these results were partly influenced by the timing of sampling and would have likely been higher if taken earlier in the year (i.e. January or February). Hypolimnetic TP and TN concentrations increased throughout the summer and into the fall, peaking at 560 $\mu g \cdot L^{-1}$ and 2,300 $\mu g \cdot L^{-1}$, respectively, in November 2005. Based on these results, Langford Lake can be classified as eutrophic.

The N:P ratio in the surface waters of Langford Lake is quite high during the growing season, indicating P-limiting conditions which are desirable from a water quality perspective. However, in the autumn the N:P ratio decreases considerably and this, in conjunction with increasing concentrations of N and P in the water column, makes conditions more favourable for blue-green algae. This is confirmed in the phytoplankton community species composition (see Section 6.3.2).

McKean and Munteanu's (1981) assessment of Langford Lake identified Hull's Field as the major input source of nutrients with average P concentrations of approximately 500 $\mu g \cdot L^{-1}$. The results from a limited number of samples reported here suggest this is no longer the case with a mean TP concentration of 24.8 $\mu g \cdot L^{-1}$ (SD = 13.6 $\mu g \cdot L^{-1}$). McKean and Munteanu (1981) reported internal loading as the next most significant source of phosphorus inputs with ortho-phosphate concentrations of 270 $\mu g \cdot L^{-1}$ measured at the deepest part of the lake in October 1980. The results from this study showed an increase in ortho-phosphate to 290 $\mu g \cdot L^{-1}$ in October 2005. Overall, the operation of the aerator in Langford Lake appears to have helped stabilize conditions, preventing any further deterioration with respect to internal nutrient loadings, but has not resulted in any long-term improvements in water chemistry.

6.3 Biological Analysis

Biological sampling is an important component of water quality monitoring. Organisms respond to a range of environmental conditions and can provide clearer understandings of

the functional relationships within an aquatic ecosystem. In this study, phytoplankton and zooplankton were collected to determine their abundance and dominance and help identify any linkages between water quality and anthropogenic activities.

Variances in phytoplankton community composition and biomass may signal changes in water chemistry and nutrient concentration. Nutrient enrichment affects phytoplankton density and diversity and can create phytoplankton surface blooms which reduce water quality (Palmer 1977). Langford Lake is not a domestic drinking water source, so the health risk concerns usually associated with eutrophic conditions (e.g., algal toxins) are generally not an issue here. There are other impairments, however, that may be issues in Langford Lake including: fish tainting or poor tasting flesh; allergic reactions (e.g., dermatitis) to certain species of phytoplankton; impacts to fish habitat (e.g., dissolved oxygen depletion, reduction in quality of food); and impacts on recreation resulting from reduced water clarity and excess algal growth (Nordin 1985).

Chlorophyll *a* is a measure of phytoplankton biomass and relates to the productivity of a waterbody. Values below 3 $\mu g \cdot L^{-1}$ are considered an indication of low productivity and values above 15 $\mu g \cdot L^{-1}$ are generally considered to indicate high productivity. Agriculture, sewage effluent, forest harvesting, urban development, and recreational activities can add nutrients to a lake, increasing chlorophyll *a* concentrations (B.C. RISC 1998). There are no provincial chlorophyll *a* lake guidelines, but phosphorus guidelines are designed to limit its concentrations.

Zooplankton hold an important intermediate position in the aquatic food web. Although they do not cause problems with water quality or aesthetics directly, they are sensitive to changes in water quality and are used as indicators. Specifically, zooplankton respond to dissolved oxygen concentrations, contaminants, and food quality/abundance.

Macrophytes (aquatic vascular plants) are an important component of a lake's littoral zone and are often the principal primary producers of those areas (Wetzel 1983). Macrophyte beds also provide an important refuge for juvenile fish and small fish

species, macrozooplankton, and benthic invertebrates from their predators. In low nutrient environments, macrophytes are able to out-compete phytoplankton; however, with increased nutrient concentrations, whole lake primary production shifts away from benthic plants and littoral communities towards the phytoplankton communities. The high growth potential of the phytoplankton community is now able to out-compete the slow growing macrophytes which are negatively impacted by the shading effects caused by the elevated phytoplankton biomass (Kalff 2002). A survey of aquatic macrophyte communities in water quality assessments is useful to provide documentation of species present and allow comparisons over time.

6.3.1 Methods

Plankton samples were collected throughout the study period (January 2005 – February 2006). Phytoplankton samples were collected by taking one litre grab samples at the surface which were preserved with Lugol's solution and shipped to the lab on ice for identification and enumeration. Chlorophyll *a* samples were collected by taking a one litre grab sample 0.5 m below the surface at the deepest part of the lake. Samples were filtered and analyzed at the lab. Zooplankton were collected using a 10 m vertical tow in a Wisconsin-style net with a mouth area of 0.07 m^2, a net opening diameter of 0.5 m and a mesh size of 80 μm. Samples were preserved with formalin and shipped to the lab on ice for species identification and enumeration. All samples were collected following Ministry of Environment approved methods (B.C. RISC 1997b). A visual survey of dominant aquatic plants was conducted in August, 2005 and the results were qualitatively compared to previous surveys.

6.3.2 Phytoplankton

Phytoplankton samples were collected at the deepest part of the lake (sites 1100944 and 1100953); the results were summarized and the dominant species are listed in Table 8. Dominant species were those that made up at least 10% of the total cells present in the sample. The complete results of taxonomic analyses for phytoplankton are provided in

Appendix 3. Historical data (1972 – 1981) for phytoplankton community species composition is provided in Appendix 4.

Phytoplankton was sampled nine times between January 2005 and February 2006. A total of 79 species were identified, the majority of which were rare. Species richness varied between 22 species and 47 species present on any one sampling date, with most days having at least 35 species present. The pennate diatoms (Order Pennales) and the green algae (Order Chlorococalles) were the best represented with 25 species and 17 species, respectively. The species that occurred the most frequently in high numbers was *Chroomonas acuta* (Order Cryptomondales) followed by *Tabellaria fenestrata* (Order Pennales), *Sphaerocystis schroeteri* (Order Chlorococcales), *Anabaena cf flos-aquae* and *Aphanizomenon flos-aquae*.

In January 2005, the phytoplankton community was dominated by the filamentous cyanobacteria *A. cf flos-aquae* (45%), cryptomonads (*C. acuta* and *Cryptomonas* spp.) (23%) and the pennate diatom *Tabellaria fenestrata* (18%). In March, the pennate diatoms *Asterionella formosa* (35%) and *T. fenestrata* (31%) were the dominant species, and by May the dominant species were *C. acuta* (35%) and *A.cf flos-aquae* (33%). The species composition shifted in the summer and the July sample was dominated by the green algae *Sphaerocystis schroeteri* (52%); others present were the blue-green *Anacystis limneticus* (10%), *C. acuta* (14%), and *T. fenestrata* (10%). The green algae continued to dominate in August, specifically *Gloeocystis ampla* (51%). Also present were the green algae *S. schroeteri*, *A. limneticus*, *C. acuta*, and the chrysophyte *Dinobryon divergens*. In September, three species dominated the phytoplankton community: *T. fenestrata* (38%), *A. flos-aquae* (24%) and *C. acuta* (20%) and by October, the phytoplankton was completely dominated by *A. flos-aquae* (82%). This was likely due to increased nutrient concentrations from the hypolimnion as the water column began to mix. In November, *A. flos-aquae* continued to dominate (63%) with the centric diatom *Melosira italica* (21%) also present. By February 2006, the phytoplankton community was again dominated by the diatoms (*Stephanodiscus niagarae* and *Fragilaria crotonensis*); other dominant species at this time included *S. schroeteri*, *C. acuta* and *A. cf flos-aquae*.

The dominant species discussed above were based on species abundance in cells•mL^{-1} which could be misleading because there is variation in cell sizes among algal species. Most of the dominant species in this study were comparable in size however there were a few exceptions. The diatoms *T. fenestrata*, *A. formosa* and *M. italica* are generally larger than the other dominant species and so would have accounted for a larger portion of the actual biomass than was indicated by the cells•mL^{-1} count. The same could be said for *Cryptomonas* and *D. divergens*.

The proposed water quality objective proposed for the phytoplankton community is the dominance (>50% cells•mL^{-1}) of non-cyanophyte species. This objective is designed to eliminate the surface cyanophyte scums that are aesthetically displeasing. This objective will likely not be met at all times in Langford Lake, especially the fall months when there is an increase in P throughout the water column. The occurrence of cyanophytes and other major taxonomic groups increases with total phosphorus and may also be influenced by other factors such as thermal regime (Watson et al. 1997).

6.3.3 Langford Lake Paleolimnological Assessment

Water quality monitoring and biological surveys are important in determining the current state of a lake but cannot provide insight to the historical trends for a given lake. Paleolimnological assessments provide this historical perspective by examining lake sediment cores and drawing conclusions from diatom assemblages preserved over time. A core is extracted from the lake bottom and sectioned into 5 mm slices. Each slice is dated and the diatom assemblage within is analyzed. Diatom assemblages can be determined because the silica shells are species specific and preserved in the sediments in layers over time. The diatom species present in the slice are indicative of conditions in the lake at the time they were deposited.

A sediment core was taken from Langford Lake in December 2003 and analyzed by the Paleoecological Environmental Assessment and Research Laboratory at Queen's University. The report is provided in Appendix 5. The results show that the diatom

assemblage was consistent from 1700 to 1900 and dominated by the mesotrophic planktonic *Stephanodiscus mediuis*, the oligotrophic planktonic *Cyclotella stelligra* and the mesotrophic benthic *Staurosirella pinnata*. Around 1940, a decline in *S. medius* began and by 1960 there was an increase in the more eutrophic planktonic diatom *Aulacoseira ambigua* which was dominant from about 1970 on (Cumming et al. 2006). It is interesting to note that *A. ambigua* was absent from the samples collected during this study and that the most abundant species in the phytoplankton samples (e.g. *Tabellaria fenestrata*) were not dominant in the historical record.

Based on this information, estimates of past total phosphorus concentrations can be made. Mid-summer concentrations prior to 1970 were representative of mesotrophic conditions ranging from 10 to15 $\mu g \cdot L^{-1}$. After 1970, the estimates increase to 14 to 19 $\mu g \cdot L^{-1}$, with the highest values occurring in the most recent sediments.

The conclusions of this analysis are consistent with the history of Langford Lake. Cumming et al. (2006) conclude that the dramatic increase in *A. ambigua* starting in 1960 and stabilizing at higher percent abundances after 1970 indicates a sharp change within the lake at that time. This is around the time that the flow of Langford Lake was changed to drain Hull's Field (Section 1.1) and illustrates the impact of the increased nutrient load on the lake.

6.3.4 Chlorophyll *a*

Historical and current chlorophyll *a* results are presented in Table 9. The highest concentrations measured in the current study were in March 2005 (16.6 $\mu g \cdot L^{-1}$) followed by 5.9 $\mu g \cdot L^{-1}$ in January. Chlorophyll *a* concentrations measured were otherwise quite low and below 3.5 $\mu g \cdot L^{-1}$. These concentrations are considerably lower than what was measured more than 20 years ago when October 1980 and July and August, 1984 concentrations were 37 $\mu g \cdot L^{-1}$, 36 $\mu g \cdot L^{-1}$ and 24 $\mu g \cdot L^{-1}$, respectively.

The proposed water quality objective for chlorophyll *a* in Langford Lake is a summer (May through August) concentration between 1.5 µg•L^{-1} and 2.5 µg•L^{-1}. Dillon and Rigler (1975) recommended this range of chlorophyll *a* in order to protect fishery and recreational values in lakes and were subsequently used for other southern Vancouver Island lakes (e.g., Elk and Beaver lakes (McKean 1992)) to protect the cold water fishery of those lakes.

6.3.5 Aquatic Macrophytes

Nuisance weed growth has been a longstanding complaint of Langford Lake shoreline residents. Surveys conducted in the 1980's showed the dominant species to be *Ceratophyllum demersum* (coontail). It was thought that *C. demersum* was so successful because this species obtains the majority of its phosphorus requirement from the water (it has no sediment bound roots) and the high phosphorus concentrations year round was a major factor promoting the growth of that plant (McKean & Munteanu 1981). In 1984, an aerator was installed in Langford Lake to help reduce water phosphorus concentrations which in turn would hopefully reduce weed densities and nuisance algal blooms.

An aquatic weed survey was conducted by Ministry staff in August 1983 documenting that *Ceratophyllum dermersum* (coontail) and *Elodea canadensis* (Canadian pondweed) were the dominant aquatic plants. *Ceratophyllum demersum* was dominant (30% - 60% bottom cover) in all embayed areas of the lake in depths greater than two metres, covering most of the bottom and growing to at least 0.5 m from the surface. *Elodea canadensis* was dominant in shallower areas less than two metres in depth. This survey also noted good overall aquatic plant diversity.

In August 1984, another survey was conducted and noted that *C. demersum* and *E. canadensis* still dominated the aquatic plant community with similar densities as in 1983, despite operation of the aerator and a weed harvester. The survey also noted poor water clarity and evidence of a recent blue-green algal bloom.

The August 1985 survey noted an overall improvement in *C. demersum* and *E. canadensis* densities with fewer weeds surfacing and generally improved water quality based on clarity and evidence of algal blooms. The survey also noted an increase in occurrence and densities of *Potamogeton amplifolius* and *P. berchtoldii*. The 1988 survey noted an increase in some areas of *E. canadensis* which was found to be growing close to the surface in some parts.

The lake was last surveyed in 1990 to look for Eurasion watermilfoil (*Myriophyllyum spicatum*), which was not found. *Ceratophyllum demersum* was the dominant aquatic plant however reductions in biomass were noted over the 1983 (pre-aerator) survey. *Elodea canadensis* was also found to in significant densities with increases in biomass noted over the 1983 baseline survey. There were also populations of the following species noted in 1988: *Potamageton amplifolius*, *Nuphar* sp. (pond lilies), *Scirpus* sp. (bull rushes) and *Typha* sp. (cat-tails).

A brief survey was conducted on August 16, 2005 with the objective of determining the current dominant aquatic macrophytes and extent of coverage to compare to the surveys of the 1980's. The survey was conducted in mid-August to be consistent with past surveys and prior to any weed harvesting efforts. Weather conditions were calm and sunny on the day of the survey and water clarity was good (Secchi depths of 4.3 m to 5.5 m). *Elodea canadensis* was found to be the dominant plant on the substrate of the littoral areas while *Potamogeton amplifolius* was the dominant plant extending to the water surface. Some *C. demersum* growth was noted within the *P. amplifolius* beds, but overall did not make up a large portion of the aquatic plant community. Filamentous green algae were noted in many of the shallow littoral areas growing on the substrate and submerged logs and branches. There were also several water lily beds (*Nympheae* sp.) noted throughout the shoreline as well as cattails (*Typha* sp.) and bulrushes (*Scirpus* sp.).

There were four primary areas of concern, with respect to *C. demersum*, noted in past macrophyte surveys and these are illustrated in Figure 18. The distribution of *P. amplifolius* in 2005 is illustrated in Figure 19. The distribution of *C. demersum* has been

significantly reduced since the 1980's and in 2005 comprised only a minor component of the aquatic plant community which was dominated by *P. amplifolius.* This shift may be due to increased water clarity (see Section 6.1.3) that has been noted since installation of the aerator in Langford Lake. Increased light levels have been shown to have a positive effect on the growth of *P. amplifolius* (Cronin & Lodge 2003).

6.3.6 Zooplankton

Zooplankton results are presented in Table 10. The zooplankton community of Langford Lake was composed predominantly of four groups: rotifers, cladocerans, calanoid copepods and cyclopoid copepods. Rotifers were the dominant group, especially *Keratella cochlearis*, *K. quadrata* and *Polyarthra* sp. Other rotifers which dominated at sometime during the year included *Asplancha* sp., *Conochilis* sp., *Hexarthra* sp., *K. longispina* and *Synchaeta* sp. McKean and Munteanu (1981) also reported a zooplankton community in Langford Lake dominated by rotifers at that time; species included *K. cochlearis*, *K. quadrata*, *Kellicotia longispina*, *Polyarthra* sp. and *Asplancha* sp.

The dominant cladocerans in 2005-2006 were *Daphnia rosea*, *D. pulex*, *D. pulicaria* and *Ceriodaphnia*. In 1980, the common cladocerans were *Bosmina coregoni* and *D. longispina*, neither of which were found in 2005. The dominant calanoid copepod in this study was *Diaptomus oregonensis*; in 1980, *Diaptomus* were also common but not identified to species. The dominant cyclopoid copepod in this study was *Diacyclops thomasi*, while in 1980 it was *Cyclops haueri*. Copepod nauplii were also very abundant in all samples collected.

There are not sufficient data to propose water quality objectives for zooplankton community structure in Langford Lake at this time. However, monitoring should continue as this information is useful in assessing water quality.

6.3.7 Biological Interpretation

Prior to aeration, the Langford Lake phytoplankton community varied seasonally from eutrophic diatoms (*Asterionella formosa*, *Fragilaria crotonensis*, *Stephanodiscus niagare* and *Synedra*) in the winter to eutrophic cyanophytes (*Anabaena* spp., *Aphanizomenon flos-aque* and *Microcystis* spp.) for the rest of the year (McKean and Munteanu 1981). Diatoms dominated after turnover when depleted silica levels were renewed which provided the diatoms with a competitive advantage over other species. After a few blooms, silica concentrations were reduced and the cyanophytes re-established dominance. Nordin and McKean (1988) noted that prior to aeration in Langford Lake, phytoplankton biomass usually decreased in August and September followed by an increase in late fall as the result of the eroding thermocline and an increase in nutrients from the hypolimnion. The late autumn blooms were significant with peaks in chlorophyll *a* of up to 35 $\mu g \cdot L^{-1}$. The historical data provided in Appendix 4 suggests that phytoplankton may have been a more significant component of the biota in the early 1970's than the early 1980's. In the fall of 1972, blue-green algae (*Coelosphaerium naegelianum*, *A. cf flos-aque*, *A. flos-aque*) and diatoms (*A. formosa*, *F. crotensis* and *Synedra cyclopum*) were dominant in the samples taken. In late summer, fall and winter samples from 1980 and 1981, most algal species were present in low amounts and no species appeared to be dominant. Phytoplankton communities are controlled by available nutrients, light and predation. During that time, *Ceratophyllum demersum* was a dominant aquatic plant in Langford Lake. This macrophyte has been shown to influence phytoplankton community composition by outcompeting phytoplankton species for available inorganic nutrients, limiting light availability and possibly inhibiting phytoplankton growth through the production of allelopathic compounds (Körner & Nicklisch 2002; van Donk & van de Bund 2002; Mjelde & Faafeng 1997).

