QUANTIFYING THE MAJOR SINKS AND SOURCES OF PHOSPHORUS IN TABOR LAKE: IMPLICATIONS FOR MANAGEMENT AND REMEDIATION

by

Sean Simmons

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Abstract

A study of Tabor Lake and its watershed during the open water season of 1995 was conducted in order to quantify the major sinks and sources of phosphorus, and the timing of phosphorus transfer between these sources and sinks. The results of this thesis are useful to help determine appropriate management strategies aimed at reducing phosphorus availability in Tabor Lake's water column and understanding some of the processes that control phosphorus availability in the lake.

An exploratory modeling exercise was undertaken to evaluate morphometric and watershed variables in predicting spring phosphorus for 39 lakes in the region, including Tabor Lake. Two multiple linear regression models were constructed. The first model uses a stepwise technique to select statistically significant variables, and the second model uses variables found in the literature which have been shown to influence phosphorus loading. These two models account for 70% and 59% of the variation in spring phosphorus, respectively, and may provide managers with an expected phosphorus loading based on easily measured morphometric and watershed variables. The models predict that Tabor Lake should have a spring phosphorus concentration of 20 and 18 μg/L, respectively. The models are useful to establish the limits of water quality improvement for Tabor Lake.

Another model was developed to estimate the quantity and timing of external sinks and sources of phosphorus in Tabor Lake for 1995. Phosphorus loading into Tabor Lake was estimated from runoff (136 kg) and atmospheric deposition (60 kg). Phosphorus output from Tabor Lake through the outlet creek was estimated at 188 kg for the entire year. Based on this input-output model, there was a net gain of 8 kg phosphorus into Tabor Lake during 1995. These results indicate that external phosphorus inputs and output is approximately balanced.

Results from a prior monitoring program indicate that Tabor Lake experienced internal loading of phosphorus during periods of hypolimnetic anoxia. However, phosphorus loading also occurred when oxygen was present in the hypolimnion, suggesting another phosphorus loading mechanism was active in Tabor Lake. A specific effort was made to evaluate the role of senescing macrophytes (primarily *Elodea canadensis*) as a source of this phosphorus during the 1995 sampling season. Two experiments were conducted, one in-vitro and the other in-situ, to estimate phosphorus leaching from senescing macrophytes. The first in-vitro experiment used weeds collected August 8, 1995 and show that 87% (S.E.= 6%) of the total plant phosphorus is leached from these macrophytes, whereas the second in-vitro experiment, using samples collected September 22, 1995, show that only 9% (S.E.= 1%) of the phosphorus is leached from macrophytes. The difference between the two leaching estimates is believed to be the result of overwintering development and expected starch accumulation in the September samples. The in-situ study provided the best prediction of phosphorus leaching from senescing macrophytes, estimating 1958 kg of phosphorus (S.E.= 867 kg) was released from senescing macrophytes in 1995.

The results from both the input-output model and the internal loading estimates show that Tabor Lake is dominated by the internal cycling of nutrients beginning in June and continuing until September. Furthermore, it quantifies two important phosphorus sources and the timing of phosphorus delivery from these sources. The quantification and timing of phosphorus delivery in Tabor Lake is useful to managers establishing remediation strategies which try to reduce the total phosphorus present in the water column. This research also shows that mechanical harvesting of macrophytes removes more phosphorus from the lake than exits through the outlet and can be useful in reducing the total phosphorus available to internal loading cycles in Tabor Lake.

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Acknowledgement

My first introduction to Tabor Lake was in May 1994, about 8 months after the widely publicized *fish kill of '93*. My new supervisor, Ellen Petticrew, brought me to Tabor Lake and introduced me to many of the Tabor Lake residents. Throughout the course of this thesis, Ellen has taught me how to conduct research and use a variety of limnological techniques. She has also spent more time on my thesis than I (or she) first thought was necessary for a supervisor. Thank you very much Ellen. And thanks to my committee members, Lito Arocena and Max Blouw, who helped me through my several revisions.

I received excellent support from many people in the community. Diana Gilbert and the Tabor Lake Management and Rehabilitation Committee were always enthusiastic about my research. Also, Bruce Carmichael at the Ministry of Environment, Lands and Parks provided me with sampling equipment and much of the background information on Tabor Lake. Thank you for your help, and may I suggest changing the name of the committee to an easily remembered acronym? I never could get it right!

During my sampling season, I recruited many friends to help me with sampling, and I would like to thank Adam Buller, Tanya Perrault, Kathy Buhler, Dan Bernier, Barry Pierce and everyone else who volunteered to help me. I was also fortunate enough to meet Stan Dmitrisanovc—he operates weed harvester. Stan has lived on Tabor Lake for more than 30 years and has told me stories about Tabor Lake that cannot be found in ministry records. I was always welcomed in his home and treated as part of the extended family, for which I am sincerely grateful.

My laboratory samples would still be frozen if it weren't for the assistance of David Dick, Richard Crombie, Jennifer Wilson, Jill Smith and Peter McEwan. They were extremely helpful getting me started and were always available when I asked. Ron Poirier was an invaluable friend who introduced me to the world of GIS and digital information, usually over several beer.

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Chapter One
Introduction to Tabor Lake

1.1 Eutrophication and management issues of Tabor Lake

Eutrophication is a common and pervasive problem in lakes and reservoirs around the world. It has been repeatedly demonstrated that increasing phosphorus concentrations in fresh water bodies is the leading cause of eutrophication (Vollenweider, 1968; Likens, 1972; Schindler, 1974; Jones and Bachmann, 1976; Schindler 1977). In general, a lake's drainage basin plays a vital role in nutrient loading because phosphorus exists almost entirely in particulate form (Rigler, 1973) and must be carried from its point of origin to a lake.

In some lakes, reversing eutrophication has been straightforward. The diversion of urban wastewater away from Lake Washington between 1963 and 1968 decreased the total phosphorus entering the lake by 72% and resulted in a rapid recovery of water quality (Edmondson and Lehman, 1981). Prior to the diversion, the average phosphorus concentration in Lake Washington was 61 μg/L and chlorophyll-a concentration was 35 μg/L. By 1975, phosphorus and chlorophyll-a concentrations were reduced to 17 μg/L and 7 μg/L, respectively (see Table 11, Edmondson and Lehman, 1981).

Eutrophication can also be caused by a process known as internal loading, where phosphorus is delivered from the sediment to the water column. This process has been identified as a major source of phosphorus in Tabor Lake (Rex and Carmichael, 1995) and is more difficult to reverse than the eutrophication of Lake Washington because the phosphorus is stored within the lake's sediment. Several mechanisms can directly or indirectly deliver phosphorus from the sediment back to the water column, such as hypolimnetic anoxia, sediment resuspension, bioturbation and macrophyte senescence (Bostrom *et al.*, 1982).

In order to determine where to focus management efforts, it is important to identify the source and magnitude of external and internal loadings of phosphorus. A study of Tabor Lake and its watershed during the 1995 open water season was conducted in order to quantify the major sinks and sources of phosphorus, and the timing of phosphorus transfer between the sources and sinks. The approach for this study was determined from an evaluation of a preliminary data set which was collected from Tabor Lake in 1994 by volunteer samplers plus data available from the Ministry of Environment, Lands and Parks (Carmichael, 1994) which had been collected intermittently over the period of 1974 to 1992. These data allowed for the identification of information gaps and problem areas in the phosphorus budget. The next two sections of this introduction evaluate these preliminary data sets in the context of phosphorus sinks and sources and show how the study questions and research focus were developed.

1.2 External loading of phosphorus into Tabor Lake: A description of the watershed

Tabor Lake is situated about 10 km east of Prince George, B.C. (Figure 1.1). It is oval in shape, approximately 3.5 km long and 2 km wide, covering an area of 408 ha. The size of the watershed is 29.0 km^2 (Ward, 1995). The Tabor Lake watershed has a common ancestry with many other lakes in this region; most were created after the retreat of the last ice-age. The surficial material is characterized primarily by unconsolidated deposits, left behind from the glaciers, which range in size from fine silts and clays up to large stones and even boulders (Dawson, 1989). More recently, the Tabor Lake watershed has been home to two small sawmills, one located near the outflow and the other located on the north-west shore (pers. comm. S. Dmitrasinvc). Also, the eastern portion of the watershed experienced an intense fire during 1961, known as the Grove Fire, which burned most of the forest on Tabor Mountain.

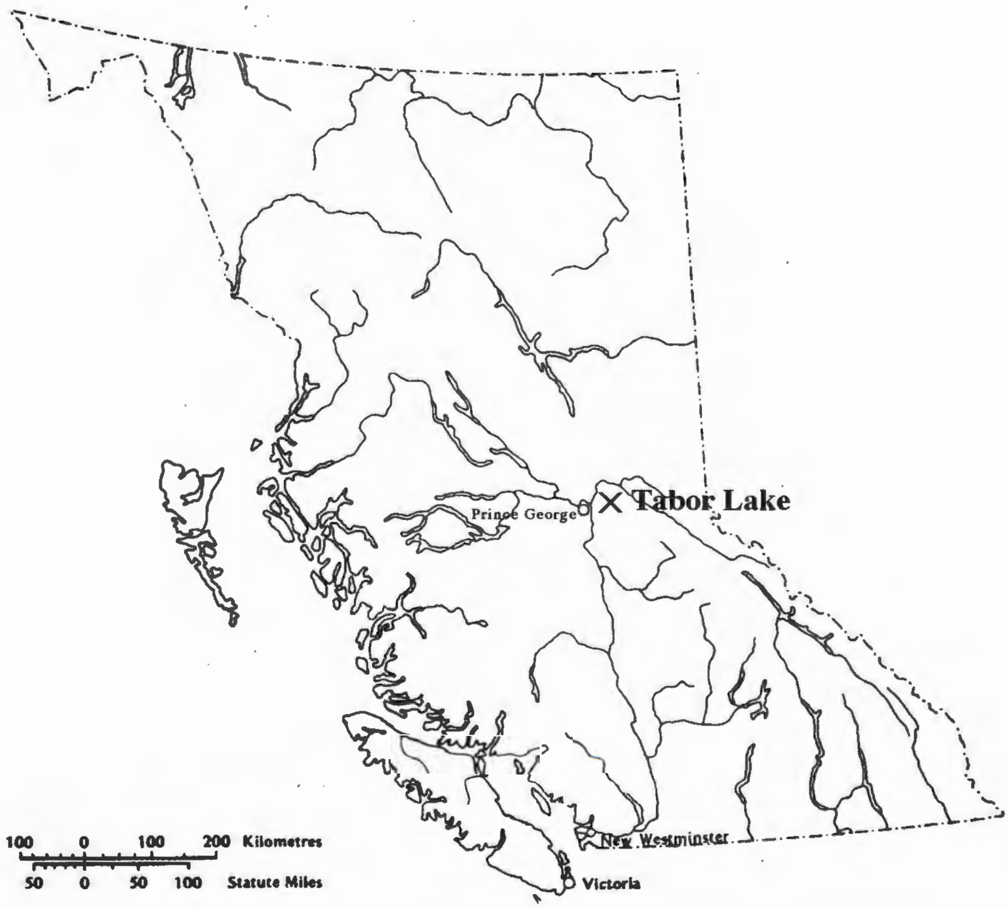


Figure 1.1 Location of Tabor Lake

Currently, agriculture in the watershed is minimal, with a small hobby farm and some pasture land interspersed among residential development. Most of the residential dwellings are in the western portion of the watershed with only sparse development near the outflow at the south-west end. This is the primary source of human activity around Tabor Lake, with an estimated 166 residences in the drainage basin (Carmichael, 1994). Most of the residences use sewage lagoons instead of septic fields because the surficial material of the Tabor Lake watershed is primarily fine grain lacustrine material and has poor drainage. In the past, the sewage lagoons have occasionally overflowed during winter, draining 'grey water' and therefore nutrients into Tabor Lake. The eastern portion of the watershed is primarily forest and logging in the watershed is minimal. Both the historical events (sawmills and fire) and the current human activity represent potentially large sources of nutrients for Tabor Lake.

Some watershed residents, concerned about the quality of their lake, state that during the early 1970's Skaret Creek was anthropogenically diverted from its original path, by-passing Tabor Lake altogether (Figure 1.2). Maps predating 1970 show Skaret Creek feeding into Tabor Lake. These residents argue that the original diversion of Skaret Creek decreased the flushing rate of Tabor Lake, thereby promoting eutrophication. They believe that by redirecting the creek back into Tabor Lake, the process of eutrophication will be reversed.

Gottesfeld (1995) conducted a watershed assessment on Skaret Creek and found several remnant channels from the alluvial fan of Skaret Creek. Preliminary studies show one or more outlets of the alluvial fan drained into Tabor Lake over 30 years ago. Gottesfeld suggests that these channels were abandoned between the mid 1960's to mid 1970's, possibly the result of debris flow deposition or logging road construction. Currently, Skaret Creek drains into Tabor Creek, downstream of the Tabor Lake outlet.

In 1993, a paleolimnological study was done on a single Tabor Lake sediment core to infer past lake-water conditions (Reavie *et al.*, 1995). The study found an increase in

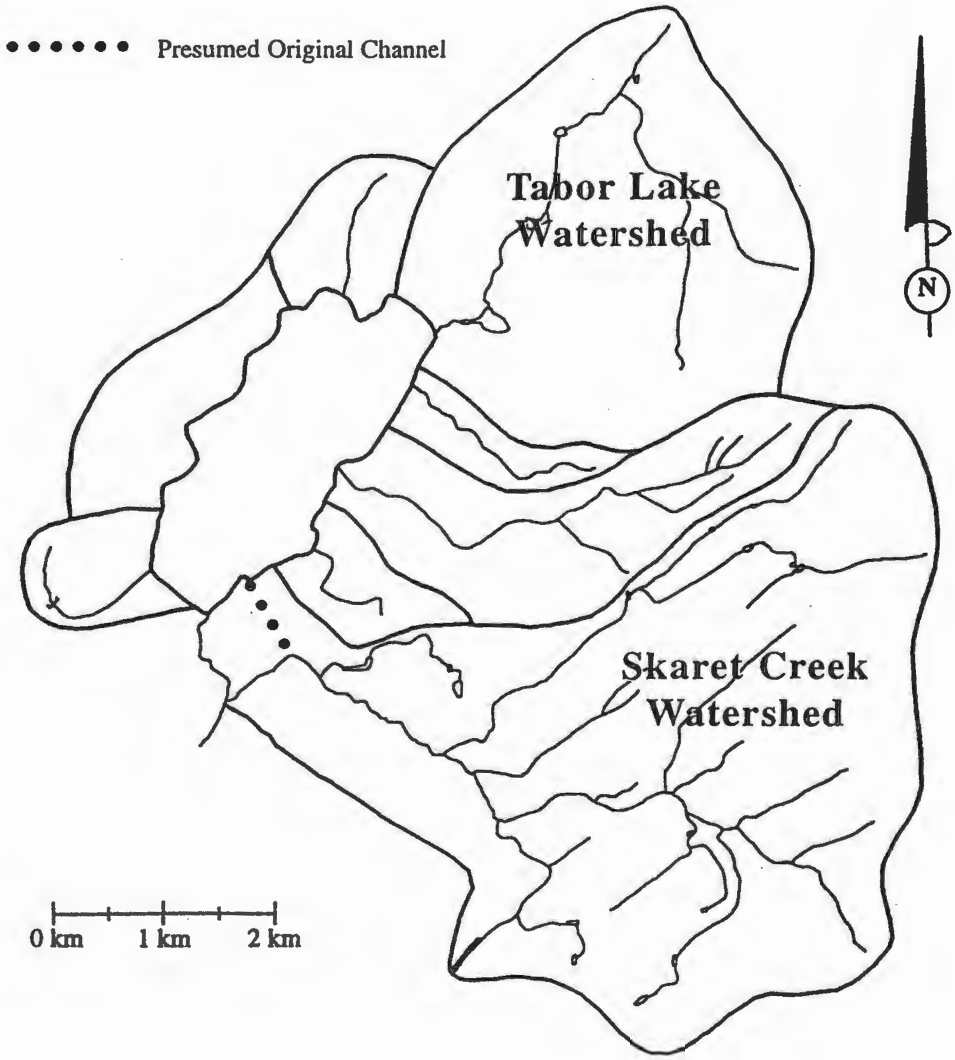


Figure 1.2 Tabor Lake and Skaret Creek watersheds. The dotted line represents where Skaret Creek presumably drained prior to diversion.

abundance of the diatom, *Stephanodiscus hanschi,* between 1960 and 1972. This species of diatom is usually found in eutrophic conditions. Reavie *et al.* (1995) have also identified a distinct ash layer in the sediment core which corresponds to the Grove Fire. The increase in *Stephanodiscus hanschi* found in the Tabor Lake sediment follows the deposition of this ash layer. Although this study cannot pinpoint the source of eutrophication, it narrows the range of historical possibilities.

1.3 Internal loading within Tabor Lake: Results from the 1994 volunteer lake monitoring program

The cumulative impact of natural and anthropogenic activities within the watershed (the two sawmills, the Grove Fire and residential development) has resulted in a large reserve of phosphorus being accumulated in the sediment. Analysis of sediment samples from depth intervals of each metre along the bottom of Tabor Lake (E. Petticrew, unpublished data) show a mean phosphorus concentration of 1318 μg/g of dry sediment (S.E.= 99 μg/g; n=10). Preliminary mass balance calculations of Tabor Lake's total phosphorus load (Figure 1.3) show that phosphorus stored in the sediment outweigh the phosphorus found in other compartments of the lake by several orders of magnitude.

Sediment phosphorus can be released into the water column through a variety of mechanisms, including diffusion, wind-induced turbulence and aquatic plant senescence (Bostrom *et al.* 1982). During hypolimnetic anoxia, diffusion rates of phosphorus from the hypolimnetic regions are often considered the dominant loading mechanisms. Bostrom *et al.* (1982) also stressed the importance of wind-induced turbulence on phosphorus release from sediments. In lakes with extensive littoral zones, wind-induced turbulence can increase release rates markedly.

Extensive growth of aquatic macrophytes can also mediate phosphorus release from the sediments. It has been demonstrated that macrophytes take up most of their phosphorus from the sediments (Barko and Smart, 1980; Carignan & Kalff, 1980) and upon

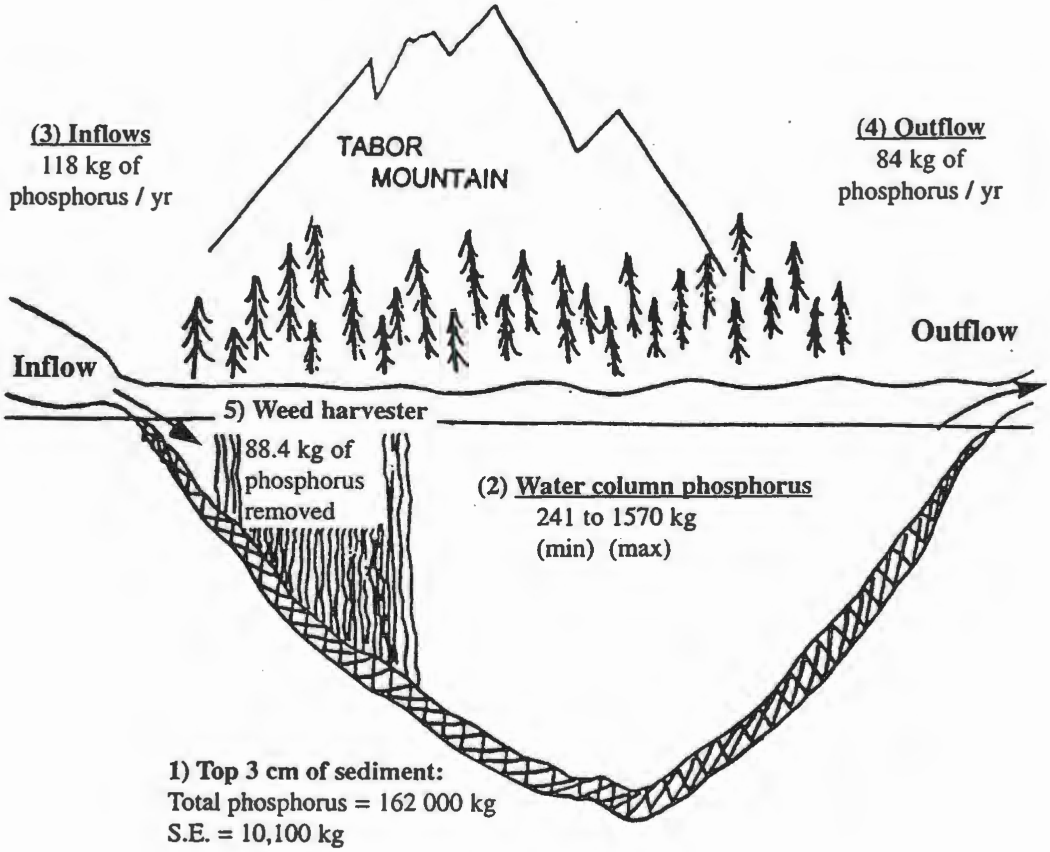


Figure 1.3 Preliminary 1994 Tabor Lake phosphorus budget.

Values were taken/calculated from:

(1) Sediment phosphorus estimate is based on samples taken during summer 1994 (Petticrew, unpublished data, 1994).

(2) Water column phosphorus: Data from 11 years of sampling, between 1973 and 1993 (Carmichael, 1994).

(3) Annual phosphorus inflows: This value from table 14 (Carmichael, 1994).

(4) Annual phosphorus outflows: This value from table 13 (Ward, 1995).

(5) Macrophyte phosphorus removal: Taken from table 18 (Carmichael, 1994).

senescence, leach phosphorus from their tissues into the water column (Carpenter, 1980; Landers, 1982; Gabrielson *et al.*, 1984; Rorslett *et al.*, 1985). Considering the extensive and dense growth of macrophytes in Tabor Lake, these plants may act as major conduits for phosphorus release from the sediments, promoting eutrophication in Tabor Lake.

Results from a volunteer monitoring program in the open water season of 1994 indicate that potentially two internal loading mechanisms occur in Tabor Lake. The first occurs in the summer months during periods of hypolimnetic anoxia. The second loading mechanism takes place during late summer/early autumn, and may be the result of phosphorus leaching from senescing macrophytes.

Phosphorus Concentrations

Weekly phosphorus samples taken from one metre above the bottom of Tabor Lake (deep hole station) show a strong inverse correlation with concentrations of dissolved oxygen (Figure 1.4). The extreme rise in phosphorus concentrations correspond with periods of thermal stratification and low oxygen concentrations. Although zero oxygen concentrations were never recorded, they are inferred because the test kit used for oxygen determination does not accurately give readings below 1.0 mg/L. Also, the water samples were taken at 1.0 metre above the bottom which does not necessarily represent conditions where the redox reactions occur—at the sediment-water interface. These results strongly suggest that classic internal loading is occurring in Tabor Lake during periods of hypolimnetic anoxia.

Observed flux in phosphorus concentrations after anoxic release

Between August 28 and October 2, 1994, dissolved oxygen concentrations at the bottom of the lake range from 7.4 mg/L to 10.4 mg/L, indicating that oxygen was introduced into bottom waters during periods of mixing. However, during this period of thermal destratification and mixing, phosphorus concentrations in the surface waters

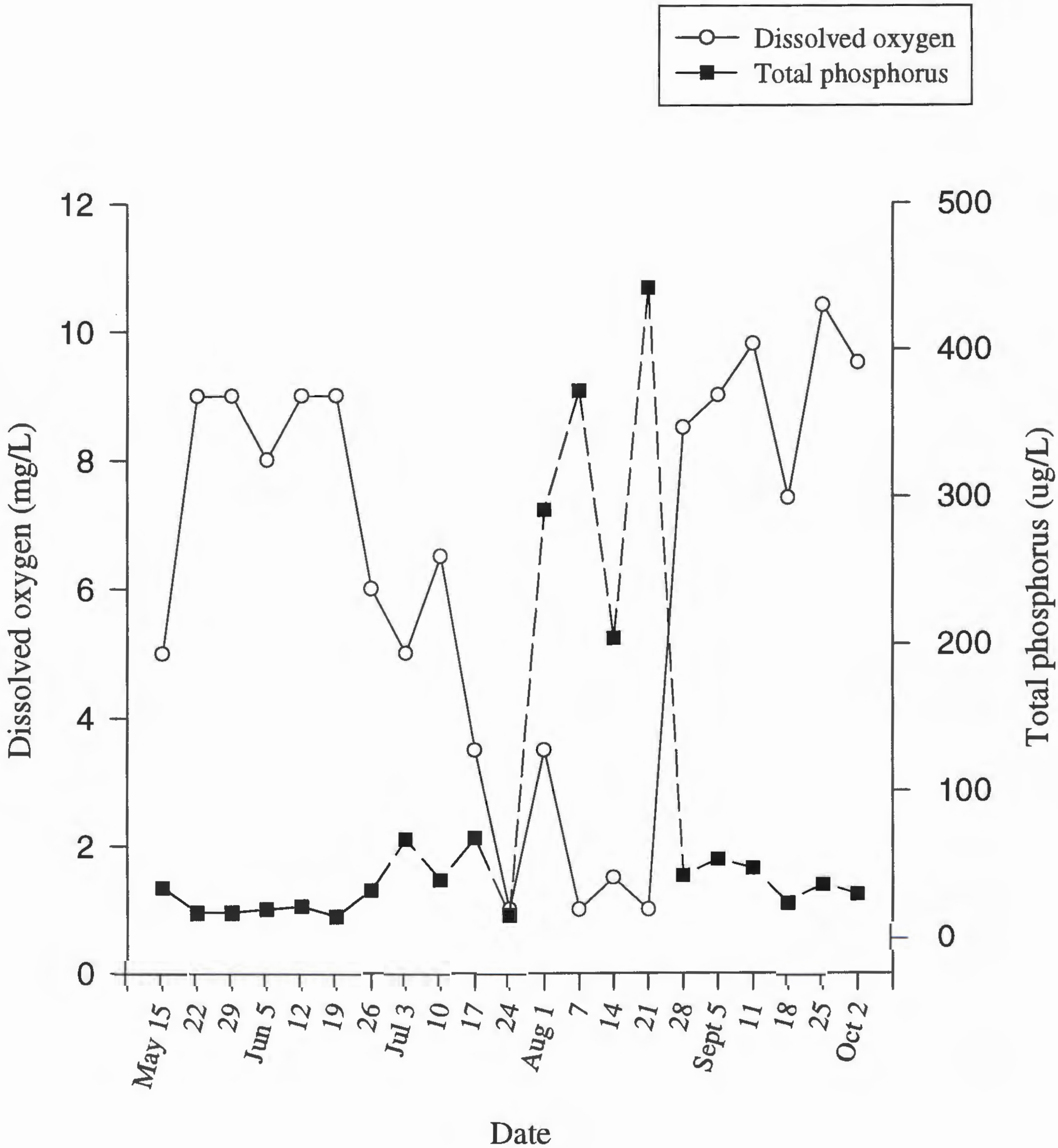


Figure 1.4 Phosphorus and oxygen concentrations one metre off the bottom at the deephole station of Tabor Lake, 1994.

fluctuate (Figure 1.5), while the bottom waters are stable around 40 µg/L. This indicates a potential source of phosphorus to the surface waters, presenting the question: *What other loading mechanism(s) can cause this flux in phosphorus levels?*

Aquatic macrophytes are known to release phosphorus from their tissue when they die. Since the phosphorus fluctuations occur during the end of the growing season, it is possible phosphorus release from Tabor Lake's extensive weed beds might account for the anomalous autumn phosphorus levels. However little is known about the role macrophytes play in the productivity of Tabor Lake (Carmichael, 1994).

The dominant macrophyte in Tabor Lake is *Elodea canadensis* and accounted for 95% of the observed biomass of the macrophyte community in Tabor Lake in 1993 (Carmichael, 1994). Information regarding the extent of the weed bed and the average biomass in Tabor Lake is limited. In 1990, distribution of the aquatic plant community in Tabor Lake was mapped by B.C. Ministry of Environment, Lands and Parks (Figure 1.6). This map shows that macrophytes do not extend beyond depths of 3 metres, but 1995 SCUBA surveys have shown macrophytes beyond 4 metres in some locations, indicating that the macrophyte community is expanding its area of colonization.

1.4 Research objectives and hypothesis

The first remediation technique used in Tabor Lake began in 1991 when a mechanical weed harvester began removing some of the dense growths of *E. canadensis* from the littoral zone. In 1994, a Tabor Lake task force was set up to determine what remediation techniques were available, appropriate and cost effective for this system. Options they reviewed included: aeration, weed blanketing, hypolimnetic withdrawal, sediment liming, sediment dredging, weed harvesting and water level control. The effectiveness of these techniques require that a quantitative estimate of the major sinks and sources of phosphorus in Tabor Lake be available. The preliminary phosphorus budget for Tabor Lake has identified that the largest compartment of phosphorus is contained within the sediment.

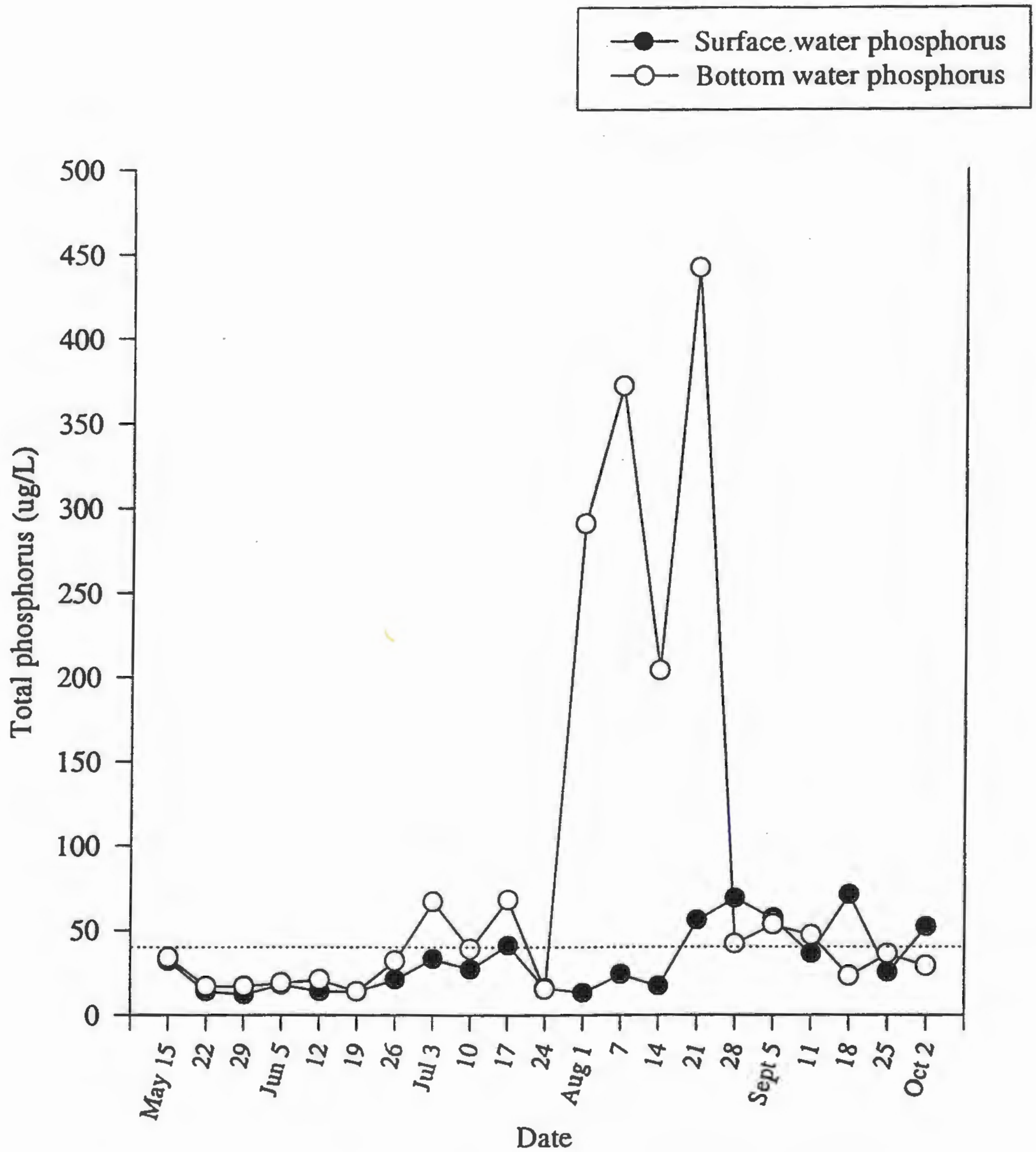


Figure 1.5 Phosphorus concentrations of surface and bottom water at Tabor Lake deep hole station, 1994. Dotted line represents total phosphorus concentrtation of 40 ug/L.

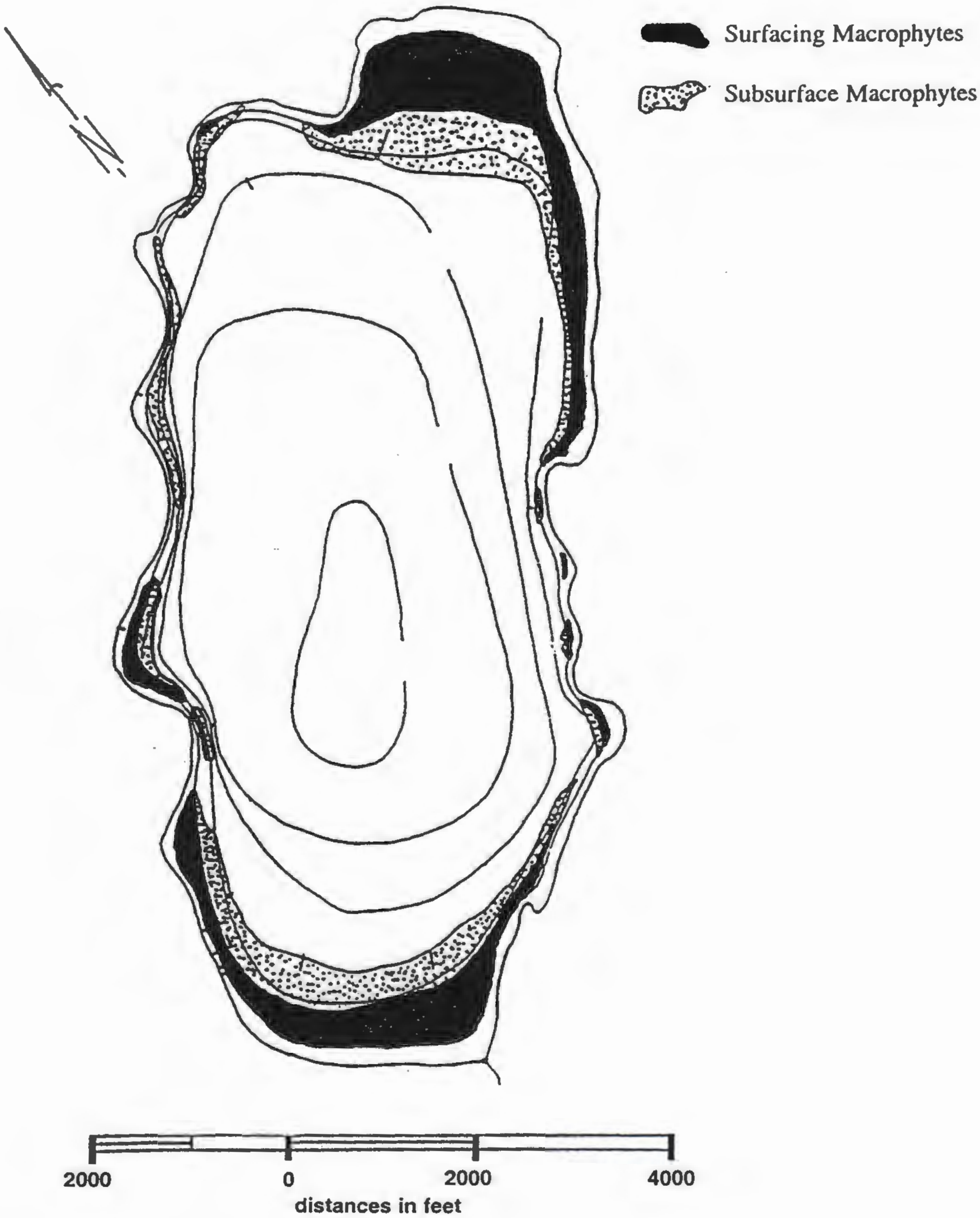


Figure 1.6 Distribution of aquatic plant communities in Tabor Lake, 1990. Bathymetric contour intervals are spaced 5 feet apart (B.C. Ministry of Environment, Lands and Parks)

However, several gaps in the data still exist. Figure 1.7 identifies five phosphorus sinks and sources which are missing from the phosphorus budget, thereby outlining the areas which need to be studied.