With the aerator running efficiently in 1985, the diatoms and green algae became dominant throughout the summer, as opposed to the cyanobacteria, and biomass increased because of higher nutrient concentrations available from the bottom waters with the destratification of the water column. When the aerator was shut off in the fall of that

year, cyanobacteria (*Anacystis*) quickly became dominant again (Nordin & McKean 1988).

The current phytoplankton assemblage generally follows the same seasonal patterns with eutrophic diatoms being common in the winter, followed by a mix of green algae, diatoms and cyanophytes in the spring and summer months, and finally a cyanophyte-dominated community in the fall. The cyanophytes now appear to be more common in the winter months; *Anabaena flos-aquae* was dominant in January 2005 and *Aphanizomenon flos-aquae* now appears to be more comman in the fall than it was immediately after the aerator was installed. The cryptomonad *Chroomonas acuta* is now a common species but was not mentioned in earlier reports. The green algae *Gloeocystis* was rare or absent in the past, yet dominated in August 2005. The variations in species composition are not uncommon and can be expected because of seasonal changes in both chemical (nutrients and other essential elements) and physical factors (e.g., light, temperature, pH). The overall biomass of phytoplankton appears to be reduced, however, as indicated by summer chlorophyll *a* levels. For example, July concentrations were reported as 36 $\mu g \cdot L^{-1}$ in July 1984, 11 $\mu g \cdot L^{-1}$ in July 1985 (after installation of the aerator) and 1.5 $\mu g \cdot L^{-1}$ in July 2005. A similar trend was seen in August data as well.

There appears to have been a change in dominance in the macrophyte community of Langford Lake. Surveys from the 1980's reported extensive beds of *Ceratophyllum demersum* in the littoral zone. A 2005 survey showed that the dominant macrophytes are now *Potamogeton amplifolius* (large-leaf pondweed) and *Elodea canadensis*. This may be an indication of increased water clarity as *P. amplifolius* is known to do well with increased light while *C. demersum* does well in turbid conditions. Nutrient levels may also play a role; *C. demersum* is non-rooted and does well in waters with high concentrations of dissolved nutrients while *P. amplifolius* is rooted and obtains nutrients from the sediments. Decreases in chlorophyll *a* concentrations since the 1980's suggest lower productivity and support for this explanation.

There is a strong relationship between the trophic status of a lake and its zooplankton community (Whitman et al. 2004). The zooplankton community found in Langford Lake is consistent with the eutrophic conditions as indicated by the water chemistry results (Section 6.2.4). For example, several rotifer genera found are typical of productive lakes: *Brachionus*; *Keratella*; *Pompholyx*; and *Trichocerca* (Gannon and Stemberger 1978). Whitman et al. (2004) found that the genera and species most highly correlated with trophic status were consistently the most abundant and typically either small grazing crustaceans (e.g. *Bosmina longirostris* and copepod nauplii) or rotifers (e.g. *Keratella* and *Polyarthra*). This is consistent with what was found in Langford Lake.

For the crustaceans, copepod and cladoceran species richness is lower in eutrophic temperate lakes than in oligotrophic lakes. However, more nutrient-enriched lakes have greater crustacean zooplankton density and biomass (Pinto-Coelho et al. 2005). In Langford Lake, the copepods were largely represented by only one calanoid species (*Diaptomus oregonensis*) and one cyclopoid species (*Diacyclops thomasi*). The cladocerans were dominated by *Daphnia* with limited numbers of other species.

It is important to note that the sampling program used in this study will only provide a snapshot at the time the sample was taken and there could be much variation between sampling dates caused by both abiotic (e.g. nutrient levels) and biotic (e.g. predation) factors. The results of this study are, however, indicative of the eutrophic condition of Langford Lake.

6.4 Microbiology

The Vancouver Island Health Authority (VIHA) routinely monitors bathing beaches throughout the Capital Regional District during summer months. Fecal coliform counts are used as an indicator of contamination from human, animal or waterfowl excrement and therefore health risks to those using these areas for primary contact recreation (e.g. swimming). Beach count warnings are posted when counts exceed a 30-day geometric

mean of 200 fecal coliforms per 100 mL of water, when values fluctuate dramatically, or when there is other evidence of beach contamination.

6.4.1 Results

Fecal coliform results were provided by VIHA and are presented in Table 11. Samples were collected at two sites: the Leigh Road bathing beach and the Goldstream Avenue bathing beach (Figure 3) between April 11 and August 26, 2005. Both sites showed low results throughout the sampling period and at no time did the geometric mean approach the recreational water quality guideline.

The proposed microbiological water quality objectives for Langford Lake are ≤200 fecal coliform•100 mL^{-1} (geometric mean). This objective is a provincial guideline to protect primary contact recreational uses. Fecal coliforms are the only microbioliogical indicator regularly sampled at Langford Lake. In future water quality assessments for Langford Lake, *E. coli* and enterococci should be included because they provide superior indicators of fecal bacterial contamimation. The presence of *E. coli* in water is a strong indication of human sewage or animal waste contamination because it rarely multiplies in the environment (Leclerc et al. 2001) and has been statistically associated with an increase in the relative risk of gastrointestinal illness in freshwater studies (Wade et al. 2003; Pruss 1998). Enterococci are considered to be a better indicator of human fecal contamination than fecal coliforms, although supplementary to *E. coli* (Griffin et al. 2001).

6.4.2 Microbiology Interpretation

The monitoring results presented here indicate that the water quality in Langford Lake is good for recreational uses.

7.0 WATER QUALITY OBJECTIVES

In British Columbia, water quality objectives are mainly based on approved or working water quality guidelines. These guidelines were established to prevent specified detrimental effects from occurring with respect to a designated water use. Identified water uses for Langford Lake that are sensitive and should be protected are primary contact recreation and aquatic life. The water quality objectives recommended here take into account the impacts from past land use and should be periodically reviewed and revised to reflect any future improvements in water quality.

The objectives are summarized in Table 12.

8.0 MONITORING RECOMMENDATIONS

The recommended water quality monitoring program for Langford Lake is summarized in Table 13. Annual spring overturn monitoring should continue on Langford Lake to determine if the water quality objectives recommended here are being attained. This monitoring should consist of full water chemistry sampling at the deep station (1100944) at three depths (surface, mid depth and bottom) and include measurement of dissolved oxygen, temperature and water clarity. This sampling should be done by mid-February before thermal stratification or increased biological activity in the water column. Samples should also be collected at this time from the inflow (E234410) to determine the water quality of the inflow from the newly constructed detention ponds.

In the event that water is withdrawn in the future (see Section 9.3), additional monitoring should be conducted. This should include sampling from three depths for nutrients (total phosphorus, ortho-phosphorus, total nitrogen, ammonia, nitrate, nitrite), temperature profiles, dissolved oxygen profiles, chlorophyll *a* and Secchi depth. The frequency of this sampling should be monthly at a minimum and should go two months past the period of withdrawal. If the aerator is operating at this time, samples should be collected from site 1100953.

In addition, microbiological indicators should continue to be part of the Vancouver Island Health Region's beach monitoring program. In future water quality assessments of Langford Lake, monitoring of *E. coli* and enterococci should be included at bathing beaches and at the appropriate frequency (i.e., five samples in 30 days) during high use periods.

9.0 MANAGEMENT OPTIONS

The following sections briefly discuss management options to help maintain and potentially improve the water quality of Langford Lake.

9.1 Aerator Maintenance

The Langford Lake aerator was installed in 1984 and has been operating ever since. The unit was designed to destratify the water column and allow mixing of the photosynthetically oxygenated surface waters with the anoxic hypolimnetic waters. Increased dissolved oxygen levels would help reduce internal P loading, but at a cost of a warmer water temperature throughout the water column. The aerator worked effectively for at least the first year after installation; however, the temperature and dissolved oxygen profiles for Langford Lake (Figures 4 – 7) are now very similar to the pre-aeration conditions (see Nordin & McKean 1988). In 2005, there was a strong thermocline established by June at the deep sites and the entire hypolimnion was anoxic throughout July and August. The high hypolimnetic concentrations of total N and total P throughout the summer and fall indicate internal nutrient loading is occurring.

Although the aeration system does not appear to be operating as efficiently as it once did, it is probably still having a positive effect on the lake and limiting the internal loading of nutrients and consequential primary production. For example, spring total phosphorus concentrations have not decreased significantly over time, but there does appear to be a reduction in chlorophyll *a* concentrations and a slight increase in water clarity. Operation and regular maintenance of the aerator should continue to prevent degradation of the water quality of Langford Lake. It is also recommended that the diffuser be inspected by someone with the appropriate expertise to determine if replacement is now required.

9.2 Sewer Upgrades

In the past, the major input of phosphorus to Langford Lake was from agricultural land use in Hull's Field. This is no longer the case and it is very likely that the actual external phosphorus inputs have been greatly reduced in comparison to historical loadings.

However there could still be significant inputs from residential septic systems around the lake. The City of Langford has recently completed several upgrades to the sewer system in this area with more planned for the future. These plans should include all areas around Langford Lake to ensure all homes have the opportunity to switch from on-site sewage treatment. Although not mandatory for existing residences, homeowners should consider connecting to the centralized system to reduce impacts on Langford Lake.

9.3 Stormwater Treatment

The City of Langford has recently constructed two new detention ponds in Hull's Field which will treat stormwater prior to its inflow to Langford Lake. As with any detention pond, ongoing maintenance of the ponds is required to prevent any contaminants from being discharged to the lake. This may include plant removal at the end of the growing season to prevent decomposition and re-release of nutrients, and occasional dredging to the ponds to remove sediment-bound nutrients and contaminants.

Any new development should include provisions for stormwater treatment (e.g., oil/water separators, catch basins) to prevent contaminants from reaching Langford Lake or any other waterbody. In addition, stormwater treatment devices (including any existing devices) must be regularly maintained to ensure proper and effective operation.

9.4 Hypolimnetic Withdrawal

Langford Lake, like most lakes in B.C., is phosphorus-limited, which means that very small amounts of P can lead to significant increases in plant and algae growth. Langford Lake stratifies in the spring each year with the warmer, less dense water forming the top layer (epilimnion) and the colder, denser water forming the bottom layer (hypolimnion). As the season progresses, organic material (algae etc.) sinks and decomposes in the hypolimnion - a process which consumes oxygen. Because the well oxygenated epilimnion is physically separated from the hypolimnion, the bottom waters can become anoxic. This presents two potential problems:

- In anoxic conditions, P is released from the sediments - this is called internal loading (as opposed to external loading from runoff, precipitation etc.). When the

lake's water column mixes in the fall, the additional P is distributed throughout the lake and there is more available for algal growth.

- The reduced oxygen levels also limit fish habitat, especially the trout. The warm epilimnion increases their metabolism, but the cooler hypolimnion doesn't have enough oxygen, so the range of suitable habitat becomes limited. In extreme cases, fish kills can occur.

In addition to P, ammonia and other compounds (e.g., methylmercury) are also released under anoxic conditions. Langford Lake is flushed about every three years, which means that most of the P stays within the systems and is recycled.

Langford Lake typically exceeds the water quality guidelines (measured at spring overturn) for aquatic life (5 $\mu g \cdot L^{-1}$ - 15 $\mu g \cdot L^{-1}$) and recreational primary contact (10 $\mu g \cdot L^{-1}$) – in 2006 it was about 33 $\mu g \cdot L^{-1}$. During 2005, hypolimnetic total P concentrations increased at the deep site throughout the summer and fall, peaking in November at 560 $\mu g \cdot L^{-1}$. Most of the phosphorus in Langford Lake is from past land use activities (e.g. agriculture) and in order to improve the current level of water quality the excess phosphorus needs to be removed from the system.

The City of Langford has considered water withdrawal from Langford Lake for irrigation purposes on playing fields and other vegetated areas. This would provide an excellent opportunity to remove some of this excess phosphorus through withdrawal of water from the hypolimnion. The key would be to withdraw water from the deep part of the lake at a time when internal loading is occurring (typically July - September). The high nutrient content of the water at this time would, at least partially, offset the need for any application of synthetic fertilizers. Ministry of Environment staff should be consulted at the design stage if this project is to proceed to ensure all environmental and limnological concerns are addressed.

Hypolimnetic withdrawal as a restoration technique has generally been very successful and the resulting improvements are potentially long-term and even permanent (Nurnberg 1987). The benefits for Langford Lake include reduced algae and nuisance plant growth,

improved water clarity, and improved fish habitat through increased dissolved oxygen and a healthier plankton community. This should also increase the effectiveness of the aerator and may even eliminate the need for it in the future.

There are considerations which must be taken into account with hypolimnetic withdrawal. Care must be taken not to withdraw too much water too quickly; this could lead to early destratification and enhanced nutrient flow from the hypolimnion to the surface, possibly resulting in algae blooms (Nurnberg 1987). In the case of Langford Lake, there will also be measures required to prevent re-introduction of the removed phosphorus (and other contaminants) back to the lake via the inflow. Irrigation rates would have to be such that there is no resulting surface runoff back to Langford Lake.

9.5 Increased Flushing

Another option to increase nutrient removal would be to increase the flushing rate of the lake. This could be achieved by increasing the flow rate of the outflow at the north end. The outflow has been periodically cleaned of vegetation in the past but re-growth seems to occur quite quickly; more frequent clearing would help maintain higher flow rates increasing flushing rates and nutrient removal during fall freshet.

9.6 Local Stewardship and Public Awareness

The Langford Lake Area Protection Society (LLAPS) provides local stewardship for this area, working with residents, other societies and all levels of government to help protect the environmental and social qualities of Langford Lake. Through regular meetings and their website (http://members.shaw.ca/wabbit/LLAPS/home.html), LLAPS helps raise public awareness on measures that can be taken to help protect the water quality of Langford Lake. Additional information for lakeshore residents is available through The Living By Water Project (http://livingbywater.ca/main.html). LLAPS also conducts volunteer lake monitoring and provides this information on their website. The efforts of LLAPS should continue to be supported through the BC Lake Stewardship Society.

10.0 CONCLUSIONS

Human land use and development (agriculture, septics and urban stormwater runoff) have contributed to the eutrophication of Langford Lake over time. The high level of productivity results in conditions within the lake that promote the ongoing cycling of nutrients, particularly phosphorus. Despite this, Langford Lake continues to provide many recreational opportunities including fishing, swimming, boating and hiking. Efforts to limit the impacts on water quality include the operation of an aerator since the mid-1980's and the local stewardship activities of the Langford Lake Area Protection Society.

Langford Lake began to show signs of nutrient enrichment in the 1960's and studies resulted in an aerator being installed in 1984. Based on the 2005 temperature profiles, the aerator (which was originally designed to destratify the water column and increase dissolved oxygen concentrations) appears to have lost its effectiveness over the years. The temperature profiles from 2005 are very similar to those for pre-aeration conditions, with a thermocline established at 7 m to 8 m. The hypolimnetic summer water temperatures were well within the proposed water quality objective of 15° C, however the dissolved oxygen concentrations were very low in the summer months and did not meet the proposed water quality guideline of 5 $mg \cdot L^{-1}$ from June to October 2005. This could create unfavourable conditions for trout if the surface waters become too warm and the trout are forced into deeper and cooler, yet unoxygenated, hypolimnetic waters.

The water chemistry parameters of concern in Langford Lake are phosphorus and nitrogen. Spring overturn P concentrations typically exceed the water quality guidelines to protect recreational activities and aquatic life. In 2005, a maximum hypolimnetic total phosphorus concentration of 560 $\mu g \cdot L^{-1}$ was measured in November demonstrating the ongoing problem of internal nutrient loading in Langford Lake. Hypolimnetic total nitrogen concentrations also peaked in November 2005 at 2,300 $\mu g \cdot L^{-1}$, however other forms of N (e.g., ammonia and nitrate) were not found in excess. The increasing fall total phosphorus and total nitrogen concentrations and the resulting change in N:P ratio resulted in conditions more favourable for blooms of cyanobacteria (blue-green algae).

Other water chemistry parameters measured in this study were low and do not suggest any problems at this time.

The aquatic plant community has changed over the past 25 years with a shift in dominance from *Ceratophyllum demersum* to *Potamogeton amplifolius*. This shift may be due to an increase in water clarity which is demonstrated in reduced chlorophyll *a* concentrations and increases (although statistically insignificant) in Secchi depth. Prior to aeration, the phytoplankton community was dominated by eutrophic diatoms in the winter and eutrophic cyanophytes throughout the rest of the year. In 2005, there was a more diverse community in the spring and summer months consisting of green algae, diatoms and cyanophytes, and a more cyanophyte-dominated community in the fall and winter. Based on these observations, it appears the water quality of Langford Lake has improved since the early 1980's. Analysis of a sediment core from Langford Lake provided evidence for a shift in trophic conditions around 1960 with the eutrophic diatom *Aulacoseira ambigua* becoming dominant from about 1970 on.

The zooplankton community is indicative of eutrophic conditions; it is dominated by small grazing crustaceans and rotifers and species richness is low for the copepod and cladoceran components, however biomass and density is relatively high.

Microbiological contamination at public beaches on Langford Lake does not appear to be an issue based on available monitoring data.

While the overall eutrophic condition of Langford Lake does not seem to be degrading at the present time this is likely the result of the ongoing operation of the aerator. Phosphorus accumulations in the sediments are recycled within the lake during periods of low dissolved oxygen concentrations in the hypolimion and contribute to the high level of productivity. To address this problem, phosphorus must be removed from the system and options available include hypolimnetic withdrawal and increased flushing through periodic vegetation removal at the outlet. Continuing operation and maintenance of the aerator will help limit conditions (i.e., low dissolved oxygen) that promote internal

nutrient loading; however, an assessment of the system by a qualified specialist would help determine if an upgrade or replacement is necessary. Proper stormwater management (including maintenance) is necessary to limit contaminant inputs to the lakes and should be part of any new development plan. Lakeshore residents can also help limit nutrient inputs to the lake by maintaining their septic systems and, if possible, switching from septic to centralized sewer service and limiting their use of fertilizers for lawns and gardens.

Increasing urban development within the Langford Lake watershed will present challenges in protecting and improving water quality in the future; however, improvements seen over the past 25 years demonstrate it is possible. Continuing efforts to protect the water quality of Langford Lake will help protect property values around the lake; protect investments in the lake, such as the annual stocking of trout for the recreational fishery; and protect the recreational value of the lake for the many users of this popular urban lake.

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TABLE 1
LANGFORD LAKE WATER QUALITY SAMPLING SITES

Site Location	Site Number	EMS Number	Site Name	Depth (m)	Latitude	Longitude
Lake	1	1100944	Southeast half, #3	14	48.4474	123.5240
Lake	2	1100953	Langford Lake Centre	15	48.4489	123.5308
Lake	3	E257395	Langford Lake #10	12	48.4478	123.5270
Lake	4	E257396	Langford Lake #9	10	44.4483	123.5305
Inflow	5	E234410	Langford Lake, #1 culvert from Hull's field		48.4455	123.5277
Outflow	6	E234414	Langford Lake outlet creek		48.4511	123.5383

TABLE 2
LANGFORD LAKE TOTAL PHOSPHORUS CONCENTRATIONS ($\mu g \cdot L^{-1}$)

	1100944		1100953		E257395		E257396	
Date	Surface	Bottom	Surface	Bottom	Surface	Bottom	Surface	Bottom
09-Nov-04	24	37			22	38	25	23
24-Jan-05	41	41	38	43	38	40	37	39
03-Mar-05	19	19	16	59	27	24	20	34
12-Apr-05	17	29	25	32	20	17	15	21
18-May-05	11	35	12	42	12	31	14	37
15-Jun-05	18	24	18	270	16	78	20	120
21-Jul-05	9	34	8	190	8	68	11	150
16-Aug-05	7	122	4	200	6	85	8	193
21-Sep-05	4	209	5	450	6	256	5	37
13-Oct-05	7	207	8	422	10	223	7	11
15-Nov-05	28	38	26	560	25	25	24	28
06-Feb-06	31	23	37	38	36	41	15	43

Surface samples were taken at 0.1 m and bottom samples were taken with 1 m of the bottom at each site.

TABLE 3
LANGFORD LAKE (SITE 1100944) MEAN TOTAL PHOSPHORUS CONCENTRATIONS ($\mu g \cdot L^{-1}$)

Date	Mean Total P (µg/L)	n	Standard Error	Confidence Intervals	
				Lower 95%	Upper 95%
17-May-73	28	2	3	22	34
22-Jan-83	29	3	3	23	34
02-Apr-84	19	6	2	16	23
07-Feb-85	37	3	3	31	42
08-Jan-86	46	1	4	37	55
27-Feb-92	11	2	3	4	17
08-Mar-93	10	2	3	4	16
15-Mar-95	10	2	3	4	16
05-Mar-97	32	2	3	26	38
02-Mar-98	35	1	4	26	44
20-Mar-00	22	2	3	15	28
21-Mar-01	26	2	3	20	32
14-Mar-02	12	2	3	5	18
13-Mar-03	9	4	2	5	14
29-Jan-04	32	1	4	23	41
24-Jan-05	41	2	3	35	47
06-Feb-06	33	3	3	26	39

TABLE 4
LANGFORD LAKE ORTHOPHOSPHATE AND TOTAL PHOSPHORUS CONCENTRATIONS ($\mu g \cdot L^{-1}$) AT THE DEEP SITES

	Site 1100944						Site 1100953					
	Epilimnion		Metalimnion		Hypolimnion		Epilimnion		Metalimnion		Hypolimnion	
	OP	TP	OP	TP	OP	TP	OP	TP	OP	TP	OP	TP
24-Jan-05	27	41			29	41						
3-Mar-05	1	19	2	29	3	19						
12-Apr-05	<1	17	<1	27	3	29						
18-May-05							2	12	2	22	5	42
15-Jun-05							<1	18	<1	21	166	290
21-Jul-05							2	8	2	14	130	190
16-Aug-05							2	4	3	14	163	200
21-Sep-05							4	5	4	5	385	450
13-Oct-05							8	8	6	11	290	422
15-Nov-05							25	26	24	29	600	560
6-Feb-06	5	31	6	44	6	23						

TABLE 5
NUTRIENT CONCENTRATIONS FROM THE MAIN INFLOW AT HULL'S FIELD TO LANGFORD LAKE (SITE E234410)

Date	Total P ($\mu g \cdot L^{-1}$)	Total N ($\mu g \cdot L^{-1}$)
November 24, 1999	115	2,800
March 15, 2000	75	890
October 10, 2000	32	680
March 20, 2001	22	310
December 18, 2001	17	670
April 12, 2005	32	1,520

November 15, 2005	24	1,220
November 24, 2005	37	1,800
January 6, 2006	6	210

TABLE 6
LANGFORD LAKE TOTAL NITROGEN CONCENTRATIONS ($\mu g \cdot L^{-1}$)

	1100944		1100953		E257395		E257396	
Date	Surface	Bottom	Surface	Bottom	Surface	Bottom	Surface	Bottom
09-Nov-04	420	430			410	420	450	370
24-Jan-05	730	710	720	620	680	520	550	560
03-Mar-05	450	573	509	795	422	533	535	524
12-Apr-05	430	570	461	661	468	526	440	437
18-May-05	410	810	360	840	400	750	390	500
15-Jun-05	350	720	330	1,030	320	890	420	570
21-Jul-05	320	760	320	810	350	770	320	1,040
16-Aug-05	320	790	330	990	320	740	400	1,100
21-Sep-05	340	850	350	1,660	360	1,210	340	670
13-Oct-05	350	730	380	1,250	340	870	380	340
15-Nov-05	430	440	400	2,300	420	410	430	410
06-Feb-06	530	510	610	600	580	570	550	560

TABLE 7
LANGFORD LAKE AVERAGE CONCENTRATIONS FOR NITROGEN PARAMETERS ($\mu g \cdot L^{-1}$)

Parameter	Jan-05	Mar-05	Apr-05	May-05	Jun-05	Jul-05	Aug-05	Sep-05	Oct-05	Nov-05	Feb-06
Ammonia	85	20	49	115	310	150	710	1,220	290	630	34
Nitrate+Nitrite	230	155	123	311	24	42	<0.02	3	13	36	212
Organic N	410	330	300	310	340	290	320	380	420	380	270
Kjeldahl N	490	340	350	430	540	450	550	800	700	1,010	310

Samples are an average of measurements taken at the top, middle and bottom of the water column, except for January 2005 which was only measured at the top and bottom. Samples were taken at 1100944 if the aerator was not operating (December – March) or 1100953 if the aerator was operating (April – November).

TABLE 8
DOMINANT PHYTOPLANKTON SPECIES FOR LANGFORD LAKE

Date	Total standing crop (cells/mL)	Chlorophyll a (ug/L)	Dominant Species		% of total	cells/mL
			Order	Species		
24-Jan-05	935.2	5.9	**Cryptomonadales**	*Chroomonas acuta*	10	94
				Cryptomonas sp.	13	122
			Nostocales	*Anabaena cf flos-aquae*	45	421
			Pennales	*Tabellaria fenestrata*	18	168
3-Mar-05	2996	16.6	**Cryptomonadales**	*Chroomonas acuta*	13	389
			Pennales	*Asterionella formosa*	35	1049
				Tabellaria fenestrata	31	929
18-May-05	442.4	3	**Cryptomonadales**	*Chroomonas acuta*	35	155
				Cryptomonas sp.	11	49
			Nostocales	*Anabaena cf flos-aquae*	33	146
15-Jun-05	no sample taken	3.4				
21-Jul-05	589	1.5	**Chlorococcales**	*Sphaerocystis schroeteri*	52	306
			Chroococcales	*Anacystis limneticus*	10	59
			Cryptomonadales	*Chroomonas acuta*	14	82
			Pennales	*Tabellaria fenestrata*	10	59
16-Aug-05	658	2	**Chlorococcales**	*Sphaerocystis schroeteri*	10	66
			Chroococcales	*Anacystis limneticus*	14	92
			Cryptomonadales	*Chroomonas acuta*	12	79
			Ochromondales	*Dinobryon divergens*	11	72
			Tetrasporales	*Gloeocystis ampla*	51	336
21-Sep-05	1156.4	2	**Cryptomonadales**	*Chroomonas acuta*	20	231
			Nostocales	*Aphanizomenon flos-aquae*	24	278
			Pennales	*Tabellaria fenestrata*	38	439
13-Oct-05	1943.7	3.4	**Nostocales**	*Aphanizomenon flos-aquae*	82	1594
15-Nov-05	562.8		**Centrales**	*Melosira italica*	21	118
			Nostocales	*Aphanizomenon flos-aquae*	63	355
6-Feb-06	876.4		**Centrales**	*Stephanodiscus niagarae*	16	143
			Chlorococcales	*Sphaerocystis schroeteri*	10	90
			Cryptomonadales	*Chroomonas acuta*	15	129
			Nostocales	*Anabaena cf flos-aquae*	10	84
			Pennales	*Fragilaria crotonensis*	40	347

TABLE 9
CHOROPHYLL *A* CONCENTRATIONS ($\mu g \cdot L^{-1}$) FOR LANGFORD LAKE

Date		1980	1984	1985	2005
January	24				5.9
March	3				16.6
May	8		5.8		
	9			3.1	
	15	5.0			
	18				3.0
	29		4.0		
June	5	5.3			
	15				3.4
	27		11.4		
July	3			11.4	
	10		36.2		
	16	12.5			
	21				1.5
	30		10.0		
August	7			9.5	
	8	6.3			
	13		6.5		
	16				2.0
	26		24.4		
	27	2.0			
	29			10.8	
September	21				2.0
October	1		11.8		
	2	37.0			
	13				3.4
	27			8.3	
November	13	4.3			

TABLE 10
ZOOPLANKTON SPECIES FOR LANGFORD LAKE
(total organisms per sample)

	Organism	Stage	24-Jan-05	3-Mar-05	18-May-05	21-Jul-05	16-Aug-05	21-Sep-05	13-Oct-06	15-Nov-05	6-Feb-06
Sub-class:	**Copepoda**										
	Nauplii		9,850.0	4,200.0	34,000.0	8,866.7	4,333.3	9,066.7	4,866.7	12,933.3	6,333.3
Order:	**Cyclopoida**										
	Diacyclops thomasi	adult	400.0	320.0	933.3	320.0	170.0	55.5	220.0	380.0	140.0
		copepodid	1,250.0	4,666.7	23,200.0	1,000.0	6,066.7	2,133.3	866.7	2,600.0	4,466.7
	Eucyclops cf agilis	adult	2.0						1.0	2.0	2.0
		copepodid							1.0	10.0	
	Macrocyclops sp.	adult									2.0
	Orthocyclops modestus	adult	1.0			2.0	10.0	180.0		1.0	
		copepodid					6.0	290.0		30.0	
	Unidentified	copepodid				1,200.0				1.0	
Order:	**Calanoida**										
	Diaptomus oregonensis	adult	78.8	57.1	100.0	3,533.3	200.0	200.0	95.0	260.0	520.0
	Diaptomus oregonensis	copepodid			240.0	800.0	1,333.3	1,600.0	1,000.0	1,000.0	
	Epischura nevadensis	adult					2.0	2.0			
Order:	**Harpacticoida**										3.0
Order:	**Cladocera**										
	Acroperus harpae										2.0
	Alona sp.		3.0		2.0						2.0
	Bosmina longirostris		18.0	4.0		95.0	135.6	80.0	2.0	2.0	4.0
	Ceriodaphnia achanthina					1,133.3			840.0	1,400.0	
	Ceriodaphnia sp.		45.0	2.0			1,333.3	1,126.7		80.0	200.0
	Chydorus sp.		3.0	1.0	200.0	6.0		11.1	4.0	4.0	10.0
	Daphnia pulex		600.0	866.7		1,466.7					240.0
	Daphnia pulex/pulcaria			40.0	2,800.0	666.7	2,333.3	4,000.0			1,666.7
	Daphnia pulicaria					800.0			1,533.3	733.3	
	Daphnia rosea		240.0	80.0	40.0	80.0	133.3	5,066.7	4,533.3	1,466.7	266.7
	Daphnia sp.		120.0	333.3							133.3
	Daphnia sp.	juvenile			933.3		666.7	2,266.7	866.7	600.0	
	Diaphanosoma brachyurum					70.0	80.0				
	Diaphanosoma sp.							165.6	24.0		
	Eurycercus lamellatus										2.0
Family:	**Chydoridae**										
	Chydorus sp.						14.3				
Order:	**Ostracoda**						1.0				
Phylum:	**Rotifera**										
	Ascomorpha sp.			66.7		266.7	266.7	533.3	133.3		
	Asplanchna sp.		2.0	2.0				2.0	10,000.0	10.0	20.0
	Brachionus patulatus						40.0			10.0	
	Callotheca sp.							133.3	66.7	133.3	600.0
	Conochilus sp.					12,400.0		11,200.0	66.7		
	Euchalnis sp.					266.7	40.0		600.0		
	Gastropus sp.						133.3		66.7		
	Hexarthra sp.					1,666.7	1,333.3	1,333.3	2,066.7	66.7	
	Kellicotia bostoniensis								66.7	66.7	266.7
	Kellicotia longispina		50.0	333.3	1,666.7	5,533.3	1,666.7	666.7			66.7
	Keratella cochlearis		350.0	333.3	4,100.0	3,800.0	333.3	20,400.0	5,133.3	3,133.3	16,000.0
	Keratella quadrata		50.0	866.7	7,700.0	13,133.3	733.3	1,066.7	200.0	2,000.0	10,900.0
	Lecane sp.								66.7		
	Macrochaetus longipes							40.0			
	Notholca cf. acuminata			66.7							400.0
	Notholca squamula			400.0							133.3
	Notholco sp.				133.3						
	Platyas quadricornis					2.0					
	Ploesoma sp.					266.7		133.3			
	Polyarthra sp.		150.0	2,000.0	266.7	2,333.3	1,533.3	8,666.7	6,600.0	17,933.3	1,133.3
	Pompholyx sp.				200.0						4,800.0
	Synchaeta sp.		800.0	20,600.0	400.0	666.7	80.0			466.7	3,933.3
	Trichocerca sp.					3,333.3	400.0	266.7	133.3	10.0	
	Unidentified		450.0	200.0	66.6	133.3		133.3	666.7	2,266.7	266.7
Order:	**Bdelloidea**			133.3						66.7	
Order:	**Chaoboridae**										
	Chaoborus sp.					30.0	20.0	9.0	1.0	1.0	
Order:	**Diptera**										
	Unidentified chironomid										10.0
Group:	**Hydracarina**				2.0	12.0	3.0	1.0			
Phylum:	**Nematoda**				2.0	12.0	3.0	1.0		66.7	66.7

TABLE 11
FECAL COLIFORM COUNTS (FECAL COLIFORMS per 100 mL) FOR LANGFORD LAKE

Date	Site		Geometric Mean
	Leigh Road	Goldstream	
April 11, 2005	1		
April 11, 2005		1	
April 19, 2005		1	
April 25, 2005		1	
April 26, 2005	1		
May 2, 2005	2		
May 2, 2005		1	
May 3, 2005	1		
May 9, 2005	14		
May 9, 2005		2	1
May 10, 2005	1		1
May 16, 2005	1		1
May 16, 2005		1	1
May 25, 2005		1	1
May 30, 2005		3	2
June 6, 2005		4	2
June 13, 2005	11		2
June 13, 2005		94	3
June 20, 2005		1	3
June 27, 2005		3	4
July 4, 2005	4		5
July 4, 2005		15	6
July 12, 2005		3	6
July 19, 2005		16	7
July 26, 2005		3	4
August 3, 2005	2		5
August 3, 2005		2	4
August 8, 2005	8		5
August 8, 2005		1	4
August 15, 2005		3	3
August 22, 2005		25	4

TABLE 12
WATER QUALITY OBJECTIVES FOR LANGFORD LAKE

Site	1100944
Designated Water Uses	Recreation (primary contact), aquatic life
Characteristics	
Temperature	≤ 15° C (summer maximum hypolimnetic temperature)
Dissolved Oxygen	≥ 5 mg•L^{-1} (at any depth throughout the year)
Secchi Depth[1]	≥ 4 m (annual mean)
Total Phosphorus (short-term : 5 – 20 years)[2]	≤ 20 µg•L^{-1} at spring overturn
Total Phosphorus (long-term: > 20 years)[2]	≤ 10 µg•L^{-1} at spring overturn
Total Nitrogen[2]	≤ 500 µg•L^{-1} at spring overturn
N:P Ratio[3]	≥ 20:1
Phytoplankton community	> 50% non-cyannophyte species (cells•mL^{-1})
Chlorophyll *a*[4]	≥ 1.5 µg•L^{-1} ≤ 2.5 µg•L^{-1}
Fecal coliforms[5]	≤ 200 •100 mL^{-1} (geometric mean)

[1]Annual means are calculated from a minimum of 4 seasonal measurements.

[2]This objective applies to the average of at least three samples taken throughout the water column (surface, mid depth, one metre above bottom).

[3]The N:P ratio is calculated using average total nitrogen and total phophorus concentrations.

[4]Values are to be growing season averages for epilimnetic water in the main basin of the lake.

[5]Geometric means are calculated from at least 5 weekly samples in a 30 day period.

TABLE 13
RECOMMENDED WATER QUALITY MONITORING FOR LANGFORD LAKE

Site	Timing	Depth	Parameters
1100944	Spring overturn (preferably before February 15)	Surface, mid-depth, bottom (1 m above bottom)	**Nutrients:** total P, dissolved P, total N, NO_3+NO_2, NO_2, total organic N, ammonia, dissolved organic C, total organic C, total inorganic C, chlorophyll *a* **Total metals** (1) **Anions:** Anion package (Cl, Br, SO_4) **Physical properties:** conductivity, pH, total solids, total dissolved solids, turbidity **General:** alkalinity, true colour, silica **Field measurements:** DO (profile), temperature (profile), Secchi depth **Biological:** phytoplankton (2), zooplankton (3), chlorophyll *a*
E234410		Surface	**Nutrients:** total P, dissolved P, total N, NO_3+NO_2, NO_2, total organic N, ammonia, dissolved organic C, total organic C, total inorganic C **Total metals** (1) **Anions:** Anion package (Cl, Br, SO_4) **Physical properties:** conductivity, pH, total solids, total dissolved solids **General:** alkalinity, true colour, silica

1. Low level metals package (ICPMS) is recommended.
2. Surface (0.5m) unconcentrated 1 L sample preserved with Lugol's solution.
3. Vertical haul from 10 m to surface. Preserved in 5% formalin. Mouth size of net must be recorded.