The objective of this thesis is to quantitatively assess the major sinks and sources of phosphorus in Tabor Lake by filling in the data gaps previously mentioned. The data from the 1994 Volunteer Lake Monitoring Program indicates that during the summer season, hypolimnetic loading is the major source of phosphorus. However, during early autumn, a second loading occurs within the lake. It is hypothesized that senescence of the extensive macrophyte community accounts for the increase in autumn phosphorus.

To test this hypothesis, two experiments, one in-vitro and one in-situ, are conducted to generate two phosphorus leaching estimates. The in-vitro leaching experiment measures the release of phosphorus from senescing macrophytes over 1-week and 3-week periods. The in-situ experiment compares phosphorus concentrations between summer and winter macrophyte samples. The results from both experiments are compared to the results from weekly phosphorus monitoring in Tabor Lake in order to identify which leaching estimate best predicted the observed phosphorus loading in Tabor Lake.

1.5 Study approach

Tabor Lake in a regional context (chapter 2)

This chapter uses exploratory data analysis to identify morphometric and watershed variables which are predictors of spring phosphorus for 39 lakes in the Prince George Region, including Tabor Lake. A total of 22 environmental variables were analysed for their ability to predict spring phosphorus. Two multiple regression models were developed, one using stepwise regression and the other using limnological theory to determine significant predictor variables. These variables are useful to lake managers in establishing the limits of water quality improvement for Tabor Lake.

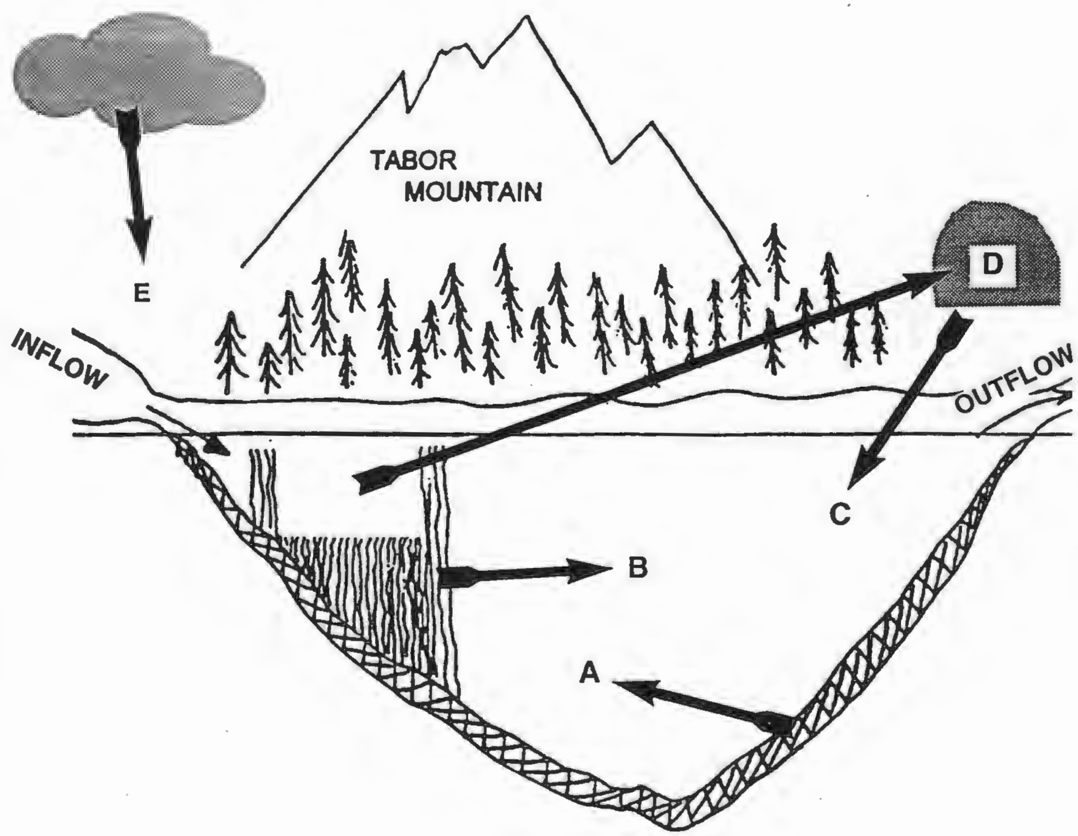


Figure 1.7 Gaps in the Tabor Lake phosphorus budget.

A: Release of phosphorus from the sediment during hypolimnetic anoxia

B: Release of phosphorus from senescing macrophytes

C: Release of phosphorus from harvested macrophytes on shore

D: Removal of phosphorus from harvested macrophytes

E: Addition of phosphorus from atmospheric deposition

External sinks and sources of phosphorus (chapter 3)

An objective of this study is to revise preliminary estimates of phosphorus loading from incoming tributaries and phosphorus release through Tabor Creek. Also, the preliminary phosphorus budget for Tabor Lake (Figure 1.3) does not include phosphorus loading from atmospheric deposition. An estimate of atmospheric loading is presented. A simple input-output model is developed to estimate the net phosphorus retention/loss for 1995.

Internal loading of phosphorus in Tabor Lake (chapter 4)

The internal loading of phosphorus from two sources is evaluated: (1) hypolimnetic loading during periods of anoxia, and (2) senescing macrophytes. Weekly monitoring data collected from the surface and bottom waters of the deep hole were used to estimate loading from the hypolimnion. To test the hypothesis that the aquatic macrophytes account for the flux in autumn phosphorus concentrations in Tabor Lake, an in-vitro experiment and an in-situ experiment were conducted. The in-vitro leaching experiment measured the release of phosphorus from senescing macrophytes in a light-sealed glass jar, and the in-situ experiment compared phosphorus concentrations between summer and winter macrophyte samples. Both experiments were used to generate whole lake estimates of phosphorus leaching from senescing macrophytes. The results from both experiments were tested against weekly phosphorus monitoring in Tabor Lake to identify which leaching estimate best predicted the observed phosphorus changes in Tabor Lake.

Analysis of macrophyte harvesting in Tabor Lake (chapter 5)

An estimate of the total phosphorus removed from Tabor Lake during the 1995 harvesting season is determined for inclusion in the final phosphorus budget. As well, a lakeside experiment was conducted on phosphorus release from harvested macrophytes along the shore to evaluate the leaching rate over a 25 hour period.

Management implications and areas of further research (chapter 6)

A review of the research is presented, with a revised phosphorus (P) budget. This new budget includes revised estimates of P loading from atmospheric deposition, surface runoff, total phosphorus in the water column, phosphorus present in macrophyte biomass, phosphorus removed from macrophyte harvesting and phosphorus loss from Tabor Creek. Several remediation options are discussed in context of the final phosphorus budget. Finally, areas of further research on Tabor Lake and lakes within the Prince George region are discussed.

Biomass estimation (Appendix A)

A prerequisite to determining the impact of senescing macrophytes on the total phosphorus budget of Tabor Lake is to estimate the biomass of macrophytes in the lake. Following the techniques outlined by Maceina and Shireman (1980) and Duarte (1987), an echosounder with chart recorder to was used to estimate total biomass of macrophytes in Tabor Lake. SCUBA quadrat harvests were collected to calibrate the echosounder chart tracings. A new technique in analysing the chart recordings was developed to improve the predictive power of this technique.

Chapter Two

Prince George regional lake survey

2.1 Introduction

Lake managers are often faced with the task of setting goals for water quality. However, unrealistic water quality objectives can be set if the natural loading rates of phosphorus are not assessed (Omernik, 1987). Also, morphometric characteristics of a lake influence how it is affected by phosphorus loading (Vollenweider, 1975; Dillon and Rigler, 1975). Therefore, assessing the factors that influence phosphorus loading from the watershed, in conjunction with the morphometric characteristics of a lake are important for managers in setting realistic water quality objectives.

A conceptual model of a lake ecosystem is required to identify the variables which influence phosphorus concentration. Likens and Borman (1974) describe a lake and its catchment area as the basic ecosystem unit because the terrestrial and aquatic portions of the watershed are intricately linked by movement of materials from land to water. This conceptual model has been useful in developing successful empirical models which predict phosphorus concentration or loading from morphometric and watershed variables (Dillon and Rigler, 1975; Vollenweider, 1975; Reckhow and Simpson, 1980). This conceptual model also provides a useful template to choose environmental variables for exploratory data analysis.

In this study, the conceptual model proposed by Likens and Borman (1974) is used to select 22 different morphometric and watershed variables which may influence the concentration of spring phosphorus in 39 lakes in the Prince George region. Two modeling techniques are used to identify which environmental variables are useful predictors of spring phosphorus. The first modeling technique uses an exploratory data analysis technique known as stepwise multiple linear regression to select statistically significant variables, and is referred to as the “Model I”. The second modeling technique selects variables which have been reported in the literature to influence phosphorus loading, and this model is referred to as "Model II".

Location of study area

The 39 lakes used in this study are located in the north-central interior of British Columbia. An approximate outline of this area is shown in Figure 2.1 by a black ovoid, which encompasses an area around the city of Prince George.

2.2 Methods and materials

Spring phosphorus (response variable) was measured in the 39 sample lakes by the Ministry of Environment, Lands and Parks (MELP) between 1976 and 1987 (unpublished data). Spring phosphorus is a useful indicator of lake trophic status because it measures the phosphorus concentration of a lake when the lake is fully mixed from spring overturn, and should have an even distribution of phosphorus throughout the water column. Also, spring phosphorus is measured when the inputs of phosphorus from the watershed are at their highest level—during spring melt.

A total of 22 predictor variables, ranging from morphometric information to natural and anthropogenic watershed variables, were measured for this study. This is not an exhaustive list of environmental variables, but were chosen to access as many of the "potentially significant" variables as possible from the sources of data which were available for this study. See Appendix B for the entire data set.

The type of data collected for each of the 22 variables falls into one of two categories—continuous or ordinal data. Continuous data was the preferred data type for this study and were used whenever available. When continuous data was not available, ordinal data were obtained from maps using visual estimates. This discrepancy in data collection introduces an inherent bias into the dataset and is discussed later.

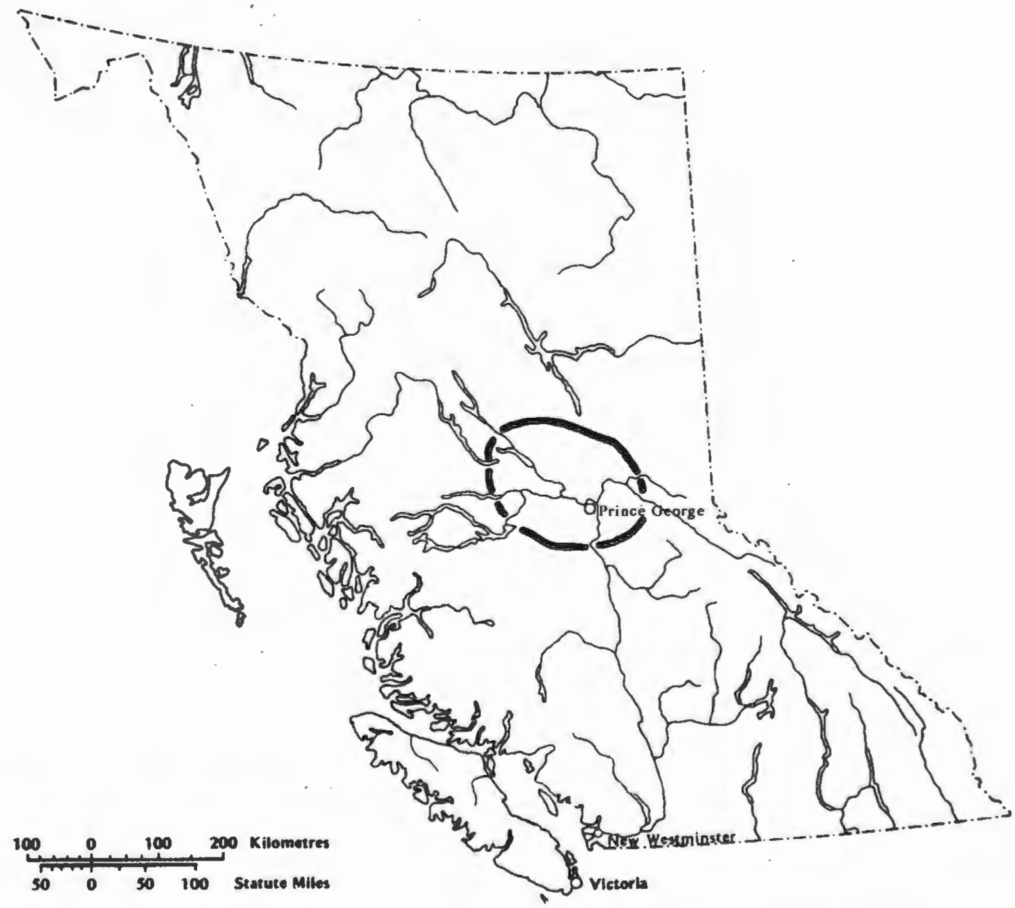


Figure 2.1 Approximate location of sample lakes (and their watersheds) used in the study

Morphometric data

Morphometric data was compiled from the MELP, Fisheries Branch database. Data for each lake includes: surface area; maximum depth; mean depth; and volume. Mean depth is calculated from lake volume divided by surface area. Continuous data was available for all morphometric variables.

Watershed data

Eight categories of watershed variables were selected as "potentially significant" predictors of spring phosphorus: wetlands, agriculture, number of habitations, forest age class, soil texture, slope, watershed area and elevation. The first four categories listed contain multiple sub-variables, such as "agriculture within the watershed", "agriculture adjacent to streams" and "agriculture around the lake perimeter", and represent locational differences within the watershed. By distinguishing the location of these variables, the additional information produced may be of some predictive value.

Wetlands:

This variable was chosen because wetlands have been shown to act as nutrient sink, thereby reducing the phosphorus load to a lake (Marble, 1992; Olson, 1993). There are two sets of data in this category: wetland areas adjacent to streams and wetland areas within the entire watershed. Wetland areas were identified on 1:50,000 scale topographic maps. The ordinal ranking was conducted as follows:

0 = less than 1 % of the stream/watershed contains wetlands;
1 = between 1% and 10% of the stream/watershed contains wetlands; and
2 = more than 10% of the stream/watershed contains wetlands.

Agriculture:

Agricultural activity within a watershed has been shown to increase phosphorus loading into a lake (Dillon and Kirchner, 1975; Omernik, 1987; Johnes *et al.*, 1996), and was chosen as a predictor variable for this reason. Agricultural areas were estimated on 1:50,000 scale topographic maps. The map does not make a distinction between agriculture, pasture or residential development. They show up as unshaded areas on these maps and are all classified as agricultural area in this study. This should not be a problem because there is minimal residential development compared to the agriculture and pasture land in these watersheds. The ordinal ranking of agriculture areas "within the watershed" and "adjacent to streams" followed the same criteria as that used for wetland classification. However, the ordinal ranking of agricultural areas surrounding lakes was stratified more extensively because visual estimates could measure this variable with greater precision. The ranking was conducted as follows:

0 = less than 5 % of the lake perimeter contains agriculture;
1 = between 5% and 10% of the lake perimeter contains agriculture;
2 = between 10% and 20% of the lake perimeter contains agriculture;
3 = between 20% and 35% of the lake perimeter contains agriculture;
4 = between 35% and 50% of the lake perimeter contains agriculture; and
5 = more than 50% of the lake perimeter contains agriculture.

Habitations

The presence of habitations is often associated with increased phosphorus loading and can be caused by several human activities, such as poor sewage treatment and the use of detergents rich in phosphates (Sawyer, 1973). Habitations were identified as black squares on the 1:50,000 topographic maps, and can include a variety of structures from barns to houses. Although, no distinction is made between the type of structure, it is assumed that human activity occurs (or at one time occurred) at these locations, and therefore all sites represent the potential for phosphorus loading. The ordinal ranking for habitations is based on the following criteria:

0 = less than 10 inhabitants adjacent to streams, the lake perimeter or in the watershed;
1 = between 10 and 100 inhabitants adjacent to streams, the lake perimeter or in the watershed;
2 = greater than 100 inhabitants adjacent to streams, the lake perimeter or in the watershed; and
3 = a community, defined by having a "settlement boundary" on the map, and is adjacent to streams, the lake perimeter or within the watershed.

Forest Age Class

Forest age class was used as a predictor variable because it allowed for the separation of recently harvested areas from unharvested areas. Separation of these forest ages is potentially valuable to this study because harvesting is known to increase the phosphorus concentration of surface runoff (Hobbie and Likens, 1973; Brownlee *et al.*, 1988). "Forest Cover Age Class" maps from the Ministry of Forests (1:250,000) were used to estimate the percent cover of three forest age classes (young, medium and old) in each watershed. The youngest forest is from age class 1 with forests being less than 20 years in age. Logging is the dominant form of disturbance in this age class (pers. comm., C. DeLong). Age classes 2, 3 and 4 represent the medium age forest, which range between 20 and 80 years old. The last grouping included age classes 5, 6 and 7, representing the old growth stands of timber. Each of the three age groups (young, medium and old) were measured ordinally using the following criteria:

0 = less than 1% coverage of watershed;
1= between 1% and 10% coverage of watershed;
2 = between 10% and 25% coverage of watershed;
3 = between 25% and 50% coverage of watershed; and
4 = greater than 50% coverage of watershed.

Soil texture and slope

These variables were extracted from the "Soils Landscapes of Canada" maps (1:1,000,000) produced by Agriculture Canada (1994). Often, several soil types were present within the same watershed and in these situations, the dominant soil within the watershed was used. "Soil texture" was selected for this study because (a) different soil textures might influence the phosphorus loading since there is greater surface area for phosphorus adsorbption to fine grain material (clays) than with coarser material (sands). Slope may also influence phosphorus loading because slope is often related to the rate of erosion in a watershed, and therefore the rate of phosphorus loading.

The variable, "soil texture" is not an intrinsically numeric variable, however enumeration of this variable followed a linear format based on the relative size the material. The two variables measured were classified using the following criteria:

"Soil texture"

1 = clay;
2 = clay loam;
3 = loam;
4 = sandy loam;
5 = sandy.

"Slope"

4 = between 4% and 10% slope;
10 = between 10% and 16% slope; and
16 = between 16% and 31% slope.

Watershed area

The area of a lake's watershed was chosen for this study because increases in phosphorus loading from the watershed have been shown to correlate with increases in watershed size (Omernik *et al.*, 1991). The watersheds were outlined on 1:50,000 scale maps and measured with a digital planimeter. Many of the watersheds traced out had a large number sub-basins, and these areas were also traced out with the digital planimeter. The number of sub-basins within each watershed and the average area of all sub-basins within each watershed are recorded.

Elevation

The elevation of each of the sample lakes was recorded in metres above sea level. This variable was chosen as a predictor variable because elevation can be a useful indicator of precipitation, which can deliver phosphorus from atmospheric deposition, and thermal characteristics, which may influence internal loading processes prior to spring turnover.

Isolating variable clusters

One problem with building models occurs when some of the predictor variables are clustered, or are from the same functional group (Hakanson and Peters, 1995). Clusters refer to families of correlated variables which describe a similar phenomenon. For example, mean depth and maximum depth both measure a slightly different aspect of lake depth. However both of these variables are highly correlated and represent a cluster. Variables which belong to the same cluster can often be replaced with other cluster variables without significantly affecting the outcome (Hakanson and Peters, 1995). To isolate important clusters, a Pearson correlation matrix was created for all parameters collected for this study, and an r-rank table has been produced from the results of the correlation. A representative variable from each cluster was chosen using in order to avoid problems multi-collinearity. Generally, variables with a correlation coefficient of $r > 0.50$ were considered part of a cluster except where correlated variables were from different functional groups.

Model construction

Two models are constructed in this study. The first model, referred to as 'Model I', uses a statistical technique known as stepwise multiple linear regression to identify "significant" variables (using probability of F-to-enter < 0.05). One of the problems which arises when using this technique is that predictor variables which increase the strength of the model may not reflect what is occurring in nature. Hakanson and Peters (1995) demonstrate this point effectively when they conducted a stepwise regression on 100 sets of random variables to predict pH. The stepwise regression found that 97% of the variation could be accounted for using 15 sets of these random numbers. This point is emphasized by Sokal and Rohlf (1991) who state that one cannot assume that the subset of variables found by a stepwise regression program necessarily corresponds to the most important set of variables.

A second model, referred to as 'Model II', is constructed as an alternative modeling approach to the stepwise technique which uses variables that have been identified in the literature as influencing phosphorus loading or retention. Five variables have been chosen and each variable is supported in the literature (Table 2.1). Model II is compared with Model I to identify similarities and differences between the predictor variables chosen, and the amount of variation in spring phosphorus explained by each model.

Both models are tested for violations of assumptions using leverage plots, residuals, Cook's distance and studentized residuals. Regression analysis assumes that the residuals are independent normal values. Analysing the residuals and studentized residuals can help detect violations of this assumption (Lewis-Beck, 1980). Cook's distance reveals observations that have a large influence on the estimates of the regression coefficients. The leverage value is similar to residual analyses, in that the leverage value describes the impact of single observations on the prediction of the regression line (Norusis, 1993).

2.3 Results

Identification of clusters and selection of representative variables

Table 2.2 is an r-rank table, showing the moderate, strong and very strong correlations among the 22 environmental variables collected in this study. This table provides a method for easily determining the degree of collinearity within variable clusters and representative(s) from each cluster were selected for use in the regression analyses. The other variables were discarded to avoid problems with multicollinearity, unless specified otherwise (see Methods section).

Five morphometric variables were collected (maximum depth, mean depth, surface area, lake volume and shoreline perimeter) and all demonstrate moderate to very strong clustering. Mean depth was selected as the representative variable, since it is used in other successful limnological models (e.g. Vollenweider, 1975; Dillon and Rigler, 1975).

Table 2.1. List of five variables chosen from the literature as important factors which influence phosphorus loading in lakes.

Variable	Reference(s)	Geographic Area of study
Agriculture	Omernik et al., 1988 Dillon and Kirchner, 1974	Wisconsin Ontario
*Forest Harvesting	Brownlee et al., 1988 Hobbie and LIkens, 1973	Northern B.C. New Hampshire
Mean Depth	Vollenweider, 1968 Dillon and Rigler, 1975	Global Ontario
Watershed Area	Omernik et al, 1988	Wisconsin
Wetlands	Reddy and Gale, 1994 Olson, 1993	Global not discussed

*Forest age class maps were used to identify young forests. These forests have been harvested over the past 20 years.

Table 2.2 An r-rank table of correlation coefficients between 22 environmental variables collected for exploratory purposes. This table divides correlation coefficients into three ranges of colinearity: Very strong, Strong and Moderate.

R > 0.90		*Very Strong Colinearity*
	LSurf	LVol, LShed
	LVol	LSurf
	LShed	LSurf
0.70 > R > 0.90		*Strong Colinearity*
	LPerim	LSurf, LVol, LShed
	LVol	LMeanD, LShed, SubNum
	LShed	SubNum, LSubArea,
	LMeanD	LMaxD, LVol
	AgLake	AgStrm,
	PeopLake	PeopStrm, PeopShed
	PeopStrm	PeopShed
	PeopShed	PeopLake
	SubNum	LVol, LShed, LPerim
	LSubArea	LShedArea
0.50 > R > .070		*Moderate Colinearity*
	AgShed	Elev, AgLake, AgStrm
	AgLake	Elev, AgStrm, AgShed
	LMeanD	LSurf, LPerim, SubNum
	LVol	LMaxD, LSubArea
	WetShed	WetStrm
	OldFor	LShed
	PeopStrm	AgStrm, PeopLake
	PeopLake	AgStrm, AgLake, PeopStrm
	LSubArea	LVol, LSurf, LPerim
	SubNum	LMeanD, LMaxD, LPerim

Definitions:

AgLake	*Agriculture around perimeter of lake*	*Midfor*	*Middle forest ageclass (20-80 years)*
AgShed	*Agriculture in the watershed*	*Newfor*	*Newest forest ageclass (< 20 years)*
AgStrm	*Agriculture along incoming stremas*	*OldFor*	*Oldest forest ageclass (over 80 years)*
Elev	*Lake elevation*	*PeopLake*	*People around perimeter of lake*
LMaxD	*Logarithm of maximum lake depth*	*PeopShed*	*People within the watershed*
LMeanD	*Logarithm of lake mean depth*	*PeopStrm*	*People around perimeter of streams*
LPerim	*Logarithm of lake perimeter*	*SubNum*	*Number of subbasins in watershed*
LShed	*Logarithm of watershed*	*WetShed*	*Wetland within watershed*
LSubArea	*Logarithm of average subbasin area*	*WetStrm*	*Wetland along incoming streams*
LSurf	*Logarithm (base 10) of lake surface area*	*Slope*	*(self explained)*
LVol	*Logarithm (base 10) of lake volume*	*Soil texture*	*(self explained)*

The variables which measure sub-basin area and number of sub-basins show strong collinearity with watershed area, and all three variables show moderate to strong collinearity with the morphometric variables. Although a correlation exists between the watershed and morphometric data (i.e. bigger lakes occur in bigger drainages), the watershed variables are considered here as a separate cluster since both sets of variables represent functionally different aspects of the ecosystem. Watershed size will be used as the representative variable because it has been shown to be a useful predictor of phosphorus loading in lakes (Omernik *et al.*, 1991).

The ordinal data collected show two sets of clusters, one with wetlands and the other combining agricultural area and number of habitations. The wetland variables—percent along streams and percent in watershed—are expected to show collinearity because wetlands which are along streams are also measured as wetlands within the entire watershed. In this study, wetlands which are adjacent to streams was the variable chosen to predict spring phosphorus. This decision is based on the premise that wetlands adjacent to streams dampen high energy flow systems by reducing flow rate, thus promoting sedimentation and phosphorus retention, whereas wetlands not adjacent to streams are not expected to achieve sedimentation to the same degree.

The second cluster of ordinal data is comprised of six different variables, three from the agricultural area measurements, and three from measurements of the number of habitations. Since both these variables were measured in a similar way to the wetland variables (i.e. area within the watershed, area adjacent to streams and area along the lake perimeter), clustering occurred in the agricultural area measurements and with measurements of the number of habitations. The correlation between agricultural area and the number of habitations can also be explained because agriculture is a form of human activity. Although many watershed models use both agriculture and human population variables for predicting water quality (Dillon and Rigler, 1975; Johnes *et al.*, 1996; Meeuwig and Peters, in press), human settlement within the study area is sparse compared to the area of agricultural activity. The representative variable chosen for use in the regression models is 'agriculture along streams'.

Based on the cluster analysis presented above, a new set of predictor variables is listed in Table 2.3. These variables are used to develop Model I and Model II, as discussed in section 2.2.

Statistical models

The stepwise regression technique used to develop Model I, identified six variables from the revised variable list (Table 2.3) and accounted for 70% of the observed variation in spring phosphorus. Model II, which was based on limnological theory, used five variables from the revised variable list and accounted for 59% of the observed variation in spring phosphorus. Table 2.4 shows the variables chosen for each model, with the respective coefficients of variation. There is considerable overlap in predictor variables. Model I (six variable) included four of the five variables used in model II. The additional variables in Model I were oldfor (the percentage of forests having the oldest age classes) and elevation. Model II included the variable 'newfor', which represents the percentage of forests having the youngest age class.

Diagnostic analyses were conducted on the data in both models. Figure 2.2 shows the diagnostic plots for the Model I regression. The residuals and student plots demonstrate homoscedasticity. Crescent Lake is shown to be an outlier in the Cook's test, having a value of 0.37. The next highest Cook's value is 0.17, for Teardrop Lake. The leverage plot also demonstrates homoscedasticity of the predicted values with respect to influence, however the points are not as evenly distributed as in the residual and student plots.

The diagnostic plots for Model II are shown in Figure 2.3. As with the Model I, residuals and student plots demonstrate homoscedasticity. The highest Cook's value in this model also corresponds to Crescent Lake having a value of 0.26. However, this value is

Table 2.3. List of collected variables before and after cluster Revised variable List uses 'representative' variables from clusters having a high degree of collinearity.

Original Variable List	Revised Variable List
AgLake	*AgStrm*
AgShed	*Elevation*
AgStrm	*LMeanD*
Elevation	*LShed*
LMaxD	*MidFor*
LMeanD	*NewFor*
LPerim	*OldFor*
LShed	*Soil texture*
LSubArea	*Slope*
LSurf	*WetStrm*
LVol	
MidFor	
NewFor	
OldFor	
Soil texture	
PeopLake	
PeopShed	
PeopStrm	
Slope	
SubNum	
WetShed	
WetStrm	

See Table 2.2 for explanations of abbreviated variables

Table 2.4. Results from two spring phosphorus loading models. Model I uses stepwise regression to identify significant variables, while Model II uses limnological theory to identify significant variables.

Model I		Model II	
Predictor Variable	Coefficient	Predictor Variable	Coefficient
Constant	-2.874	*Constant*	-1.497
AgStrm	0.112	*AgStrm*	0.065
Elevation	0.002	*LMeanD*	-0.548
LMeanD	-0.586	*LShed*	0.200
LShed	0.200	*NewFor*	0.082
OldFor	0.091	*WetStrm*	-0.136
WetStrm	-0.204		
	R-square = 0.70 S.E. est.= 0.164 F = 12.62		R-square = 0.59 S.E. est.= 0.189 F = 9.51

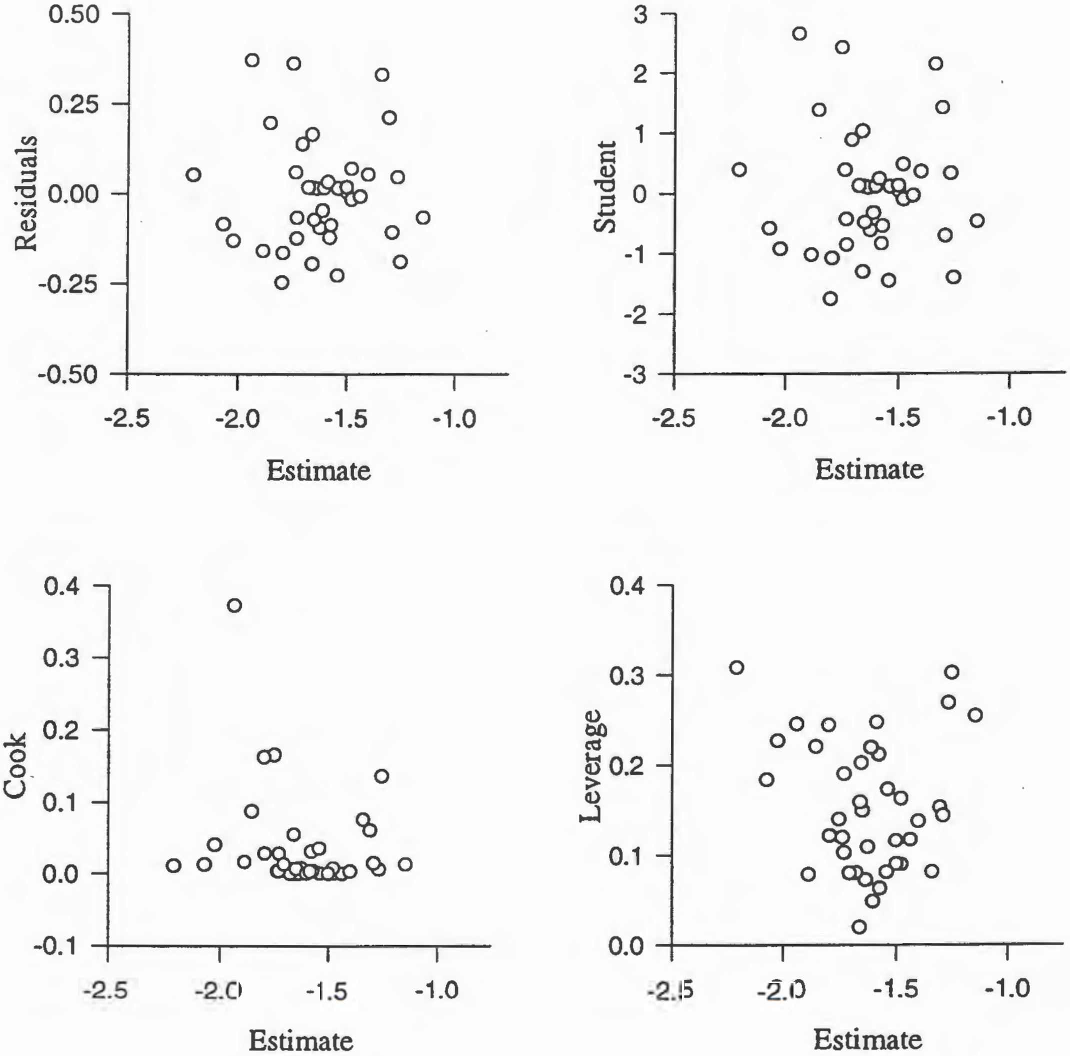


Figure 2.2 Residuals, studentized residuals, Cook's distance and leverage diagnostic plots for Model I (stepwise regression model).

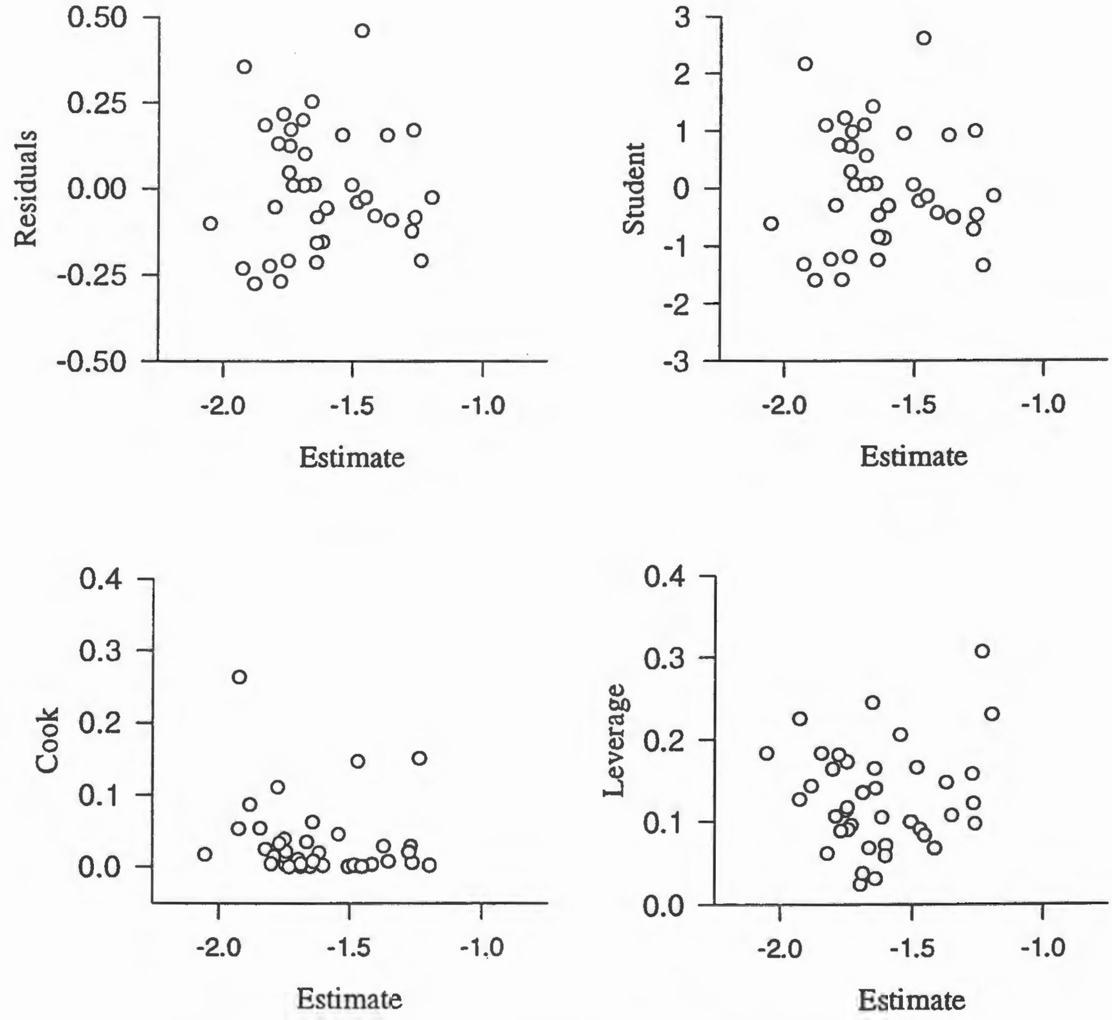


Figure 2.3 Residuals, studentized residuals, Cook's distance and leverage diagnostic plots for multiple linear regression, Model II.

lower than the Model I (0.37). The Leverage plot for model II demonstrates homoscedasticity of the predicted values with respect to influence on the regression line.

Since Crescent Lake was found to be the most significant lake which influenced the regression, a brief description of the lakes characteristics is provided. Crescent Lake's spring phosphorus was measured as 27 μg P/L (the average spring phosphorus for all lakes in the data set is 24 μg P/L), while the predicted estimate is 11 μg P/L and 12 μg P/L from Model I and Model II, respectively. This lake has a slightly higher mean depth than the average (6.5 m compared to 6.0 m) and is within a relatively small basin (6.4 km^2) compared to the average (33.5 km^2). Both wetlands and agriculture are found to make up greater than 10% along streams. The majority of forests are within the 20-80 year age group (more than 50%). Less than one percent of the watershed contains forests less than 20 years and between 1% and 10% of the forests are more than 80 years old. Further analysis and interpretation of the Cresent Lake data is presented in the next section.

2.4 Discussion

Model I and Model II represent two different exploratory modeling approaches, in terms of the *process* of selecting predictor variables. The first model selects variables using a stepwise approach while the second model selects variables using limnological theory; in other words, the choice of predictor variables is independent between the two models. Results from the modeling exercise found that the same four variables (lake mean depth, watershed area, wetland area adjacent to streams and agricultural area adjacent to streams) are predictive of spring phosphorus in both models. Since each modeling technique has inherent differences in the selection of predictor variables, the convergence of both models on the same variables strengthens the claim that these variables are useful predictors of spring phosphorus.

The variables which did not appear in both models are lake elevation, the area of old growth forests (in Model I) and the area of young forests (Model II). The selection of these variables in predicting spring phosphorus is also valuable within the context of their respective model. The functional role of these variables in predicting spring phosphorus (as well as the four variables listed in the previous paragraph) is discussed in following section.

Functional analysis of predictor variables from regression models

Both models selected the variable mean depth as a predictor of spring phosphorus. This is not unusual, since the importance of mean depth in predicting phosphorus levels in lakes is integral to several lake phosphorus models (Dillon and Rigler, 1975; Vollenweider, 1975; Reckhow and Simpson, 1980). Model I and II found spring phosphorus to be negatively correlated to mean depth, i.e. deeper lakes have lower phosphorus concentrations. The usefulness of this variable is that combines two important morphometric variables which regulate phosphorus response in a lake---volume and surface area. Volume is expected to play a role in the dilution of phosphorus delivered to a lake. However, large volumes can also occur in shallow lakes with a large surface area. In these situations the value of phosphorus dilution can be offset by increased phosphorus availability from larger littoral zones. Mean depth has the effect of balancing these two lake characteristics, and is a useful predictor of spring phosphorus concentration.

Watershed area was also found to play a significant role in spring phosphorus concentrations because as the watershed size increases, the total phosphorus input also increases (Dillon and Rigler, 1975; Omernik *et al.,* 1991; Marble, 1992). Another variable which showed a positive relationship with spring phosphorus levels was agriculture along streams. Phosphorus loading from agricultural areas has been studied for several decades, and the results from this study agree with the findings of other authors. Dillon and Kirchner (1975) found that phosphorus export from pasture lands was significantly higher than from forested lands. Omernik *et al.* (1991) found that lakes within high agricultural areas had the largest spring phosphorus values, whereas lakes within forested areas had the lowest spring phosphorus values.

The results from both models show a negative relationship between the proportion of upstream wetlands and spring phosphorus values. Wetlands process runoff through two basic systems—sedimentation and nutrient assimilation (Reddy and Gale, 1994). As phosphorus is assimilated into the biological component of the wetland ecosystem, it is prevented from traveling further downstream. The role of wetlands as nutrient sinks is also demonstrated by the current use of natural and constructed wetlands to control a variety of high nutrient pollution problems (Marble, 1992; Olson, 1993).

The newfor variable, which represents the percent of forests which have been harvested at some time over the past 20 years, was selected as a predictor variable for Model II, but was not chosen for Model I. This variable was selected because recently harvested forests have been shown to increase phosphorus loading in tributaries, compared to intact forests (Hobbie and Likens, 1973; Brownlee *et al.*, 1988). The inconsistency between the results of Model I and the literature might be explained by the 20 year time frame of this variable. Harvested areas generally do not remain without vegetation for prolonged periods. Alder and aspen tend to colonize recently harvested areas and achieve rapid growth rates a few years after an area is harvested. It is possible that this vegetative growth reduces phosphorus transport to a receiving water by consuming the available nutrients.

Model I found old growth forests to be positively related to spring phosphorus, but was not selected in Model II. It is possible that old growth forests represent the final life stage of a forest, when the nutrient uptake by the forest is less than the loss of organic material through death and decay. This theory implies a net loss of phosphorus from the forest and could enter a lake via surface runoff during spring melt. However, this theory requires further study, including greater precision in the measurement of this variable (both spatially and temporally).

The stepwise model found the predictor variable, elevation, to be a significant determining variable with a positive coefficient for predicting phosphorus concentrations. This implies that as elevation increases, so does a lake's phosphorus level. No literature was found to support these findings. This does not imply that elevation is unimportant in determining spring phosphorus, but suggests further studies are required to assess whether elevation is truly an important variable in determining spring phosphorus. Perhaps the relationship between elevation and spring phosphorus reflects variable amounts of orographic precipitation and is therefore related to erosion and transport. This is difficult to assess because limited data was collected on watershed slopes and their orientation to storm tracks.

Statistical violations and sources of error

One of the rules of multiple regression analyses is that observations of the response variable (spring P) must be independent of each other. Some of the lakes used in this study were within sub-basins of other lakes also used in this study, which may have created a dependency between these lakes. The statistical problem with dependency between response variables represents an inflated type I error (pers. comm. B. Zumbo, 1996). Zumbo suggested that a crude way to correct for this error is to start with 0.01 p-value, instead of 0.05. However, since the purpose of this model is more descriptive than predictive, this error should not affect the usefulness of the models.

Spring phosphorus data was collected over a 12 year period between 1976 and 1987. The various maps used in this study to collect data were produced between 1961 and 1993. Changes in agriculture, settlement and forestry may have occurred between the time at which spring phosphorus was sampled and the date when the maps used to collect these variables were produced.

The collection of ordinal and continuous data for use in the same model introduces a bias as a result of using measurements with two distinct levels of precision. The variables using continuous data provides the greatest precision (compared to the ordinal data) and is more likely to detect correlations with spring phosphorus. This has the overall effect of strengthening the predictive value of the continuous data versus the ordinal data.

The plot of Cook's distance (Figures 2.2, 2.3) revealed that Crescent Lake has a large influence on the regression coefficient for both models. When this data point is removed from both the stepwise model and model II, the R^2 increases from 0.70 to 0.78 and 0.59 to 0.65, respectively. Although removing this observation improves the predictive power of the model and removing outliers from a model can help establish more accurate general patterns, it should only be eliminated if the observation is erroneous (Hakanson and Peters, 1995) or if there is good evidence to suggest the lake is subject to different processes.

Model I and Model II predict the spring phosphorus concentration of Crescent Lake to be 11 μg P/L and 12 μg P/L, respectively, which is much lower than the phosphorus concentration measured by the MELP. However, a closer analysis of the agricultural and wetland attributes along the main tributary suggest the location and distance of these variables from the lake might explain the anomalous prediction of spring phosphorus.

The main tributary for Crescent Lake has a total of three sub-basins including the Crescent Lake sub-basin. Between Obert Lake (the middle sub-basin) and Crescent Lake, there is roughly 30% wetlands and 30% agriculture along the creek. However, the agricultural land occurs downstream of most of the wetland area. This suggests that the agricultural runoff is not subject to the nutrient assimilation effects of wetlands, which may explain why Crescent Lake has a higher than anticipated phosphorus value. If we change the wetland variable for Crescent Lake to zero, the predicted phosphorus concentrations rise to 29 μg/L and 22 μg/L for Model I and Model II, respectively. These predicted values are much closer to the observed phosphorus concentration of 27 μg/L, compared to the unmodified predictions. Although the wetland variable recorded for Crescent Lake probably overestimates the effect of nutrient assimilation by wetlands, it should not be removed from the data set. Instead, it represents a source of information about this model, revealing weaknesses in the modeling procedure, and should be used to construct new modeling techniques. This will be discussed later.

Conclusion

Model I and Model II account for 70% and 59% of the observed variation in spring phosphorus, respectively. These values are higher if Crescent Lake is removed from the data set (78% and 65%, respectively). Model I and Model II also show a “convergence” towards similar predictor variables, which strengthens the claim that these variables are useful predictors of spring phosphorus. The predictor variables which were not common to both models are more difficult to interpret because they are limited to only one modeling approach. However, they suggest a focus for further research about the role of old growth forests and elevation in determining spring phosphorus levels. The uncertainty about the influence of recently harvested forests on spring phosphorus concentrations might be resolved if this variable is measured within 2 or 3 years after harvesting occurs, rather than the range of 20 years used in this study.

Model I and Model II provide valuable insight into the importance of mean depth and watershed variables in predicting spring phosphorus. Furthermore, these models provide quantitative estimates of spring phosphorus based on easily measured variables. These estimates can be used to represent natural and anthropogenic phosphorus loading estimates for a water body and provide managers with a baseline phosphorus value in setting realistic water quality objectives for a lake.

Suggested modeling changes

The models developed in this study account for a large percentage of the variation observed in spring phosphorus. However, the accuracy of these models may be compromised by inconsistencies in the scales used to collect the data and the dates when the maps were produced. Three recommendations are made to improve the models developed in this study.

(1) The use of ordinal data in this study removes a considerable amount of the natural variation between the sampled lakes and introduces a bias between the ordinal data and the continuous data (see previous section—Statistical violations and sources of error). These variables have served their purpose in this model, identifying significance with respect to spring phosphorus. To improve the accuracy of these models in predicting spring phosphorus, these variables should be measured with greater precision and on similar scales to avoid a biased data set.

(2) The relative location of wetland and agriculture along a stream is likely to cause problems if ignored (see previous section—Statistical violations and sources of error). It might be useful to develop a measurement which prioritizes the variables based on its distance from the lake. For example, wetlands and agriculture located close to the lake are ranked higher than those further away.

(3) The spring phosphorus data were collected between 1976 and 1987, while the watershed data were collected from maps which were printed between 1973 and 1992. This should not have much of an influence on the wetland variable, however agriculture and forest age classes are more likely to have changed over this time frame. Since some of these variables can change rapidly it is crucial that measurements be made with as little time lag between variables as possible.

2.5 How does Tabor Lake fit in?

Tabor Lake is a shallow (8.5 m), eutrophic lake situated about 10 km east of Prince George. The watershed is influenced by both natural and anthropogenic variables. Half of the watershed (eastern portion) is located on Tabor Mountain and is primarily forested. The other half of the watershed (western portion) has been partially developed, blending residential development with the forested land. Some of this area is also being used for pasture and agriculture.

How well do the models predict spring phosphorus in Tabor Lake?

Table 2.5 shows the observed spring phosphorus value from Tabor Lake with the predicted values using Model I and Model II. Both models underestimate the value of spring phosphorus, however the observed value is within one standard error of both estimates. This suggests that spring phosphorus in Tabor Lake is predicted by the same variables which influence spring phosphorus in other lakes within this region. However, the observed spring phosphorus value from Tabor Lake was included in the original data used to construct both models, and thus the data are not independent from the model.

To further analyse Tabor Lake in context of both models, nine years of spring phosphorus data were averaged and compared to the predictions (Table 2.5). These spring phosphorus values range between 13 µg/L to 32 µg/L, with an average value of 24 µg/L (S.E.= 2 µg/L). Both models predict a lower concentration of phosphorus (on average), but most of the observations are within one standard error of the predictions. Model I and Model II predict spring phosphorus concentrations within one standard error 7 out of 9 times and 8 out of 9 times, respectively. These values also support the claim that Tabor Lake is controlled by the same parameters which influence spring phosphorus in other lakes from this region. Figure 2.4 compares Tabor Lake (single point) to the sample population (box plot) for spring phosphorus, elevation, mean depth, watershed size. Tabor Lake is close to the median for all variables except elevation, where it is located at the lower end of the scale.

From the results of this study, it is clear that the factors which determine spring phosphorus in Tabor Lake are similar to other lakes in the region. However, the usefulness of spring phosphorus in predicting the summer trophic status of a lake is limited if phosphorus loading occurs after spring phosphorus is sampled. As discussed in chapter

Table 2.5. Comparison of observed and predicted spring phosphorus values.

	Observed		Predicted	
	P value used in the model (ug P / L)	Tabor Lake 9 yr avg (ug P / L)	Model I (ug P / L)	Model II (ug P / L)
[P]	27	24	20	18
Std Err of [P]	na	2	+S.E.=29 - S.E.=14	+ S.E.=28 - S.E.=12

+/- S.E. refers to plus or minus one standard error of the estimate

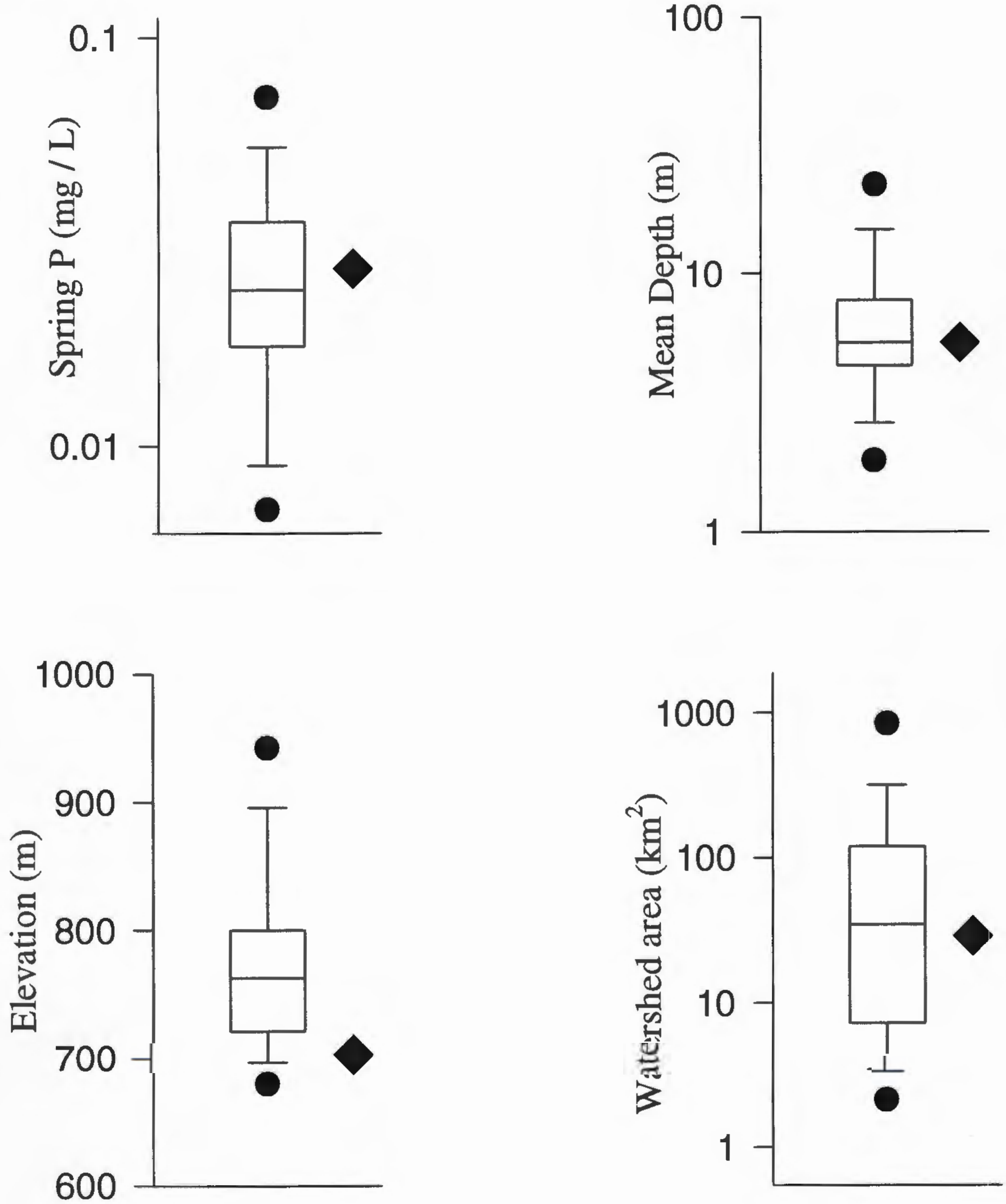


Figure 2.4 Comparison of environmental variable measurements from Tabor Lake (diamond shape) with observations of the same variables from the sample population (box plot).

one, the trophic status of Tabor Lake is believed to be dominated by internal loading during the summer months, after spring phosphorus is taken (Carmichael, 1994; Rex and Carmichael, 1995). The following two chapters address the roles of external and internal phosphorus loading in Tabor Lake.

Chapter Three
External loading: input-output phosphorus model for Tabor Lake

A mass balance phosphorus model for lakes uses inputs and outputs to quantify phosphorus loading, where external loading of phosphorus represents the input and phosphorus release through an outlet creek represents the output. The difference between a lake's input and output is the net phosphorus retention (or release). For lake managers, this is often the most practical model of phosphorus cycling, where the lake is treated simply as a "black-box" (Reckhow and Simpson, 1980). The objective of this chapter is to develop a simple phosphorus input-output model for Tabor Lake, and to improve the preliminary phosphorus loading estimates presented in chapter one (Figure 1.3) using 1995 data.

3.1 Phosphorus loading from incoming tributaries

To estimate total phosphorus loading from tributaries, two parameters need to be measured: the total volume of water entering the lake (V); and the concentration of phosphorus in the water (C_{in}). These two measurements are multiplied together to get the total mass of phosphorus ($V * C_{in}$).

Tabor Lake inflow (V_{in})

The annual surface water entering Tabor Lake was estimated by Ward (1995) using Water Survey of Canada records from a monitoring station on Tabor Creek and an equal area contribution method. Ward found that total runoff into Tabor Lake varies considerably from year to year, and used three different years (low, medium and high volumes) to demonstrate the variability in discharge. To best estimate the surface runoff into Tabor Lake during the 1995 sampling season, precipitation records during 1995 were gathered from the Prince George weather office and compared to the precipitation from each of the three flow years shown in Table 3.1. The precipitation during the medium flow year (1985) best matched the 1995 precipitation records, and will be used as an estimate of runoff (V_{in}).

The majority of runoff into Tabor Lake occurs over a relatively short period during spring melt. The estimated 1995 discharge volume into Tabor Lake is 5.92 Mm^3 (million m^3), of which 4.52 Mm^3 (76%) is released during an 11 week period between mid March and the end of May. During the summer months (June through August), runoff of water into Tabor Lake is negligible, increasing slightly in autumn.

Phosphorus concentration of inflow (C_{in})

Water samples were collected from two creeks flowing into Tabor Lake, 4-Culvert Creek and Pumphouse Creek, and analysed for total phosphorus following APHA (1993) guidelines. These two creeks were chosen as representative of two different watershed characteristics, either developed or undeveloped. Four-Culvert Creek was chosen for two reasons. First, it is the major tributary entering Tabor Lake accounting for the majority of its runoff, and second because it is representative of the undeveloped portion of the drainage basin, consisting primarily of wetland and forested area. Pumphouse creek is situated on the western side of the watershed, and was chosen as representative of runoff from the developed portion of the watershed.

Water samples were collected between March 26 and May 1, 1995. These dates were chosen to coincide with period of peak runoff. The phosphorus concentration in both creeks varied over the course of the sampling period, however Pumphouse creek (51 µg/L; n=6; S.E. = 14 µg/L) was consistently higher in phosphorus than Four-Culvert Creek (14 µg/L; n=6; S.E.= 5 µg/L). Carmichael (1994) reported similar patterns in phosphorus concentration between these two creeks during 1990 and 1991 (March through May), but found the mean phosphorus concentrations to be slightly higher (69 µg/L; n=10; S.E.= 6

Table 3.1 Monthly precipitation records for three flow years and the 1995 sampling year. Annual precipitation totals are calculated from November to October.

Month	high (76) *total (mm)*	low (78) *total (mm)*	mid (85) *total (mm)*	1995 *total (mm)*
Nov	94	36	48.3	47.7
Dec	41.7	41.7	45	22.8
Jan	79.2	29.2	33.7	25.4
Feb	39.8	7.6	40	22.8
Mar	49.5	53.3	8.5	21.8
Apr	24	18.4	32.2	72.7
May	78.4	44.5	30.5	20
Jun	122.1	37	34.1	66.6
Jul	80.8	37.9	20.3	111
Aug	68.7	62.8	49	60.5
Sep	41.4	52.4	79.5	19.6
Oct	52.9	48.6	112.6	64.4
Total	772.5	469.4	533.7	555.3

μg/L and 25 μg/L; n=11; S.E.= 3 μg/L, respectively). The 1995 phosphorus concentrations will be used to estimate average phosphorus concentrations from the developed and undeveloped portions of the watershed.

Carmichael (1994) sampled both creeks during autumn 1991, and found that phosphorus concentrations almost doubled from spring values to 116 μg/L (6 μg/L; n=3) in Pumphouse Creek and 42 μg/L (6 μg/L; n=4) in Four-Culvert Creek. However, this autumn increase will not be used in the 1995 loading estimate because phosphorus concentrations were not sampled during this period. This omission should not greatly influence the total loading estimate since most of the water is delivered during spring melt.

Total phosphorus from runoff

Since two different phosphorus estimates are available, the watershed is divided up depending on whether it is a developed area or an undeveloped area. These areas represent 21% and 79% of the drainage basin, respectively (Figure 3.1). The total surface water runoff (V_{in}) is divided according to the percentage of developed versus undeveloped area and is multiplied by the representative phosphorus concentration.

The weekly total phosphorus loading from the Tabor Lake watershed is shown in Figure 3.2. The phosphorus loading from the developed and undeveloped sections of the watershed are distinguished in the figure. Although the undeveloped section delivers 79% of the water to the lake, the phosphorus loading from each section is almost evenly divided with 71 kg coming from the developed area and 65 kg from the undeveloped area. The total phosphorus delivered to Tabor Lake from all tributaries during 1995 was 136 kg.

The annual variation in phosphorus loading can be divided into three sections: spring, summer and autumn. Since 76% of the runoff takes place during an 11 week period known as the spring freshet (mid March to end of May), this is when most of the phosphorus enters the lake. During the summer months (June, July and August) there is virtually no

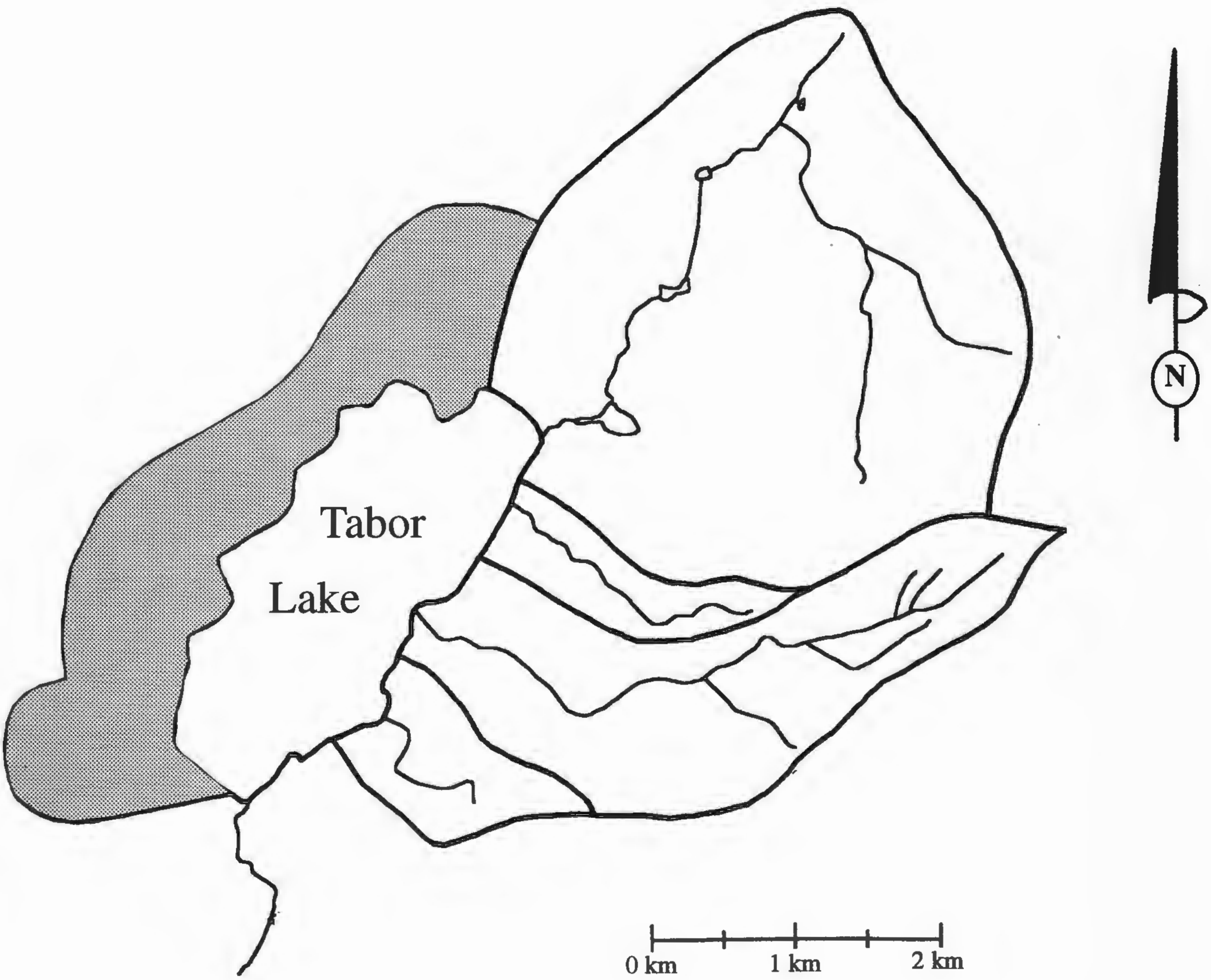


Figure 3.1 Tabor Lake watershed divided into regions with residential development (shaded) and without residential development.

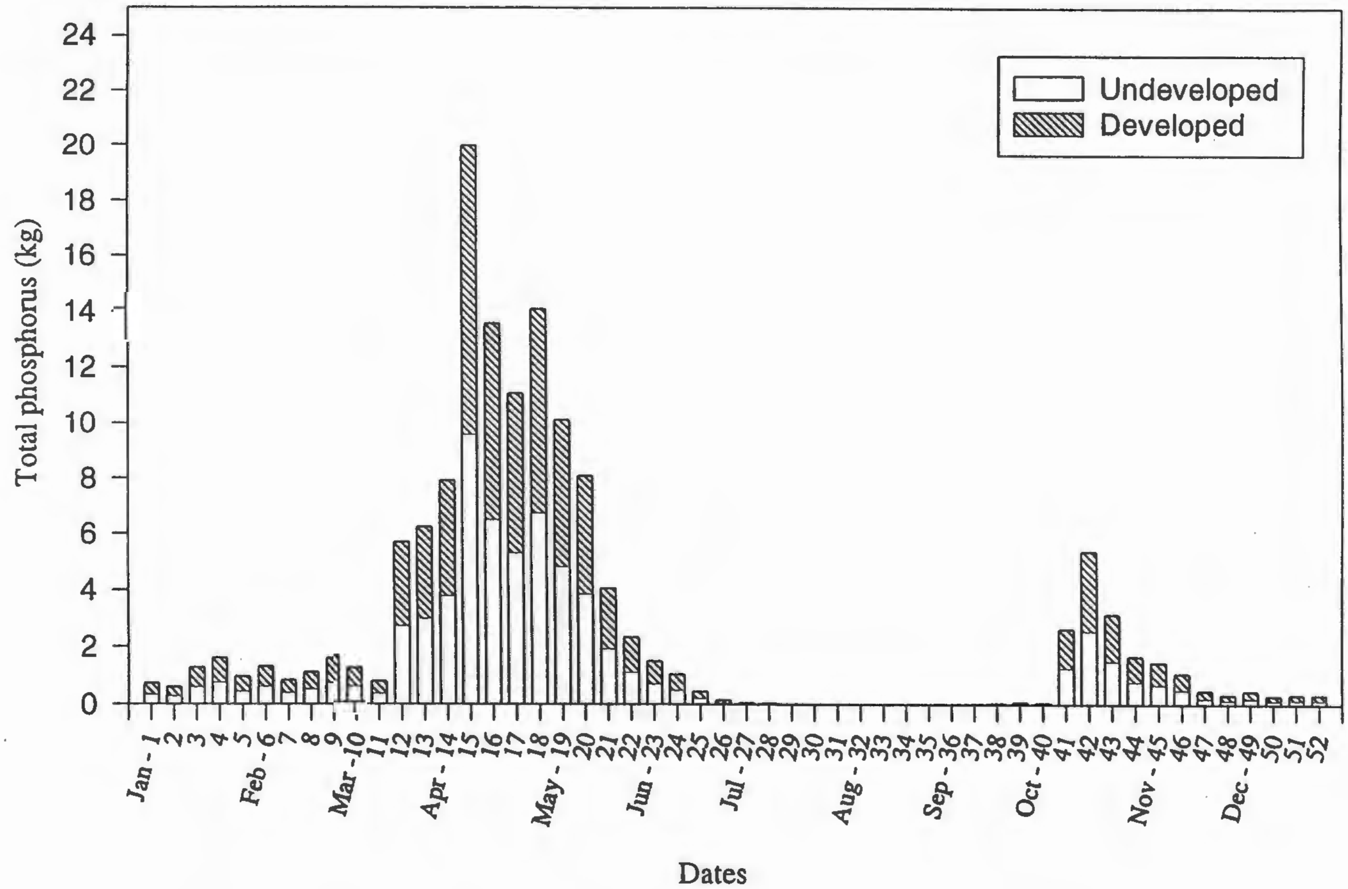


Figure 3.2 Weekly total phosphorus loading from Tabor Lake watershed. Total is divided into loadings from developed and undeveloped areas.

phosphorus loading as there is minimal or no stream flow. In autumn (September, October and November), increased precipitation causes an increase in surface runoff, thereby increasing phosphorus loading. However, the autumn phosphorus loading only accounts for small portion of total phosphorus load in 1995.

3.2 Phosphorus loading from other external sources

On March 17, 1996, snow cores were collected and tested for phosphorus content. These snow surveys are used to estimate total phosphorus deposition in Tabor Lake from the atmosphere for the period of mid November to mid March. The average phosphorus input is 5.0 mg/m^2 (S.E.= 0.7 mg/m^2; calculated from the mean of 17 triplicate samples from Tabor Lake watershed). The area of Tabor Lake is 408 Ha, which means 20 kg (S.E.= 3 kg) of phosphorus is deposited directly into the lake from the atmosphere during the 4 month period (Petticrew, 1997). Assuming an equal rate of deposition during winter and summer, over a 12 month period it is estimated that 60 kg (S.E.= 9 kg) of phosphorus falls directly into Tabor Lake from atmospheric deposition. Although the summer months may have a higher rate of atmospheric loading, the increase would likely be within one factor and not affect the total lake phosphorus budget significantly (pers. comm, R. Nordin).

Ground water flow and its associated phosphorus loading into Tabor Lake was not measured and represents a gap in the mass balance phosphorus budget for Tabor Lake. However, it is suspected that this variable does not represent significant phosphorus loading to the lake due to the nature of the surficial material in the developed portion of the watershed (pers. comm. E. Petticrew)

3.3 Phosphorus release through Tabor Creek

Following the same techniques outlined at the beginning of section 3.1, total phosphorus removed from the lake will be calculated by multiplying discharge volume (V_{out}) by phosphorus concentration (C_{out}).

Tabor Lake outflow (V_{out})

The annual discharge volume leaving Tabor Lake was estimated by Ward (1995) using Water Survey of Canada records from a station on Tabor Creek, downstream of the outlet. Ward presents two flow estimates, based on either a low flow (1978) or a medium flow year (1985). As determined in section 3.1, the most representative discharge for 1995 is the medium flow estimate (1985). This assumption was verified using discharge measurements taken during the spring of 1995 at the Tabor Lake outlet (Figure 3.3). Both the medium and low flow years are plotted against the measured 1995 spring discharge measurements, showing a closer match between 1995 and the medium flow year, than with the low flow year.

The annual estimated discharge for Tabor Lake is 5.68 Mm^3, which is slightly less than the annual runoff into Tabor Lake. Evaporation during the summer accounts for the net difference between runoff into Tabor Lake and discharge through Tabor Creek. The period of peak discharge from Tabor Lake occurs in the spring time during spring melt, and follows a similar pattern to the inflow. Ward (1995) found the peak outflow is about 1.5 weeks after the peak inflow, as a result of lake storage. Between mid March and the end of May, 3.82 Mm^3 (67%) of water leaves Tabor Lake, compared to the annual surface runoff into Tabor Lake of 76% during the same period. However, summer time flows out of Tabor Lake are larger than inflows. The autumn flows are considered negligible since most of the precipitation which falls during this period is used to offset evaporation losses from the summer.