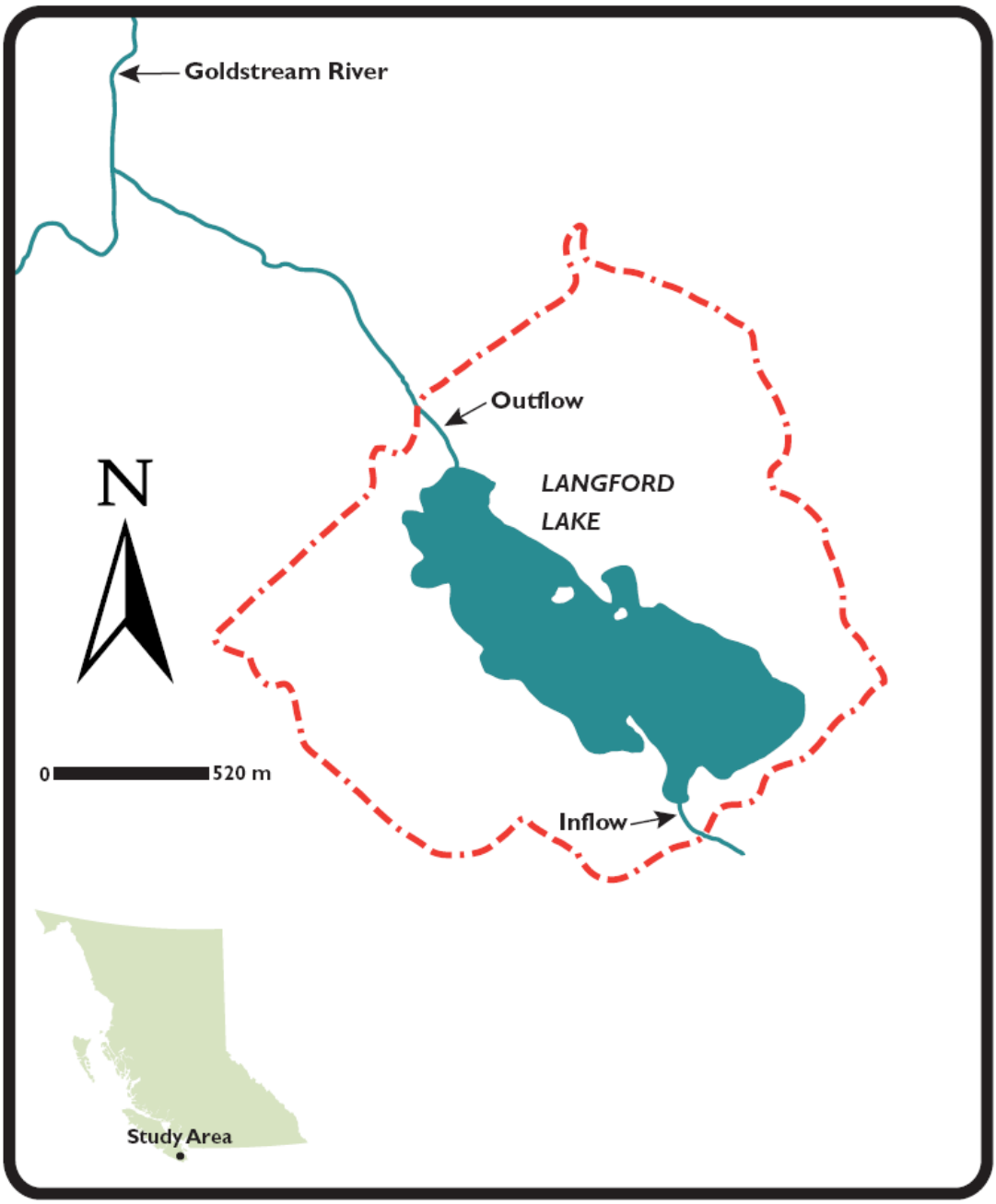

Figure 1: Location of Langford Lake and watershed boundaries.

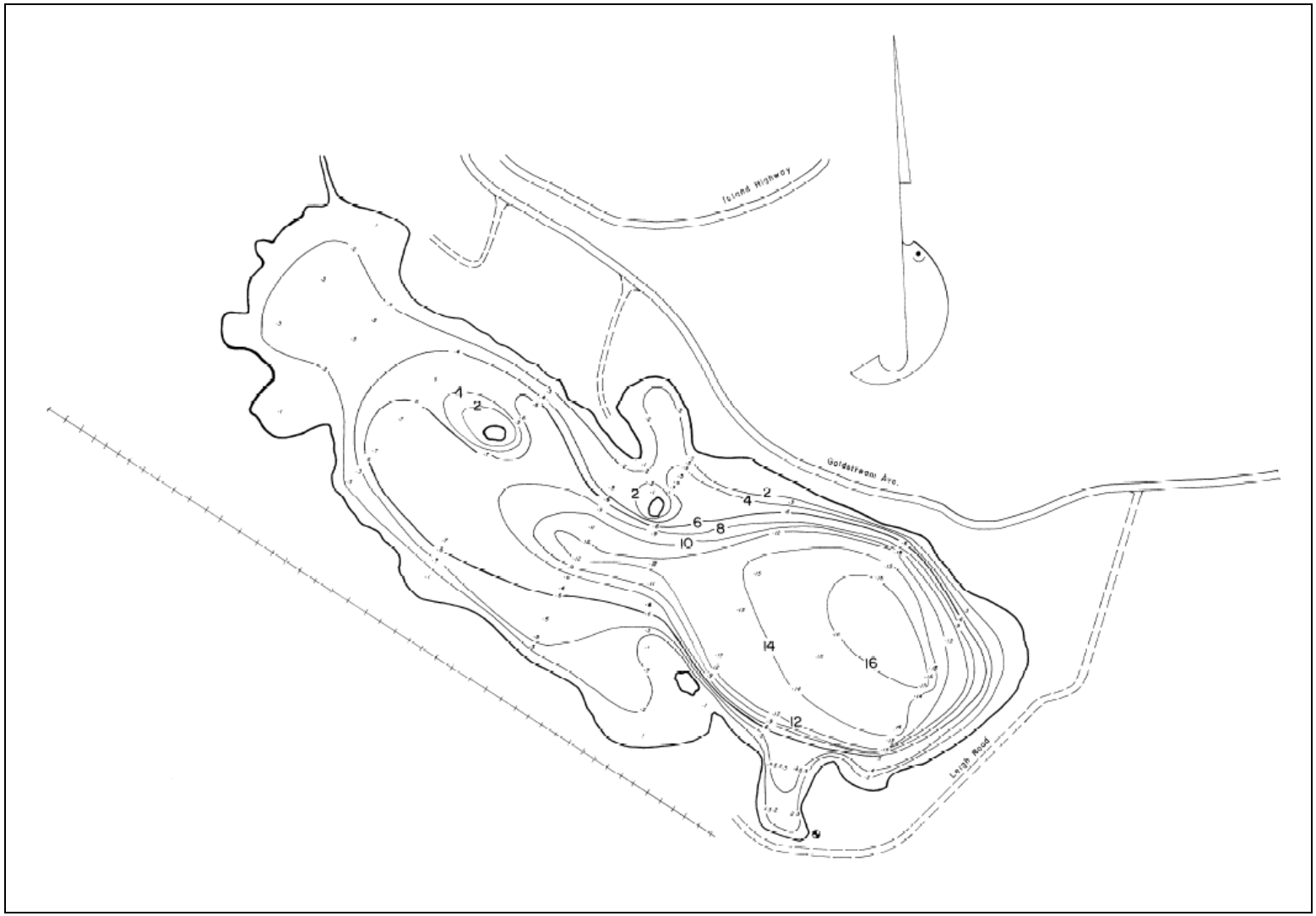

Figure 2: Bathymetric map of Langford Lake.

Figure 3: Location of Langford Lake sampling sites:

- 1: 1100944 Langford Lake SE half;
- 2: 1100953 Langford Lake Center;
- 3: E257395 Langford #10
- 4: E257396 Langford #9
- 5: E234410 Langford Lake Site 1
- 6: E234414 Langford Lake Outlet

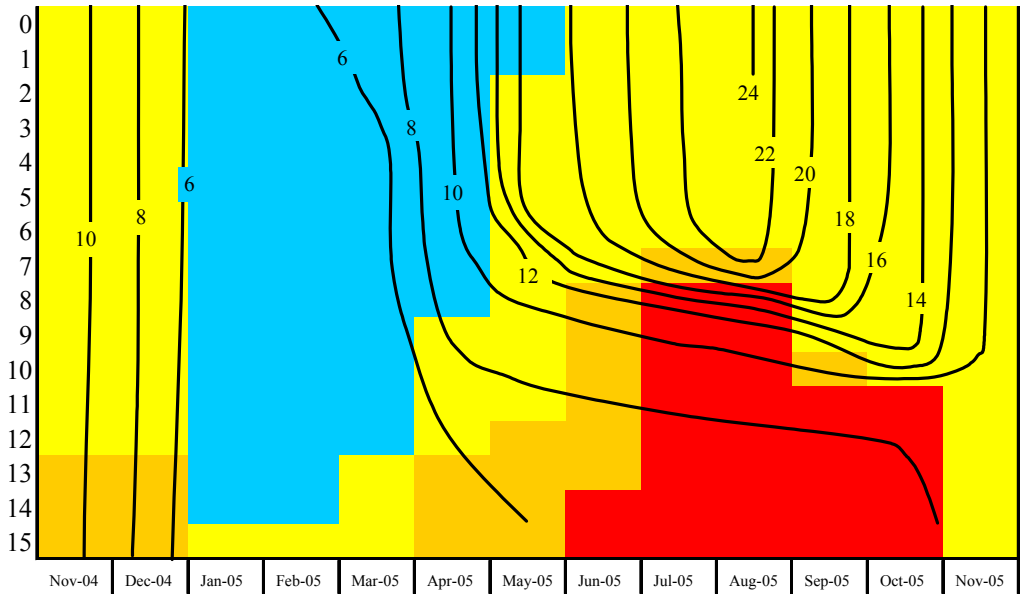

Figure 4: Site 1100944 Time/depth profiles for temperature and dissolved oxygen. Isotherms represent temperature (ºC) and colours represent dissolved oxygen (mg/L): < 1 mg/L DO 1 - 5 mg/L DO 5 - 10 mg/L DO >10 mg/L DO

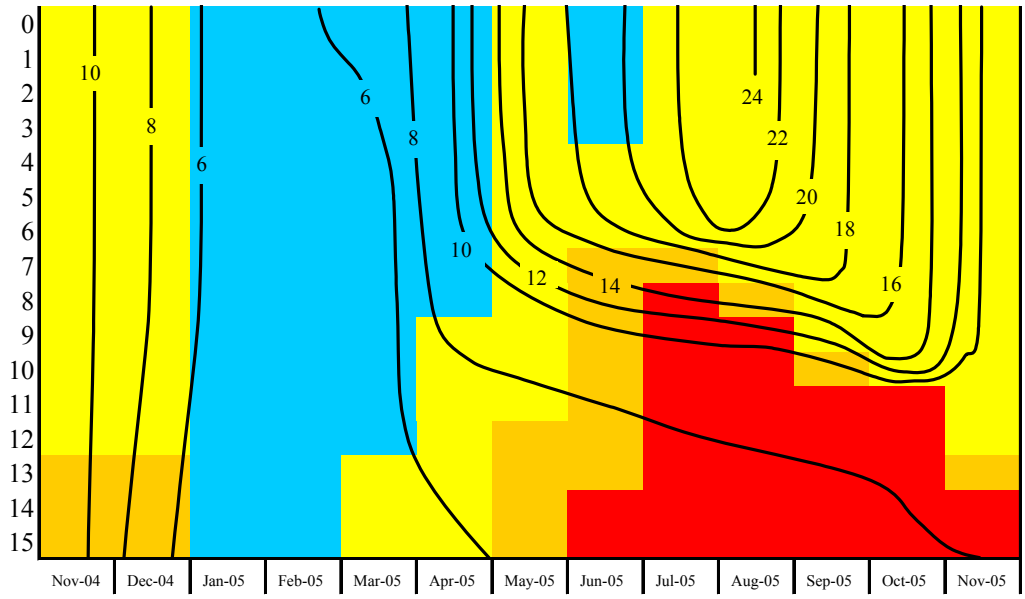

Figure 5: Site 1100953 Time/depth profiles for temperature and dissolved oxygen. Isotherms represent temperature (ºC) and colours represent dissolved oxygen (mg/L): < 1 mg/L DO 1 - 5 mg/L DO 5 - 10 mg/L DO >10 mg/L DO

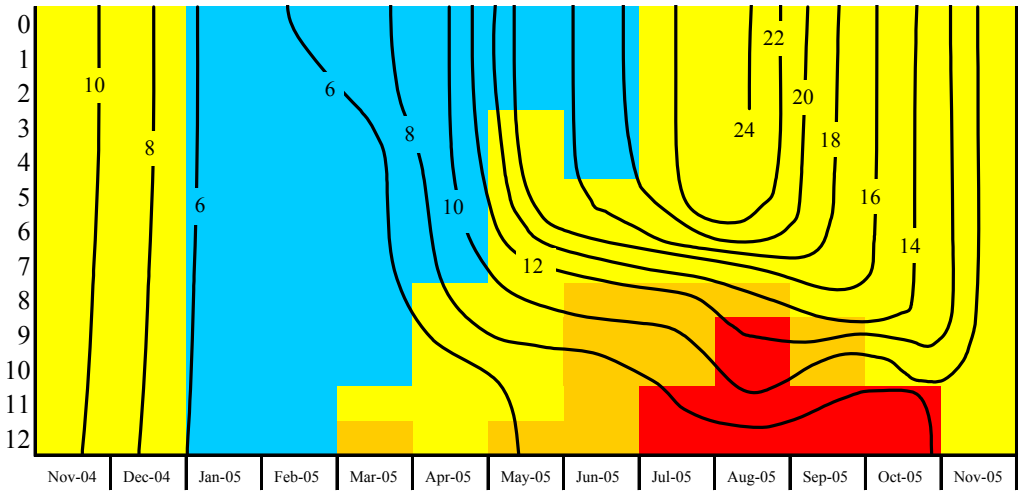

Figure 6: Site E257395 Time/depth profiles for temperature and dissolved oxygen. Isotherms represent temperature (ºC) and colours represent dissolved oxygen (mg/L):
< 1 mg/L DO 1 - 5 mg/L DO 5 - 10 mg/L DO >10 mg/L DO

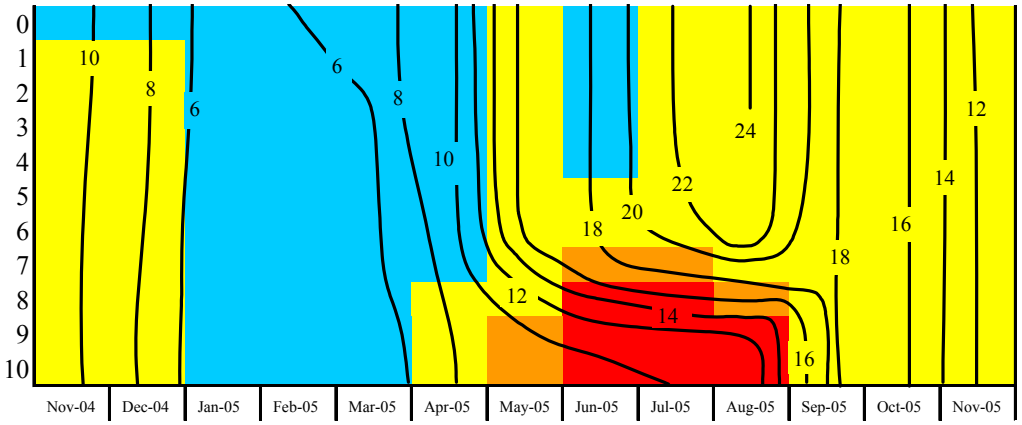

Figure 7: Site E257396 Time/depth profiles for temperature and dissolved oxygen. Isotherms represent temperature (ºC) and colours represent dissolved oxygen (mg/L):
< 1 mg/L DO 1 - 5 mg/L DO 5 - 10 mg/L DO >10 mg/L DO

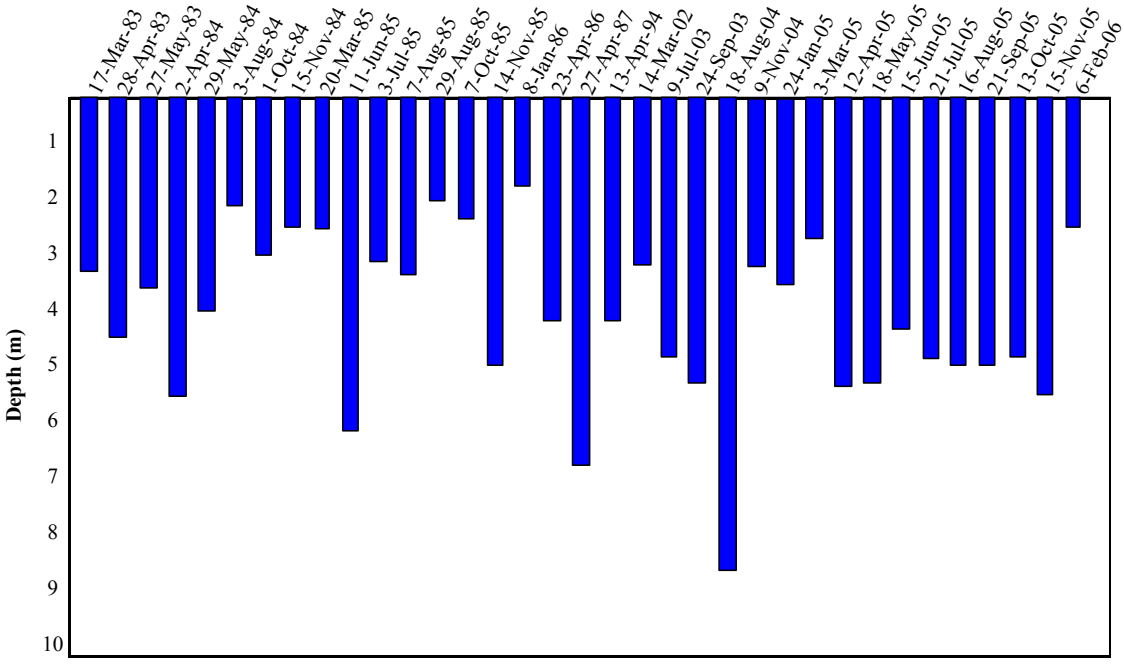

Figure 8: Average secchi depths (m) for Langford Lake over time.

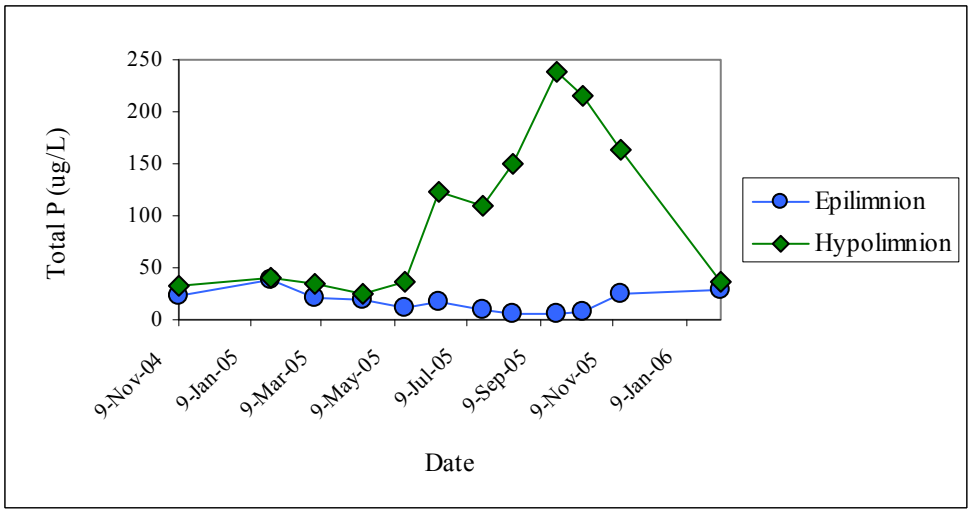

Figure 9: Mean total phosphorus concentrations for Langford Lake (November 2004 – February, 2006).