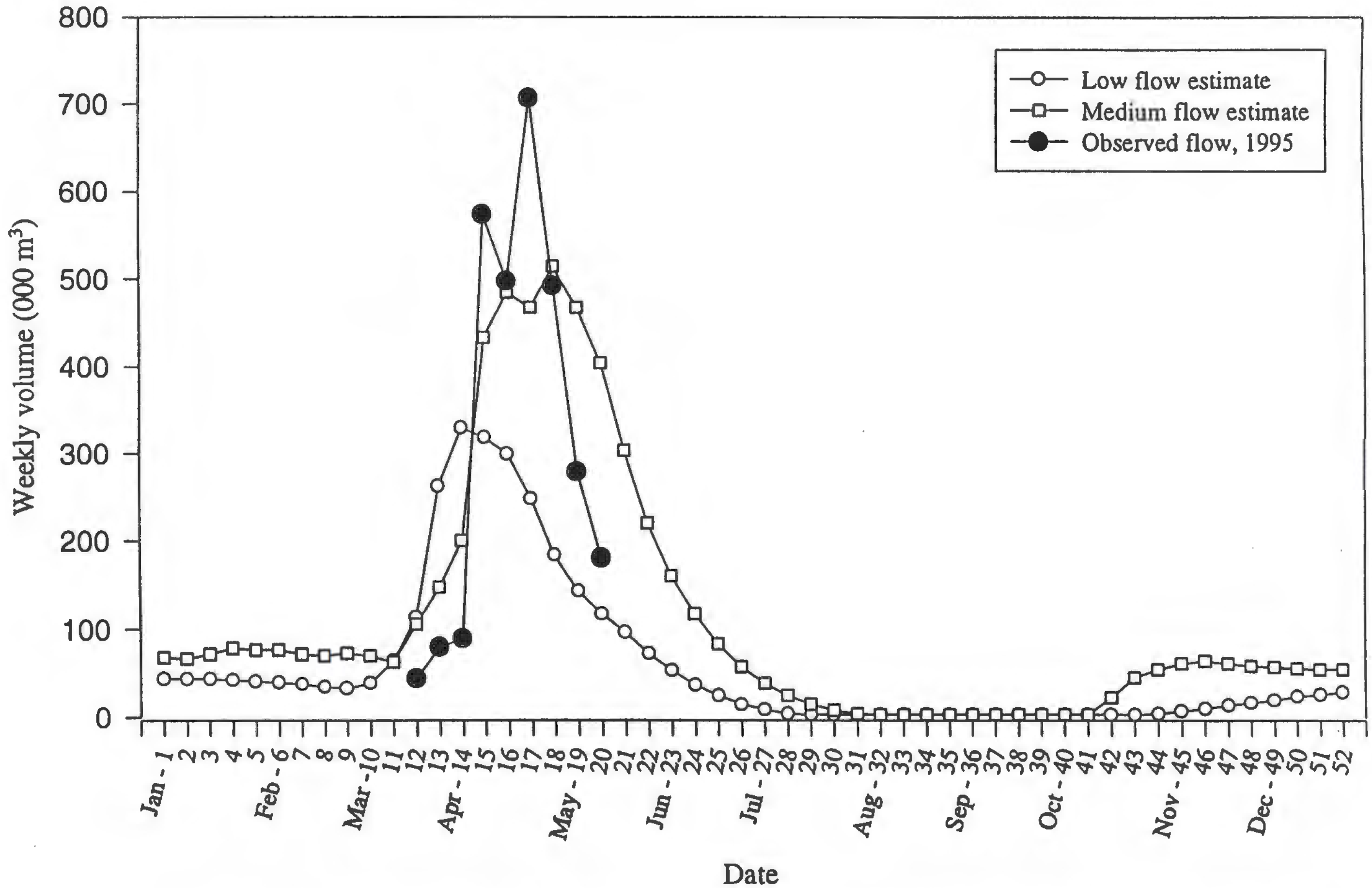


Figure 3.3 Low and medium flow estimates from Ward (1995) compared to observed discharge from Tabor Creek, 1995.

Phosphorus concentration of outflow (C_{out})

Tabor Creek was sampled for total phosphorus concurrently with sampling of 4-Culvert and Pumphouse Creek. An additional sample was collected on May 28 since discharge into Tabor Creek at that time was still high. The average phosphorus concentration in Tabor Creek between March 26 and May 28, 1995 was 33 μg/L (S.E.= 4 μg/L; n=7). No phosphorus measurements were taken during summer and autumn 1995, and therefore spring phosphorus values are used for the entire season. As noted earlier, any error associated with this assumption is small due to the fall discharge, representing only a small percentage of the annual flow. For the inflow estimate, using spring phosphorus should not affect the total phosphorus budget since most of the lake's discharge occurred during Tabor Creek's sampling period.

Total phosphorus leaving Tabor Lake through outflow

The total mass of phosphorus which left Tabor Lake during 1995 is estimated to be 188 kg. The weekly total phosphorus load leaving Tabor Lake is shown in Figure 3.4. The pattern of phosphorus release through the Tabor Lake outlet into Tabor Creek, is slightly different than inflow patterns. From the end of October until the middle of March, discharge of phosphorus from Tabor Creek is fairly consistent, releasing 2-3 kg of phosphorus each week. From mid-March and the end of May, 67% of the total phosphorus output of phosphorus is released. During June, discharge tapers off to near zero and does not increase again until the end of October.

3.4 Phosphorus mass balance equation using input-output model

The input-output mass balance model for phosphorus provides an estimate of the external sources of phosphorus in Tabor Lake and is useful in determining the net external phosphorus sources and sinks in Tabor Lake. The total phosphorus inputs are calculated by adding total phosphorus entering Tabor lake through runoff (136 kg) and phosphorus

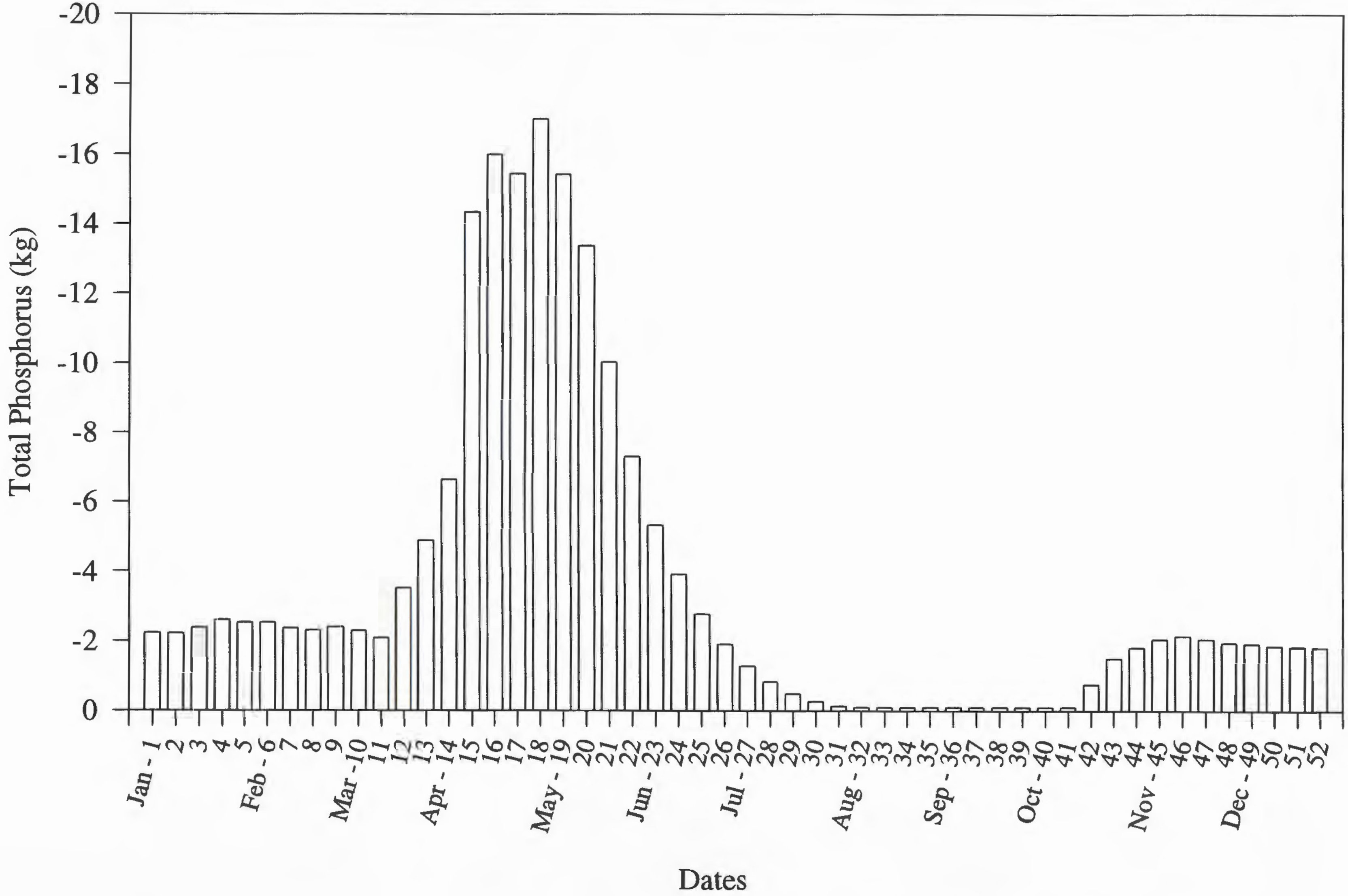


Figure 3.4 Weekly total phosphorus export from Tabor Lake through outflow.

entering Tabor Lake through atmospheric deposition (60 kg). The total phosphorus output is measured using discharge volume of into Tabor Creek (188 kg). Equation 3.1 describes mathematics of the basic input-output phosphorus model:

$$(V_{in} * C_{in}) + Atmospheric\ P = (V_{out} * C_{out}) + Net\ Retention \qquad (3.1)$$

The net retention of phosphorus in Tabor Lake is calculated by rearranging equation 3.1:

$$Net\ Retention = (V_{in} * C_{in}) + Atmospheric\ P - (V_{out} * C_{out}) \qquad (3.2)$$

During 1995, the model estimates there was a net retention of 8 kg of phosphorus from Tabor Lake, which indicates a near balance of phosphorus in this simplified input-output model. Figure 3.5 plots the weekly balance between phosphorus input and output in Tabor Lake. From January until the beginning of April, phosphorus input and output are approximately balanced. Between the middle of April and the middle of June, the maximum flux in phosphorus into and out of Tabor Lake occurs; a function of peak discharge volume (V). Beginning in July, the model shows Tabor Lake changing from a net source of phosphorus into a net sink and takes place during low stream discharge periods. An increase in phosphorus loading into Tabor Lake is observed between the end of September and the middle of October and represents a net retention of phosphorus in Tabor Lake predicted by the model. From the middle of October until the end of December, phosphorus input and output approaches an equal balance.

The input-output model identifies periods and duration of phosphorus loading into and out of Tabor Lake. The weekly input-output estimates will be useful to compare to the weekly lake monitoring program, to identify if and when external loading is responsible for increases in lake phosphorus.

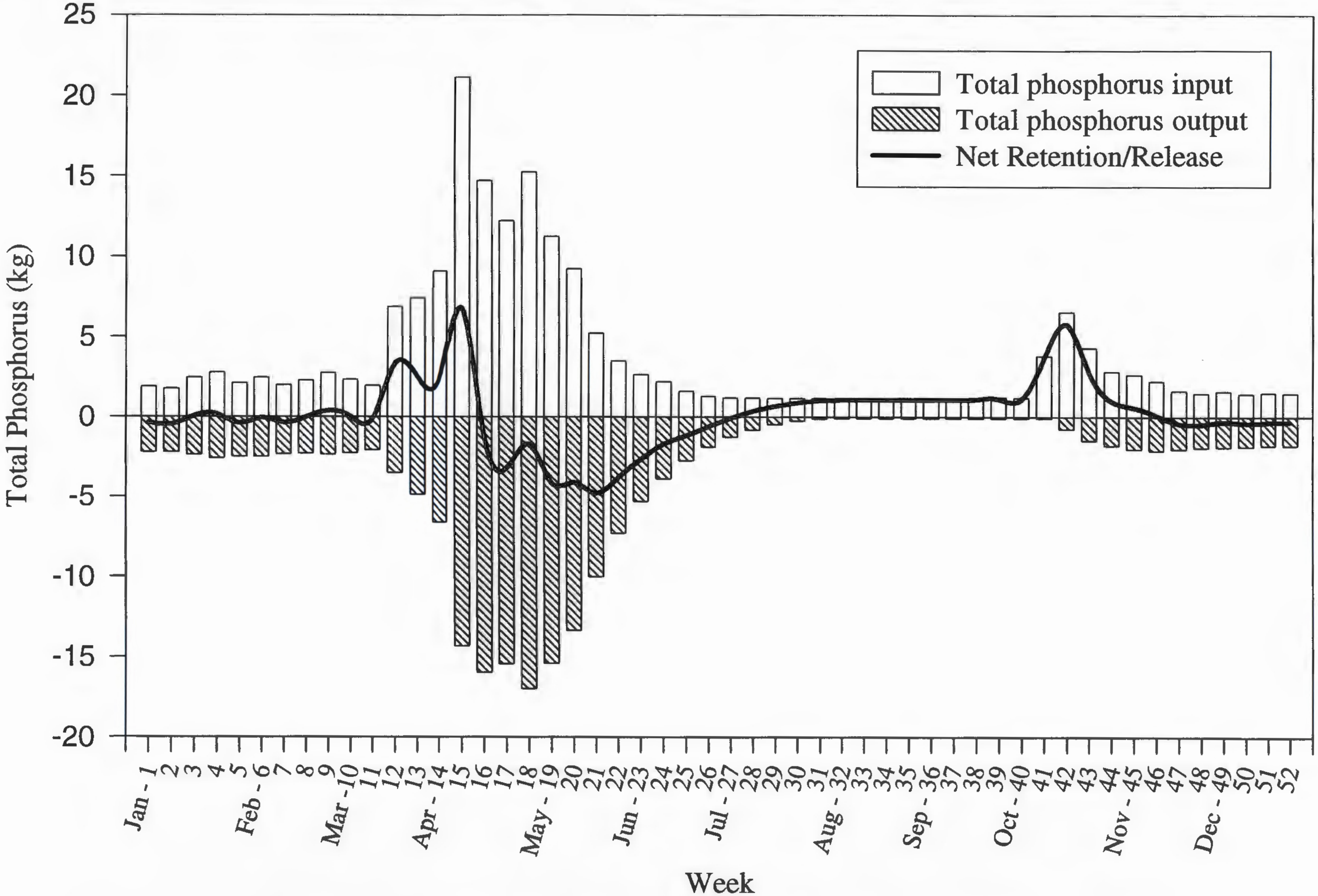


Figure 3.5 Weekly total addition and loss of phosphorus from Tabor Lake.
The net retention/release is represented by the line graph.

Limitations of the model

The assumption of equal phosphorus concentrations throughout the season of runoff may not be appropriate. This potential problem is not expected to be large source of error since most of the inflow and outflow occurred during the period of phosphorus sampling.

The input-output model provides a quantitative estimate for two external sources of phosphorus, runoff and atmospheric deposition, and an estimate of the annual net flux of phosphorus through outflow into Tabor Creek. An obvious limitation of this model is that it ignores the internal phosphorus dynamics. However, it provides us with an estimate of the external inputs and losses so that we can determine their significance in the context of the internal sources of phosphorus loading presented in the next chapter.

Chapter Four
Internal Loading of Phosphorus in Tabor Lake

The Tabor Lake phosphorus budget presented in chapter one identified the lake sediment as the single largest compartment of phosphorus in Tabor Lake by several orders of magnitude (Figure 1.3). Therefore, the sediments represent a potentially large source of phosphorus to Tabor Lake. Rex and Carmichael (1995) claim that some form of internal loading occurs in Tabor Lake during the summer, and analysis of the 1994 volunteer monitoring data indicates that during periods of hypolimnetic anoxia, phosphorus concentration in the hypolimnion increases. The 1994 data also reveal that increases in epilimnetic phosphorus loading observed during the late summer and early autumn cannot be explained by sediment release into the hypolimnion as oxygen is found in high concentrations in this period. It is hypothesized that the extensive macrophyte community in Tabor Lake accounts for this increase.

In this chapter, two mechanisms of internal loading are analysed— phosphorus loading from anoxic hypolimnion and phosphorus leaching from senescing macrophytes. Loading estimates from each source are calculated.

4.1 Weekly phosphorus observations in Tabor Lake

This section describes the weekly variation of phosphorus concentrations observed in Tabor Lake at two sampling stations, the deep hole and the littoral zone, during the 1995 sampling season. The purpose of this section is to identify periods of rapid change in the lake, and quantify the changes in total phosphorus within different compartments during the open water season.

Methods

Figure 4.1 shows the locations of the deep hole sampling station and the littoral zone sampling station. The deep hole was chosen to represent pelagic conditions and was located near the centre of the lake. Temperature and dissolved oxygen measurements were taken using a YSI calibrated meter at 0.5 metre intervals from the surface to the sediment. Water samples were collected in triplicate at the surface (0.5 metres deep) and 8.0 metres deep, and sent to Zenon Laboratories for phosphorus analysis. The ascorbic acid method with persulfate digestion was utilized to measure total phosphorus, following APHA (1993) guidelines. These two depths were chosen to identify periods of internal loading and instances of lake turnover.

The littoral sampling station was located along a forested section of the shoreline to minimize the influence of residential development on littoral zone phosphorus. Also, this location provided protection from frequent southern winds, minimizing the influence of wind disturbance on littoral phosphorus concentration. Sampling was conducted approximately midway between the shore and the outer most region of the macrophyte weed bed at a depth of approximately 2.5 metres. Water samples were collected in triplicate from 0.5 metres below the surface and analysed for total phosphorus.

Volume calculations based on bathymetric analysis (Ward, 1995) estimates that the total lake volume is 21.95 x 10^6 m^3. The lake has been subdivided into three compartments for comparison. The first is the littoral zone which comprises a ring around the shoreline up to a water depth of 4 metres (approximately the maximum extent of rooted plants). The pelagic core of water left in the lake (inside of the littoral ring) was divided into the epilimnion and hypolimnion. The volume of the pelagic epilimnion and hypolimnion is determined using the depth of the thermocline and a hypsographic curve of the pelagic volume to estimate percent total volume of each compartment. The one-degree-per-metre rule was used to locate the thermocline (Cole, 1994). However, at higher temperatures (above 15 degrees), water density differences increase and thermoclines can become

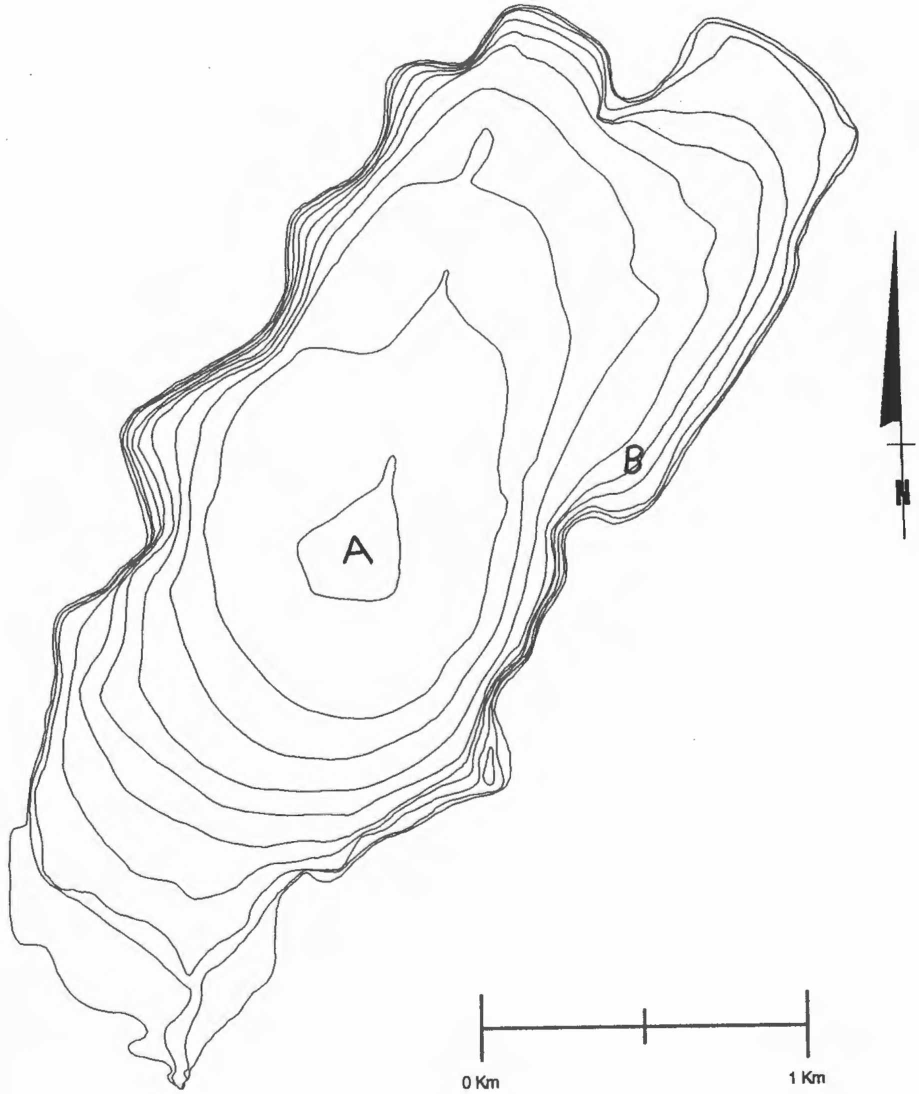


Figure 4.1 Bathymetric map of Tabor Lake. Location of sampling sites shown as letters. **A** refers to the deep hole station and **B** refers to the littoral station.

established at less than one-degree-per-metre depths. In these situations, the depth at which dissolved oxygen concentration drops rapidly is used in conjunction with the depth where maximum temperature change occurs to estimate depth of thermocline. During periods of isothermy, it will be assumed that the epilimnion represents the top four metres of water (58%) and the hypolimnion represents the water beneath 4 metres depth (42%).

Results and discussion

The 1995 open water season began on Tabor Lake during the last week of April. Deep hole samples were taken on April 30, in order to capture spring phosphorus concentration in Tabor Lake, when the lake was still isothermal. Spring phosphorus concentration was 13 μg/L at the surface and 20 μg/L at the bottom. Although littoral zone phosphorus sampling was not conducted, mixing of the littoral zone water and pelagic water was unimpeded as macrophyte growth was minimal. The total mass of spring phosphorus in Tabor Lake was 342 kg.

Weekly changes in phosphorus

Weekly monitoring of the three compartments in Tabor Lake during the 1995 sampling season began on June 5 and continued until October 18. Figure 4.2 plots the total phosphorus in Tabor Lake during the 1995 sampling season for each phosphorus compartment (littoral zone, epilimnion and hypolimnion). The maximum weekly increase in total phosphorus, as calculated by summing the three lake compartments, was observed on September 13, increasing 936 kg from the previous week. The maximum decline in total phosphorus was observed on September 27, falling by 1497 kg of total lake phosphorus from the previous week. These rapid changes highlight two important points about Tabor Lake. First, that phosphorus loading during the summer and autumn months can be rapid and second, phosphorus removal from the water column may also be rapid.

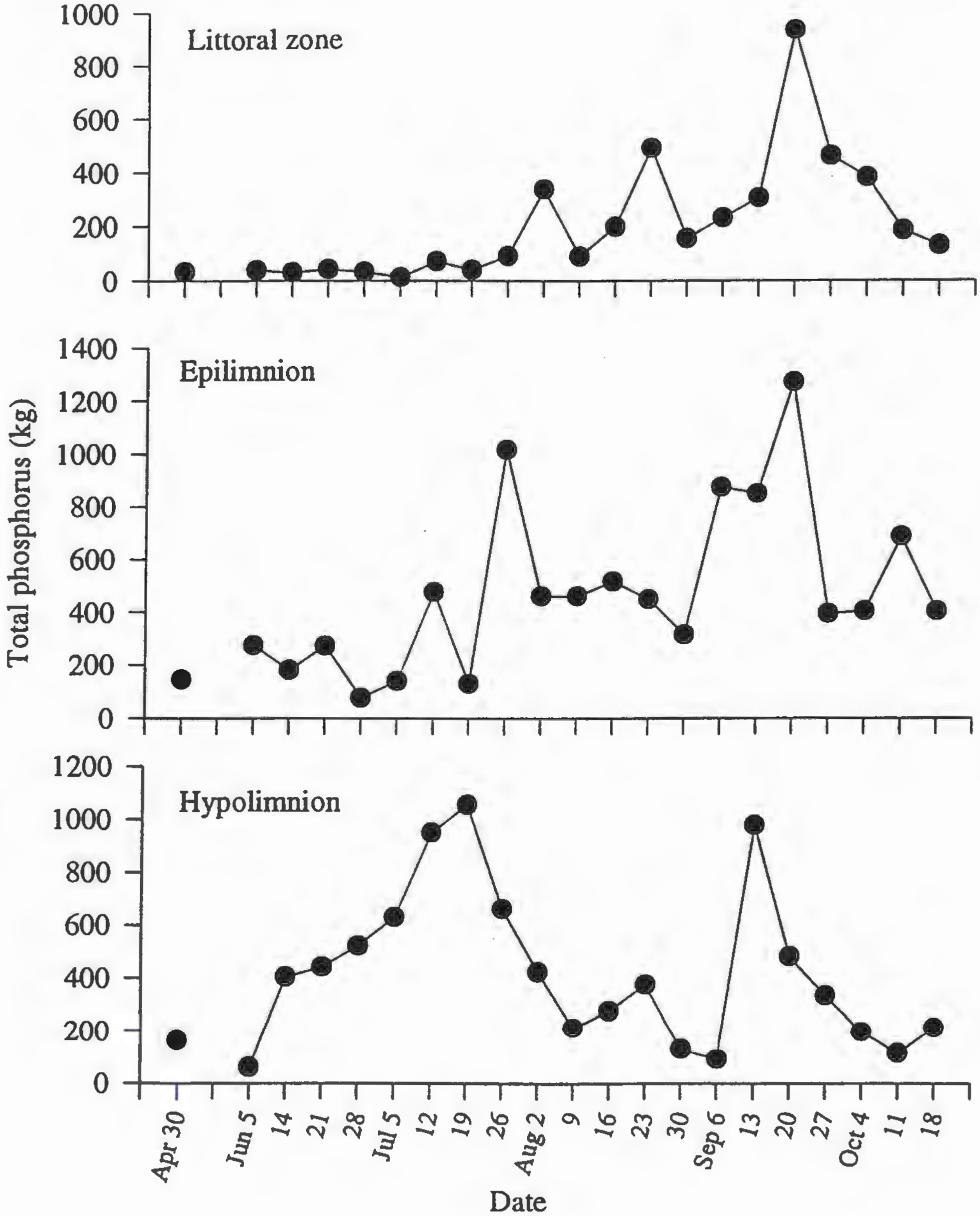


Figure 4.2 Weekly total phosphorus in Tabor Lake, divided into three compartments: the littoral zone, the epilimnion and the hypolimnion.

The two loading mechanisms suspected of delivering phosphorus to Tabor Lake, anoxic hypolimnion and macrophyte senescence, are known to rapidly deliver phosphorus to the water column. Nurnberg (1984) presents 15 studies of anoxic hypolimnia release rates, ranging from 42 mg P/m^2/week to 196 mg P/m^2/week. During the summer months, the maximum extent of anoxia in Tabor Lake was at a depth of 5 metres, extending over an area of 2.42 million m^2. Based on Nurnberg's release estimates, the loading rates could lie between 102 kg and 474 kg of phosphorus/week. Carpenter (1980), observed rapid release rates of phosphorus from senescing macrophytes. He found the majority of tissue phosphorus was released within the first two days after death.

Rapid loss of phosphorus from the water column has also been reported in the literature. Rigler (1964) observed a loss of 77% of radio-labeled phosphorus over a four-week period. Salonen *et al.* (1994) found that 40%-54% of radio-labeled phosphorus disappeared from the water column over two weeks. These studies demonstrate that lakes can rapidly lose phosphorus which might explain the decline in total lake phosphorus observed in Tabor Lake. The maximum observed decline in phosphorus over a one week period was 56%, occuring between September 20 and 27, 1995.

Analysis of the phosphorus compartments

Figure 4.2 shows the total phosphorus in each of the three lake compartments (littoral zone, hypolimnion and epilimnion) and is useful in identifying which compartments are responsible for changes in total lake phosphorus at various times in the open water season. Throughout the 1995 sampling season, the hypolimnion appears to be the dominant phosphorus compartment in the lake between mid June and the end of July. The remainder of the sampling season does not show a dominant phosphorus compartment, but reveals large changes in total lake phosphorus on a weekly basis.

Between mid June and the end of July, the hypolimnion accounted for an average of 64% of the total phosphorus in Tabor Lake, while only containing 6% to 18% of the total lake volume. From August until the end of the sampling season (Oct. 18), the average phosphorus mass present in the hypolimnion was 26%. These results indicate that phosphorus loading from the hypolimnion dominated the total loading of Tabor Lake from mid June until the end of July, but was not the dominant phosphorus loading mechanism in August and September.

During June and July, the littoral zone did not show a large portion of phosphorus storage, ranging between 2% and 11% of total lake phosphorus. Because the littoral zone only accounts for 12% of the total lake volume, the littoral zone does not appear to be a source of phosphorus during this period. From August 2 until the end of the sampling season, phosphorus concentrations in the littoral zone experience three periods of increasing phosphorus levels: July 26–August 2, August 16–23 and September 6–20. During these periods, total littoral zone phosphorus ranges between 5% to 37% of total lake phosphorus. The first period of increased phosphorus levels (August 2) corresponds with decreasing lake phosphorus and might represent the net migration of phosphorus from the pelagic zone to the littoral zone (Rigler, 1973). However, the remaining two periods of increasing littoral zone phosphorus (August 16–23 and September 6–20) correspond with increasing lake phosphorus levels. Furthermore, these periods of high littoral phosphorus levels occur during late summer/early autumn—when macrophyte senescence and phosphorus leaching is expected to occur.

The epilimnion, which links all three compartments together, tends to reflect the conditions of both the hypolimnion and/or the littoral zone. Periods of large increases in epilimnetic phosphorus correspond to previous increases in hypolimnetic phosphorus, indicating complete or partial turnover of pelagic water column during the week and subsequent loading of the epilimnion. Also, increases in the epilimnion phosphorus correspond to increasing levels of littoral zone phosphorus. These results indicate that phosphorus levels in the epilimnion are likely caused by loading in the other two compartments.

4.2 Estimating internal phosphorus load from anoxic hypolimnion

Phosphorus release from the sediments can be prompted by changing the environment at the sediment-water interface from oxic to anoxic (Bostrom *et al.*, 1982). In oxic environments, phosphorus is sorbed to iron(III), mainly in the form of strengite. Under anoxic conditions, iron is reduced to iron(II), at which point both iron and phosphate enter solution. The results from section 4.1 show total phosphorus in the hypolimnion increasing from mid June to the end of July, indicating input to the hypolimnion. The purpose of this section is three-fold, first to identify periods of increasing phosphorus mass in the hypolimnion, second to estimate conservative and liberal hypolimnetic release rates and third to generate an estimate of the total phosphorus input to Tabor Lake's anoxic hypolimnion.

Figure 4.3 plots the phosphorus concentration versus dissolved oxygen concentration at the deep hole (8.0 metre) station. The figure shows an inverse relationship between dissolved oxygen and phosphorus concentrations. When the hypolimnion becomes anoxic, phosphorus concentrations rise and when oxygen replenishes the bottom waters, phosphorus concentrations fall rapidly. This pattern was repeated twice during the 1995 sampling season and is strong evidence for the existence of phosphorus input to the water column during hypolimnetic anoxia. Figure 4.4 shows the calculated stability for the water column at the deep hole (Cole, 1994). This value of stability reflects the resistence to mixing or the degree of stratification. These data, used in conjunction with the mean daily wind speed (Figure 4.5), allow an evaluation of the mixing processes which could have occurred between the periods of sampling.

The first period of hypolimnetic anoxia occurred between June 14 and July 26, during which time the total phosphorus mass in the hypolimnion increased from 129 kg to 580 kg. However, two sampling dates show increases in epilimnetic phosphorus (July 12 and July 26), indicating decreased stability and release of phosphorus into the epilimnion. This could not be detected from the sampling program since it was limited to weekly monitoring,

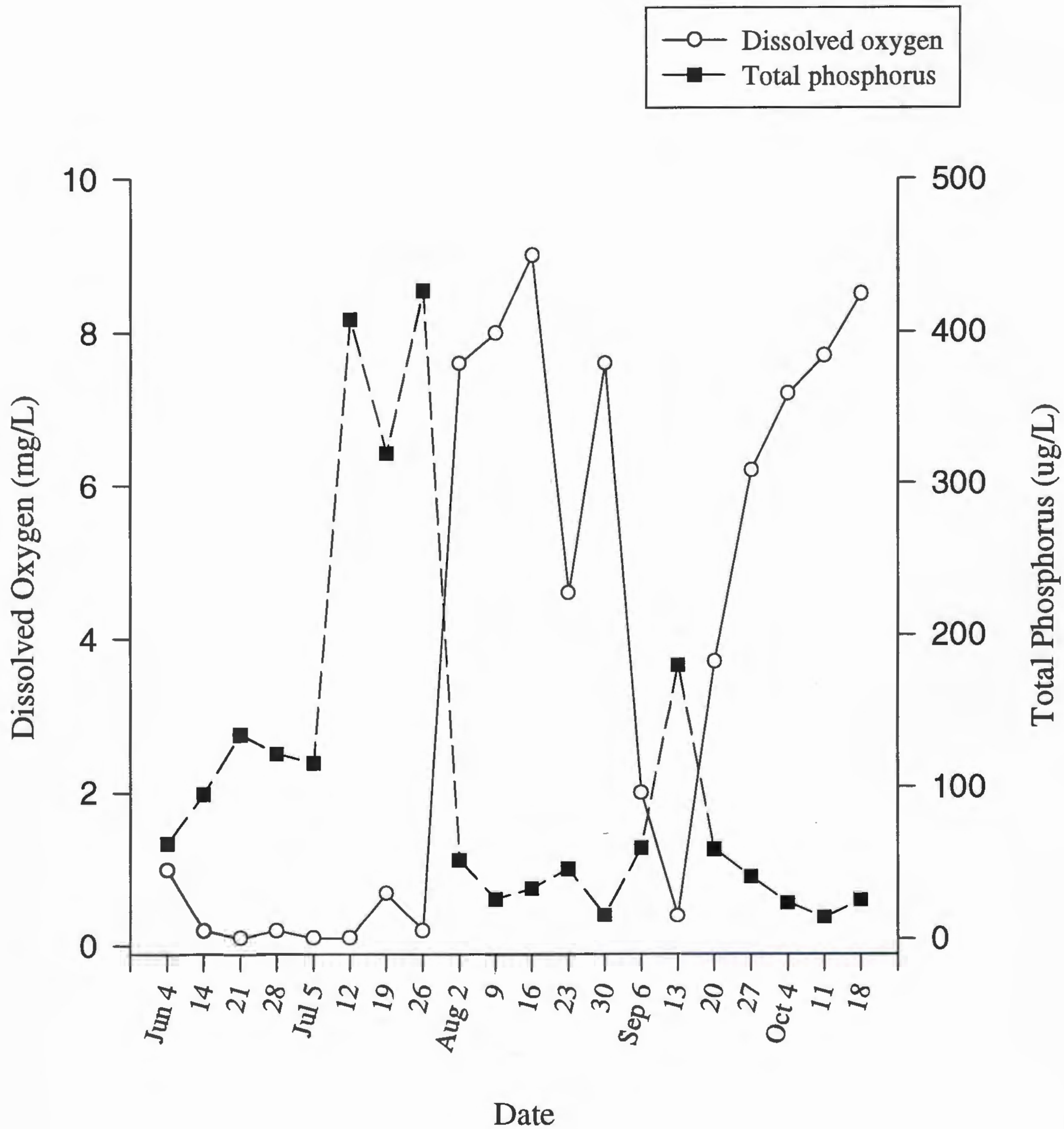


Figure 4.3 Phosphorus and oxygen concentrations at 8 metre depth at the deephole station of Tabor Lake, 1995.