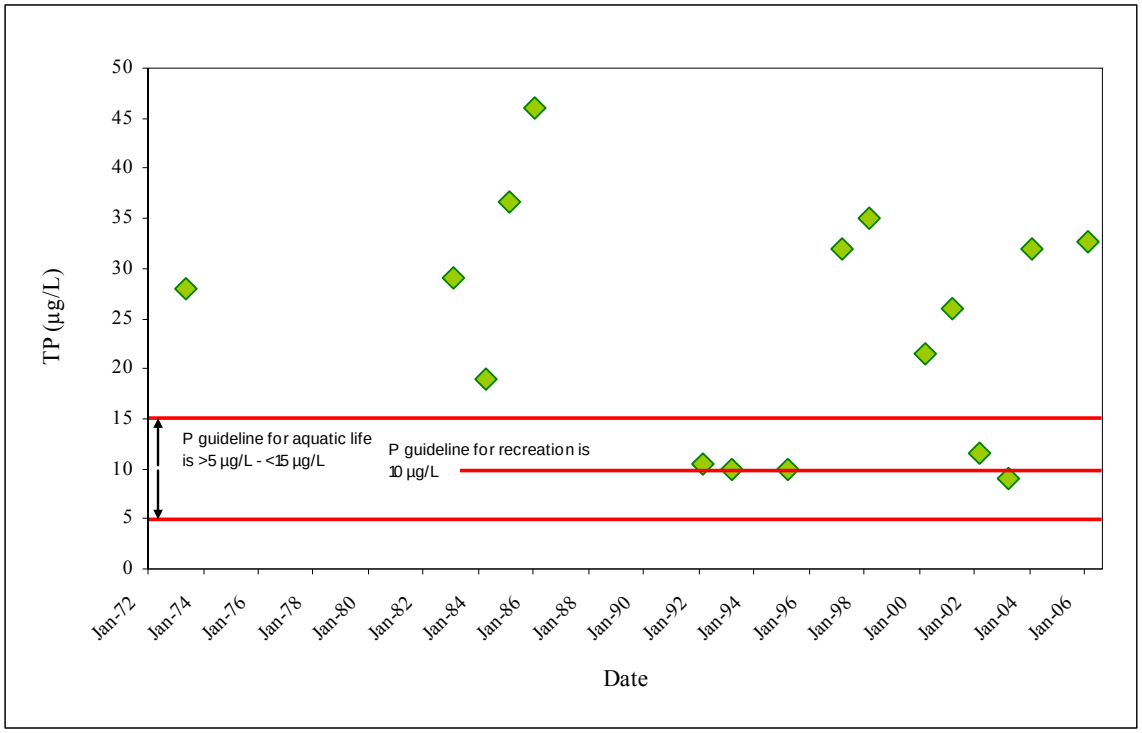

Figure 10: Average spring overturn total phosphorus concentrations ($\mu g \cdot L^{-1}$) at site 1100944.

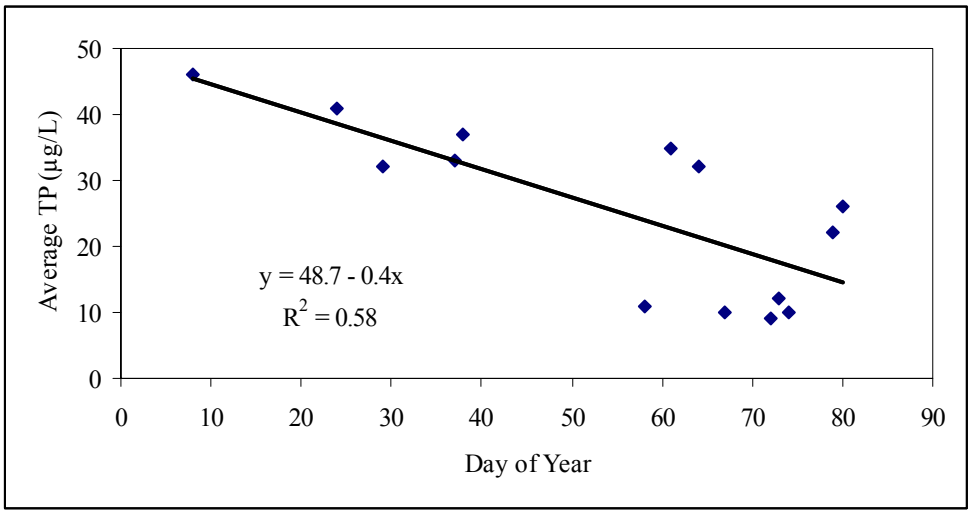

Figure 11: Average total phosphorus as a function of sampling date.

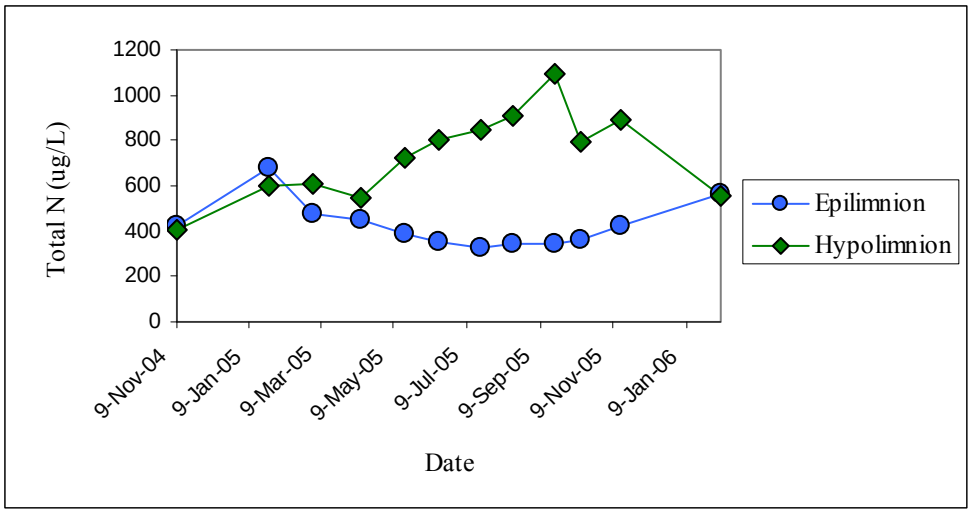

Figure 12: Average total nitrogen concentrations for Langford Lake (all sites)

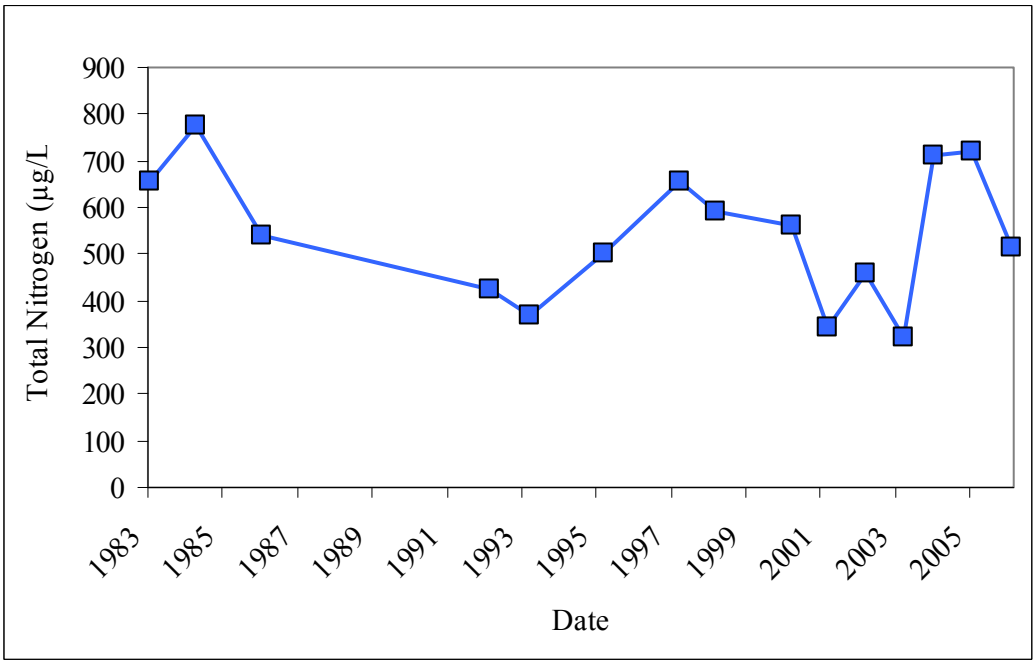

Figure 13: Average spring overturn total nitrogen concentrations ($\mu g \cdot L^{-1}$) for Langford Lake (site 1100944).

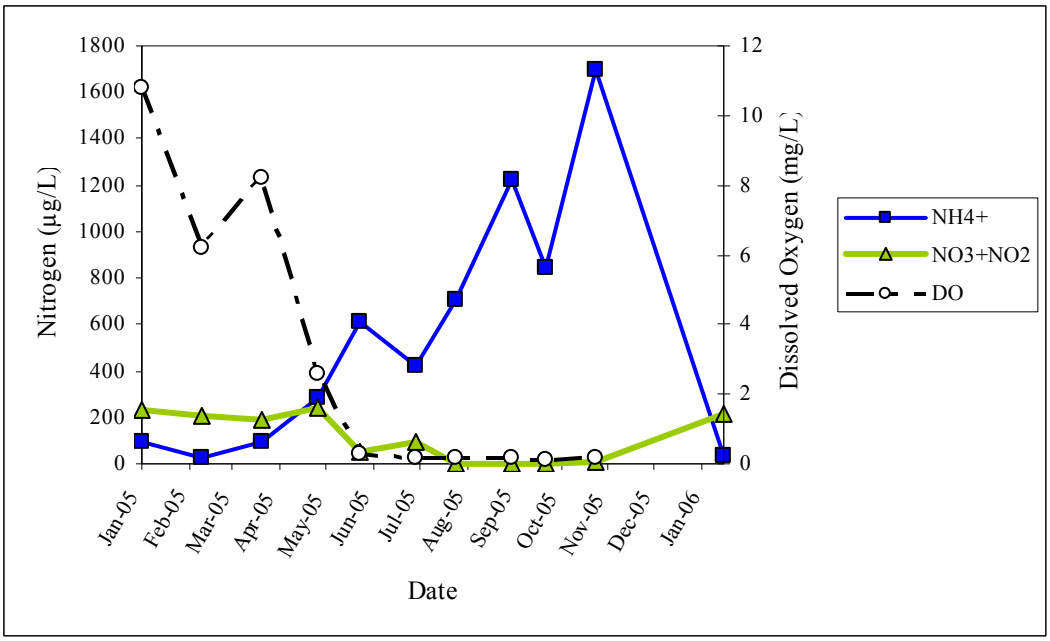

Figure 14: Hypolimnetic ammonia, nitrate+nitrite and dissolved oxygen concentrations at the deep stations (1100944, 110953) of Langford Lake.

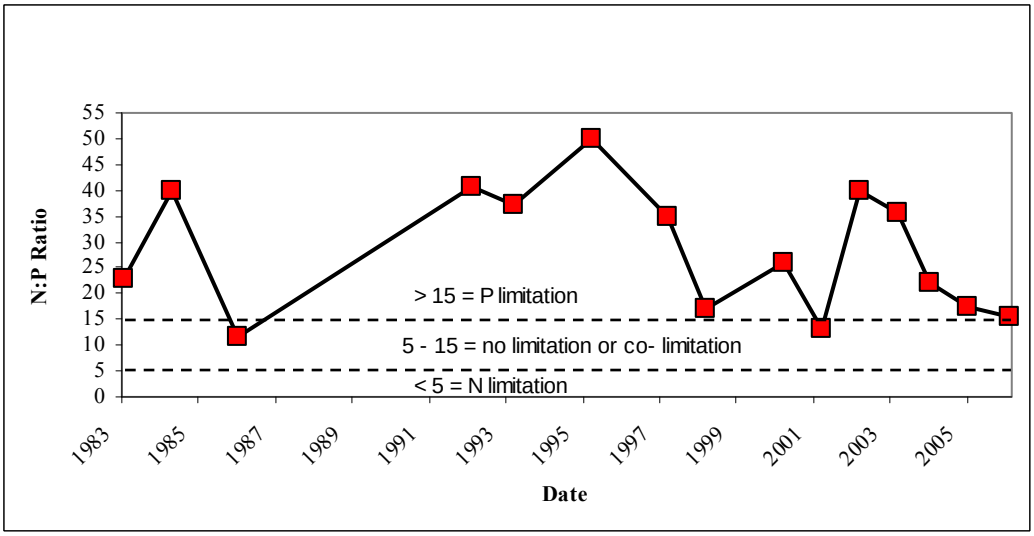

Figure 15: N:P ratios at spring overturn for Langford Lake (site 119044).

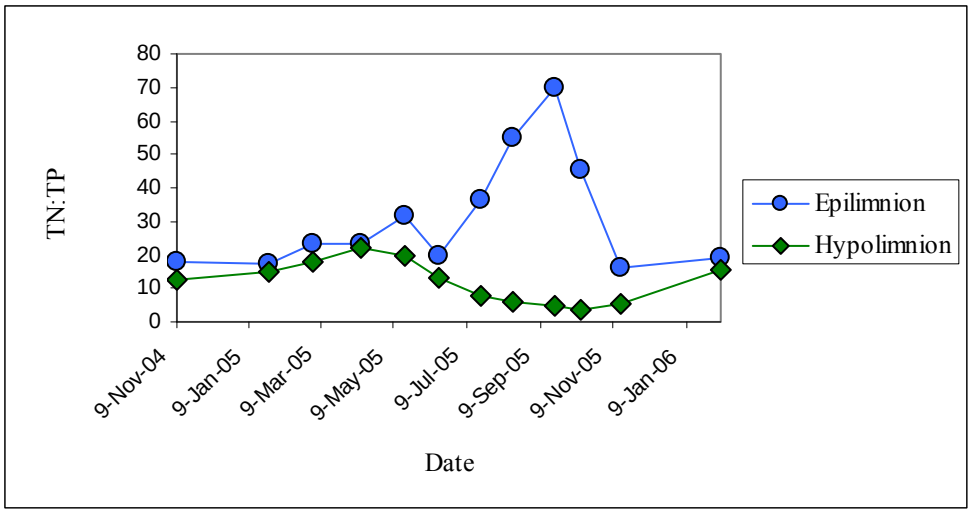

Figure 16: Average epilimnetic and hypolimnetic N:P ratios Langford Lake.

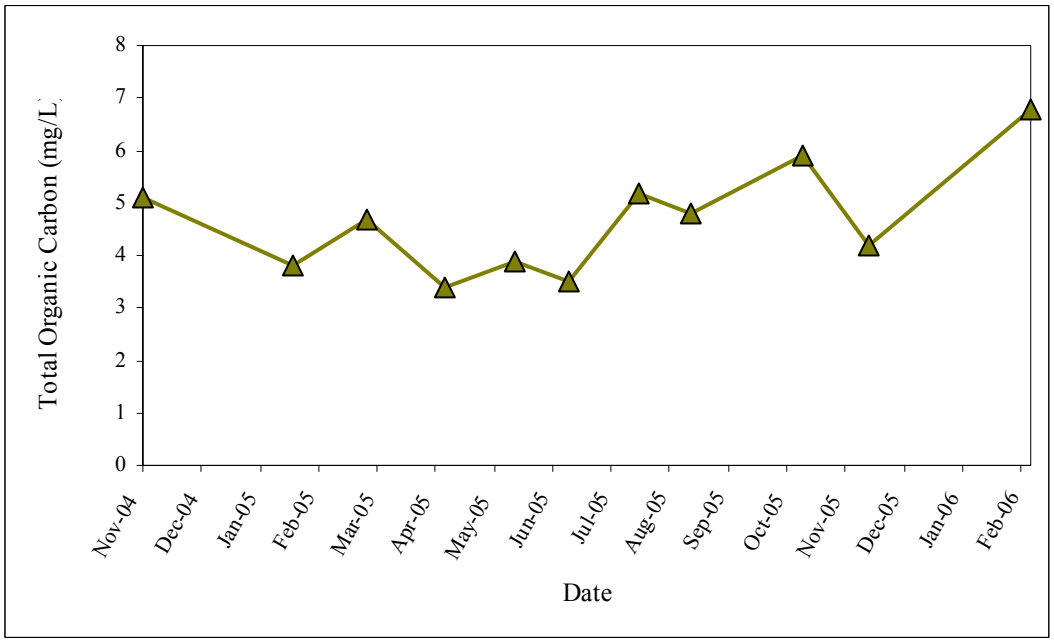

Figure 17: Total organic carbon concentrations ($\mu g \cdot L^{-1}$) for Langford Lake (sites 1100944 and 1100953).

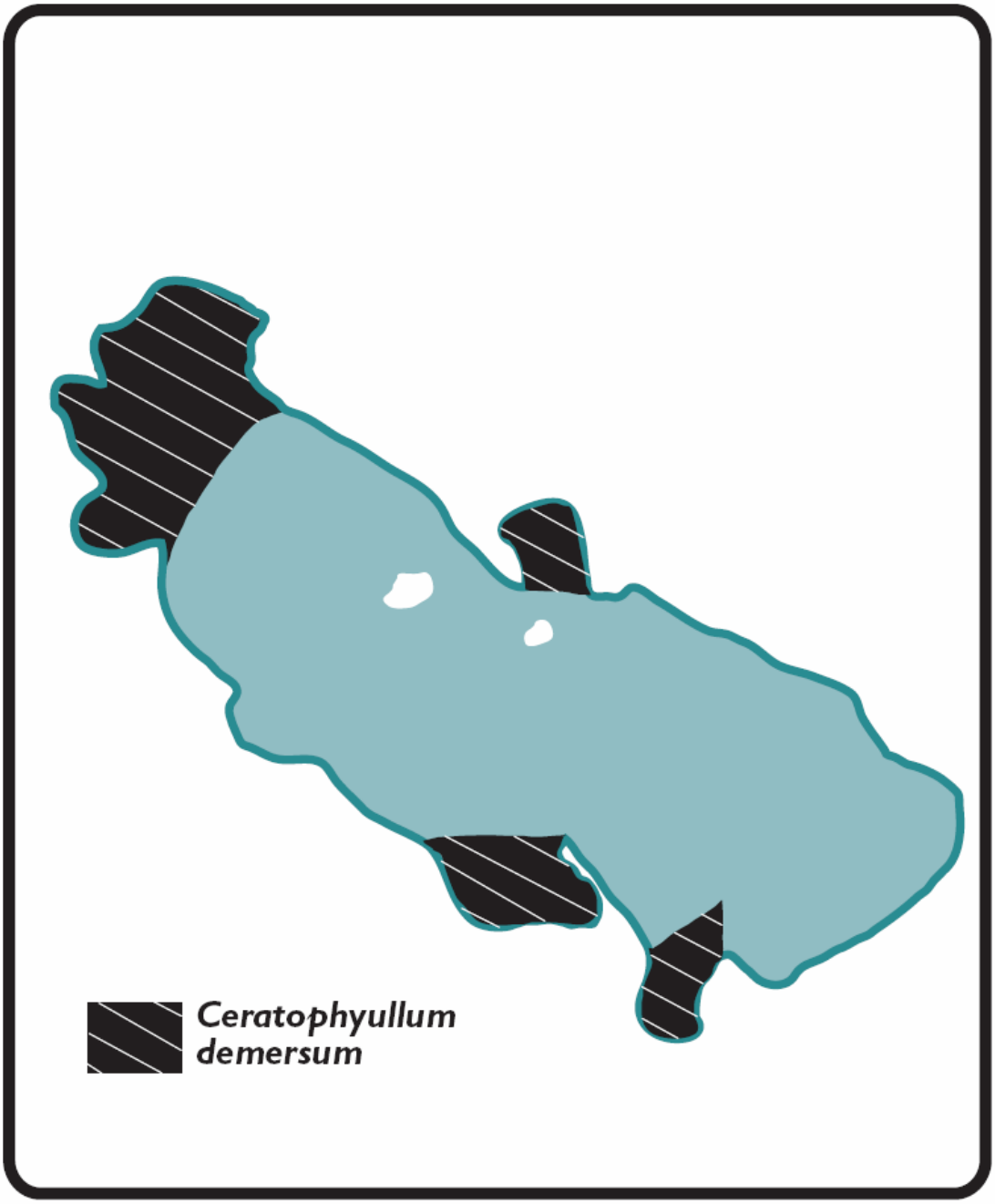

Figure 18: Distribution of *Ceratophyllum demersum* in Langford Lake – 1984.

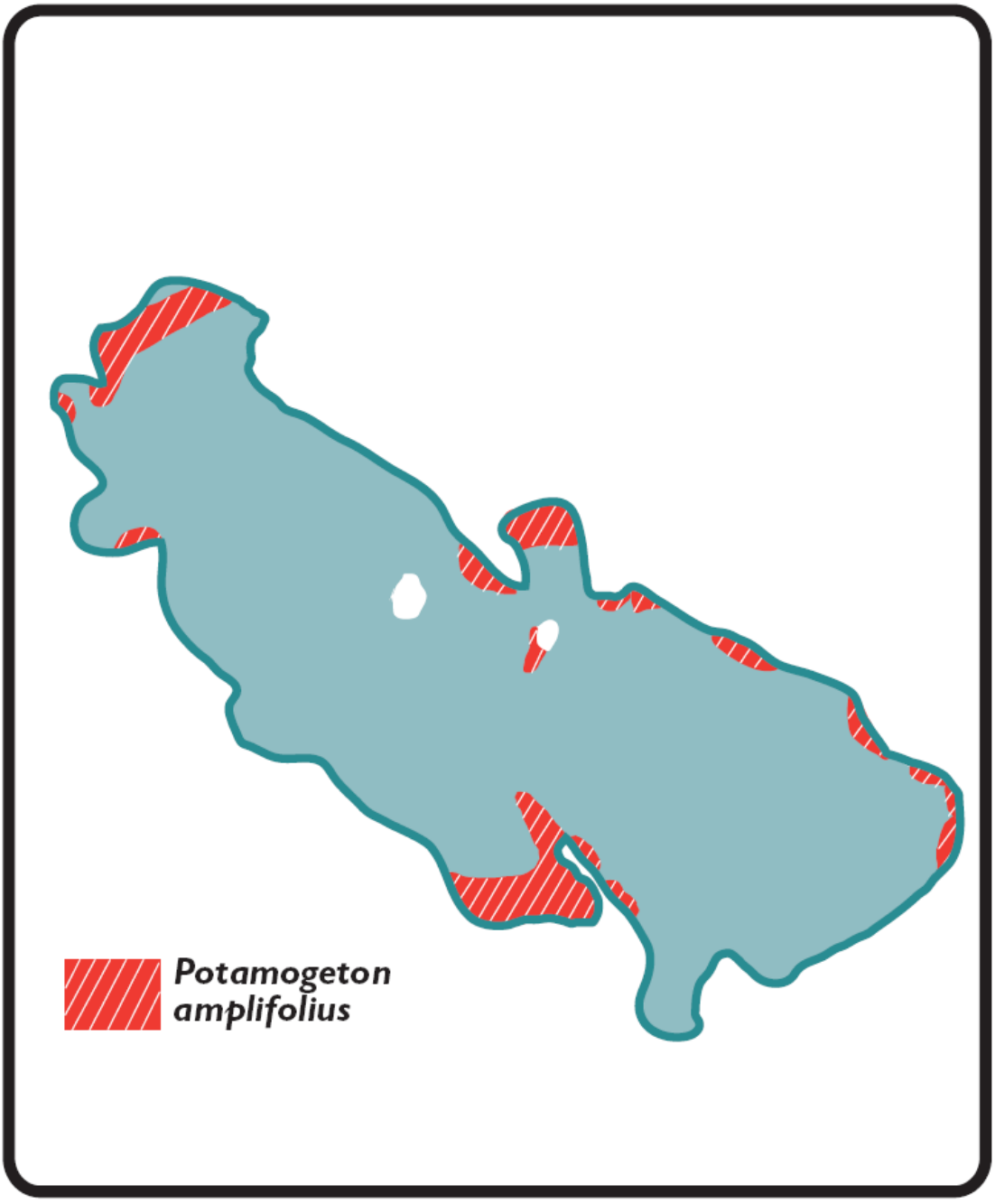

Figure 19: Distribution of *Potamogeton amplifolius* in Langford Lake – 2005.

Appendix 1
Langford Lake Water Quality Data

Summary of water chemistry results for Langford Lake site 1100944 (deep station). All results are reported in mg/L unless otherwise noted.

	1970 - 1979			1980 - 1989			1990 - 1999			2000 - Present		
	Mean	Std. Dev.	n =	Mean	Std. Dev.	n =	Mean	Std. Dev.	n =	Mean	Std. Dev.	n =
General												
Alkalinity Total 4.5	52.4	0.9	2	63.8	11.0	10	58.7	0.3	2	61.7	2.9	9
Color True (Col. Unit)				7	2	10	9	5	13	12	5	19
Color TAC (TAC)												
Hardness Total				67.8	6.4	34	70.3	5.6	4	70.6	5.2	8
pH (pH units)	7.1		1	7.7	0.5	61	7.9	0.2	11	7.5	0.3	35
Residue Nonfilterable				3	2	40	4	4	5	4	0	7
Residue Total				109	9	43	134	51	9	105	12	13
Specific Conductance (uS/cm)				174	16	58	176	7	13	188	19	35
Sulfate Dissolved				8.1	0.7	12	15.8	7.1	5	10.8	0.5	9
Silica Dissolved				2.1	2.3	22	5.7	2.5	11	3.1	4.3	19
Turbidity (NTU)	0.5	0.1	2	1.2	0.6	19	1.0	0.5	7	1.2	0.5	15
Nutrients												
Ammonia Dissolved				0.260	0.510	119	0.024	0.019	16	0.025	0.026	19
Carbon Total							17	1	4	18	1	3
Carbon Total Inorganic	12	1	2	17	4	18	14	1	4	15	1	14
Carbon Total Organic	6	0	2	3	3	18	3	0	4	4	1	29
Carbon Dissolved Organic										4	1	6
Nitrate Dissolved							0.249	0.102	5	0.127	0.148	6
Nitrate + Nitrite Dissolved	0.110	0.130	2	0.090	0.100	119	0.140	0.100	14	0.140	0.097	21
Nitrogen Total				0.850	0.570	87	0.630	0.050	5	0.510	0.160	45
Nitrogen Kjeldahl Total				0.670	0.540	119	0.320	0.050	11	0.370	0.080	8
Organic Nitrogen Total	0.35	0.08	2	0.42	0.13	87				0.33	0.06	8
Ortho-Phosphate Dissolved	0.009	0.008	2	0.031	0.069	115	<0.050	0	3	0.008	0.010	11
Phosphorus Total	0.028	0.001	2	0.06	0.089	122	0.015	0.009	16	0.033	0.043	46
Phosphorus Total Dissolved	0.021	0.006	2	0.043	0.075	105	0.007	0.004	16	0.01	0.012	17
Halides												
Bromide Dissolved							0.06	0.01	5	<0.08	0.03	9
Chloride Dissolved				11.0	0.3	8	10.5	0.6	13	10.0	2.4	9
Flouride Dissolved										0.03	0.01	4
Flouride Total							0.03	0.01	5			
Metals												
Aluminum Total				0.04	0.02	46	0.05	0.02	12	0.046	0.0802	11
Antimony Total							0.033	0.02	12	0.0391	0.02546	11
Arsenic Total				0.24	0.04	46	0.05	0.01	12	0.0392	0.0252	11
Barium Total							0.01	0.013	12	0.009	0.008	11
Beryllium Total							<0.001	0	12	0.00037	0.00041	11
Bismuth Total							<0.02	0	8	0.03126	0.02587	8
Boron Total							0.032	0.013	12	0.023	0.004	8
Cadmium Total				0.009	0.003	52	0.003	0.002	12	0.00255	0.00238	11
Calcium Total				19.9	2	46	20.9	1.7	15	21.3	1.7	8
Cobalt Total				0.1	0.02	46	0.005	0.001	12	0.0043	0.00306	11
Copper Total				0.009	0.002	51	0.004	0.003	12	0.00397	0.00245	11
Chromium Total				0.01	0.01	46	0.004	0.003	11	0.004	0.0025	11
Iron Total				0.08	0.14	46	0.063	0.063	12	0.047	0.017	8
Lead Total				0.09	0.03	51	0.04	0.02	12	0.03001	0.02323	11
Magnesium Total				4.4	0.4	46	4.2	0.4	15	4.26	0.25	8
Manganese Total				0.26	0.51	46	0.021	0.014	12	0.028	0.018	10
Mercury Total				0.00069	0.00143	31				0.00006	0.00001	8
Molybdenum Total				<.01	0	46	0.006	0.003	12	0.00503	0.00383	11
Nickel Total				0.05	0.01	52	0.013	0.005	12	0.00914	0.00776	11
Potassium Total							0.6	0.1	7	0.8	0.2	8
Selenium Total							0.04	0.01	12	0.0301	0.0232	11
Silicon Total							2.7	1.27	15	0.94	1.06	3
Silver Total							0.02	0.01	12	0.00728	0.00466	11
Sodium Total							7.8	0.3	9	8.0	0.7	8
Strontium Total							0.057	0.006	12	0.06	0.007	11
Sulfur Total							4.14	0.49	6	3.72	0.14	8
Tellurium Total							<0.02	0	8	<0.05	0	5
Thallium Total							0.023	0.012	12	0.01875	0.01552	8
Tin Total							0.03	0.02	12	0.02546	0.02381	11
Titanium Total							0.003	0	12	0.0028	0.0005	8
Vanadium Total				0.01	0	46	0.005	0.003	12	0.0052	0.00361	11
Zinc Total				0.03	0.08	43	0.011	0.01	12	0.0055	0.007	11
Zirconium Total							<0.003	0	8	<0.005	0	5

Appendix 2
Langford Lake Water Quality Data
November 2004 – February 2006

Site 1100944

Parameter	Units	9-Nov-04			24-Jan-05		
		0.1 m	12 m	avg.	0.1 m	12 m	avg.
Physical							
Conductivity	uS/cm						
pH	pH units						
Total Suspended Solids	mg/L						
Total Solids	mg/L						
Anions							
Silica	mg/L						
Nutrients							
Inorganic C - Total	mg/L	15.3	16.1	15.7	14.5	15.2	14.9
Organic C - Total	mg/L	5.1	5.1	5.1	3.5	4.1	3.8
Total Kjeldahl N	mg/L				0.5	0.47	0.49
Total N	mg/L	0.42	0.43	0.43	0.73	0.71	0.72
Total organic N	mg/L				0.43	0.38	0.41
Ammonia N	mg/L				0.077	0.092	0.085
Nitrate+Nitrite	mg/L				0.227	0.233	0.230
Ortho-P	mg/L				0.027	0.029	0.028
Total P	mg/L	0.024	0.037	0.031	0.041	0.041	0.041
General Biology							
Chlorophyll a	ug/L				5.9		5.9
Miscellaneous							
Alakalinity	mg/L						
True colour	Col. Units						

Parameter	Units	3-Mar-05				12-Apr-05			
		0.1 m	6 m	12 m	avg.	0.1 m	6 m	12 m	avg.
Physical									
Conductivity	uS/cm	180	181	181	181				
pH	pH units	7.4	7.3	7.2	7.3				
Total Suspended Solids	mg/L	5	5	4	5				
Total Solids	mg/L	104	110	104	106				
Anions									
Silica	mg/L	<0.5	1.4	2.7	2.1				
Nutrients									
Inorganic C - Total	mg/L	15.5	15.4	15.5	15.5	13.3	13.7	14.8	13.9
Organic C - Total	mg/L	4.9	4.6	4.6	4.7	4.1	3.3	2.7	3.4
Total Kjeldahl N	mg/L	0.30	0.35	0.36	0.34	0.35	0.33	0.38	0.35
Total N	mg/L	0.450	0.506	0.573	0.510	0.430	0.42	0.57	0.473
Total organic N	mg/L	0.33	0.34	0.33	0.33	0.32	0.3	0.29	0.30
Ammonia N	mg/L	0.016	0.014	0.029	0.020	0.027	0.024	0.095	0.049
Nitrate+Nitrite	mg/L	0.102	0.153	0.210	0.155	0.086	0.091	0.191	0.123
Ortho-P	mg/L	0.001	0.002	0.003	0.002	<.001	<0.001	0.003	<0.003
Total P	mg/L	0.019	0.029	0.019	0.022	0.017	0.027	0.029	0.024
General Biology									
Chlorophyll a	ug/L	16.6			16.6				
Miscellaneous									
Alakalinity	mg/L	<.5	<.5	<.5	<.5				
True colour	Col. Units	10	15	15	13				

Site 1100944 cont.

Parameter	Units	18-May-05			15-Jun-05		
		0.1 m	12 m	avg.	0.1 m	12 m	avg.
Physical							
Conductivity	uS/cm						
pH	pH units						
Total Suspended Solids	mg/L						
Total Solids	mg/L						
Anions							
Silica	mg/L						
Nutrients							
Inorganic C - Total	mg/L						
Organic C - Total	mg/L						
Total Kjeldahl N	mg/L						
Total N	mg/L	0.41	0.81	0.61	0.35	0.72	0.54
Total organic N	mg/L						
Ammonia N	mg/L						
Nitrate+Nitrite	mg/L						
Ortho-P	mg/L						
Total P	mg/L	0.011	0.035	0.023	0.018	0.024	0.021
General Biology							
Chlorophyll a	ug/L						
Miscellaneous							
Alakalinity	mg/L						
True colour	Col. Units						

Parameter	Units	21-Jul-05			16-Aug-05		
		0.1 m	12 m	avg.	0.1 m	12 m	avg.
Physical							
Conductivity	uS/cm						
pH	pH units						
Total Suspended Solids	mg/L						
Total Solids	mg/L						
Anions							
Silica	mg/L						
Nutrients							
Inorganic C - Total	mg/L						
Organic C - Total	mg/L						
Total Kjeldahl N	mg/L						
Total N	mg/L	0.32	0.76	0.54	0.32	0.79	0.56
Total organic N	mg/L						
Ammonia N	mg/L						
Nitrate+Nitrite	mg/L						
Ortho-P	mg/L						
Total P	mg/L	0.009	0.034	0.022	0.007	0.122	0.065
General Biology							
Chlorophyll a	ug/L						
Miscellaneous							
Alakalinity	mg/L						
True colour	Col. Units						

Site 1100944 cont.