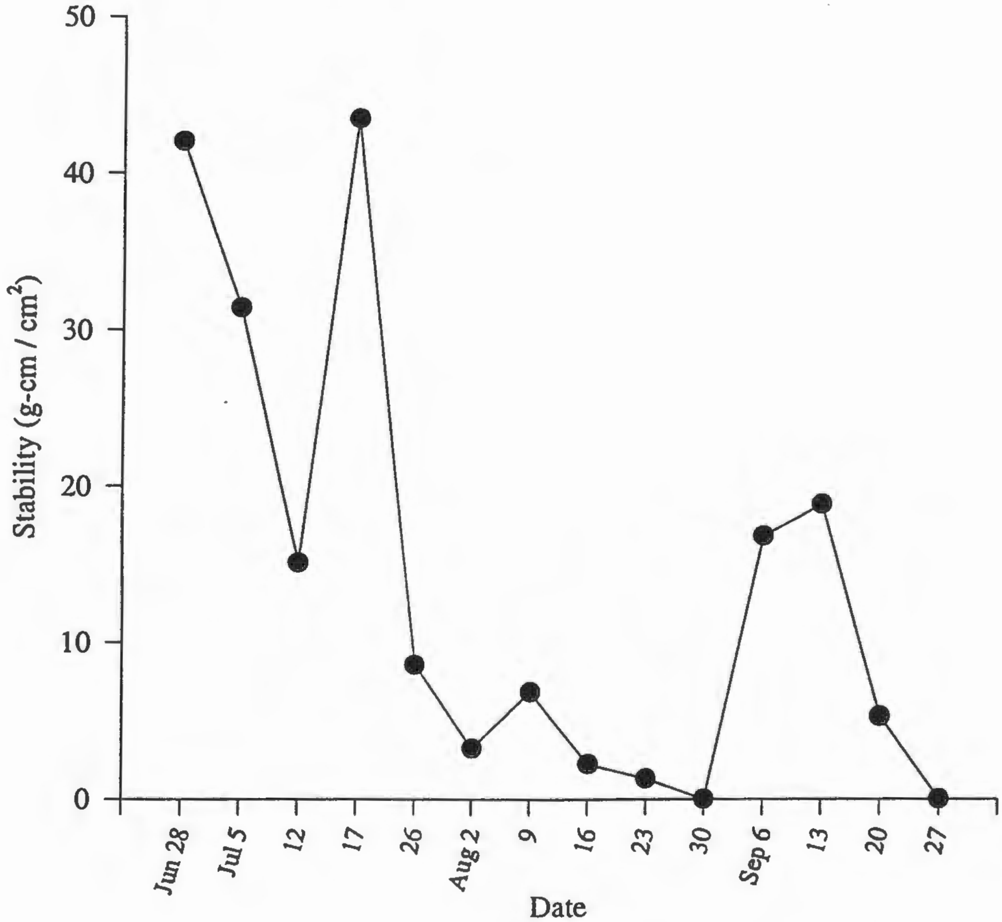


Figure 4.4 Weekly stability values for Tabor Lake, 1995 calculated from deephole temperature measurements.

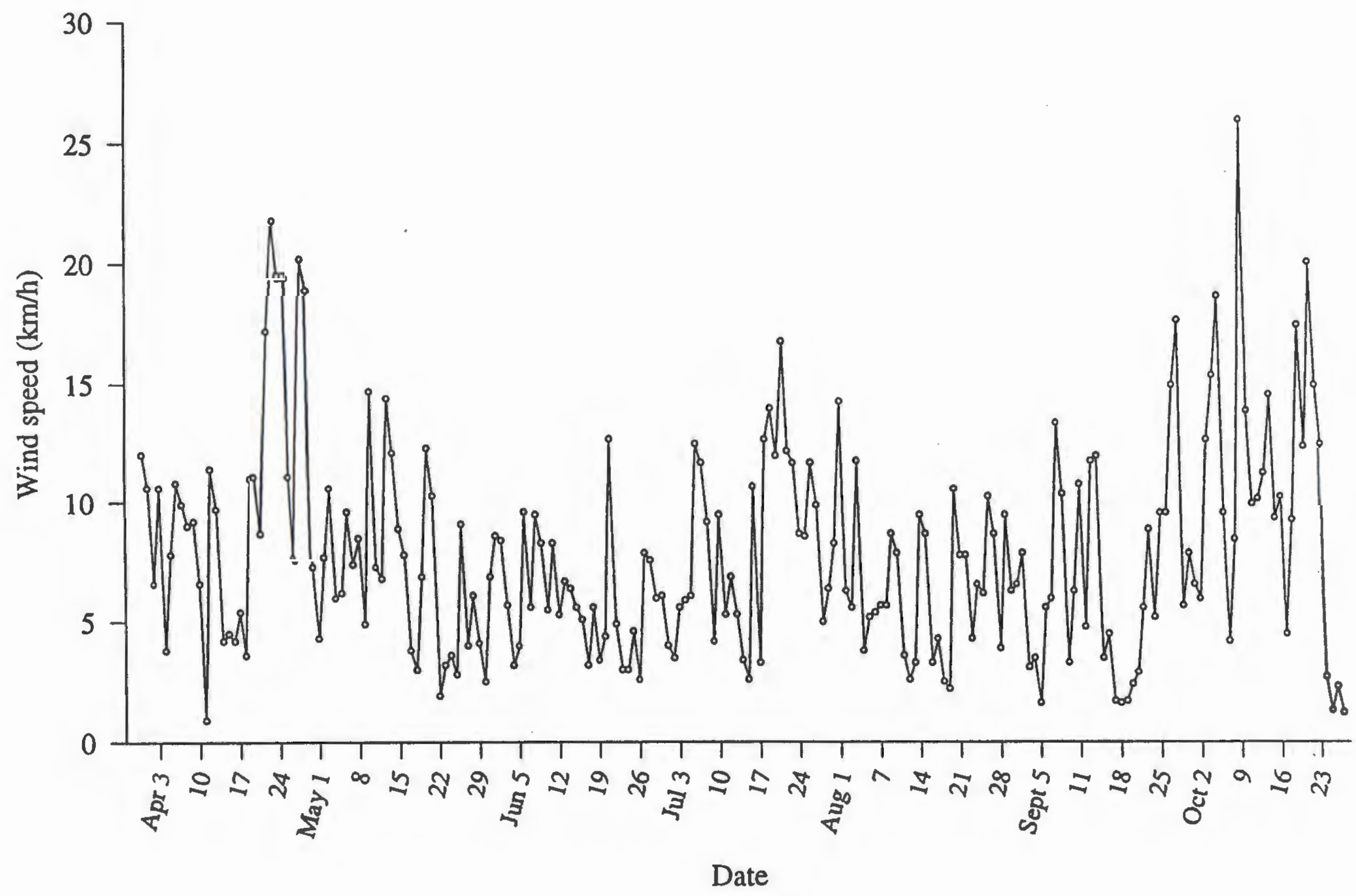


Figure 4.5 Mean daily wind speed recorded at the Prince George airport, 1995.

but was inferred from changes in phosphorus between different compartments in Tabor Lake (see Figure 4.2) and stability values (see Figure 4.4). The second period of hypolimnetic anoxia was measured on September 13, which is also the same period when macrophyte loading is expected to occur.

<u>Estimating conservative and liberal phosphorus release rates in the hypolimnion</u>

Two estimates of hypolimnetic phosphorus loading are determined here. The first, representing a conservative release rate, is calculated using phosphorus increases in <u>only</u> the hypolimnion during periods of anoxia. The second rate represents a liberal release estimate as it is calculated using the whole lake change in phosphorus during periods of anoxia. This was considered necessary as there were two periods of low stability and wind events during the June 14–July 26 anoxic period which apparently mixed the hypolimnetic water and released phosphorus into the other compartments. Assuming low phosphorus loading from other measured sources (creeks, atmosphere, littoral weeds), the observed increases in total lake phosphorus are attributed to anoxic delivery and represents a liberal estimate of hypolimnetic phosphorus loading.

This method used the seven week period of continuous anoxia between June 14 and July 26, when phosphorus loading from senescing macrophytes is assumed to be low. To generate the phosphorus release rates discussed above, the weekly total phosphorus increase in the hypolimnion (conservative estimate) and the entire lake (liberal estimate) were divided by the estimated surface area of anoxia. The surface area was calculated using the thermocline depth and surface area calculations for each contour interval (Ward, 1995). The weekly anoxic release rates are calculated for the period between June 14 and July 26 and averaged to generate a single release rate for the conservative estimate (84 mg $P/m^2/wk$) and the liberal estimate (203 mg $P/m^2/wk$). Figure 4.6 compares the conservative and liberal release rates calculated for Tabor Lake with the 15 release rates presented by Nurnberg (1984). The conservative release rate estimates for Tabor Lake is close to Nurnberg's median value, while the liberal release rate is higher than Nurnberg's values.

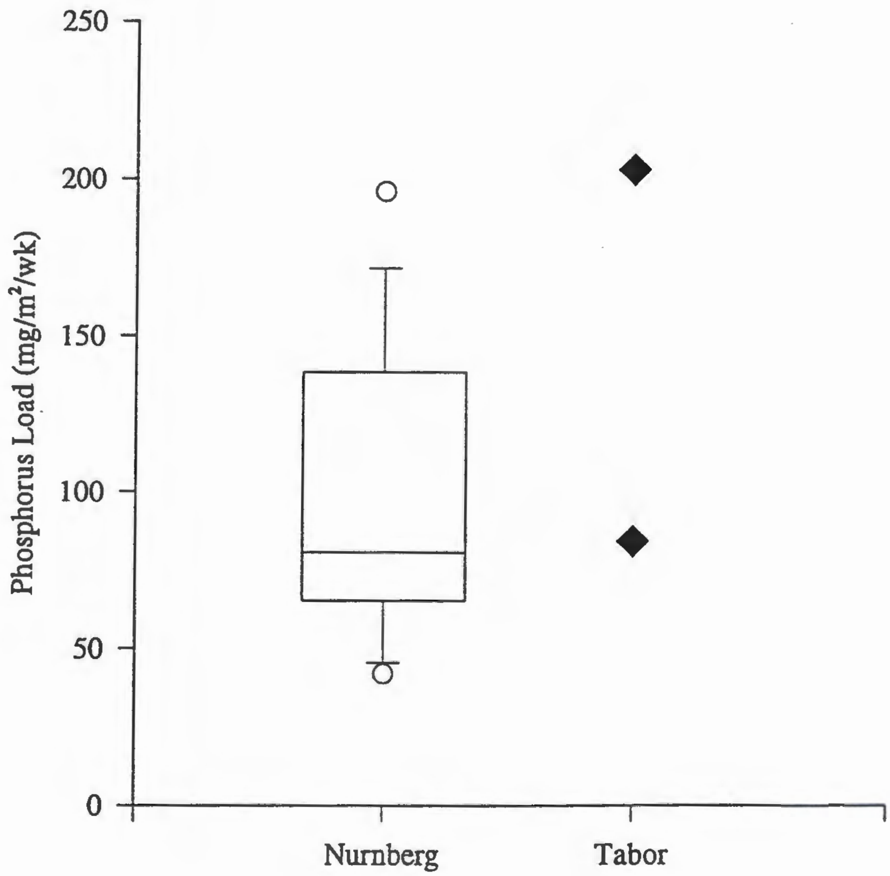


Figure 4.6 Comparison of phosphorus release rate estimates from 15 lakes (Nurnberg, 1984), and two estimates from Tabor Lake (conservative estimate and liberal estimate).

The two Tabor Lake release rate estimates using either hypolimnetic phosphorus or whole lake phosphorus, are designed to bracket the potential phosphorus delivery from Tabor Lake's anoxic hypolimnion. Throughout the discussion, these two release rates are used in place of a mean and standard error. During the six weeks of continuous hypolimnetic anoxia, two sampling dates revealed an increase in epilimnetic phosphorus (July 12 and 26), while phosphorus in the hypolimnion remained high. It is possible that a partial disruption of the thermocline could have released some portion of the hypolimnetic phosphorus into the epilimnion without completely destratifying Tabor Lake, and therefore go undetected by the weekly monitoring program. In this scenario, the "conservative" phosphorus release rate would underestimate the actual release of phosphorus. Alternatively, other unaccounted sources of phosphorus loading might contribute to the increase of total lake phosphorus in Tabor Lake (such as wind resuspension or bioturbation). If these processes did contribute to lake loading, the "liberal" release rate would have the effect of overestimating the phosphorus loading from the anoxic hypolimnion. Although both release rate estimates may not accurately represent the actual loading from Tabor Lake's anoxic hypolimnion, they provide an upper and lower bound of phosphorus loading.

<u>Total phosphorus load from anoxic hypolimnion during sampling season</u>

During the 1995 sampling season, Tabor Lake experienced eight weeks of anoxia in the hypolimnion. The first seven weeks were discussed above, however hypolimnetic loading during the week of September 13 was concurrent with macrophyte senescence and observed increases in littoral phosphorus. Therefore, this date was not used in the calculation of the average hypolimnetic release rate. In order to account for the total amount of phosphorus delivered from the hypolimnion at this time, the two mid-summer release rate predictions developed above were used in conjunction with the estimated areal extent of the hypolimnetic anoxia on Sept. 13 (2.42 Mm^2) to calculate hypolimnetic loading. The conservative and liberal hypolimnetic release of phosphorus for this date are 203 kg and

491 kg, respectively. Adding the phosphorus loading estimates from mid-summer and September 13, results in the annual hypolimnetic loading estimate in Tabor Lake, which is between 1197 kg (conservative estimate) and 2250 kg (liberal estimate).

4.3 *Elodea canadensis* as a conduit for internal phosphorus loading in Tabor Lake: literature review

Elodea canadensis is the dominant macrophyte species in Tabor Lake (Carmichael, 1994) and is hypothesized to release phosphorus into Tabor Lake when it senesces during the late summer/early autumn. This section reviews the literature on *E. canadensis* regarding its ecological life history and nutrient cycling, in order to better understand and interpret the results from the in-vitro and the in-situ leaching experiments discussed in later sections.

Ecological life history of *Elodea canadensis*

Elodea canadensis is a submerged macrophyte, native to North America (Spicer and Catling, 1986) and is found in many fresh water habitats throughout British Columbia (Figure 4.7). Sculthorpe (1967) has documented the invasion and spread of *E. canadensis* throughout many parts of the world where it has become a serious nuisance. Reproduction of this plant is achieved primarily through asexual, shoot fragmentation. Detached fragments settle to the sediment where they take root. This reproductive strategy provides *E. canadensis* with the potential to rapidly colonize new territory. Once the plant has become established in an area, it often grows to nuisance levels forming dense monocultures (Nichols and Shaw, 1986).

Rorslett *et al.* (1986) described the life cycle of this plant as having three stages of growth. During the initial stage, *E. canadensis* forms a creeping mat along the sediment as it expands its territory. Once an area has been colonized, the second stage sends up numerous erect shoots, rapidly filling in the overlying water. The final stage begins when

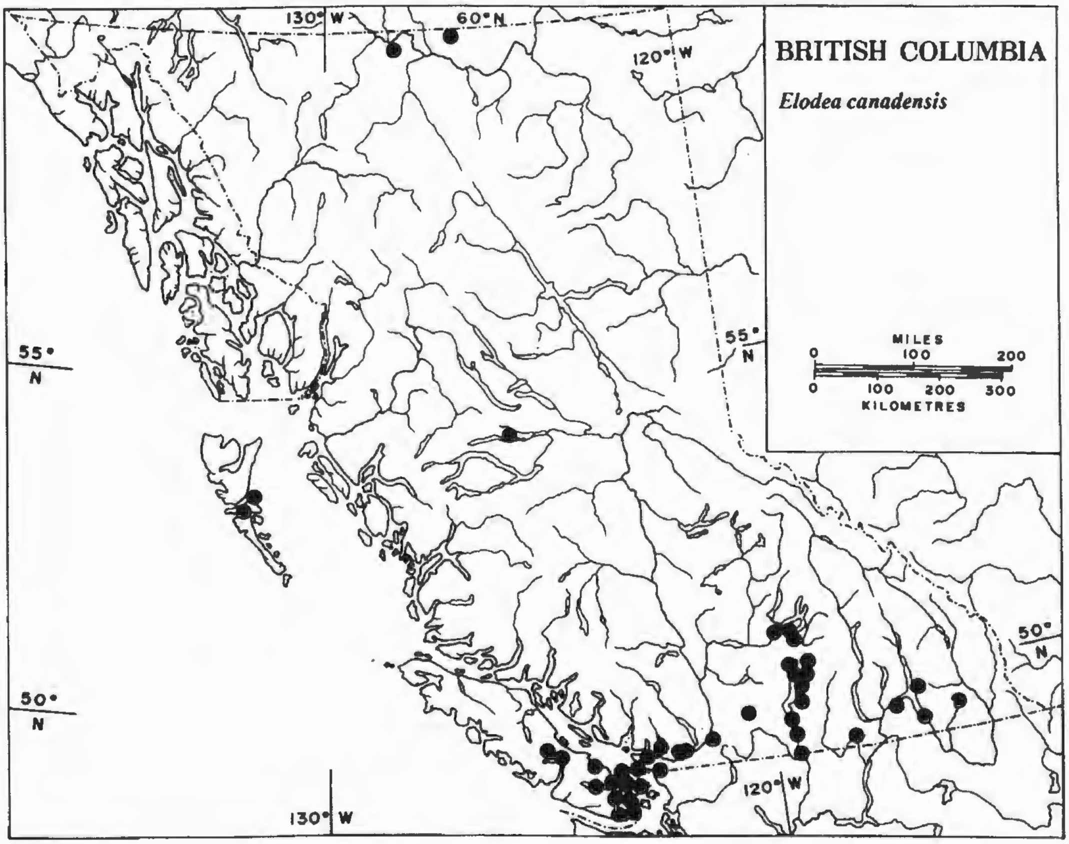


Figure 4.7 Map of British Columbia showing points where collections of *Elodea canadensis* have been made (Brayshaw, 1985).

the erect shoots reach the surface and profuse branching occurs. The formation of multiple meristems improves photosynthetic production – an adaptive advantage since light can often be limiting in dense stands of macrophytes (Westlake, 1971).

E. canadensis has two overwintering strategies. Towards the end of the growing season, *E. canadensis* forms specialized structures known as turions or dormant apices (Sculthorpe, 1967). These perennation structures are produced in large numbers with the onset of cool weather and can number up to 5000 per m^2 of sediment (Catling and Wojtas, 1986; Bowmer *et al.*, 1984). During this transformation, starch is accumulated as a food reserve for growth in the spring (Best, 1977; Janauer, 1981). *E. canadensis* has also been found to overwinter as entire plants underneath a cover of ice (Spicer and Catling, 1988). Haag (1978) investigated dormancy of this plant during winter and found that growth was easily promoted in the lab regardless what season the sample was collected. This flexible strategy allows the plant to initiate growth early in the growing season.

Population dynamics of *E. canadensis* is a source of ongoing debate. Typically, this plant exhibits a population explosion when it is first introduced to new aquatic territory and several years later the population declines just as rapidly. One theory suggests that low genetic diversity (from asexual reproduction) might explain rapid population declines. Sculthorpe (1967) rejects this hypothesis because it does not explain the regularity at which the populations collapse. Instead, he suggests that the rapid growth depletes the sediment of nutrients, causing a population crash. Eventually, an equilibrium is reached where population levels are balanced by the availability of nutrients. This theory is supported by Bowmer *et al.* (1979) who found that the population size is related to iron availability. Spicer and Catling (1988) discuss yet another theory which suggests that a reduction in water transparency might affect this "light-demanding" species, thus leading to a population decline.

Nutrient cycling

Whether macrophytes access nutrients primarily from the sediment or from the water column has been an evolving debate for many years. *E. canadensis* has a sparse rooting structure, making up only 2.6% of total plant biomass on average, but during the growing season abundant root hairs form (Sculthorpe, 1967). By tracing the path of radio labeled phosphorus from the sediments to the above ground tissue of nine different aquatic plants, Carignan and Kalff (1980) helped solve the sediment versus water column debate. Using the ratio of labeled phosphorus to unlabelled phosphorus they were able to determine the percentage of phosphorus coming from the sediments and concluded that all nine species studied, including *E.canadensis*, access most of their phosphorus from the sediments. Other studies have found similar results. Gabrielson *et al.* (1984) found that *Elodea densa* obtained 83% to 85% of its total phosphorus from root absorption. Smith and Adams (1986) estimated the rate of phosphorus uptake from sediments in *Myriophyllum spicatum* to be 3.0 g/m^2 annually. Sediment samples taken from Tabor Lake during the summer of 1994 show that phosphorus concentrations in the littoral sediments (< 4.0 metres deep) are lower than in the pelagic zone (684-1480 μg/g vs. 1390-1800 μg/g, respectively).

During the growing season macrophytes act as temporary storage sites of phosphorus, constantly relocating phosphorus from the sediments to their tissue, yet most estimates of trophic status in lakes ignore the phosphorus stored in macrophytes. Open-water nutrient concentrations are frequently used as an estimate of a lakes trophic state primarily because of their simplicity and general effectiveness in application. Unfortunately, this measurement only accounts for one compartment of phosphorus within a lake—in the ambient water, while missing phosphorus stored in the macrophytes.

Canfield *et al.* (1986) have developed a trophic state classification system for lakes which incorporate aquatic weeds. Instead of characterizing a lakes trophic status as a measurement of conditions in the open water, a "potential water column nutrient concentration" is proposed which includes the phosphorus held within the macrophytes.

By multiplying total biomass of macrophytes in a lake with the average phosphorus concentration found in macrophytes, an estimate of total macrophyte phosphorus levels are given. Combining this estimate with total phosphorus concentration in the open water, the result is a value of Water Column Phosphorus (WCP) for the whole lake.

An important reason for considering the phosphorus concentration in aquatic plants is that once they begin to senesce, much of their phosphorus is leached into the ambient water (Carpenter, 1980; Jacoby *et al.*, 1982; Landers, 1982; Gabrielson *et al.*, 1984). The portion of phosphorus not released into the water column returns to the sediment surface to become part of the organic sediment or to be consumed by detritus feeding organisms. In other words, macrophytes can serve as a conduit for phosphorus cycling within lakes.

Results from the 1994 volunteer sampling program on Tabor Lake indicated that some mechanism of internal loading, other than anoxic release, might account for the pattern of phosphorus levels after thermal destratification (see Chapter 1). The extensive bed of macrophytes in Tabor Lake certainly provide a potential conduit for phosphorus transport from the sediments. Therefore, it is important to evaluate the potential phosphorus load held within the aquatic macrophytes of Tabor Lake and estimate loading values from these macrophytes.

An estimate of phosphorus release from the macrophyte community to the water column of Tabor Lake was analysed using in-vitro and in-situ experiments. The in-vitro experiment was conducted to quantify the release of phosphorus from senescing *E. canadensis* in a controlled laboratory environment over both 1-week and 3-week periods. The in-situ experiment was conducted to measure differences in standing stock summer and winter samples of *E. canadensis*, to identify temporal changes in phosphorus content.

4.4 In-vitro experiment of phosphorus leaching from *Elodea canadensis*

Measuring the total mass of phosphorus released from senescing macrophytes and the rate at which leaching occurs is important in developing a mass balance budget for phosphorus in Tabor Lake. However, directly measuring this exchange in-situ is complicated by other factors which also influence phosphorus concentrations (e.g. hypolimnetic phosphorus loading) which cannot be isolated from the phosphorus leached out of senescing macrophytes. Therefore, an in-vitro study was conducted to estimate the leaching rate of phosphorus from *E. canadensis*, where senescence was artificially induced.

Methods and materials

Seven sets of *E. canadensis* grab samples were collected from the littoral station (see Figure 4.1) on August 8 (3 sets), August 22 (1 set) and September 22 (3 sets). Each set of samples were stored on ice until they arrived at the lab. Water from the littoral station was returned to the laboratory and filtered (0.7 μm) for use in the in-vitro experiments.

At the lab, healthy stems were separated from unhealthy stems based on their colour (healthy stems were green and unhealthy stems were typically brown). Once separated, the healthy stems were rinsed in tap water to remove loosely attached epiphytes and calcium deposits. The samples were then spun dry in a lettuce spinner to remove excess water.

Five grams (wet weight) of healthy stems of *E. canadensis* were placed in one litre, light-sealed mason jars with 750 ml of filtered lake water. To prevent the experiment from becoming anoxic, air was bubbled through the water using a Pasteur pipette at a rate of 3 to 8 bubbles per second. The experiment was run at 20 degrees C. Figure 4.8 shows the apparatus and construction design used in this experiment.

Light sealing the jars was accomplished by painting the outside of the jar black, and then placing all the jars underneath a black plastic curtain. This approach was chosen after preliminary experiments demonstrated that *E. canadensis* survived when placed in clear jars. Freeze dried plants were also tested following Carpenter's (1980) method, however the extent of cellular damage from freeze-drying became apparent when the plants fell apart soon after the experiment began. Placing the weeds in light sealed containers provided the

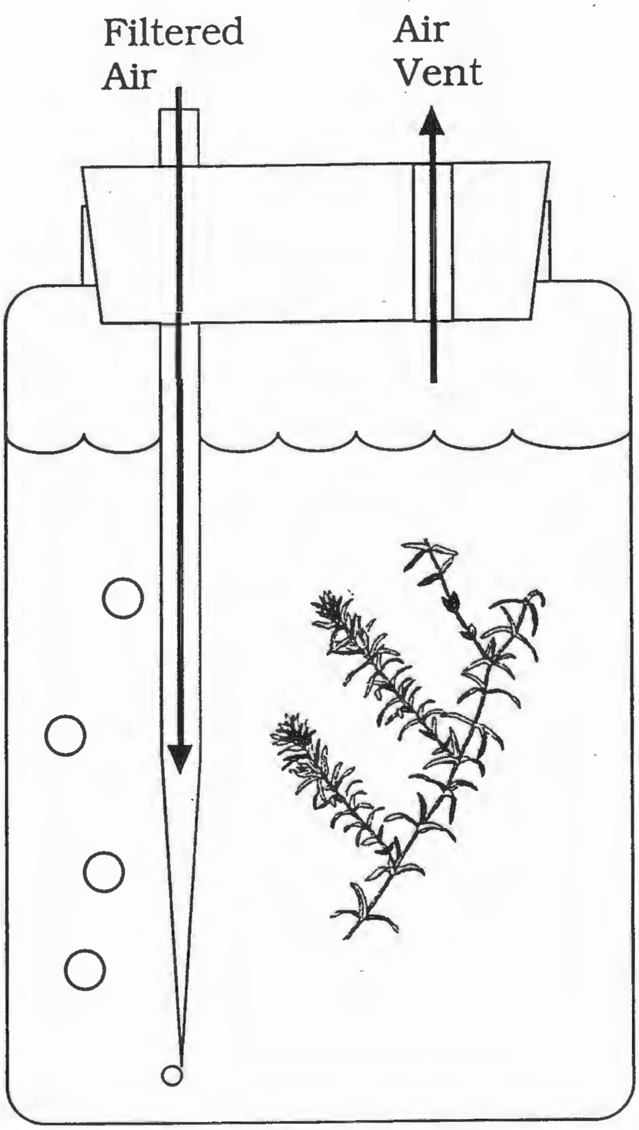


Figure 4.8 Construction design of in-vitro experiments used to induce senescence in *Elodea Canadensis.*

best available technique for inducing senescence. Without light for photosynthesis, the plants began to die without requiring freeze drying.

Each set of *E. canadensis* samples used for this experiment were conducted in triplicate, to identify the statistical variation within each set of samples. As mentioned, each set used a total of nine 5.0 g samples (wet weight). Three of the samples were immediately dried and analysed to establish initial phosphorus content of the weeds (see below). The remaining six samples were used in the senescence experiment that were conducted over both a 1-week intervals (3 samples) and a 3-week interval (3 samples).

After the senescence experiment ended, the water-plant mixture was poured through a 355 μm Nitex mesh screen to remove the larger particles. The water volume was measured and then sent to Zenon Laboratories for total phosphorus analysis using the ascorbic acid method, following APHA (1993) guidelines. *E. canadensis* was also analysed for total phosphorus, however a different technique was employed. First, samples were oven dried to a constant weight and then mixed with 4 ml Nitric Acid and 2 ml hydrogen peroxide (30%) and digested in a Milestone microwave digestion (Model mls 1200 mega). Next, the digested samples were passed through a 0.45 μm filter to remove undigested silicates and then analysed for total phosphorus using a Leeman Labs ICP-AES (Model PS1000UV).

Results and discussion

Figure 4.9 plots the relationship between initial phosphorus concentration and percent leached over 1-week and 3-week periods. Each plotted value shows the mean and standard error of triplicate measurements. Average initial phosphorus concentrations for each of the 7 sets of *E. canadensis* range between 3.5 mg to 9.8 mg phosphorus / g Dry Weight. Wallsten (1980) found Tissue phosphorus in *E. canadensis* ranged, on an annual basis, between 2.0 and 7.0 mg / g D.W. in Lake Tamnaren, Sweden. Also, Rorslett *et al.* (1985) reported tissue phosphorus of *E. canadensis* in Lake Steinsfjord (Norway) ranging between 3.0 mg to 7.0 mg P / g D.W.

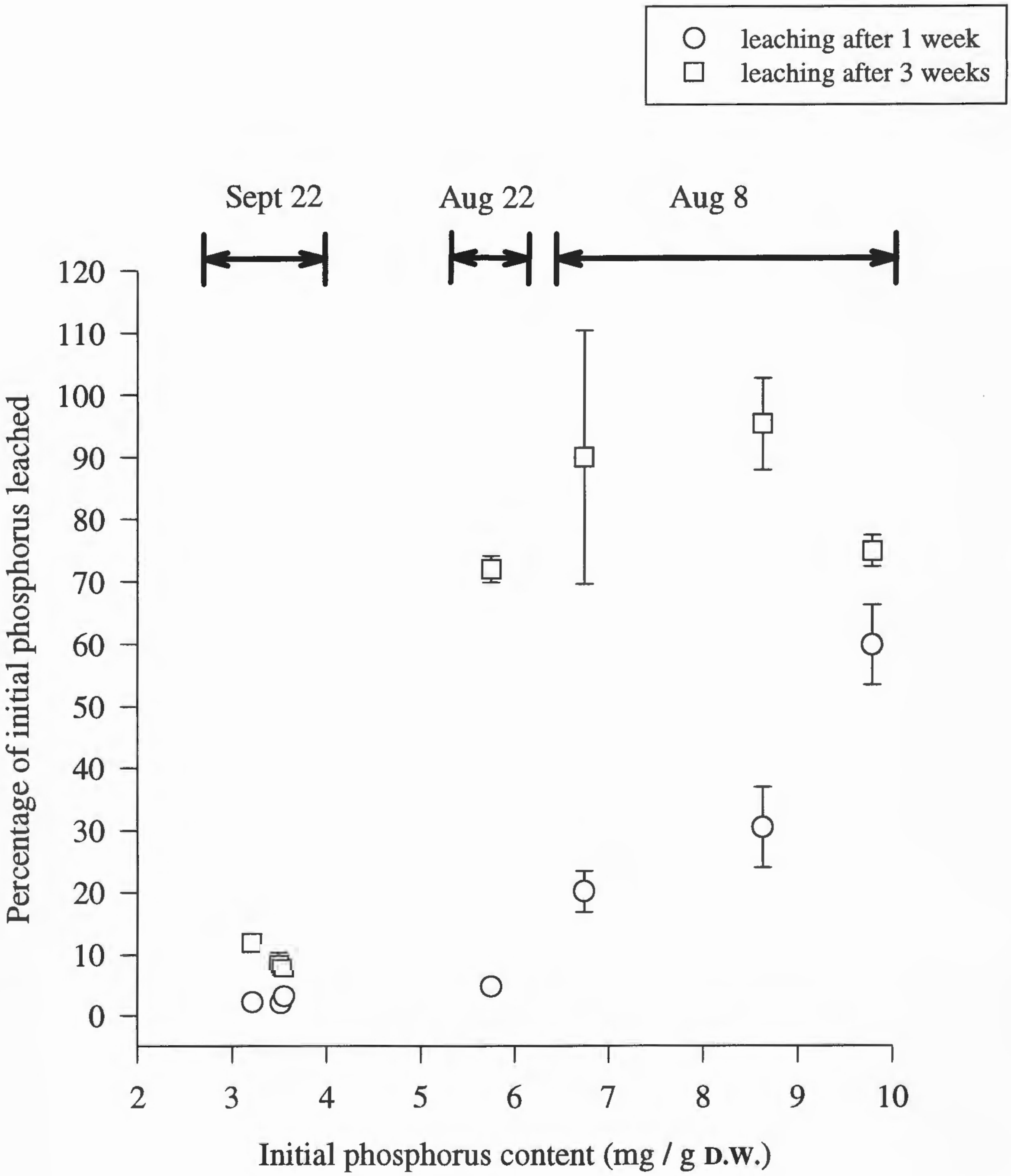


Figure 4.9 Relationship between initial phosphorus content of *E. canadensis* and the percent phosphorus leached after one week and three week experiments. The seven sets represent samples from three different dates (shown above).

Data shown in Figure 4.9 reveal two clusters - a tightly grouped cluster around 3.5 mg P / g D.W., and a second, more loosely arranged cluster between 5 mg and 10 mg P / g D.W. The first cluster represents weeds sampled from Tabor Lake on September 22 while the other cluster represents August samples. Virtually no phosphorus is leached from the September 22 samples after one-week of senescence, and only 10 % of initial P is released after three weeks of senescence. The August samples leach phosphorus much more rapidly. After one week of senescence, phosphorus release ranges between 5% and 60% initial P content. After 3-weeks of senescence the August samples exhibit release rates ranging between 70% and 95% initial P content.

The observed condition of *E. canadensis* after one week of in-vitro senescence varied considerably, depending on when the plants were sampled. After one week of senescence the plants collected on August 8 had undergone considerable decomposition. Most of the leaves were detached, the stems were beginning to fragment and only traces of chlorophyll (green coloration) was seen in the stems and detritus. On August 22, only one set of samples were collected and the decay observed after one week in this set of triplicate samples was minimal, with green leaves still attached to the stem and very little detritus formed. The final sampling date was on September 22 and one week senescence experiment revealed minimal decay (similar to the August 22 samples).

After three weeks of in-vitro senescence, the samples from August 22 showed the most change compared to the one week senescence experiments. Although the stem was mostly intact, there was a considerable increase in the amount of detritus. The samples from August 8 had formed a sludge-like detritus, with only a few stalk remnants distinguishable, while the September samples were still green and showed no signs of decay, except for a few leaves which were detached from the stem.

The clustering of data in the Tabor Lake study indicates that two different responses to induced senescence occurred. The September samples were low in initial phosphorus and did not release much phosphorus into the water, exhibiting no signs decomposition or fragmentation. This is in contrast with the August samples that did exhibit decomposition, and whose stem was fragmented into many parts. The September samples apparently had the ability to withstand a three week period without light.

The samples collected on September 22 differ from the August samples in two ways, first they were collected later in the sampling season when temperature and light availability generally declined, and second they were collected from shallower depths. This creates a problem interpreting the different responses as a result of date or depth.

Resolving the date vs. depth dilemma

The differences found between the August 8 and September 22 samples may have arisen either from the different depths at which these plants were harvested or from the different dates when they were harvested. Table 4.1 compares the sampling dates, depths, initial phosphorus concentration and the percent of phosphorus leached after both one week and three weeks for the seven sets of in-vitro experiments.

The idea that depth may be the factor which influences both initial P content and P release became an issue when the initial phosphorus concentration of *E. canadensis* appeared to be correlated to the depth at which the samples were collected. Carpenter (1980) claims that initial P content determines the percentage of P leached. If depth plays a role in regulating initial P content then it might explain why the plants collected at a shallower depth did not release as much phosphorus.

Carpenter (1980) found that lower initial P concentrations leached a smaller percentage of the initial phosphorus. To ensure that his plants were indeed dead, he freeze-dried them prior to the experiment. The plants used in the Tabor Lake experiment were not physically killed, but relied on the absence of light energy to effectively "starve" the plants to death. After the 3 weeks enclosed in total darkness, the three sets of samples collected from September 22, lost minimal amount of their tissue to detritus, and were still a healthy green colour. These plants still maintained many of the characteristics attributed to "living plants",

Table 4.1 Seven sets of macrophytes collected for leaching experime *(Elodea canadensis)*

Sampling dates	Sampling depths (metres)	Initial phosphorus (mg P / g DW)	1 week leaching (percent)	3 Week leaching (percent)
Aug 8	3.25	9.79 (0.52)	59.8 (6.4)	74.8 (2.5)
Aug 8	2.80	8.63 (0.24)	30.4 (6.5)	95.3 (7.4)
Aug 8	2.40	6.74 (0.08)	20.1 (3.3)	90.0 (20.4)
Aug 22	3.05	5.05 (0.06)	4.6 (0.8)	72.0 (2.1)
Sept 22	1.65	3.55 (0.35)	1.9 (0.3)	8.5 (2.0)
Sept 22	1.30	3.20 (0.17)	3.1 (0.7)	7.8 (0.7)
Sept 22	1.00	3.20 (0.17)	2.1 (0.2)	11.9 (na)

* numbers in brackets represent the standard error

especially in contrast to the August experiments where death was clearly observed after 3 weeks.