Parameter	Units	21-Sep-05			13-Oct-05		
		0.1 m	12 m	avg.	0.1 m	12 m	avg.
Physical							
Conductivity	uS/cm						
pH	pH units						
Total Suspended Solids	mg/L						
Total Solids	mg/L						
Anions							
Silica	mg/L						
Nutrients							
Inorganic C - Total	mg/L						
Organic C - Total	mg/L						
Total Kjeldahl N	mg/L						
Total N	mg/L	0.34	0.85	0.60	0.35	0.73	0.54
Total organic N	mg/L						
Ammonia N	mg/L						
Nitrate+Nitrite	mg/L						
Ortho-P	mg/L						
Total P	mg/L	0.004	0.209	0.107	0.007	0.207	0.107
General Biology							
Chlorophyll a	ug/L						
Miscellaneous							
Alakalinity	mg/L						
True colour	Col. Units						

Parameter	Units	15-Nov-05			6-Feb-06			
		0.1 m	12 m	avg.	0.1 m	6 m	12 m	avg.
Physical								
Conductivity	uS/cm				181	181	182	181
pH	pH units				7.5	7.4	7.5	7.5
Total Suspended Solids	mg/L							
Total Solids	mg/L							
Anions								
Silica	mg/L				3.8	3.9	3.9	3.9
Nutrients								
Inorganic C - Total	mg/L				14.3	14.4	14.5	14.4
Organic C - Total	mg/L				6.4	6.7	7.3	6.8
Total Kjeldahl N	mg/L				0.32	0.31	0.30	0.31
Total N	mg/L	0.43	0.44	0.44	0.53	0.52	0.51	0.52
Total organic N	mg/L				0.29	0.26	0.26	0.27
Ammonia N	mg/L				0.023	0.044	0.034	0.034
Nitrate+Nitrite	mg/L				0.211	0.212	0.214	0.212
Ortho-P	mg/L				0.005	0.006	0.006	0.006
Total P	mg/L	0.028	0.038	0.033	0.031	0.044	0.023	0.033
General Biology								
Chlorophyll a	ug/L							
Miscellaneous								
Alakalinity	mg/L				60.2	59.9	<5	<5
True colour	Col. Units				5	<5	5	<5

Site 1100953

Parameter	Units	24-Jan-05			3-Mar-05			12-Apr-05		
		0.1 m	14 m	avg.	0.1 m	14 m	avg.	0.1 m	14 m	avg.
Nutrients										
Inorganic C - Total	mg/L									
Organic C - Total	mg/L									
Total Kjeldahl N	mg/L									
Total N	mg/L	0.72	0.62	0.67	0.509	0.795	0.652	0.461	0.661	0.561
Total organic N	mg/L									
Ammonia N	mg/L									
Nitrate+Nitrite	mg/L									
Ortho-P	mg/L									
Total P	mg/L	0.038	0.043	0.041	0.016	0.059	0.038	0.025	0.032	0.029

Parameter	Units	18-May-05				15-Jun-05			
		0.1 m	7 m	14 m	avg.	0.1 m	7 m	14 m	avg.
Nutrients									
Inorganic C - Total	mg/L	16.9	15.3	20.3	17.50	14.4	16.3	21.7	17.5
Organic C - Total	mg/L	4.3	4.1	3.4	3.93	3.7	3.5	3.4	3.5
Total Kjeldahl N	mg/L	0.34	0.35	0.59	0.43	0.32	0.33	0.98	0.54
Total N	mg/L	0.36	0.39	0.84	0.53	0.33	0.34	1.03	0.57
Total organic N	mg/L	0.32	0.31	0.31	0.31	0.32	0.33	0.37	0.34
Ammonia N	mg/L	0.024	0.035	0.285	0.115	<0.005	0.006	0.61	0.31
Nitrate+Nitrite	mg/L	0.24	0.45	0.243	0.311	0.005	0.013	0.053	0.024
Ortho-P	mg/L	0.002	0.002	0.005	0.003	<0.001	<0.001	0.166	<0.166
Total P	mg/L	0.012	0.022	0.042	0.025	0.018	0.021	0.27	0.103
General Biology									
Chlorophyll *a*	ug/L	3			3.0	3.4			3.4

Parameter	Units	21-Jul-05				16-Aug-05			
		0.1 m	7 m	14 m	avg.	0.1 m	7 m	14 m	avg.
Nutrients									
Inorganic C - Total	mg/L	14.6	15.6	18.6	16.3	14	14.1	20.3	16.1
Organic C - Total	mg/L	4.1	8.1	3.4	5.2	4.8	5	4.5	4.8
Total Kjeldahl N	mg/L	0.31	0.31	0.72	0.45	0.33	0.33	0.99	0.55
Total N	mg/L	0.32	0.33	0.81	0.49	0.33	0.33	0.99	0.55
Total organic N	mg/L	0.3	0.29	0.29	0.29	0.33	0.33	0.29	0.32
Ammonia N	mg/L	0.007	0.025	0.423	0.15	<0.005	<0.005	0.71	0.71
Nitrate+Nitrite	mg/L	0.016	0.014	0.097	0.042	<0.002	<0.002	<0.002	<0.002
Ortho-P	mg/L	0.002	0.002	0.130	0.045	0.002	0.003	0.163	0.056
Total P	mg/L	0.008	0.014	0.19	0.071	0.004	0.014	0.2	0.073
General Biology									
Chlorophyll *a*	ug/L				1.5				2

Site 1100953 cont.

Parameter	Units	21-Sep-05				13-Oct-05			
		0.1 m	7 m	14 m	avg.	0.1 m	7 m	14 m	avg.
Nutrients									
Inorganic C - Total	mg/L	16.1	16.1	24.5	18.9	15.7	15.5	24.2	18.5
Organic C - Total	mg/L	15	15	16	15.3	6.2	5.4	6.2	5.9
Total Kjeldahl N	mg/L	0.35	0.36	1.7	0.80	0.37	0.54	1.2	0.70
Total N	mg/L	0.35	0.36	1.66	0.79	0.38	0.56	1.25	0.73
Total organic N	mg/L	0.35	0.36	0.43	0.38	0.35	0.52	0.4	0.42
Ammonia N	mg/L	<0.005	<0.005	1.22	1.22	0.014	0.022	0.84	0.29
Nitrate+Nitrite	mg/L	<0.002	0.002	0.004	0.003	0.018	0.019	0.003	0.013
Ortho-P	mg/L	0.004	0.004	0.385	0.131	0.008	0.006	0.290	0.101
Total P	mg/L	0.005	0.005	0.45	0.153	0.008	0.011	0.422	0.147
General Biology									
Chlorophyll *a*	ug/L				2				3.4

Parameter	Units	15-Nov-05				6-Feb-06		
		0.1 m	7 m	14 m	avg.	0.1 m	14 m	avg.
Nutrients								
Inorganic C - Total	mg/L	16.9	17	27.4	20.4			
Organic C - Total	mg/L	3.3	4.2	5	4.2			
Total Kjeldahl N	mg/L	0.35	0.38	2.3	1.01			
Total N	mg/L	0.4	0.43	2.3	1.04	0.610	0.6	0.605
Total organic N	mg/L	0.26	0.3	0.57	0.38			
Ammonia N	mg/L	0.092	0.086	1.7	0.63			
Nitrate+Nitrite	mg/L	0.049	0.048	0.01	0.036			
Ortho-P	mg/L	0.025	0.024	0.600	0.216			
Total P	mg/L	0.026	0.029	0.560	0.205	0.037	0.038	0.038
General Biology								
Chlorophyll *a*	ug/L				3.3			

Site E257395

Parameter	Units	9-Nov-04			24-Jan-05		
		0.1 m	12 m	avg.	0.1 m	12 m	avg.
Inorganic C - Total	mg/L	16.1	16.3	16.2			
Organic C - Total	mg/L	5.2	4.9	5.1			
Total N	mg/L	0.41	0.42	0.415	0.680	0.520	0.600
Total P	mg/L	0.022	0.038	0.030	0.038	0.04	0.039

Parameter	Units	3-Mar-05			12-Apr-05		
		0.1 m	12 m	avg.	0.1 m	12 m	avg.
Inorganic C - Total	mg/L						
Organic C - Total	mg/L						
Total N	mg/L	0.422	0.533	0.478	0.468	0.526	0.497
Total P	mg/L	0.027	0.024	0.026	0.02	0.017	0.019

Parameter	Units	18-May-05			15-Jun-05		
		0.1 m	12 m	avg.	0.1 m	12 m	avg.
Inorganic C - Total	mg/L						
Organic C - Total	mg/L						
Total N	mg/L	0.400	0.750	0.575	0.320	0.890	0.605
Total P	mg/L	0.012	0.031	0.022	0.016	0.078	0.047

Parameter	Units	21-Jul-05			16-Aug-05		
		0.1 m	12 m	avg.	0.1 m	12 m	avg.
Inorganic C - Total	mg/L						
Organic C - Total	mg/L						
Total N	mg/L	0.350	0.770	0.560	0.320	0.740	0.530
Total P	mg/L	0.008	0.068	0.038	0.006	0.085	0.046

Parameter	Units	21-Sep-05			13-Oct-05		
		0.1 m	12 m	avg.	0.1 m	12 m	avg.
Inorganic C - Total	mg/L						
Organic C - Total	mg/L						
Total N	mg/L	0.360	1.210	0.785	0.340	0.870	0.605
Total P	mg/L	0.006	0.256	0.131	0.01	0.223	0.117

Parameter	Units	15-Nov-05			6-Feb-06		
		0.1 m	12 m	avg.	0.1 m	12 m	avg.
Inorganic C - Total	mg/L						
Organic C - Total	mg/L						
Total N	mg/L	0.420	0.410	0.415	0.580	0.570	0.575
Total P	mg/L	0.025	0.025	0.025	0.036	0.041	0.039

Site E257396

Parameter	Units	9-Nov-04			24-Jan-05		
		0.1 m	10 m	avg.	0.1 m	10 m	avg.
Inorganic C - Total	mg/L	15.9	16.3	16.1			
Organic C - Total	mg/L	4.7	3.8	4.3			
Total N	mg/L	0.45	0.37	0.410	0.580	0.560	0.570
Total P	mg/L	0.025	0.023	0.024	0.037	0.039	0.038

Parameter	Units	3-Mar-05			12-Apr-05		
		0.1 m	10 m	avg.	0.1 m	10 m	avg.
Inorganic C - Total	mg/L						
Organic C - Total	mg/L						
Total N	mg/L	0.535	0.524	0.530	0.440	0.437	0.439
Total P	mg/L	0.02	0.034	0.027	0.015	0.021	0.018

Parameter	Units	18-May-05			15-Jun-05		
		0.1 m	10 m	avg.	0.1 m	10 m	avg.
Inorganic C - Total	mg/L						
Organic C - Total	mg/L						
Total N	mg/L	0.390	0.500	0.445	0.420	0.570	0.495
Total P	mg/L	0.014	0.037	0.026	0.02	0.12	0.070

Parameter	Units	21-Jul-05			16-Aug-05		
		0.1 m	10 m	avg.	0.1 m	10 m	avg.
Inorganic C - Total	mg/L						
Organic C - Total	mg/L						
Total N	mg/L	0.320	1.040	0.680	0.400	1.100	0.750
Total P	mg/L	0.011	0.15	0.081	0.008	0.193	0.101

Parameter	Units	21-Sep-05			13-Oct-05		
		0.1 m	10 m	avg.	0.1 m	10 m	avg.
Inorganic C - Total	mg/L						
Organic C - Total	mg/L						
Total N	mg/L	0.340	0.670	0.505	0.380	0.340	0.360
Total P	mg/L	0.005	0.037	0.021	0.007	0.011	0.009

Parameter	Units	15-Nov-05			6-Feb-06		
		0.1 m	10 m	avg.	0.1 m	10 m	avg.
Inorganic C - Total	mg/L						
Organic C - Total	mg/L						
Total N	mg/L	0.430	0.410	0.420	0.550	0.560	0.555
Total P	mg/L	0.024	0.028	0.026	0.015	0.043	0.029

Appendix 3
Langford Lake Phytoplankton Results
January 2005 – February 2006

2005 Langford Lake phytoplankton results (cells/mL).

	Organism	24-Jan-05	3-Mar-05	18-May-05	21-Jul-05	16-Aug-05	21-Sep-05	13-Oct-05	15-Nov-05	6-Feb-06
		1100944	1100944	1100953	1100953	1100953	1100953	1100953	1100953	1100953
Order:	**Centrales**									
	Cyclotella bodanica			present	present					
	Cyclotella sp.			present						present
	Melosira granulata	present		present			47.6	present		present
	Melosira italica	present	present						120.4	
	Melosira sp.	39.2	11.2	present	present	present	present	39.9	present	42
	Stephanodiscus niagarae	5.6	95.2	present	present	present		5.7	present	142.8
Order:	**Chlorococcales**									
	Ankistrodesmus cf falcatus					present				
	Ankistrodesmus sp.	present	42.0	present	1.9	2.8	present	5.7	8.4	present
	Botryococcus braunii		present	present	present		present	present	present	
	Closteriopsis sp.									present
	Coelastrum microporum				present					
	Coelastrum sp.			present						
	Crucigenia rectangularis				present	present				
	Dactylococcopsis Smithii								present	
	Dictyosphaerium pulchellum		present						present	
	Elakatothrix gelatinosa	present	present		present	2.8	present		present	14
	Nephrocytium sp.				present					present
	Oocystis sp.		present		present	5.6			present	present
	Pediastrum sp.	present		present	present	present	present	present	present	
	Quadrigula closterioides					present	present		present	
	Quadrigula lacustris			present		present	present			
	Scenedesmus sp.	present		present	present					
	Schroederia setigera			42.0	present	present	16.8	28.5	5.6	2.8
	Sphaerocystis schroeteri			present	304.0	67.2	present		present	89.6
	Tetraedron minimum	present					present		present	
	Tetraedron sp.							present		
Order:	**Chroococcales**									
	Anacystis cf aeruginosa			present	present					
	Anacystis elachista						present	present	present	
	Anacystis limneticus				60.8	89.6	present	22.8	present	present
	Anacystis sp.			present	present				present	
	Gomphosphaeria sp.	present		present	present					
	Unidentified								present	
Order:	**Cryptomonadales**									
	Chroomonas acuta	92.4	389.2	154.0	83.6	78.4	226.8	91.2	50.4	128.8
	Cryptomonas ovata/erosa	50.4	5.6	22.4	9.5	2.8	14.0	22.8	8.4	present
	Cryptomonas sp.	120.4	22.4	47.6	present	present	14.0	5.7	2.8	22.4
Order:	**Dinokontae**									
	Ceratium hirundinella			present		present	2.8	5.7		present
	Perinidinium / Glenodinium	present	present							
Order:	**Euglenales**									
	Euglena sp.			present			present			
	Phacus sp.							present		
	Trachelomonas sp.	2.8	present	present	1.9	present	present	present	present	present
Order:	**Nostocales**									
	Anabaena cf affinis					present	present			
	Anabaena cf circinalis			present	present	present	present	present		
	Anabaena cf flos-aquae	417.2	224.0	147.0						84
	Anabaena spiroides						70.0	present	present	
	Anabaena sp.				present	present	present	present	present	
	Aphanizomenon flos-aquae					present	274.4	1584.6	352.8	
	Nostoc sp.						present			
Order:	**Ochromonadales**									
	Dinobryon divergens			present	3.8	72.8	5.6	11.4	present	2.8
	Dinobryon sertularia		present	present						
	Dinobryon sp.			present	present					present
	Mallomonas akrokomas									present
Order:	**Oedogoniales**									
	Oedogonium sp.			present						

2005 Langford Lake phytoplankton results (cells/mL) cont.

Order:	**Oscillatoriales**											
	Lyngbya Birgei				present	present	present	present				
	Lyngbya limnetica				present		present					
	Lyngbya sp.	present										
	Oscillatoria cf tenuis				present			present	present			
	Oscillatoria sp.					present	present	present	present	present	present	
	Phormidium mucicola		present	present	present							
Order:	**Pennales**											
	Achnanthes minutissma	present		1.4		present		5.7				
	Asterionella formosa	25.2	1041.6	19.6	32.3	present	5.6	17.1	present	present		
	Ceratoneis sp.			present	present							
	Cocconeis cf pediculus									present		
	Cocconeis placentula	present		present	present							
	Cocconeis sp.				present							
	Cymbella sp.	present					present			present		
	Denticulata sp.	present							present	present		
	Diatoma elongatum											
	Diatoma sp.			present						present		
	Epithemia sorex					present				present		
	Epithemia turgida				present		present					
	Eunotia sp.					present						
	Fragilaria crotonensis	11.2	224.0	present	3.8	present	42.0	5.7	8.4	347.2		
	Fragilaria sp.	present		present			present		2.8			
	Gomphonema cf augur			present	present							
	Gomphonema constrictum	present						present				
	Gomphonema olicaveum						present					
	Gomphonema sp.	present		present								
	Meridion sp.	present										
	Navicula cf radiosa			present	present							
	Navicula sp.	present		present	present	present	present	present	present	present		
	Nitzschia sp.	present		present	present		present	present	present	present		
	Pleurosigma / Gyrosigma							present				
	Rhoicosphenia curvata	present										
	Rhopalodia gibba					present	present	present				
	Synedra capitata	present										
	Synedra ulna	present	present			present						
	Tabellaria fenestrata	170.8	940.8	2.8	57.0	present	436.8	91.2	present	present		
	Tabellaria flocculosa			present		present						
	Unidentified				present							
Order:	**Tetrasporales**											
	Gloeocystis ampla				30.4	336.0			present			
Order:	**Volvocales**											
	Chlamydomanas sp.						present					
	Eudorina elegans	present	present									
	Volvox sp.						present					
	Unidentified				present	present		present	present	present		
Order:	**Zyngematales**											
	Cosmarium sp.			present	present	present	present					
	Mougoetia sp.			present		present	present					
	Spondylosium sp.			5.6		present			present			
	Staurastrum paradoxum	present	present	present	present	present	present	present		2.8	present	
	Staurastrum sp.					present						
	Unidentified					present						
	Unidentified filamentous algae	present										
	Unitdentified unicellular Chlorophyte						present					

Appendix 4
Langford Lake Historical Phytoplankton Results
1972 – 1981

Historical data for Langford Lake phytoplankton community compositions (an explanation of the codes is provided at the bottom of the table).

Order	Family	Genus/species	20-Oct-72	27-Oct-72	11-Nov-72	24-Nov-72	1-Jan-73	23-Feb-73	15-May-80	5-Jun-80	7-Aug-80	6-Oct-80	1-Jan-81
Chroococcales	Chroococcaceae	*Aphanocapsa* sp.			R	R		R					
		Coelosphaerium naegelianum	A	C	C	A	R	C-A	A-D	F	F	R	F
		Anacystis aeruginosa							R	R	R-F	R	
Nostocales	Nostocaceae	*Anabaena* sp.							R	R-F	R		
		Anabaena flosaquae	D	A	A	R			C-A				
		Anabaena limnetica	F	C			R						
		Anabaena spiroide							F-C				
		Aphanizomenon flosaquae	D	D	D	D			A	F	C	C	R+
	Oscillatoriaceae	*Oscillatoria* sp.								R-F			
		Oscillatoria rubescens							F-C				
	Rivulariaceae	*Gloeotrichia echinulata*							R	R			R
Centrales	Melosiraceae	*Melosira italica*		C				R					
	Stephanodiscaceae	*Stephanodiscus niagarae*	C-A	C	C	C	R	C			R-F	R	R
Pennales		*Asterionella formosa*				R	D	D	C				C
		Fragilaria capucina							R			R	R
		Fragilaria crotenensis	A-D	A	A	D		A-C	F-C	A-D	C-A		C
		Synedra cyclopum	R		R	D	A	A					
		Synedra ulna				R		R					R+
	Synuraceae	*Mallomonas* sp.									R-F		R-F
		Synura elegans											R+
	Ochromonadaceae	*Uroglena* sp.							R				
Peridinales		*Ceratium hirundinella*	R	R	R	R				R	C	F	
		Peridinium pusillum							R-F	R			R
Chlorococalles		*Ankistrodesmis fractus*					R						
		Botryococcus braunii				R		F	R-F				
		Chlorella sp.								R			R
		Closteriopsis longissima								F			
		Dictyosphaerium pulchellum							R	C	R		
		Gloeocystis echinulata											
		Gloeocystis sp.											R
		Schroederia sp.				R							
	Oocystaceae								R				
Oedogoniales		*Oedogonium* sp.							R				
Volvocales		*Chlamydomonas* sp.					C						
		Eudorina elegans	C	C	C	C	C	C	F-C	F-C	R		
		Pandorina sp.							R	F	F		
		Volvox aureus		A		R	C						
		Volvox sp.							A				
Zygnematales		*Closterium* sp.									A	R	R-F
		Micrasterias americana				R							
		Pleurotaenium trabecula			R								
		Spirogira sp.							R				
		Staurastrum gracilis							F-C		F		R+
		Staurastrum limneticum				R							
		Staurastrum paradoxum	C	C	R	C	R	R		A			

D Dominant, found in over 80% of the counts, usually very dense
A Abundant, found in 60% or more of the counts, fairly dense
C Common, found in 50% or more of the counts, not dense
F Frequent, found in 10% or more of the counts, sparse
R Rare, encountered once or a few times, very sparse

Appendix 5
Langford Lake Paleolimnological Assessment

ASSESSMENT OF CHANGES IN TOTAL PHOSPHORUS IN LANGFORD LAKE, B.C. : A PALEOLIMNOLOGICAL ASSESSMENT (Spring 2006)

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List of Figures:

BACKGROUND

A ~43-cm sediment core was retrieved from Langford Lake with a modified K-B corer in December 2003 by R. Nordin and colleagues. Nineteen samples were sent to PEARL at Queen's University where they were prepped and analyzed for diatom composition were every 1.0 cm for the top 6.0 cm, then approximately every 1.5 cm to 23.5 cm and then a final sample at 42.5cm (sample selection was based on samples received from R. Nordin). A preexisting core chronology, established by MyCore Scientific using the CRS model, was used provide a time-depth chronology in the core.

METHODS

Diatom Preparation and Enumeration

Slides for diatom analysis were prepared using standard techniques (Cumming et al. 1995). Briefly, a small amount of wet sediment was suspended in a 50:50 (molar) mixture of sulfuric and nitric acid in a 20-ml glass vial for 24 hr. prior to being submersed at 70˚C in a water bath for approximately 5 hr. The remaining sediment material was settled for a period of 24 hr, at which time the acid above the sample was removed. The sample was rinsed with distilled water and allowed to settle once again for 24 hrs. The procedure was repeated approximately 8 times until the sample was acid free (litmus test). The samples were settled onto coverslips in a series of four 100% dilutions, which when dry, were mounted onto glass slides using a high-resolution mounting media called Naphrax®. For each sample, at least 400 diatom valves were enumerated with a Leica DMRB microscope equipped with DIC optics at 1000X magnification (Numerical Aperature of objective = 1.3). These analyses were based on the references of Krammer and Lange-Bertalot (1986, 1988, 1991a,b), Patrick and Reimer (1966, 1975) and Cumming et al. (1995).

Diatom-based Reconstructions of Total Phosphorus

Inferences of total phosphorus from the diatom assemblages in the core are based on a phosphorus model developed from 268 freshwater lakes from across British Columbia. This dataset includes lakes from several regions within British Columbia. This model is based on estimates of the optima of taxa from weighted-averaging regression on non-transformed relative percentage data. The coefficient of determination (r^2) of this model is 0.62, and the bootstrapped r^2 is 0.51. This model is superior to the earlier models developed by Reavie et al. (1995) for several reasons including its better predictive ability and the larger number of samples which provide more analogs for downcore reconstructions.

The total phosphorus inferences (Fig. 1B) were critically assessed to determine if they tracked the main direction of variation in the diatom species assemblages (Fig. 1A). If the diatom-based phosphorus reconstructions match the main direction of variation in the diatom assemblages in the core, then we can be fairly confident that the diatoms are tracking changes that are related to phosphorus, or correlated variables. If the correlation between the main direction of variation and the diatom-inferred phosphorus values is weak or nonexistent, then other environmental variables (e.g., water depth, conductivity, turbulence, etc), or interactions between environmental variables, are likely responsible for the observed changes in diatom assemblages.

Determination of the Main Direction of Variation

The main direction of variation in the diatom assemblages in the Langford Lake core was determined from the axis-1 scores from a principal components analysis (PCA) ordination using non-transformed species abundance data (Figs. 1A). A PCA was chosen to represent the main direction of variation of the diatom assemblages in these cores based on the small gradient length (< 1.5 standard deviation units) obtained in an initial detrended correspondence analysis (DCA) ordination.

Cluster Analysis

Cluster analysis provides a means of grouping those samples that are most similar to each other. In the clustering algorithm, the samples were stratigraphically constrained and a chord distance was used as the measure of similarity between samples. The programs, TILIA and TGVIEW 2.02 (Grimm, unpublished), were used to generate the stratigraphic profile of the diatom assemblages and the cluster dendogram (Fig. 2.).

RESULTS AND DISCUSSION

Approximately ninety-four taxa were documented in the core from Langford Lake. However, most of these taxa were rare. Cluster analysis suggests two major periods of diatom assemblages in the recent history of the lake (Fig. 2). In Zone B, representing the time period prior to 1970 AD, the diatom assemblage is dominated by the mesotrophic planktonic *Stephanodiscus medius*, oligotrophic planktonic *Cyclotella stelligera* and mesotrophic benthic taxon *Staurosirella pinnata.* Subdominant taxa include the benthic taxa *Achnanthidium minutissima*, *Staurosira construens* and *Pseudostaurosira brevistriata* and planktonic taxa *Fragilaria tenera*, *Cyclotella michiganiana*, *Fragilaria nanana* and *Aulacoseira subarctica*. The bottommost sample analyzed was at 42.5 cm, but for ease of graphing is placed at 26 cm. This sample is similar to the four samples analyzed above it from 23.5 to 19.0 cm, indicating that the assemblage was extremely similar for at least 200 years prior to 1900 AD. Beginning around 1940 there is a consistent decline in *Stephanodiscus medius* and by approximately 1960 an increase in the more eutrophic planktonic *Aulacoseira ambigua*. Zone A is characterized by the dominance of *A. ambigua* which reaches percent abundances between 35 to 53%. Whereas *S. medius* declines to 11% or less.

S. medius and the more eutrophic *Stephanodiscus minutulus* were difficult to differentiate in this core and thus were combined in the final analyses. Recounts of several samples suggest that *S. minutulus* comprises 10% or less of this combined group during Zone B where it dominates and thus if separated would only increase the total phosphorus (TP) estimates slightly in this zone.

Diatom-inferred total phosphorus (TP) estimates indicate mid-summer mesotrophic conditions that vary between 10 to 19 μgL^{-1} (Fig. 1B). Estimates of TP, prior to 1970 AD (Zone B), range from 10 to 15 μgL^{-1}, with an average of 12.7 μgL^{-1}. Post-1970 AD (Zone A), TP estimates increase to 14 to 19 μgL^{-1} (average of 16.7 μgL^{-1}), with the highest values occurring in the most recent sediments (post 2000 AD). The in inferred TP in Zone A is driven by the increases the more eutrophic planktonic, *A. ambigua.* The increase in *A. ambigua* may also indicate more turbulent conditions, since this taxon is a large chain former and thus needs more turbulent conditions to remain within the photic zone.

The correspondence between the first main direction of variation in taxa (i.e., PCA axis-1

scores, Fig. 1A) and the log TP inferences (Fig. 1B) is high ($r^2 = 0.61$) indicating that the changes seen in the diatom assemblages are consistent with the changes seen in the TP estimates. All dominant taxa, which are driving the reconstructions of TP, are well represented in our modern-day calibration set, thus providing evidence that the TP estimates are reliable.

SUMMARY

In summary, the diatom-inferred TP level of Langford Lake indicates a relatively productive lake during the recent history, with mesotrophic estimates ranging between 10 to 19 μgL^{-1}. A dramatic increase in the planktonic taxon *A. ambigua* starting in 1960 and stabilizing at higher percent abundances post 1970 indicates a sharp change within the lake at this time. The increase in *A. ambigua* indicates an increase in phosphorus conditions, which is collaborated with the high correspondence between TP estimates and the main direction in variation. However, some of this increase in *A. ambigua* may also be associated with an increased mixing regime.

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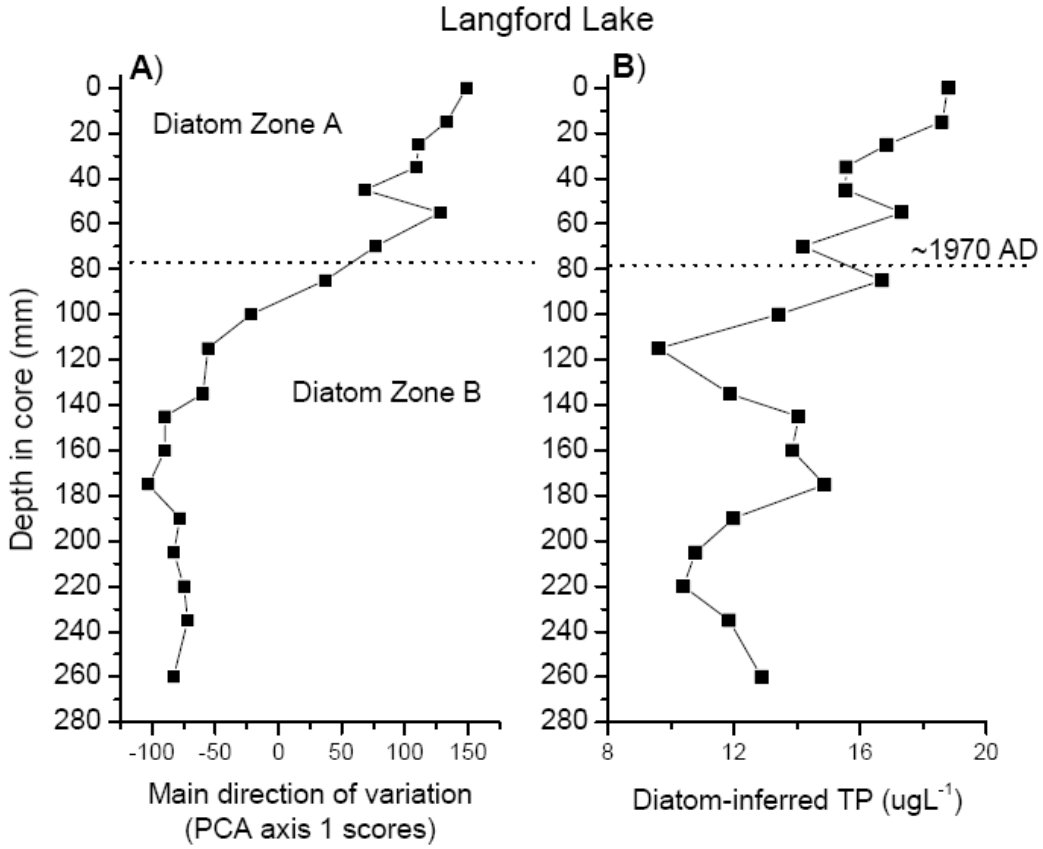

Figure 1. Summary diagram for Langford Lake showing: A) the main direction of variation in the diatom assemblage data; and B) diatom-based estimated of late-summer total phosphorus.

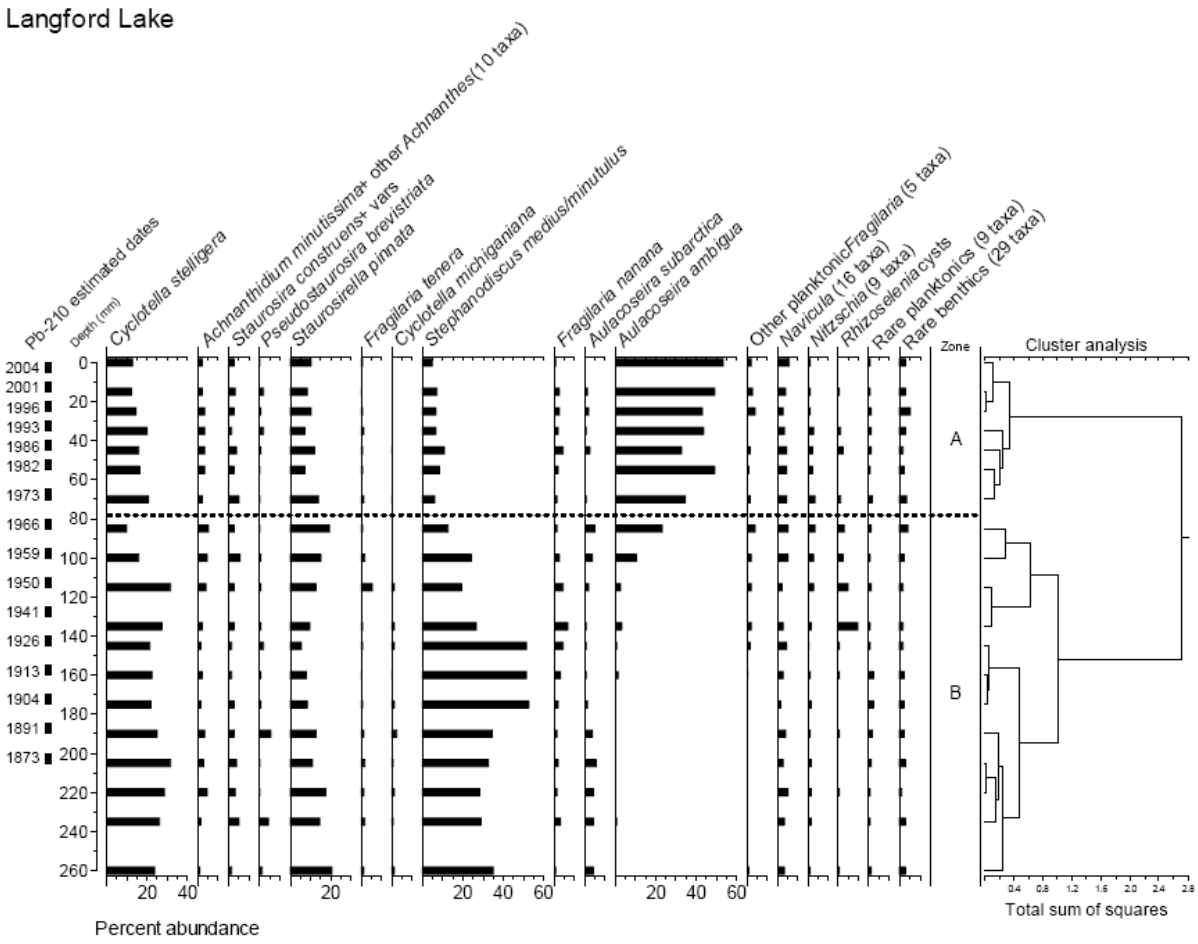

Figure 2. Stratigraphy of the most abundant diatom taxa found in the sediment core from Langford Lake. The diatom taxa are arranged in order of increasing late-summer total phosphorus (TP) optima.

Appendix 6
Glossary

Ambient
Refers to conditions in the surrounding environment.

Ammonia
A measure of the most reduced inorganic form of nitrogen in water and includes dissolved ammonia (NH_3) and the ammonium ion (NH_4^+).

Bathymetric map
A map showing the depth (bottom contours) of water in lakes, streams, or oceans.

Cyanobacteria (blue-green algae or cyanophytes)
A group of phytoplankton which often cause nuisance conditions in water, so called because they contain a blue pigment in addition to chlorophyll. Blue-green algae are often associated with problem blooms in lakes. Some produce chemicals toxic to other organisms, including humans. They often form floating scum as they die. Many can fix nitrogen from the air to provide their own nutrients.

Chlorphyll *a*
The primary green-coloured pigment found in plants and algae which traps and converts light energy to chemically stored energy.

Colour, true
A measure of the dissolved colouring compounds in water. The colour of water is attributed to the presence of organic and inorganic materials which absorb different light frequencies.

Designated water use
A water use that is to be protected at a specific location.

Disinfection by-products
Chemicals (e.g. trihalomethanes) formed when a disinfectant (e.g., chlorine) is added to water containing organic matter. Such by-products are suspected to be human carcinogens.

Dissolved oxygen (DO)
Oxygen dissolved in water and essential for respiration by most aquatic organisms.

Enterocci
Bacteria species inhabiting the gut of humans and other warm blooded animals which are used as an indicator of water contamination. Some forms can be pathogenic.

Epilimnion
The surface layer of a thermally stratified lake.

Eutrophication
Increasing nutrient content in a body of water over time. This natural process may be accelerated by nutrient-rich discharges from agriculture or sewage, resulting in algal blooms, excessive growth of macrophytes or undesirable changes in water quality.

Fall freshet
A sudden increased period of stream flow as a result of heavy rainfall typical of coastal areas in the fall.

Fecal coliforms
Enteric bacteria inhabiting the gut of humans and other warm blooded animals which are used as an indicator of water contamination.

Geometric mean
The N^{th} root of the product of N observations.

Grab sample
A single sample taken at a given place and time.

Hypolimnion
The cooler, deeper waters of a thermally stratified lake.

Isotherm
A line drawn on a map or chart linking all points of equal or constant temperature.

Kjeldahl nitrogen
A measure of both the ammonia and the organic forms of nitrogen.

Littoral
The region along the shore of a non-flowing body of water.

Macrophyte
The larger aquatic plants, including aquatic mosses, liverworts, larger algae and vascular plants.

Morphometry
The physical characteristics of a lake such as size and shape of a lake basin, mean depth, maximum depth, volume, drainage area, and flushing rate.

Nitrate + nitrite (NO_3 + NO_2)
A measure of the most oxidized and stable form of N in a water body (NO_3) and an intermediate form (NO_2) that occurs in the biological conversion of NH_4 to NO_3.

Oligotrophic
A water body with limited nutrient input or cycling, resulting in low levels of biomass production.

Ortho-phosphorus
A measure of the inorganic oxidized and biologically available form of soluble phosphorus.

pH
A measure of the hydrogen ion concentration of a solution which provides a quantitative expression of its acidity or alkalinity ranging, from 0 to 14. pH 7 is neutral, less than 7 is acidic and more than 7 is alkaline or basic.

Phytoplankton
An assemblage of small plants suspended in the water column with little or no powers of locomotion.

Primary productivity
A measure of algal producitivity or rate of growth in a body of water; the primary productivity measures the mass of carbon used annually by algae per unit area of lake surface.

Recreational primary contact
Activities like swimming and water sports where a person has or risks direct contact with water through immersion or ingestion.

Secchi disc
A black and white disk used to measure the transparency or clarity of water in a lake by measuring the maximum depth at which it can be seen.

Stratification
The vertical temperature stratification of a lake which consists of: (a) the upper layer (**epilimnion**), (b) the middle layer (**thermocline**) and (c) the bottom layer (**hypolimnion**).

Thermocline
A well defined vertical temperature change or boundary; often associated with **thermal stratification** in lakes.

Total nitrogen
A measure of all forms of nitrogen (organic and inorganic).

Total Phosphorus
A measure of all forms of phosphorus (organic and inorganic).

Trophogenic zone
The area of a body of water where organic production from mineral substances takes place on the basis of light energy and photosynthetic activity

Water column
The portion of an aquatic or marine environment extending from the water surface to the bottom or the surface of the sediment.

Water Quality Guideline
Numerical value(s) for a physical, chemical or biological characteristic of water, biota or sediment which must not be exceeded to prevent specified detrimental effects from occurring to water use.

Water Quality Objective
A water quality guideline adapted to protect the most sensitive designated water use at a specified location with an adequate degree of safety, taking local circumstances into account.

Water residence time
A measure of measure of how often, usually in years, water is replaced in a lake based on flows into and out of the system.

Watershed
All lands enclosed by a continuous hydrologic drainage divide and lying upslope from a specified point on a waterbody.

Zooplankton
Microscopic animals which swim freely in the water column or are carried about by water currents. Many feed on **phytoplankton** and are in turn a staple diet of small fish.