Alternatively, the two clusters may be the result of sampling dates. Samples were collected on three separate dates: August 8, August 22 and September 22, 1995. At the end of the growing season, *E. canadensis* begins forming large numbers of overwintering buds, known as turions (Januauer, 1981; Sculthorpe, 1967; Spicer and Catling, 1988). During this phase, the plant increases its starch reserve in order to survive poor environmental conditions such as low light and cool temperatures. If the plants sampled on September 22, had undergone this overwintering stage, then it is likely that these plants would be able to survive the zero light conditions in the in-vitro experiment by consuming their starch reserves.

The strongest argument favoring the sampling *date* theory over the sampling *depth* theory are the results from samples taken on August 22 (depth of 3.05 m). The sampling depth theory should result in a higher concentration of initial phosphorus than was measured in these plants since these plants were the second deepest group sampled. However, this value represented the median phosphorus value measured from all seven sets of samples (5.05 mg P / g D.W.; SE = 0.06 mg P / g D.W). This median phosphorus value does agree with the theory that initial P is related to sampling date. The first set of samples collected on August 8 all contain higher initial P concentrations and the standard errors of P are small. This sample set brackets the depth value of Aug. 22 and yet its initial phosphorus concentration is statistically lower than those in the same depth bracket from August 8 (Figure 4.10).

Changes in seasonal tissue phosphorus concentrations has also been observed in other lakes. Rorslett *et al.* (1985) observed the highest phosphorus concentration in *E. canadensis* during the summer months and was low in the winter months. Janauer (1981) found that inorganic anions of PO_4 were also higher in summer plants than in both the entire winter plant and the dormant apices.

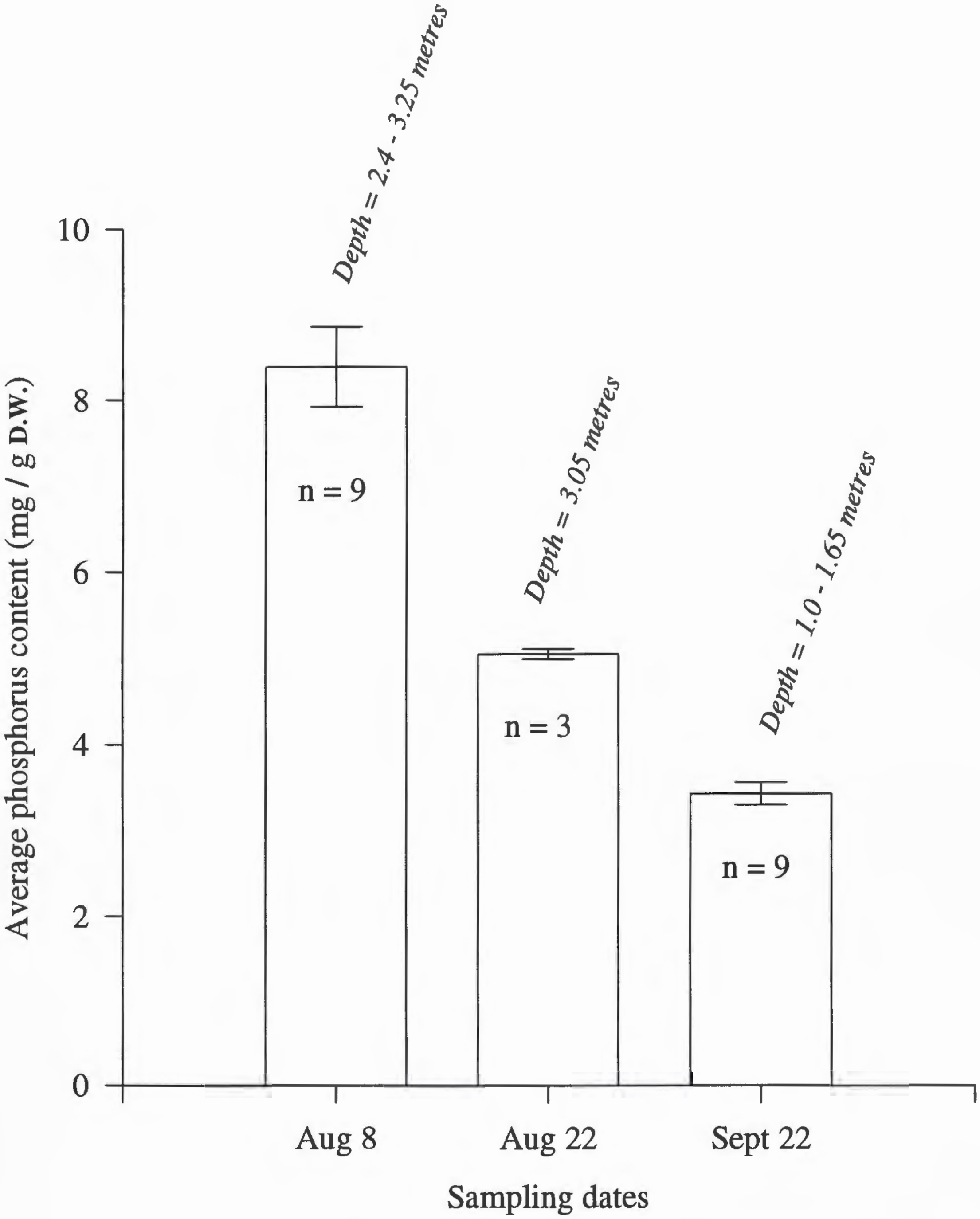


Figure 4.10 Comparison of phosphorus concentrations between different sampling periods. The first three samples were collected for in-vitro analysis.

The results from the leaching experiments on August 22 samples suggest that these plants were in transition between the growth stage and the overwintering stage. After one week of senescence, the phosphorus leached was less than 5 %, and the plants appeared to be alive, placing these samples in the same category as the samples from September 22. However after three weeks of senescence, decomposition occurred and 72% of the initial phosphorus was leached, placing these samples in the same category as the samples from August 8. The August 22 set of samples represented the median rate of decomposition, since the plants remained healthy after one-week, but decomposed after 3 weeks. It is possible that starch accumulation had already began when these samples were harvested, but the accumulation was insufficient to supply the plants energy requirements for three weeks.

Comparison of three macrophyte leaching studies

Carpenter (1980) found that *M. spicatum* leached most of its phosphorus, two days after the plant was killed. Carpenter concluded that rate of leaching depends inversely on time since death and directly on initial phosphorus concentration. From his experiments, a formula for leaching was presented which incorporated these two factors in estimating the rate of leaching over time. This model is not an appropriate comparison for the Tabor Lake samples as Carpenter killed his plants by freeze drying and then began his experiment. Tabor Lake plants were light deprived but death was not immediate and so the rate of phosphorus loss is modified accordingly.

Pieczynska and Jachimowicz-Janaszek (1988) studied two aspects of decomposition in *E. canadensis*: fragmentation of the stem and release of ions from senescing plants. Their experiment was conducted in-vitro and used similar apparatus to the Tabor Lake in-vitro study (dark bottles with an aeration system). Samples were collected between May and November near Warsaw, Poland. Although their experiment did not measure phosphorus release directly, the release of ions (measured as changes in the electrolytic conductivity of the water) was measured at daily intervals throughout the decomposition experiment. The greatest increase in conductivity occurred between the second and sixth day of the experiment. It can be interpreted that phosphorus leaching was likely correlated with the increase in electrolytic conductivity. Pieczynska and Jachimowicz-Janaszek observed that fragmentation of the stem was greatest 8 days after senescence began (2 days after the peak ion release occurred). These results are similar to those observed in the Tabor Lake in-vitro study (in-vitro experiment using August 8 samples).

Figure 4.11 shows the leaching results observed in the Tabor Lake experiment and the electrolytic conductivity results measured by Pieczynska and Jachimowicz-Janaszek (1988). Whereas Carpenter's model for *M. spicatum* predicts a continuous decrease in the rate of P leaching, Pieczynska and Jachimowicz-Janaszek's leaching rates for *E. canadensis* follow a logistic curve. Three data points for the August samples can be plotted for the Tabor Lake initial P, percent leached after one week and percent leached after three weeks. It is likely that leaching follows the same logistic type curve measured by Pieczynska and Jachimowicz-Janaszek, however more data points are required to confirm this pattern.

Applicability of Results

Generally, the purpose of in-vitro studies is to isolate some variable(s) from the natural environment so that they can be studied with some degree of control. However, it has been found that in-vitro studies are often too different from the natural environment to provide meaningful results. For example, Schindler (1977) discusses why the literature reported conflicting results as to which factor was limiting the productivity in lakes. All three elements (carbon, nitrogen and phosphorus) were reported to control primary productivity. However, many of these studies were conducted in-vitro and the isolation of these variables from the natural environment inaccurately described the role of each variable in the context of the aquatic ecosystem. Only after conducting whole lake studies was

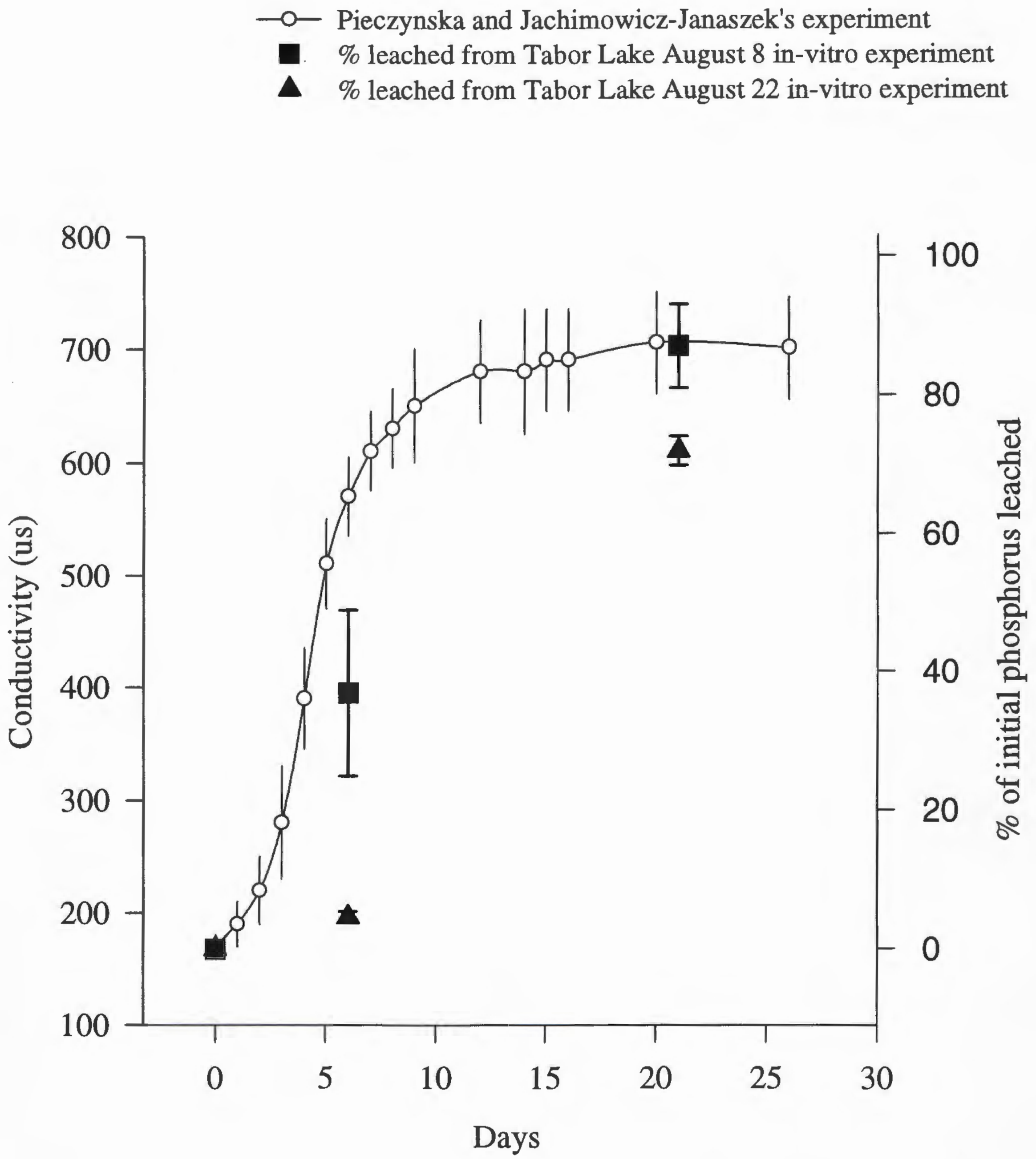


Figure 4.11 Comparison of changes in electrolytic conductivity of water during decomposition (Pieczynska & Jachimowicz-Janaszek, 1988) and percent phosphorus leached from in-vitro leaching experiments on *E. canadensis*.

phosphorus determined to be the limiting factor. The same dilemma applies to the Tabor Lake in-vitro experiment.

The purpose of the in-vitro experiment was to determine leaching rates from senescing *E. canadensis* and the experiment was designed so that this natural process could be reproduced in the lab. The results produced from this study, however, demonstrate some short-comings with this procedure. The most striking observation was the difference between samples collected in August and samples collected in September. Although the August samples behaved as expected (i.e. they died and proceeded to decompose, which followed patterns reported in the literature), the fact that September samples did not decompose illustrates the enormous physiological differences which can occur in plants over a six week period, between August 8 and September 22.

Field observations of *E. canadensis* demonstrate that turion formation is extensive (Bowmer *et al.,* 1984) and starch accumulation prepares the plant for prolonged periods of poor growing conditions (Sculthorpe, 1967; Janauer, 1981; Spicer and Catling, 1988). Using the in-vitro measurements of P leaching (from August 8 samples) to predict macrophyte phosphorus loading likely overestimates the true loading occurring in the lake since a portion of the plant community did not decompose. Alternatively, if the September 22 samples were used to predict macrophyte phosphorus loading, they would discount any phosphorus leached from senescing macrophytes and therefore underestimate the influence of macrophytes on the phosphorus concentration in Tabor Lake.

Another unexpected observation was the decline of initial phosphorus in macrophytes collected after overwintering occurred (Figure 4.10). These findings show a decrease in tissue phosphorus between the growing months and the overwintering period and may represent another pathway by which phosphorus is released from the macrophyte into the water column. Similar observations have been reported by Rorslett *et al.* (1985) in lake Steinsfjord, where a decrease in tissue phosphorus concentration between summer and autumn was observed. Table 4.2 shows the changes in tissue phosphorus for *E.*

Table 4.2 Comparison of phosphorus content in summer, autumn and winter macrophyte samples between Tabor Lake and Lake Steinsfjord, Norway. *(Elodea canadensis)*

Tabor Lake		Lake Steinsfjord*	
Sampling Dates	Phosphorus Content (%)	Sampling Dates	Phosphorus Content (%)
Aug 8	0.84	Summer	0.7
Sept. 22	0.33	Autumn	0.4
December	0.31	Winter	0.3

* Lake Steinsfjord data taken from Rorslett et al. (1985)

canadensis obtained from Tabor Lake during August, September and December, as well as the summer, autumn and winter tissue phosphorus observed by Rorslett *et al.* (1985). Also, Best (1978) has shown the concentration of adenosine tri-phosphate (ATP) in *E. canadensis* declines between summer and winter months. ATP is the primary energy molecule and during summer growth the high energy requirements for the plant are supplied by ATP molecules (Salisbury and Ross, 1992). Although ATP is also required during winter, metabolic processes are slowed dramatically and therefore the amount of ATP is found in decreased amounts. This seasonal change in metabolic activity would explain the decrease in ATP found by Best and might explain why tissue phosphorus decreases during the fall and winter season.

An alternative explanation for the decrease in tissue phosphorus levels of plants between August 8 and September 22 samples is caused by a relative increase in total starch, and therefore a relative decline in total phosphorus. Titus (1977) studied the allocation of total non-structural carbohydrates (TNC), in *Myriophyllum spicatum* over an entire year. He found that between early summer and October, TNC in *M. spicatum* increased from 5% to 20% indicating the plant was accumulating starch in preparation for overwintering. If this process occurred in the population of *E. canadensis* in Tabor Lake, this might account for the observed decline in tissue phosphorus. However, the decline in tissue phosphorus between August 8 and September 22 samples was greater than 50% of the August 8 samples and would require a corresponding increase in TNC of more than a 100% (at minimum) since the plant would have to double its mass. This increase is much larger than the increase of TNC from 5% to 20% reported by Titus.

<u>Estimating macrophyte leaching using August 8 and September 22 results</u>

There are two leaching values for each in-vitro experiment; leaching after one week and leaching after three weeks. The three week leaching estimate will be used because the one week leaching represents incomplete leaching. The mean percentage leached from the

August 8 samples is 86.7% (S.E.= 6.1%) of the initial phosphorus content while the mean phosphorus content of these samples is 8.6 mg P/g D.W. (S.E.= 0.47 mg P/g D.W.), resulting in a final leaching estimate of 7.5 mg P/g D.W. (S.E.= 0.93 mg P/g D.W.). The mean percentage leached from the September 22 samples is 9.4% (S.E.= 1.3 %) of the initial phosphorus content which was, on average 3.3 mg P/g D.W. (S.E.= 0.12 mg P/g D.W.) resulting in a mean leaching estimate of 0.31 mg P/g D.W. (S.E.= 0.05 mg P/g D.W.). Propagation error was calculated by multiplying the initial P content of the samples by the standard error of mean percentage leached, to account for the error around the leaching estimate. Next, the percentage leached was multiplied by the standard error of the initial P content to account for the error around the initial P estimate. These two errors are then added together to produce the standard error of phosphorus leaching from senescing macrophytes.

The total macrophyte biomass in Tabor Lake at the end of August, 1995 was estimated using an echosounder with chart recorder to generate a profile of the macrophyte population from 23 transects around Tabor Lake. SCUBA quadrat samples were also collected and weighed in order to calibrate the echosound tracings (for a detailed explanation of this technique as well as an explanation of how standard error was calculated, see Appendix A). The macrophyte biomass estimate is 376,500 kg D.W. (S.E.= 124,000 kg D.W.). Multiplying this estimate of biomass with the August 8 and September 22 leaching estimates results in a total phosphorus leaching load of 2824 kg and 117 kg, respectively.

To account for the propagation of errors between the biomass estimate and the leaching estimate, the standard errors of both estimates were added together, and calculated in three steps. First, the standard error of macrophytes (124,000 kg) was multiplied by the phosphorus leaching estimate (7.5 mg P/g D.W.) to account for the error around the macrophyte estimate. Next, the standard error of the leaching estimate (0.93 mg P/g D.W) was multiplied by the macrophyte biomass (376,500 kg) to account for the error around the leaching estimate. Finally, the two standard errors were added together. The standard error of the August 8 in-vitro estimate is 1280 kg, while the standard error of the September 22 is 57 kg.

The next section presents an alternative macrophyte leaching estimate which is obtained by comparing the total phosphorus in macrophytes between summer and winter samples to estimate a net release of phosphorus to the lake's water column.

4.5 In-situ leaching estimate of phosphorus from *E. canadensis*

An alternative approach to estimating phosphorus release from *E. canadensis* is presented because of limitations in using laboratory experiments to explain the natural world. Results from the in-vitro experiments conducted on August 8 and September 22 to estimate the phosphorus leaching rates from healthy plants underscores the limitations of controlled experiments which try to explain a natural process (see previous section). The purpose of this section is to provide an in-situ estimate of phosphorus loading by subtracting the phosphorus concentration in healthy summer macrophytes from grab samples of both living and dead macrophytes in the winter. This estimate of phosphorus release is based on the assumption that the difference between the August samples and the winter samples represents the release of phosphorus into the lake's water column.

The summer samples use the phosphorus concentrations measured in the August 8 in-vitro samples because they represent a healthy, growing plant, which has not yet prepared for overwintering. The winter samples were collected in triplicate during the first week of December from 8 locations around Tabor Lake between 0.75 m and 2.25 metres in depth, approximately. The sampling procedure for the December samples was similar to the August 8 samples, except no separation of healthy and unhealthy stems was conducted in order to include both surviving and senescent plants.

The average tissue phosphorus concentration of the August 8 and December samples are 8.4 mg/g D.W. (S.E.= 0.47 mg/g) and 3.2 mg/g D.W. (S.E.= 0.12 mg/g), respectively. These results show a 62% decrease in tissue phosphorus between sampling dates.

Subtracting the mean phosphorus concentration of the August samples from the December samples gives an average leaching value of 5.2 mg P/g D.W., and the standard error of both estimates are added together, so that S.E. = 0.59 mg P/g D.W. Multiplying the biomass value (376,500 kg) with the leaching estimate provides a total phosphorus load of 1958 kg (S.E.= 867 kg) from senescing macrophytes. Standard error was calculated following the same procedure used for the in-vitro estimate.

4.6 Comparing leaching estimates to the whole lake phosphorus budget

The three macrophyte leaching estimates developed in the previous sections differ greatly in their prediction of total phosphorus release from senescing macrophytes. In this section, these estimates are compared to the whole lake phosphorus budget in order to assess which leaching estimate best predicts the observed changes in lake phosphorus. This comparison is complicated by sediment phosphorus release during periods of anoxia and therefore phosphorus increases as a result of input from the sediment (as determined in section 4.2) will be subtracted from the total observed increases in lake phosphorus. Although external loading during the summer and autumn also adds to the total phosphorus in Tabor Lake, the total contribution from surface runoff and atmospheric deposition between June and October is considered negligible, contributing approximately 25 kg over the 5 months (see chapter 3).

<u>Comparing phosphorus leaching estimates to observed changes in lake phosphorus</u>

Since the leaching estimates only provide a "bulk-sum" delivery estimate, an additional parameter must be added to account for the rate at which leaching occurs. The in-vitro experiments are based on three week intervals and are not constrained to any specific time period past August 8. Therefore, they are applied to the time frame when maximal phosphorus loading is observed. However, since the time frame of the in-situ estimate is about 4 months, the model will be applied to the remainder of the sampling season after August 8, when the first set of samples were collected.

Figure 4.12 shows weekly observed values of total lake phosphorus increase (total lake load $week_n$ – total lake load $week_{n-1}$) in Tabor Lake during the 1995 sampling season and the predicted phosphorus loading from the hypolimnion calculated in section 4.2. The largest three week increase in lake phosphorus occurred between August 30 and September 20, when a cumulated 2087 kg of phosphorus was introduced into the water column. This three week loading will be used to compare in-vitro leaching estimates with field observations. As shown in Figure 4.12, hypolimnetic phosphorus loading during this period accounted for 203 kg (conservative estimate) to 491 kg (liberal estimate), leaving between 1596 kg to 1884 kg of phosphorus unaccounted.

The first leaching estimate (using the August 8 in-vitro samples) predicts that 2824 kg of phosphorus (S.E.= 1280 kg) is leached from senescing plants. This overestimates the total unaccounted phosphorus in Tabor Lake by 940 kg to 1228 kg, but is within one standard error. The second in-vitro leaching estimate (September 22 in-vitro samples) predicts only 117 kg of phosphorus (S.E.= 57 kg) is leached from senescing plants and greatly underestimates the observed phosphorus increase in Tabor Lake.

The in-situ leaching estimate applies to the sampling period after August 8. During this 10 week period, the total phosphorus increase is 2647 kg, with hypolimnetic phosphorus loading accounting for 203 kg (conservative estimate) to 491 kg (liberal estimate). This leaves between 2156 kg to 2444 kg of phosphorus unaccounted. The in-situ leaching estimate predicts 1958 kg of phosphorus (S.E.= 867 kg) is leached from the plants, and is within one standard error of the unaccounted phosphorus increase.

Of the three leaching estimates presented above, only the September 22 in-vitro estimate can be discarded as a poor predictor of phosphorus leaching. The August 8 in-vitro leaching estimate and the in-situ leaching estimate are within one standard error of the observed increases in Tabor Lake phosphorus during late summer and early autumn. However, the August 8 in-vitro estimate overestimates the total phosphorus load into Tabor Lake by 964 kg (using liberal hypolimnetic loading in the calculation) and 1222 kg (using

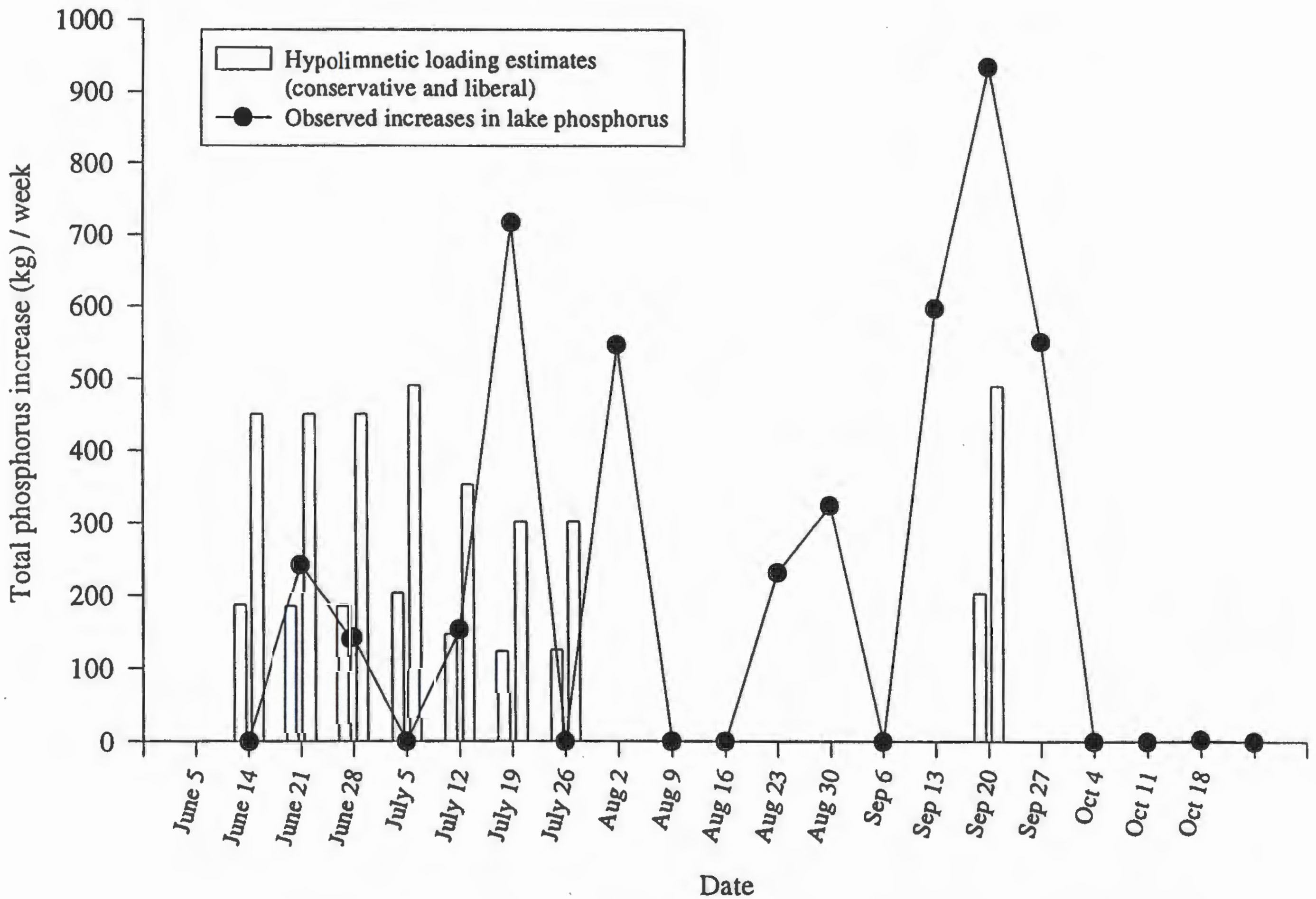


Figure 4.12. Weekly increases in total lake phosphorus (line graph), with estimates of internal loading from hypolimnion (both conservative and liberal release estimates).

conservative hypolimnetic loading in the calculation). The in-situ estimate underestimates the total phosphorus increase by 486 kg (liberal) and 198 kg (conservative). The in-situ phosphorus release estimate represents the closest prediction of all three estimates to the observed changes in lake phosphorus. Furthermore, the August 8 in-vitro experiment assumes almost complete decomposition of the macrophytes in Tabor Lake, which was not observed as many turions had formed in preparation for overwintering.

Although the standard error associated with the in-situ release rate is large, the contribution of senescing macrophytes to internal phosphorus loads represents 44% (S.E.= 20%) of the total observed increase in Tabor Lake during the 1995 sampling season. As the bulk of this phosphorus is delivered to the lake during the late summer and early autumn, the calculated proportion of phosphorus loading during this period (August to October) is 74% (S.E.= 34%).

Limitations on whole lake estimates of phosphorus dynamics

Changes in the total phosphorus of Tabor Lake are estimated to illustrate the weekly pattern over the course of the sampling season. The phosphorus dynamics in Tabor Lake are much more complex and occur over shorter time periods than the weekly sampling interval. There are two points to recall when interpreting the results: the technique used to describe whole lake phosphorus changes provide only coarse estimates and secondly, other potentially significant internal loading processes were not investigated.

The sampling technique used to estimate total lake phosphorus is limited in two ways—temporally and spatially. The rate (temporal) at which phosphorus accumulates in the hypolimnion can be difficult to determine accurately on a weekly basis when stratification in Tabor Lake is disrupted by high winds and cooling. Some of the hypolimnetic loading estimates are suspected of having temporary thermocline disruption between sampling periods, which is why a conservative and a liberal estimate was included.

The spatial design of Tabor Lakes sampling program was limited to the deep hole and one littoral station. The deep hole should adequately represent the pelagic portion of Tabor Lake since it is a single basin with bowl-like morphology. However, the extent of macrophyte colonization throughout the littoral zone creates a sampling problem. Macrophyte beds typically act as a sink for particulate phosphorus (Carpenter and Lodge, 1986) and this was observed in Tabor Lake when large accumulations of algae were trapped around thick mats of *E. canadensis,* especially at the windward end of the lake. To accurately estimate phosphorus in the littoral zone, many more sampling points should be chosen to account for the variation within Tabor Lake's littoral zone.

Although it has been demonstrated that senescing macrophytes leach phosphorus into the water column (Carpenter, 1980; Landers, 1982; Gabrielson *et al.*, 1984), there are several other internal loading mechanisms which have not been studied. Bostrom *et al.* (1982) has written a comprehensive review of internal loading mechanisms, which may apply to Tabor Lake. Wind resuspension of the sediment is one mechanism that can contribute significant amounts of phosphorus to the water column. On Tabor Lake, winds frequently run along the length of the lake, possibly resuspending the fine grain sediment. However, this loading mechanism was not quantified in this project as the littoral zone supports a very dense and deep (4 m) macrophyte bed. This extensive bed should act to dampen the wave action and retain sediments (Petticrew and Kalff, 1992). Another process of internal loading, known as bioturbation, is caused by sediment dwelling organisms that resuspend sediment phosphorus into the overlying water. During SCUBA surveys of Tabor Lake, abundant populations of tube-dwelling chironimids were found in the sediment, representing another potential source of phosphorus loading.

The hypolimnetic loading at the deep hole is believed to be the result of internal loading, where zero or even low (<1 mg/L) oxygen conditions affect the redox potential of the sediment-water interface. When the surficial sediments become reduced, phosphorus which is chemically bound to iron (strengite) dissociates and enters solution. This phenomenon is not limited to the hypolimnion and can occur anywhere in the lake, as long as anoxic conditions exist. Typically this occurs in the hypolimnion, however *E. canadensis* occasionally can also cause littoral oxygen depletions (Buscemi, 1958). Daytime oxygen conditions at the littoral station in Tabor Lake never recorded anoxia, however during the night when plants respire, oxygen consumption could have resulted in temporary anoxia.

The changes of whole lake phosphorus in Tabor Lake show that some process other than hypolimnetic loading contributes to the phosphorus loading in this lake. Although the internal loading mechanisms mentioned above have not been studied in Tabor Lake, the results from field and lab studies show that the extensive macrophyte population in the littoral zone and periods of hypolimnetic anoxia play significant roles in the internal cycling of phosphorus.

Chapter Five

Mechanical harvesting of macrophytes in Tabor Lake.

Macrophyte harvesting operations began on Tabor lake in 1991 in order to address the significant increase in the macrophyte population (Carmichael, 1994). Mechanical harvesting of macrophytes can be effective in removing nutrients prior to their release during senescence, but is sometimes only useful for cosmetic purposes (Cooke *et al.*, 1993). The purpose of this chapter is to discuss the effects of mechanical harvesting which has been reported in the literature, estimate the total phosphorus in the macrophyte population and show the results of a leaching experiment conducted on mechanically harvested plants from Tabor Lake and discuss the usefulness of mechanical harvesting as an effective remediation tool for managers to use.

5.1 Impacts of weed harvesting

The effects of mechanical weed harvesters on lake water quality was investigated by Carpenter and Gasith (1978) in Lake Wingra, Wisconsin. The parameters measured were conductivity, temperature, biological oxygen demand, dissolved organic carbon, nitrogen, phosphorus and oxygen. These parameters were studied in both harvested and unharvested plots. Their results show that minimal disturbance of the littoral zone was caused by mechanical harvesting. Although the study was conducted in a small area, this study indicates that harvesting the weeds should not contribute to the poor water quality in Tabor Lake.

Crowell *et al.* (1994) studied the impact of harvesting on growth rates of *Myriophyllum spicatum* and found that harvesting increased the plants growth rate. They found that plant biomass between the harvested and unharvested sections was the same after six weeks. Other studies have also found that plant biomass returned to pre-harvested levels 3-4 weeks after harvesting (Cooke *et al.*, 1990; Rawls, 1975). Although, these results were based on studies of *M. spicatum* which is not the dominant macrophyte in Tabor Lake, discussion with the harvester operator on Tabor Lake revealed that some areas which were harvested early in the summer (1993 and 1994) needed to be harvested again in late summer/early fall. This indicates that rapid regrowth of harvested areas in Tabor Lake is site specific.

5.2 Total phosphorus in Tabor Lake macrophytes

The total macrophyte biomass in Tabor Lake at the end of August, 1995 is estimated to have been 376,500 kg (S.E.= 124,000 kg)—see Appendix. The estimated phosphorus concentration is 8.2 mg/g D.W. (S.E.= 0.47 mg/g D.W.) from the August samples (Chapter 4, section 4). The estimated mass of phosphorus held within the macrophyte community is 3087 kg (S.E.= 1194 kg/g D.W.). This average concentration of phosphorus is calculated using healthy plants collected prior to overwintering. Using this phosphorus estimate assumes that the entire population of macrophytes in Tabor Lake were healthy at the time the biomass estimate was made. This assumption is not entirely accurate because unhealthy plants were observed at all times over the course of the sampling season, but it is reasonable to assume that all biomass in Tabor Lake was healthy at one time during the summer.

5.3 Phosphorus removal from weed harvesting

During the 1995 harvester season, 882 loads of aquatic macrophytes were harvested from Tabor Lake. Weight measurements taken during July show that each harvester load removes approximately 1130 kg (S.E.= 115 kg) of weeds from the lake—about 113 kg of dry weeds (S.E.= 12 kg), assuming 90% moisture content (Westlake, 1965). The estimated total dry mass of macrophytes removed from Tabor Lake during the 1995 harvesting season is 99,700 kg (S.E.= 10,600 kg).

The average tissue phosphorus concentration found in healthy macrophytes during the 1995 open water season varied considerably between the August 8 and September 22 samples. Since harvesting was conducted during both periods, the average phosphorus concentrations from all periods (Aug. 8, n=9; Aug. 22, n=3; Sep. 22, n=9) are used to estimate average phosphorus in weeds. The average phosphorus concentration from all dates is 5.8 mg/g D.W. (S.E.= 0.56 mg/g D.W.). Multiplying the total macrophyte biomass harvested, by the average phosphorus content measured in the plants sampled during the 1995 season results in the total phosphorus removed from Tabor Lake during this season, which is 578 kg. To estimate the error of this estimate, the two standard errors given above are added together. The error around total biomass removed and average phosphorus content are 61 kg and 56 kg of phosphorus, respectively which yields a total standard error of 117 kg of phosphorus.

5.4 Phosphorus leaching from harvested weeds on shore

Once the macrophytes are harvested, the standard disposal practice is to dump the weeds on shore, where they will sit until a loader and dump truck are available to remove them from the lakeside. They are then transported to a quarry a few hundred metres from the lake and left to decompose. The weeds will often sit on the shore for more than a day before a dump truck removes them. During this period some of the water and nutrients in the weeds leaches back into the lake. If a large percentage of phosphorus is released back into the lake when the weeds are on shore, then the harvesting operation represents a new phosphorus loading mechanism to the lake and does not assist in reducing the nutrient load in Tabor Lake.

To study the effect of this leaching process on the water quality of Tabor, a full harvester load of weeds were placed on a polyethylene sheet. Over a 25 hour period, the volume of runoff water from the weeds was measured periodically and tested for total phosphorus concentration. This estimate of phosphorus leaching was used to estimate total phosphorus release into Tabor Lake from lakeside weeds throughout the summer of 1995.

Figure 5.1 shows the rate of water runoff from the sitting weeds. Also, phosphorus concentrations of the runoff water is shown. Over the 25 hour period, more than 50% of the water and more than 70% of the total phosphorus is leached within the first hour. After five hours on the lakeside, the leaching rate of water and phosphorus remained fairly constant for the duration of the experiment. Although this may appear large at first glance, the total phosphorus released in the runoff represents only a fraction of the phosphorus remaining in the macrophytes. The total water released from the harvested weeds was estimated at 198 litres, and contained 426 mg of phosphorus. The total phosphorus content present in a harvester load is 655 g (113 kg D.W. biomass * 5.8 mg P/g D.W.).

To minimize phosphorus release from harvested macrophytes, the weeds need to be removed from shore immediately. However, the total phosphorus leached back into the lake by the 882 loads from 1995 is considered negligible, leaching approximately 0.375 kg of phosphorus into the lake during the harvesting season. Although this study did not quantify release beyond 25 hours, the results indicate that water release in inversely related to time since harvesting and phosphorus release remains relatively constant, five hours after harvesting.

This study was conducted on September 11, 1996; a warm and sunny day. These results may not be congruent with cloudy days, or days with high precipitation. It is likely that runoff will increase if it rains, and phosphorus release may also increase.

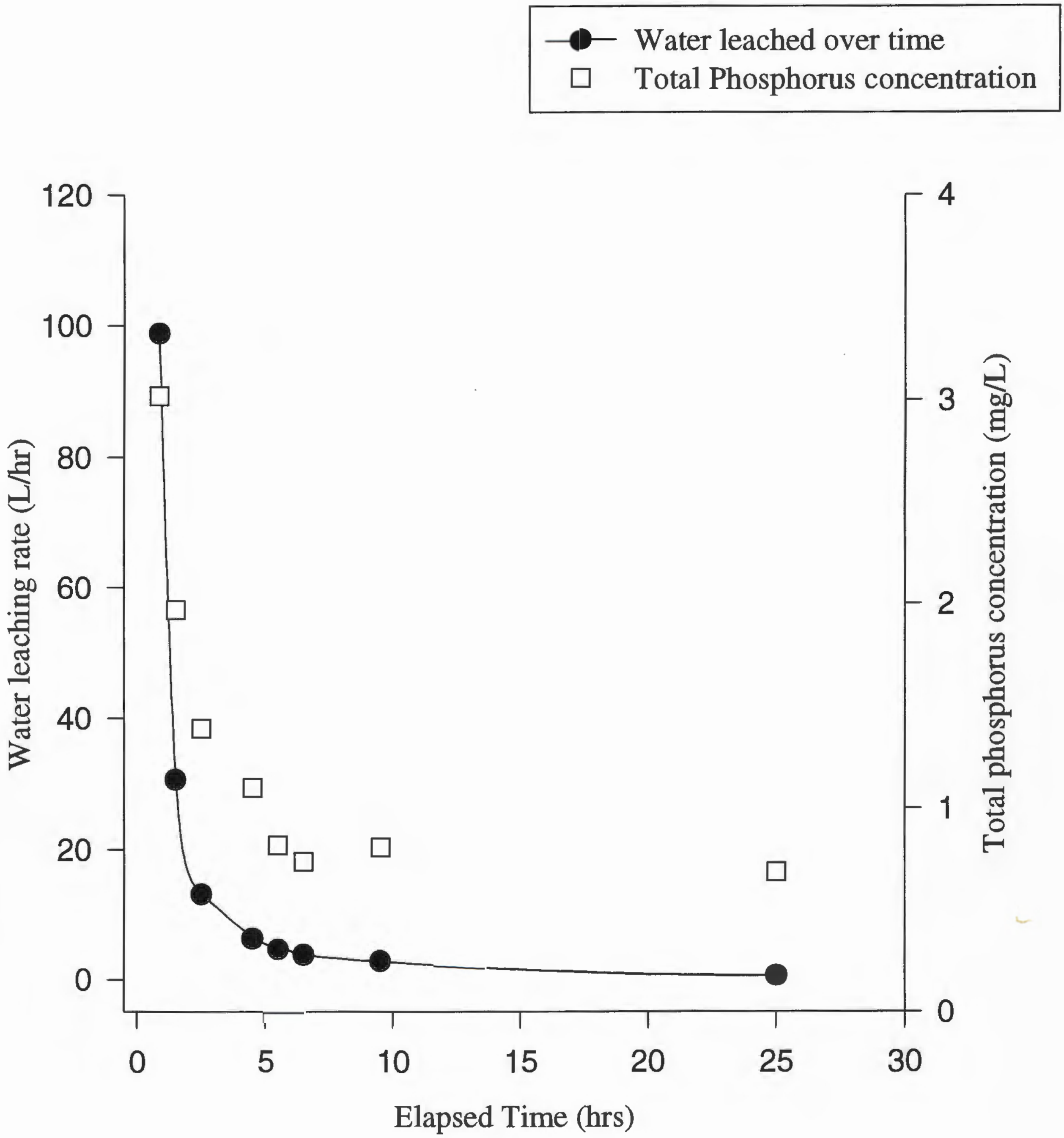


Figure 5.1 Water loss from weeds harvested by mechanical harvester. Experiment commenced immediately after weeds were dumped on shore. Phosphorus concentration of the water is also shown.

Chapter Six
Major sinks and sources of phosphorus in Tabor Lake

The purpose of this chapter is to relate the three types of phosphorus loading models developed in chapters 2 (regional lake survey), 3 (external loading estimate) and 4 (internal loading estimates) to provide a holistic perspective of phosphorus loading in Tabor Lake, and is used to revise the preliminary 1994 phosphorus budget presented in chapter one. Phosphorus removal from weed harvesting is also included in the budget. This revised phosphorus budget provides quantitative estimates of phosphorus loading that will assist in the decision making for Tabor Lake management. In conclusion, some remediation strategies are discussed and recommendations for further research are presented based on questions raised during the course of this thesis.

6.1 Description of phosphorus loading estimates

The regional lake survey places Tabor Lake in a regional context with other lakes, identifying patterns in the factors which are useful in predicting spring phosphorus levels. Spring phosphorus is a useful indicator of lake trophic status because it measures phosphorus concentration of a lake when the lake is fully mixed following spring overturn, and therefore should have an even distribution of phosphorus throughout the water column. Spring phosphorus measurements are also taken when the inputs of phosphorus from the watershed are usually at their highest levels.

Exploratory statistics used in chapter 2 determined common characteristics between Tabor Lake and other lakes in the Prince George region, based on spring phosphorus. Morphometric and watershed variables were used to predict spring phosphorus concentrations in 39 lakes throughout the region. Two multiple regression models were generated, the first using stepwise methods (Model I) and the second using limnological theory (Model II), to select significant variables. Although these two modeling approaches represent different pathways for identifying significant variables, they both converge towards the collection of similar predictive variables. The variables which were found to significantly influence spring phosphorus in both models are mean depth, percentage of wetlands along tributaries, percentage of agriculture along tributaries and watershed area. The lone variables (not common to both models) were elevation and old growth forests in Model I, and recently harvested forests in Model II. Model I accounted for 70% of the variation in spring phosphorus while Model II explained 59% of the variation.

Both regression models predicted spring phosphorus in Tabor Lake within one standard error. Furthermore, a nine year average of spring phosphorus values in Tabor Lake was also within one standard error of both model predictions. Based on these results, Tabor Lake is not considered anomalous to other lakes in the region in terms of spring phosphorus. However, the usefulness of this model is restricted since it does not adequately describe the behavior of phosphorus in Tabor Lake during summer and autumn. The proceeding two phosphorus loading estimates address this issue by quantifying the external and internal loading of Tabor Lake in 1995.

The external sinks and sources of phosphorus for 1995 were estimated in chapter 3 using a simple input-output model based on measured phosphorus concentrations and water discharge estimates. Figure 6.1 shows the weekly Tabor Lake input and output of phosphorus, as well as the net retention/release of phosphorus in Tabor Lake. The annual input and output of phosphorus in Tabor Lake is 196 kg and 188 kg, respectively, resulting in a net flux of 8 kg of phosphorus into Tabor Lake.

The internal sinks and sources of phosphorus were estimated during the 1995 open water season from weekly phosphorus measurements taken from Tabor Lake, combined with two experimental estimates of phosphorus release from senescing macrophytes. The weekly monitoring program showed that during summer stratification, phosphorus concentrations in the hypolimnion increase dramatically. The monitoring program also revealed that stratification in Tabor Lake is often disrupted, increasing the concentration of

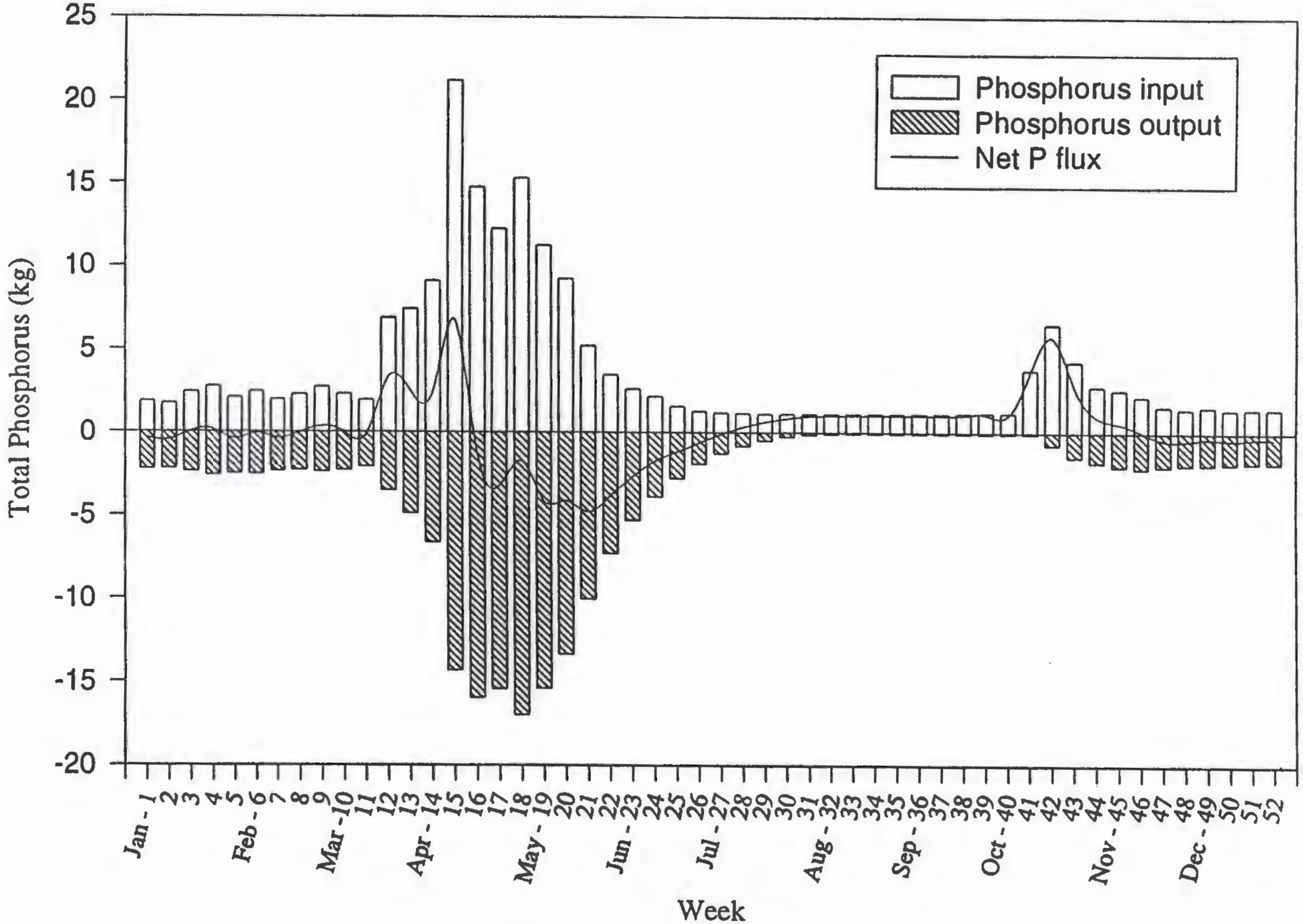


Figure 6.1 Weekly total addition and loss of phosphorus fromTabor Lake. The net retention/release is represented by the line graph.

phosphorus in the epilimnion and prompting extensive algal blooms. These results demonstrate that internal loading from the hypolimnion occurs in Tabor Lake. Results from the 1995 sampling season were used to estimate the load of phosphorus from the anoxic hypolimnion. Two estimates were generated, a conservative and a liberal estimate, which predicted between 1197 kg and 2290 kg of phosphorus was released from the hypolimnion during the 1995 sampling season.

Large increases in lake phosphorus were also observed during late summer and early autumn, 1995 which could not be explained solely by hypolimnetic phosphorus loading. Two macrophyte leaching experiments were conducted, one in-vitro and the other in-situ, to estimate the total phosphorus release from senescing macrophytes. These predictions were then compared with whole lake phosphorus calculations from weekly phosphorus measurements taken during the summer of 1995, to determine which model best predicted the observed changes in total lake phosphorus. The in-situ estimate, which compares summer and winter phosphorus concentrations in the macrophytes to estimate total phosphorus release, best predicted the observed increase in lake phosphorus. The in-situ estimate predicted 1958 kg (S.E.= 867 kg) of phosphorus was released from senescing macrophytes in 1995. The observed late summer increase in phosphorus when hypolimnetic loading is discounted is between 2156 kg and 2444 kg, which is within one standard error of the phosphorus leaching estimate.

Figure 6.2 shows the weekly increases of total phosphorus in Tabor Lake (solid line), and compares it to the predictions of internal loading estimates (bar graph). Although leaching estimates from macrophyte senescence are evenly averaged at 338 kg of phosphorus per week, it is expected that phosphorus delivery from senescing macrophytes is less constant, following environmental cues such as temperature and light availability.

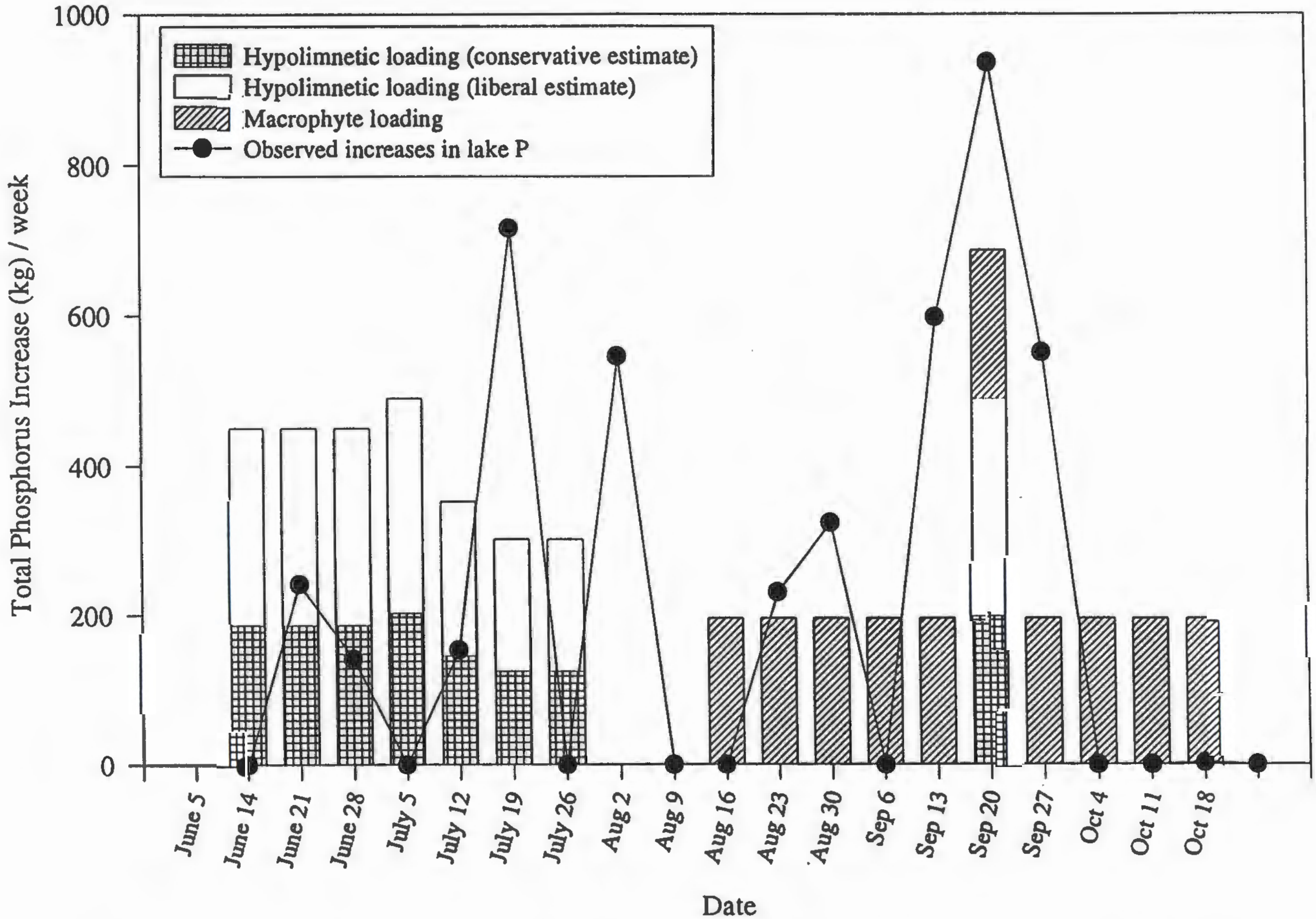


Figure 6.2 Weekly increases in total phosphorus observed in Tabor Lake (line graph), with the estimates of (i) conservative hypolimnetic loading, (ii) liberal hypolimnetic loading, and (iii) macrophyte loading.

6.2 Revised phosphorus budget for Tabor Lake

In chapter 1, a preliminary phosphorus budget was presented and several gaps in the budget were also identified (Figures 1.3 and 1.6, respectively). The results from chapters 3, 4 and 5 provide estimates to revise the original phosphorus budget for 1995 values and also includes phosphorus estimates for the identified data gaps (Figure 6.3).

Revisions of original phosphorus budget

The sinks and sources of phosphorus estimated for the 1995 sampling season found inflow values to be slightly more than calculated in the original budget, while outflow values were about double the original estimate. The underestimate of total phosphorus discharged through Tabor Creek is the result of underestimating the phosphorus concentration in the discharge water of Tabor Creek, and also underestimating the total discharge of water. The original estimate (Ward, 1995) presumed phosphorus concentrations were 20 μg/L (S.E.= 7 μg/L) during peak runoff from April to mid June, while measurements of total phosphorus taken from Tabor Creek from March 26 to May 28, 1995 were much higher, at 33 μg/L (n=6; S.E.= 4 μg/L). The total discharge of water during 1995 was estimated at 5.7 Mm^3, which was also higher than the preliminary estimate of 4.2 Mm^3.

The original phosphorus budget estimated the minimum and maximum phosphorus mass in Tabor Lake at any one time was 241 to 1570 kg, respectively. This value was estimated for the 1995 sampling season, and was found to range from 362 kg to 2634 kg of phosphorus. The minimum value of 362 kg was measured on April 30, 1995 a few days after the ice melted and is similar to the minimum value originally estimated. However, the largest instantaneous value of phosphorus measured during 1995 is considerably larger than previously recorded. Several factors contribute to this abnormally large phosphorus estimate. First, water column phosphorus concentrations at the deep hole during 1995 were the highest ever recorded. Also, total phosphorus present was calculated slightly differently

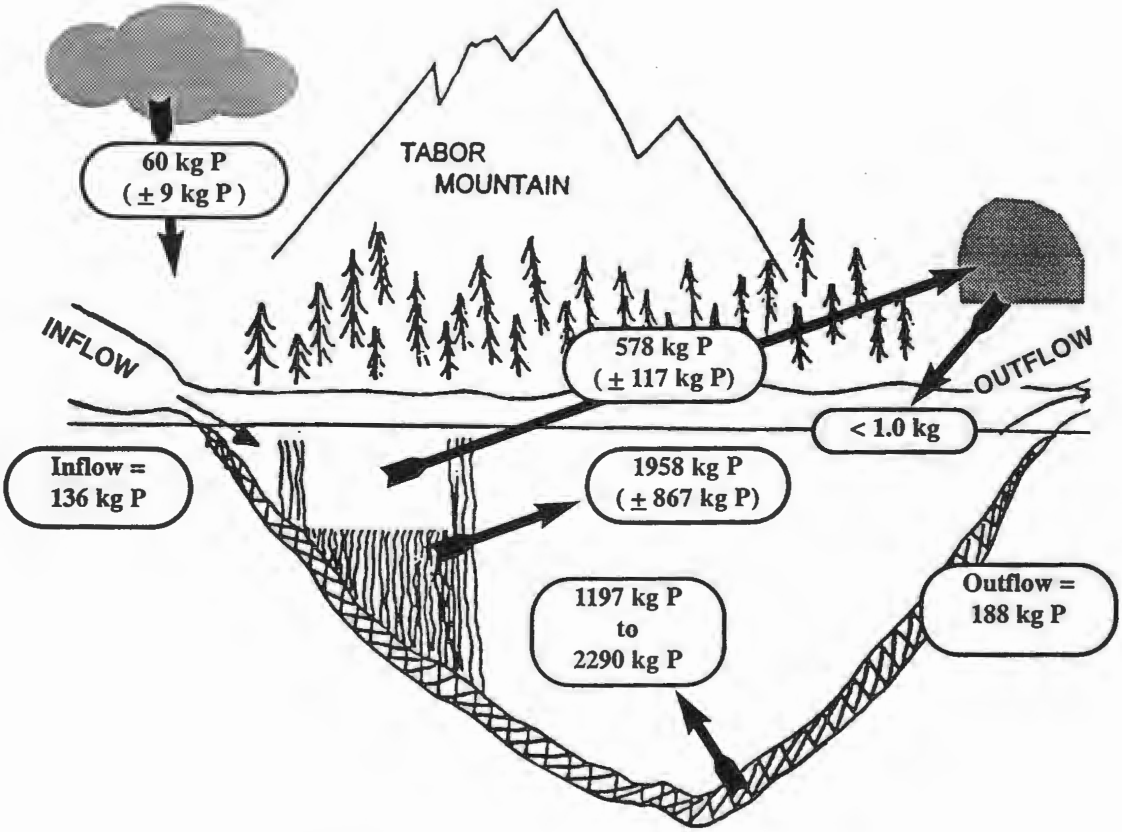


Figure 6.3 Revised phosphorus budget for Tabor Lake, 1995.

Time frame of phosphorus estimates presented above:

- Inflow and outflow estimates are for an entire year (November to October);
- Phosphorus loading from hypolimnion and macrophytes apply to summer and early autumn
- Phosphorus loading from precipitation represents annual load directly into Tabor Lake;
- Phosphorus removal from harvesting applies to summer and early autumn.

(based on 1995 data)

than previously calculated. This new technique divided the lake into three different compartments (hypolimnion, epilimnion and littoral zone). During the late summer and early autumn, the littoral zone was often found to have considerably higher concentrations of phosphorus than the pelagic zone. Although this new technique accounts for some of the increase in maximum lake phosphorus, 1995 was a year with abnormally high phosphorus levels.

The last item revised from the original phosphorus budget is total phosphorus removed from harvested macrophytes. In 1995, 882 loads of macrophytes were harvested from Tabor Lake removing approximately 578 kg of phosphorus (S.E.= 117 kg). This revised estimate is much higher than the original estimate of 88.4 kg because of increased harvesting and greater precision in the estimate of total macrophyte biomass removed and mean total phosphorus measured in the plants was higher than previously observed.

Phosphorus estimates for data gaps in original budget

During 1995, internal loading of phosphorus from the hypolimnion and senescing macrophytes, were estimated. The total phosphorus load released into Tabor Lake during 1995 from hypolimnetic anoxia was between 1197 kg and 2290 kg, while the estimated total phosphorus release from senescing macrophytes was 1958 kg (S.E.= 867 kg). These internal loads explain the large changes in total lake phosphorus observed during the sampling season.

A study of atmospheric phosphorus deposition was conducted during 1996 and found that Tabor Lake received an estimated 60 kg of phosphorus directly from the atmosphere, annually (Petticrew, 1997). Although this estimate was based on 1996 data, it provides a baseline estimate of total phosphorus deposition and demonstrates that atmospheric deposition is small relative to other sources.

Also, a study of the total phosphorus released from harvested macrophytes on shore was conducted. The results show that maximum release of phosphorus from recently harvested macrophytes occurs during the first hour after harvest. However, the total phosphorus released is considered negligible (less than 1 kg from 882 loads).

6.3 Management implications

Determining appropriate remediation strategies for Tabor Lake first requires that a comprehensive phosphorus budget for the lake is available in order to identify the major sinks and sources of this nutrient. The present total external phosphorus load is not likely to degrade lake water quality by itself because the input-output model used in chapter three, estimates only a small net loss of phosphorus to the sediment between Tabor Lake's inflow and outflow. Estimates of the internal phosphorus loading in Tabor Lake show phosphorus release during hypolimnetic anoxia and from senescing macrophytes are important factors regulating the trophic status of Tabor Lake. For improved lake water quality, remediation strategies need to focus on reducing the internal loading of phosphorus in Tabor Lake. Several management options are presented below which discuss the anticipated effects of each option on water quality, based on the 1995 phosphorus budget. The management options do not represent an exhaustive list of possible remediation strategies, but were selected based on existing and planned remediation for Tabor Lake. Also, the potential usefulness of sediment removal is briefly discussed since the ultimate source of Tabor Lake's internal loading is derived from sediment phosphorus.

No action

If no remedial action is taken, the result on Tabor Lake's phosphorus budget will be primarily determined by the input-output estimate developed in chapter 3. Based on the results from 1995, approximately equal quantities of phosphorus (a net flux of 8 kg of phosphorus into Tabor Lake) enter and exit Tabor Lake. Furthermore, any slight changes in the flux of phosphorus from external inputs and outputs is outweighed by the quantity of phosphorus within the lake sediments by almost 4 orders of magnitude.

Aeration

The process of aeration involves introducing oxygen into the hypolimnetic water, with the desired effect of preventing anoxia. This approach might be effective in controlling phosphorus release from the hypolimnion. Studies on other lakes which have aeration systems installed show a decrease in hypolimnetic phosphorus concentrations compared to pre-aeration conditions (Cooke *et al.*, 1993). Generally these systems serve to reduce the phosphorus concentration in the hypolimnion but do not eliminate phosphorus loading. Cooke *et al.* (1993) present phosphorus reduction estimates for four lakes with hypolimnetic aeration (Jarlasjon, Waccabuc, Wesslinger and Amisk), ranging between 30% to 57%. It is hypothesized that phosphorus remaining in the hypolimnion during oxic conditions is due to insufficient concentrations of iron, which binds to phosphorus causing both elements to precipitate out of solution. Lean *et al.* (1986) added iron to an aerated, 15-m deep enclosure and found soluble reactive phosphorus concentrations decreased sharply.

There are several environmental implications of aeration, beyond the reduction of phosphorus concentrations. Presumably, reduced phosphorus loading will reduce the phytoplankton population and increase the depth of light penetration in the water column. Since macrophyte populations are often limited by light availability (Barko *et al.*, 1986), a reduced algal bloom may expand macrophyte habitat in Tabor Lake. Another impact may arise from the zooplankton population which are normally restricted to the epilimnion during periods of hypolimnetic anoxia. An aeration system might increase the available habitat for zooplankton populations and as they move into the hypolimnion, there may be an increase in the protection from predators since there is less light available at deeper depths. In Hemlock Lake, the abundance of *Daphnia pulex* increased by a factor of 90 after a hypolimnetic aeration system was introduced (Fast, 1971). However, an increase in zooplankton may not occur in Tabor Lake due to its shallow nature and frequent summer destratification.

Even though aeration may reduce phosphorus concentrations in the hypolimnion, it only acts as a band-aid solution because it does not remove the phosphorus from the sediments. Also, the cost of purchasing, installing and operating an aeration system is prohibitive since it requires a continual infusion of funds for its operation. Assuming an aeration system was installed, phosphorus loading from senescing macrophytes would still occur during late summer/early autumn.

Macrophyte harvesting

The macrophyte community in Tabor Lake is the second largest compartment of phosphorus in Tabor Lake, holding an estimated 3087 kg during peak biomass in August, 1995. The release of phosphorus from senescing macrophytes represents a large source of phosphorus to the lake during late summer and early autumn. In theory, the removal of these macrophytes prior to senescence should improve the water quality of Tabor Lake, however three factors make this objective difficult to achieve. First, only 26% of the total estimated macrophyte biomass was removed from Tabor Lake during the harvesting season of 1995. To remove the entire biomass of Tabor Lake would require more machines.

The second factor deals with the time frame at which harvesting takes place. Macrophyte harvesting was conducted between June and early October, while macrophyte senescence occurred from the end of August to the middle of September. The plants which were harvested after August 31 are assumed to have undergone senescence, at which point harvesting does not serve to reduce internal loading from macrophytes. The harvesting log book shows more than 200 harvester loads were removed after the macrophytes began senescing and releasing phosphorus into the water column in late August. In other words, about 6% of the harvested macrophytes had already released phosphorus into Tabor Lake.

The third factor which complicates macrophyte removal prior to senescence is that rapid regeneration of macrophytes was observed in some areas of Tabor Lake, particularly in front of the Log House restaurant (personal observation). In these areas, the macrophyte population may return to pre-harvest levels prior to senescence. For harvesting to eliminate (or substantially reduce) the internal loading from senescing macrophytes, most of the macrophytes in the lake would need to be harvested in a short period of time (less than 3 weeks) prior to senescence. This is an expensive undertaking and would be required each year to prevent phosphorus loading from senescing macrophytes.

Over several years of harvesting, the phosphorus which is removed by this technique reduces the total phosphorus available for biological production within the lake. Since the ultimate goal of remediation in Tabor Lake is to reduce the total level of phosphorus, harvesting the macrophytes is useful for this purpose. In 1995, 882 loads of macrophytes were harvested, which removed 578 kg (S.E.= 117 kg) of phosphorus from Tabor Lake. Although it does not reduce the immediate phosphorus loading from macrophytes, it provides a technique to remove phosphorus from the internal loading cycle.

Hypolimnetic withdrawal

During the 1995 sampling season, loading from the hypolimnion contributed between 839 kg and 2291 kg of phosphorus to Tabor Lake's water column. Theoretically, the removal of phosphorus enriched hypolimnetic water prior to its release into the epilimnion prevents the large blooms of algae from forming, thus improving the lakes water quality. Two estimates of phosphorus removal have been calculated for Tabor Lake using a 24 inch diameter pipe (Ward, 1996). The two estimates, 130 kg and 312 kg of phosphorus, are based on a gravity driven system and a pump driven system, respectively. If this system were in place during the 1995 sampling season, the total phosphorus removed would not have been sufficient to remove all of the hypolimnetic phosphorus from the lake. The gravity driven system would have removed between 6% to 15% of the total hypolimnetic phosphorus, while the pump driven system would have removed 14% to 37% of total hypolimnetic phosphorus. The remaining phosphorus would have been cycled into the epilimnion and become available for algal uptake.

It is difficult to predict the how long it will take the hypolimnetic withdrawal system to improve the water quality of Tabor Lake. Nurnberg (1987) found that at least five years of operation was necessary before many lakes in her study experienced noticeable improvements in water quality. The large storage of phosphorus within the sediments and the relatively small proportion removed from the proposed hypolimnetic withdrawal indicates that more than five years of operation will be necessary before the sediment phosphorus is noticeably reduced. However, it is still unclear as to how much sediment phosphorus is available from hypolimnetic release. If most of the phosphorus is insoluble under normal conditions, five years may be a reasonable time frame to notice water quality improvements. Further study on sediment phosphorus fractions would be useful to help answer this question. This system achieves the same long term goals as macrophyte harvesting, removing phosphorus from the internal loading cycle of Tabor Lake.

An undesirable side effect of a hypolimnetic withdrawal system is that it transfers anoxic, phosphorus enriched water from one location to another. If the receiving end of a hypolimnetic withdrawal system is sensitive to low levels of oxygen and/or increased phosphorus levels, a hypolimnetic withdrawal system may just be transferring the problem of one aquatic system to another. The problem of anoxia can be easily solved by aerating the hypolimnetic before discharging it downstream. Removing the phosphorus prior to releasing the water downstream is more difficult, but may be solved by constructing a wetland immediately after the water is released from the hypolimnetic withdrawal pipe. Constructed wetlands are known to remove nutrients, which should prevent the export of phosphorus downstream.

<u>Water level control</u>

A weir was installed at the Tabor Lake outlet in the winter of 1996, to control the water level in Tabor Lake, thereby having some control over Tabor Lake discharge. The usefulness of regulating lake water levels is that during periods of high phosphorus, a greater volume of water can be discharged downstream. This strategy does not focus on a specific internal loading mechanism, but can increase the lake's flushing rate after internal loading occurs. Although this method will not likely improve the water quality of Tabor Lake noticeably, it serves the long term objectives of increasing the rate of phosphorus loss from the lake.

Skaret Creek diversion

Another remediation option for Tabor Lake is the diversion of Skaret Creek directly into the lake. This technique, known as dilution and flushing, can achieve improved water quality by reducing the concentration of phosphorus in the lake and increase the flushing rate. This approach is only feasible if water is available during the periods of internal loading. In 1995 maximum discharge of water occurred during April and May, whereas internal loading in Tabor Lake occurred between mid June and October. Skaret Creek flows during the period of internal loading are too small to achieve the objectives of dilution and flushing and will not achieve the desired effect.

Sediment removal

The phosphorus budget constructed for Tabor Lake shows that most of the phosphorus is stored within the lakes sediment and ultimately, is the source of internal loading in Tabor Lake. Removal of this phosphorus compartment from the lake should achieve two goals: control of algal blooms and control of macrophyte growth. The estimate of phosphorus stored in Tabor Lake's sediment (162,000 kg; S.E.= 10,100 kg) is based on sediment samples taken from the top 3 cm of the sediment, however it is unknown how much phosphorus is stored in sediments beneath the 3 cm layer and the depth of organic and fine grained sediment is unknown. Sediment coring of other lakes have shown 6 to 10 metres of sediment at the centre (pers. comm. R. Nordin). SCUBA surveys at the deep hole have shown at least 2 metres of soft sediment is present. Carignan and Flett (1981) have shown that sediment phosphorus often migrates along a redox gradient, concentrating near the sediment-water interface. If this is the case, then removing just the surface layer may achieve adequate phosphorus reduction and improve lake water quality. However, if the concentration of phosphorus is high throughout the sediment profile, then removal of all sediment becomes necessary and more costly. A study of the sediment depth and phosphorus concentration through the sediment profile should answer these questions.

Summary

The two options most suitable to manage Tabor Lake phosphorus loading, macrophyte harvesting and hypolimnetic withdrawal, may not result in an apparent water quality improvement in the short term (less than 5 years). This is a difficult prediction to make as primary production responds to phosphorus availability, which in turn is regulated by climatic conditions (solar heating and wind stress affect the water column stability and lake mixing). Therefore, short term improvement could be observed if the summer weather consisted of cool, gray, non-windy days. Alternatively, hot, sunny and windy summers creates a worst case scenario where phosphorus release from the hypolimnion into the epilimnion is frequent. The uncertainty of future summers makes it difficult to predict potential short term change.

Over the longer term, macrophyte harvesting and hypolimnetic withdrawal techniques serve to remove phosphorus from the internal loading cycles exhibited in Tabor Lake. Predicting when noticeable improvements in Tabor Lake's water quality occur is difficult because of the large reserves of phosphorus in the surface layer of sediment. Based on the estimated 162,000 kg of phosphorus (S.E.= 10,100 kg) in the top 3 cm of sediment, many decades may pass before there is a noticeable improvement in lake quality. Alternatively, if most of the sediment phosphorus is insoluble under normal conditions, then only a fraction of the 162,000 kg of phosphorus needs to be removed which would shorten the time required for the remediation process to have a noticeable effect.

If most of the sediment phosphorus is contained within the top 3 cm of sediment, the removal of this layer would prove to be the most effective, permanent solution to prevent internal loading. However, this technique has many undesirable side effects, such as phosphorus resuspension during removal, potential release of toxic substances, destruction of benthic fish-food organisms and disposal concerns (Cooke *et al.*, 1993). Furthermore, the cost of dredging a lake the size of Tabor is prohibitive. It is likely that the sediments beneath the surface layer may also harbour large quantities of phosphorus waiting to be released after the surface phosphorus is depleted. If this is the case, the option of dredging becomes even more costly.

Developing a cost per unit phosphorus removal for the macrophyte harvesting and hypolimnetic withdrawal is useful for comparison purposes. In 1995, about $35,000 was spent on the macrophyte harvesting operation, which removed an estimated 578 kg of phosphorus. This works out to about $60/kg phosphorus. The proposed hypolimnetic withdrawal system, using high density polyethylene pipe would have a life span of at least 30 years at a cost of $216,000 (Ward, 1996). This system would be gravity fed and therefore does not require an operational budget. It is estimated that this system will remove about 130 kg of phosphorus from the lake each year. Assuming no repair costs and no salvage value after 30 years, the estimated cost per unit of phosphorus is $55/kg. The phosphorus removal costs are similar for the two remediation strategies discussed above.

6.4 Areas of further research

(1) <u>Population cycle of *E. canadensis*</u> — Rorslett *et al.* (1986) discuss the typical life cycle of *E. canadensis* in most lakes. When these plants are first introduced to an area, growth and colonization of new territory is rapid. This is followed by growth of very dense floating "mats" of *E. canadensis*, and finally sloughing of the macrophytes to a small, but relatively stable population. Sculthorpe (1967) proposes that this well established boom-bust cycle may be related to nutrient deficiencies. Spicer and Catling (1988) discuss several

studies which identify iron as a limiting nutrient. If iron is the limiting nutrient for this plant in Tabor Lake, another process which occurs in Tabor Lake — classic internal loading, where phosphorus and iron both enter solution — may serve to prolong the growth of *E. canadensis* in Tabor Lake. During periods of anoxia, it is hypothesized that both iron and phosphate enter solution due to the change in redox conditions. By cycling some of the iron from the profundal sediments during into the littoral sediments, the large population of *E. canadensis* in Tabor Lake may survive for long periods of time.

(2) <u>Wetlands</u> — In chapter two, the two multiple regression models identified wetlands to be inversely correlated with phosphorus concentration. The importance of wetlands as a tool for removing phosphorus has been well documented in the literature, however no studies were found which use wetlands to predict nutrient concentrations in watershed loading models. The results from this study indicate that wetlands can be used as an effective predictor of nutrient concentrations, and may be useful for determining if a body of water is resilient to impacts of various land uses, such as forestry and agriculture.

(3) <u>Old growth forests and elevation</u> — In the stepwise multiple regression model generated in chapter two, the variables oldfor (old growth forests) and elevation were found to be significant predictors of spring phosphorus. Although this was unanticipated, the results raise interesting questions into the role of both of these in predicting spring phosphorus.

(4) <u>Phosphorus in overwintering plants</u> — In chapter four, two possible explanations were presented which might explain the decrease in tissue phosphorus observed in healthy macrophyte stems between August and September. It is assumed that these plants entered an overwintering phase between these two dates, however it is unclear why the September samples had less than 50% of the tissue phosphorus found in the August samples. One explanation is that as plants enter the overwintering phase phosphorus is lost from the tissue because it is not all needed for metabolic processes during the winter. This implies another phosphorus loading pathway, since turion formation is not expected to deliver

phosphorus to the water column. An alternative explanation is that the assumed increase of starch in the turion would also increase the percentage of total non-structural carbohydrates (TNC) in the plant. Titus (1977) observed an increase in TNC from 5% to 20% in *Myriophyllum spicatum* between summer and autumn. If this occurred in the *E. canadensis* plants sampled from Tabor Lake in 1995, it would require a substantially larger increase in TNC than observed by Titus.

(5) <u>Bioturbation</u> — During the 1995 field season, SCUBA dives over the littoral and profundal sediments revealed a large number of tube dwelling establishments. Bostrom et al (1980) briefly discussed the role that sediment organisms can play in resuspending phosphorus. The large chironimid population in the Tabor Lake sediment might also represent an important conduit for phosphorus cycling.

6.5 Conclusion

The revised phosphorus budget presented in section 6.2 of this chapter identifies the internal loading mechanisms in Tabor Lake which appear to dominate the lake's trophic status from June until September. The two internal loading mechanisms studied, phosphorus leaching from senescing macrophytes and hypolimnetic loading during periods of anoxia, account for the majority of observed phosphorus increases in Tabor Lake during the open water season. Phosphorus loading from macrophyte senescence represents 44% (S.E.=20%) of the total observed phosphorus increases in Tabor Lake. The two estimates for hypolimnetic loading (conservative and liberal) are 27% and 52%, respectively. Phosphorus loading from surface runoff and atmospheric deposition does not appear to influence the trophic status of Tabor Lake. The annual contribution of phosphorus from these two sources represent about 3% and 2% of the observed increases of phosphorus in Tabor Lake. The results from this study are useful for lake managers deciding appropriate management and remediation strategies aimed at reducing phosphorus availability in Tabor Lake.

The most significant compartment of phosphorus in Tabor Lake is the sediment, where 162,000 kg of phosphorus (S.E.= 10,100 kg) is stored in the top three cm. The only remediation technique which might remove this phosphorus compartment over a relatively short time frame is sediment dredging. However, this option has not been explored in any detail because the financial and ecological costs are high. Alternatively, macrophyte harvesting has been used in Tabor Lake for the past several seasons, and is useful in removing phosphorus from the lake. Furthermore, removing the dense growths of macrophytes from recreational and residential areas improves the aesthetic value of the lake. Although this technique will not likely improve water quality in the near future, it is useful over the long term in removing phosphorus from the lake.

Another remediation strategy proposed for Tabor Lake is the hypolimnetic withdrawal, which removes phosphorus rich hypolimnetic water prior to its circulation into the epilimnion. Ward (1996) estimates that a gravity driven system would remove 130 kg of phosphorus from Tabor Lake during a 12 week period in the summer (using a 24 inch diameter pipe). The hypolimnetic withdrawal will not prevent summer algal blooms for the first few years of operation since the total phosphorus released from the hypolimnion during 1995 was between 1197 kg and 2290 kg. However, over many years this system should achieve a reduction of sediment derived phosphorus in Tabor Lake.

Focusing remediation efforts on the two phosphorus loading compartments quantified in this study, macrophyte senescence and hypolimnetic anoxia, should provide the best approach to managing lake phosphorus without physically removing the sediment. Over the long term, water quality improvements should be noticed since the annual removal of phosphorus from Tabor Lake will exceed the annual contribution from external sources.

Appendix A

Estimating the biomass of macrophytes in Tabor Lake

A.1 Comparison of two biomass estimation techniques

Deciding which technique should be used to quantify the standing crop of aquatic macrophytes in a lake usually involves a tradeoff between the accuracy and cost of using the technique. To estimate the mean macrophyte biomass with a permissible error of 10 %, typically requires more than 200 sampling stations and is extremely labour intensive (Canfield *et al.*, 1990; Downing and Anderson, 1985). Alternatively, some estimation techniques employ simple variables (such as slope and fetch) to estimate biomass, but have standard errors covering two or three orders of magnitude around the estimate (Duarte *et al.*, 1986). This tradeoff between accuracy and cost has spurred the development of alternative techniques which attempt to minimize the cost while maximizing the accuracy.

Canfield *et al.* (1990) suggest that maximum standing biomass in a lake be used as a descriptor of mean standing biomass. They found the maximum standing biomass was strongly related to the mean standing biomass (R^2=0.79) and had a comparable error estimate to the average standing crop. They claim their technique is relatively simple to employ since emergent vegetation can be spotted from shore and maximum biomass in submerged stands of macrophytes can be easily found using an echosounder. This technique offers the potential of significantly reducing the number of samples required for accurately estimating biomass, but the authors suggest that their model only be considered tentative as additional studies are needed.

Several recent innovations have also been employed to estimate biomass. Rorslett *et al.* (1985) used a combination of stereo-photographs, regular quadrat harvests and an underwater television camera (with recorder) to estimate the biomass and areal distribution of *E. canadensis* in Lake Steinsfjord, Norway. Three cover classes were used (0-10%, 10-30%, and >30%) and biomass within each class was estimated by harvesting 1 m^2

quadrats. More recently, Lehmann *et al.* (1994) used GIS, an echosounder and SCUBA quadrat harvests to quantify the littoral zone of Lake Geneva, Switzerland. They combined an in-situ study of three macrophyte species with GIS to show annual spatial variation in vegetation structure. These techniques offer new approaches for macrophyte research, however it is unclear if their techniques reduce the overall costs compared to the standard random quadrat harvesting commonly used.

Maceina and Shireman (1980) developed an alternative technique, using an echosounder (also known as a recording fathometer and a fishfinder) to estimate biomass. This technique recorded the abundance of macrophytes on a chart recorder, calibrating specific values on the chart with actual samples using a dredge. They found that dividing the echosound tracings into "thick" and "sparse" vegetative areas and using a separate multiple regression model for each area improved the overall accuracy of their biomass prediction. The resulting equations produced an $r = 0.796$ and 0.807, respectively. The variables used to predict thick vegetation were the height of the vegetation and the distance from the top of the plant to the water surface. Sparse vegetation included a third variable which was the percent vertical cover of vegetation on the echosound tracings.

Thomas *et al.* (1990) compared SCUBA quadrat harvesting to the acoustic method proposed by Maceina and Shireman to determine the relative costs associated with each technique. Over three years of research, the quadrat harvesting method was shown to be 1.9 times more expensive than sampling with an echosounder. This greater expense was primarily attributed to the greater labour costs associated with harvesting. Thomas *et al.* (1990) also found the acoustic technique had a higher precision than from quadrat harvests, providing a 5-to-18 fold greater capability to detect a change in the mean. However, the two techniques were comparing slightly different characteristics of the macrophyte community. The acoustic technique estimated biovolume (m^3), whereas the quadrat harvest technique estimated biomass (g / m^2) .

Duarte (1987) modified the echosound regression model proposed by Maceina and Shireman, using only two variables, the height of vegetation and vegetation type, to predict biomass. Duarte reasoned that since the technique relied on a biomass per unit height, different plant forms might account for a major source of the variance. Duarte first tested the accuracy of this technique using only one variable, *height*, to predict biomass and generated an R^2=0.56. He then included vegetation form as his second variable and using a multiple regression model generated an R^2=0.89. Duarte pointed out that biomass to height values are a function of plant growth form, and created three classes of plants. The first class were tall, canopy forming plants, like *Myriophyllum*, which grow to the surface to flower. The second class form short under-stories, like *Utricularia*. The third class either had submerged flowers or did not form flowers and included *Elodea canadensis*. This last class had the highest biomass to height values.

Maceina and Shireman (1980), Duarte (1987) and Thomas *et al.* (1990) have demonstrated that an echosounder can be used to estimate aquatic biomass (or biovolume) with as much, or greater, precision as quadrat harvesting techniques. In this experiment, an echosounder is used to estimate the biomass of aquatic vegetation in Tabor Lake.

Several discrepancies exist between the echosound tracings collected in Tabor Lake and those used by Maceina and Shireman (1980). These differences (which are discussed in greater detail in the methods section) made it necessary to develop a new model to estimate biomass. The purpose of this study is to evaluate and compare the predictive power of two single-variable biomass estimation models. The first model uses height as a predictor variable, while the second model employs digital imaging of the echosound tracings to predict biomass.

Methods And materials

Figure A.1 displays the 23 transects which were established around Tabor Lake between the outer littoral zone (approximately 5 metres in depth) and as close to shore as

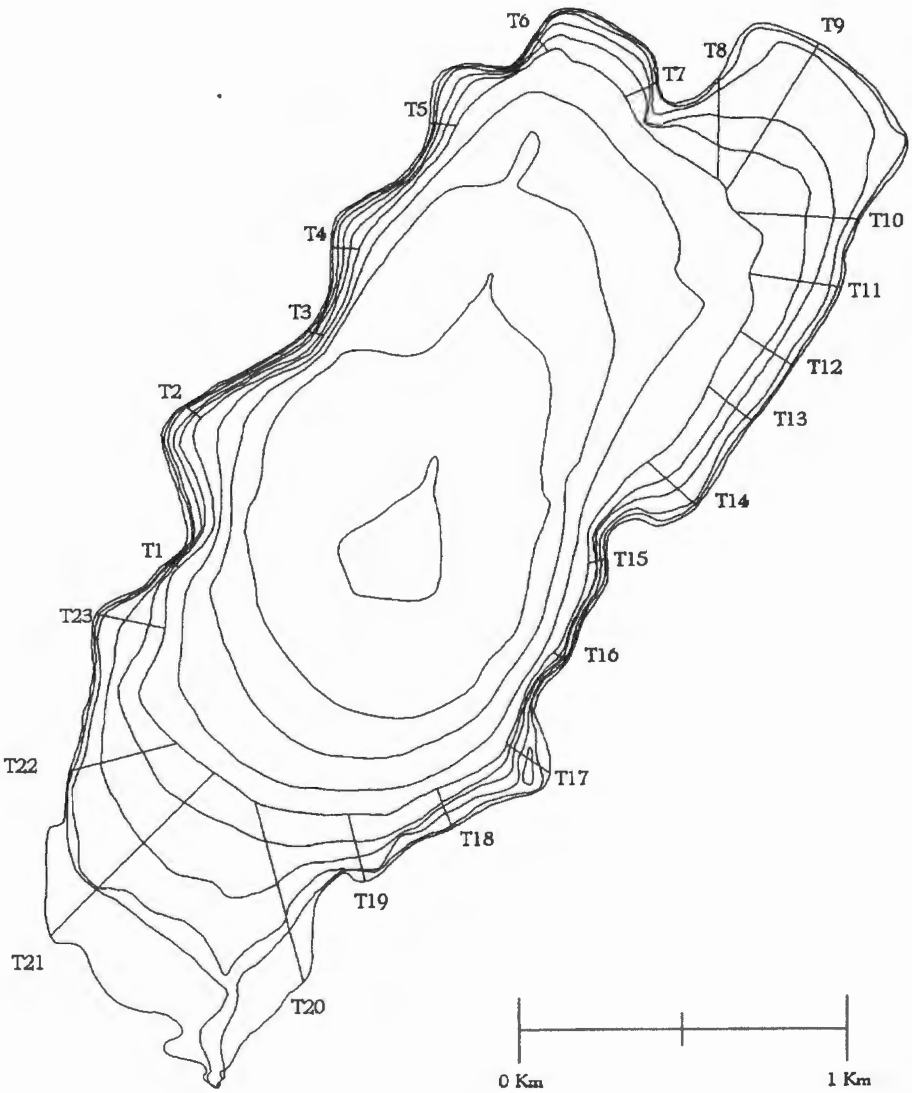


Figure A.1 Bathymetric map of Tabor Lake, showing locations of the 23 transects where echosound tracings and SCUBA samples were taken.

was permitted by boat access (between 0.75 and 1.0 metres). Field observations and a bathymetric map were used to determine transect locations so that the natural variability of macrophyte biomass in Tabor Lake was surveyed. The field observations identified areas where surface vegetation densities changed, and the bathymetric maps were used to identify changes in littoral slope, which has been shown to influence biomass (Duarte *et al.*, 1986).

Scuba quadrat harvesting

Along the first six transects intensive SCUBA sampling was conducted at approximately 1 metre contour intervals. For transects seven through 23, SCUBA sampling was conducted at every second transect (8,10,12, etc.) and only at one, randomly selected depth. A marker buoy was dropped at each sampling site. Sampling was conducted within a 5 metre radius of the buoy.

Macrophyte samples were collected using a 30cm x 30cm steel quadrat. SCUBA divers randomly placed the quadrat over the weeds and harvested everything enclosed by the quadrat. The number of replicate samples taken at each site was determined using the techniques developed by Downing and Anderson (Table 7; 1985) to achieve a standard error of 20% of the mean value. The number of samples required to achieve an S.E./x of 0.2 depended on the density of macrophytes. Downing and Anderson suggest that macrophyte biomass is spatially aggregated and that fewer replicates are required in higher densities. As density declines, more samples are required to achieve the same degree of accuracy.

Once harvested, the weeds were placed in a mesh bag and brought to the boat, where the samples were labeled and placed on ice. The macrophytes were then brought back to the lab and rinsed in cold water to remove loosely attached epiphytes and mineral deposits. After washing, the samples were dried in an oven and weighed to the nearest 0.1 g. Samples not immediately processed were placed either in the refrigerator (if processed within a few days) or in the freezer.

Echosounder with chart recorder

Along each transect, a DE-719 Precision Survey Fathometer Depth Recorder from Raytheon Company was used to collect continuous echosound tracings. A barium titanate type 2445 AD transducer was used, with a beam width of 8 degrees. Before transects were conducted, the rate of the echosound chart recordings was calibrated with the speed of the boat using a foresters hip chain, marking the echosound recordings every 10 metres traveled. This was done so that echosound tracings could be calibrated to the distance traveled.

At each SCUBA sampling site, a buoy was dropped and, concurrently, a check point was made on the echosounder chart. This was used as a reference for calibrating the echosound recordings with SCUBA collected biomass.

Developing an alternative technique

The echosound tracings generated from Tabor Lake did not produce the results which were necessary to use Maceina and Shireman's (1980) original model. Figure A.2 compares the echosound tracings shown by Maceina and Shireman in their paper (a) with a typical echosound tracing generated from Tabor Lake (b). The original model used two or three variables (depending on plant density) to predict biomass - height of vegetation and distance between surface water and the top of the plant (plus cross sectional area in "sparse" stands). It is relatively simple to measure these variables using the data from Maceina and Shireman, however the Tabor Lake echosound tracings do not allow for accurate measurements of these variables because the tracings show discontinuous weed heights. Different climatic conditions and the composition of macrophyte species might be responsible for the observed discrepancy in echosound tracings between these two studies.

Although the echosound tracings from Tabor Lake cannot be used in Maceina and Shireman's biomass estimation model, the Tabor Lake tracings still contain valuable

(a)

(b)

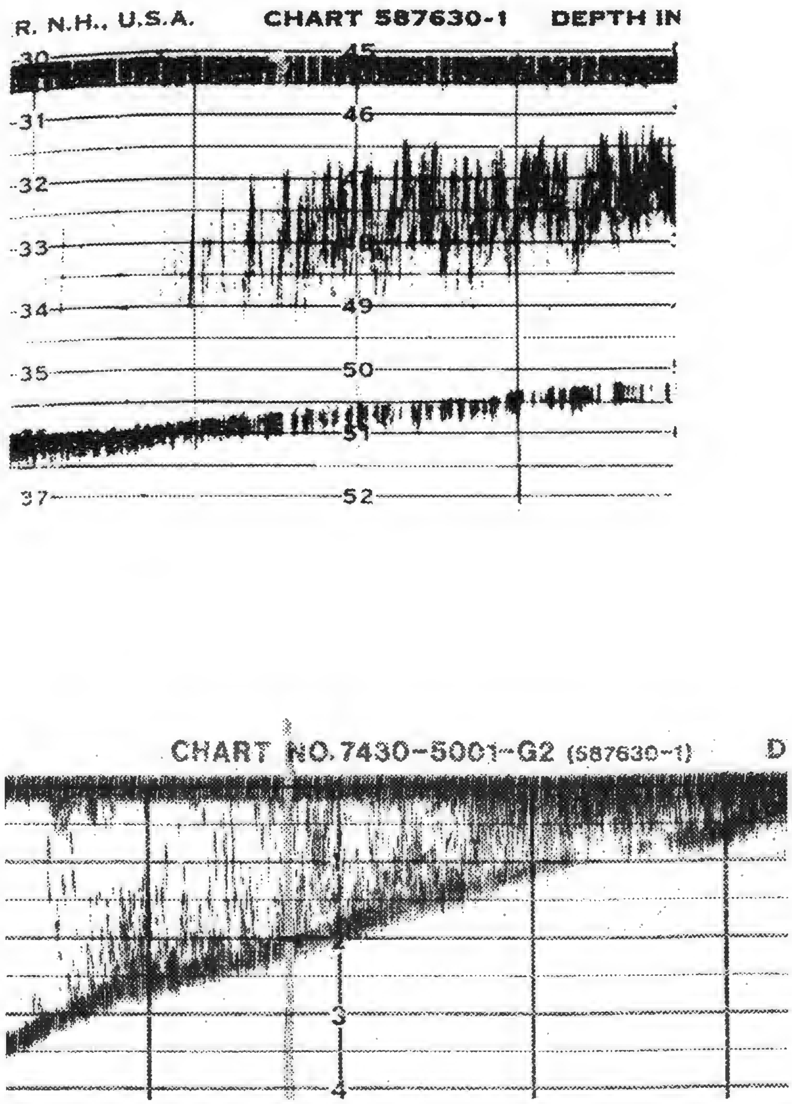


Figure A.2 Comparison of echosound tracings between (a) Lake Baldwin (Maceina and Shireman, 1980) and (b) Tabor Lake, August 1995.

information about the density of macrophytes. To quantify this density information, a computer scanner and imagery software is used. The echosound chart tracings were digitized using an HP II scanner on an IBM P.C. at 32 dots per inch (dpi). This resolution was chosen after comparing scanned images at 16, 32, 50 and 200 dpi At 16 dpi, the resolution was not strong enough to easily differentiate between weeds and bottom. Increasing the resolution translated to an exponential increase in the amount of memory, time and effort necessary to conduct the analysis. 32 dpi provided the best compromise between resolution and available time. After the image was scanned in, it was transferred to PCI Works, where it was visually checked and weed biomass areas were identified from the sediment. Finally, it was transferred to a spreadsheet where the pixel counts were conducted within a 5 metre radius of the SCUBA sampling sites. These values are plotted against the SCUBA sampled biomass and a regression analysis was conducted.

Also, the height of vegetation at each sample site was estimated as the mid-height between the shortest and tallest echosound tracing within a 5 metre radius. This follows Duarte's (1987) technique for estimating average height of vegetation.

Results and discussion

Scuba quadrat harvesting

There were 23 mean biomass values used in the calibration of the echosound tracings. Sampling depths were chosen between 1.3 and 5.0 metres. Biomass values also ranged widely from 6 g/m^2 to 594 g/m^2 (all biomass values reported as dry weight). Extensive field sampling revealed that *Elodea canadensis* was the dominant macrophyte between the shoreline and approximately 3.0 m depth, growing in dense monocultures around the entire lake perimeter. *E. canadensis* was also found growing beyond 3.0 metres, however it was sparsely distributed and rarely grew to the surface. Beyond 2.5 metres in depth *Ceratophyllum desmerus*, *Potomogeton* species and *Myriophyllum spicatum* grew in small isolated patches, often rising to the surface. Very few plants were found growing past 4.5 metres in depth.

Figure A.3 plots the relationship between depth and biomass from SCUBA collected samples. Depth of maximum observed biomass (594 g/m^2, S.E.= 140 g/m^2) was 1.8 m, and biomass values above 300 g/m^2 were observed at depths between 1 m and 3 m. Although the highest biomass values were observed between 1 m and 3 m, five samples within this depth range had biomass values at 200 g/m^2 or lower. Extensive weed harvesting by the mechanical weed harvester between 1m and 3m depths might explain these low biomass values. Also, natural populations of macrophytes tend to be lower as slope increases (Duarte *et al.*, 1986).

By the time quadrat sampling was conducted, 642 loads of harvested weeds were removed from Tabor Lake (pers. comm. S. Dmitrasinvc) which is roughly 71,000 kg of macrophytes. Recent harvesting can be determined from a visual inspection of the echosound tracings, where the macrophyte community is missing square sections from areas of otherwise dense weeds. Sampling stations T5-B and T8 both show signs of being recently harvested, and both these sites had biomass values of 203 g/m^2 and 150 g/m^2, respectively. The other three sections with low biomass values were T3-A, T3-B and T16. Both of these transects are located along the steepest littoral slopes found in Tabor Lake.

Figure A.4 plots S.E./x vs. depth, showing that the level of precision with each biomass estimate is related to depth. Downing and Anderson (1985) have shown that the deeper, less colonized areas have a much greater variation about the mean, and therefore require a larger number of samples to represent the mean with a desirable level of precision. As the S.E./x exceeds 60% at depths greater than 4 metres, it is clear that we underestimated the number of samples required to adequately represent the mean standing biomass at deep sites. However, the maximum biomass occurs predominantly between 1 m and 3 m depths, where the average S.E./x was 0.19, which is an acceptable level of

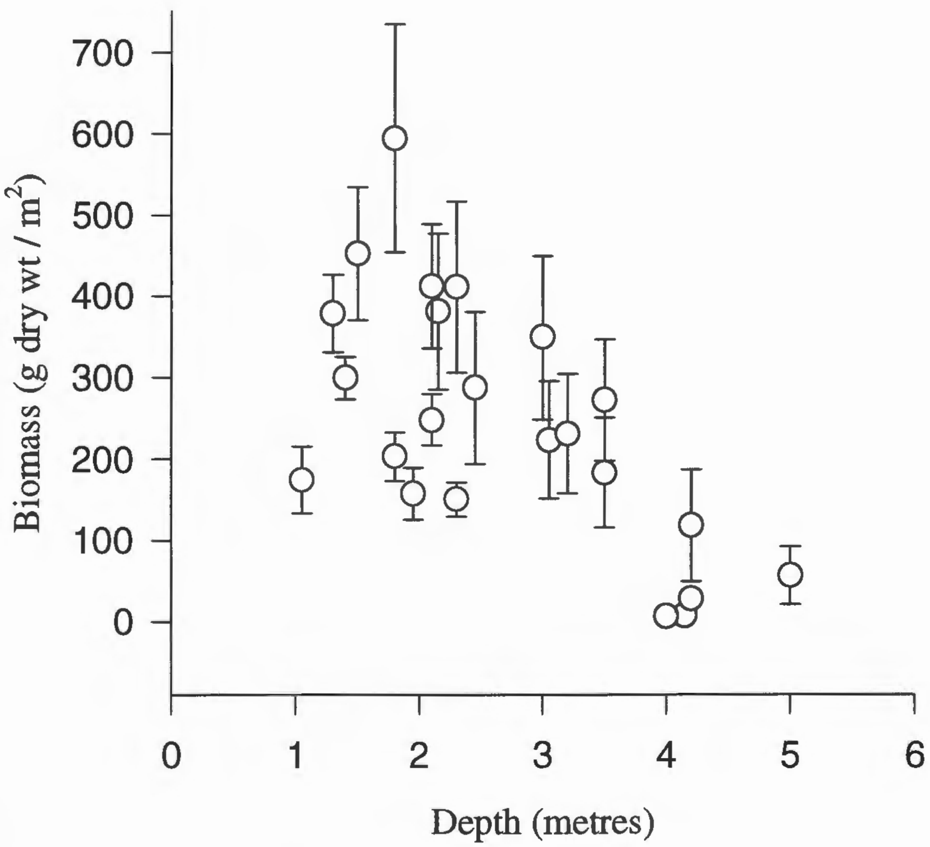


Figure A.3 Plot of Mean Biomass (plus/minus one standard error) vs. depth from SCUBA collected quadrat samples taken August, 1995 from Tabor Lake.

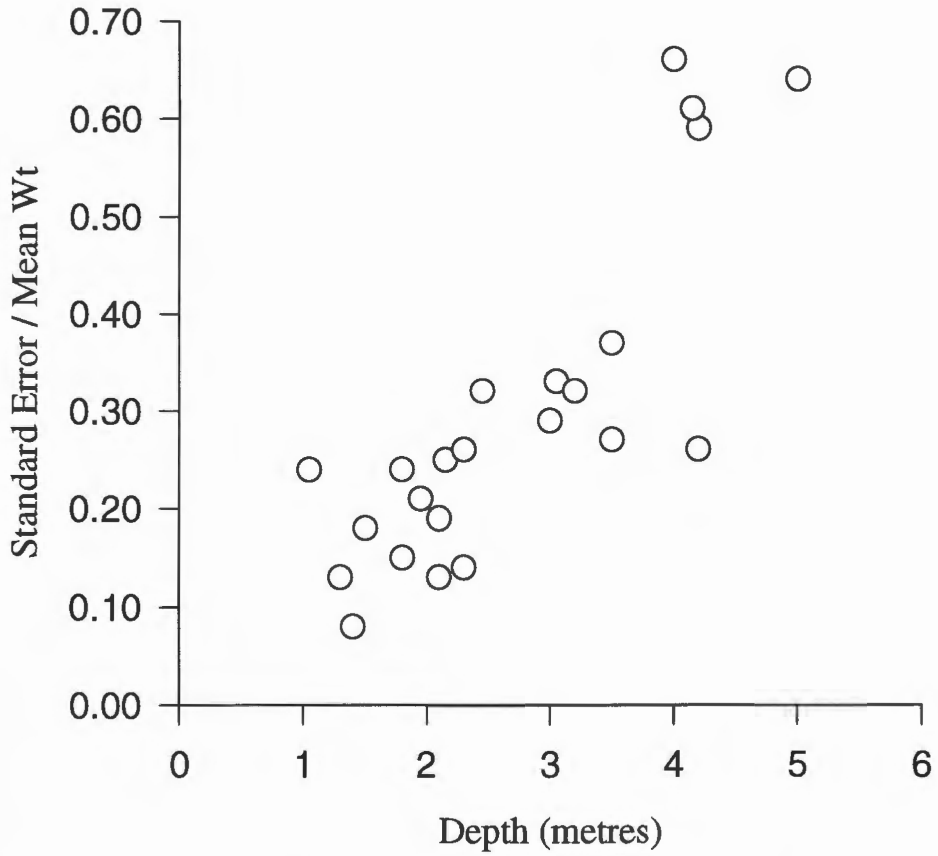


Figure A.4 Plot of standard error / mean biomass vs. Depth from SCUBA collected quadrat samples taken August, 1995 from Tabor Lake.

precision for these depths. The high variability of samples collected beyond the 4 metre depth is not considered a major problem since it represents only a small fraction of the total biomass.

Two models predicting biomass

Table A.1 shows the results from both height-to-biomass and pixel count-to-biomass regression analyses. The height-to-biomass relationship produces an $R^2 = 0.15$ with a standard error of biomass to be 144 g/m^2. This relationship is not considered significant ($p=0.065$; $F=3.79$). The results from pixel count-to-biomass regression analysis provides a better result than height, yielding an $R^2 = 0.49$ and a standard error of 112 g/m^2. This relationship is considered significant, ($p<0.001$; $F=19.94$). The pixel count model accounts for a greater proportion of the variation than the height model, and the relationship is considered significant.

The height-to-biomass relationship from the Tabor Lake data is much lower than Duarte's (1987) height-to-biomass regression, which generated a $R^2 = 0.55$ and a standard error of the biomass estimate to be 83 g/m^2 (estimated from 831 g wet weight / m^2). This might be explained by some of the methodological differences which exist between this study and in Duarte's study. Duarte used a transducer with a cone beam of 50 degrees, where plant height represented clusters of plants about 80 cm in diameter. The cone beam used in the Tabor Lake study was only 8 degrees and therefore capable of distinguishing greater variability of the stand height. The overall effect is that Duarte's echosound tracings generate a more continuous recording of weeds, and less variability in height distinction compared to the Tabor Lake tracings. Also, Duarte ran six replicate transects with the echosounder, calculating average height to increase the statistical reliability of the tracings.

The results found in this study indicate that height is a poor predictor of biomass, while the pixel count method accounts for almost 50% of the variation in mean biomass, and the

Table A.1 Results from two biomass estimation models, using either the height of plant or pixel count to estimate biomass.

Biomass Estimation Model	R-squared	Std Err of Biomass *(g / m2)*
Height of plant	0.15	144
Pixel count	0.49	112

relationship is considered statistically significant. The pixel count results show that this method can be used to estimate biomass values from the Tabor Lake echosound tracings.

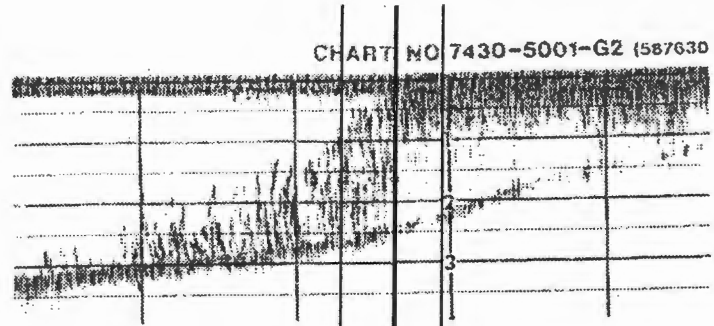
A.2 Merging areas of "sparse" and "dense" vegetation: the *Maximum Pixel Density* technique.

Maceina and Shireman (1980) found it necessary to use two models with their echosound tracings because areas of thick hydrilla did not permit a clear reading of the bottom. Thus two models were used depending on the density of macrophytes. Thomas *et al.* (1990) found that sediment was often undetected by the echosounder, especially in dense macrophyte stands. This problem also occurred in the Tabor Lake study. Since the transducer was mounted on the side of the boat and extended below the bottom of the hull, the surface vegetation would became entangled around the transducer. In areas of very dense vegetation, the acoustic signal was often disrupted and the echosounder recorded a washed out signal (Figure A.5(a)). In areas where this occurred, the echosound tracings would not provide a representative tracings of the macrophytes.

From the 23 sample sites where SCUBA collected vegetation was harvested, three sites exhibited this echosound disfunction. This problem would tend to greatly underestimate the biomass using a pixel-count method since most of the vegetation was not adequately represented. Although Maceina and Shireman's solution is not transferable to the Tabor Lake experiment, a technique was developed to solve this problem.

Generally, the acoustic signal became disrupted once a critical density of vegetation was encountered. Furthermore, this critical density also appeared to be depth-related, indicating that depth plays a role in determining when the disruption occurs. Assuming that each depth has a critical density which clutters the transducer signal, it should be possible to regress maximum density against depth and generate an equation to predict densities when the acoustic signal becomes disrupted. This regression equation will be tested against the

(a)



(b)

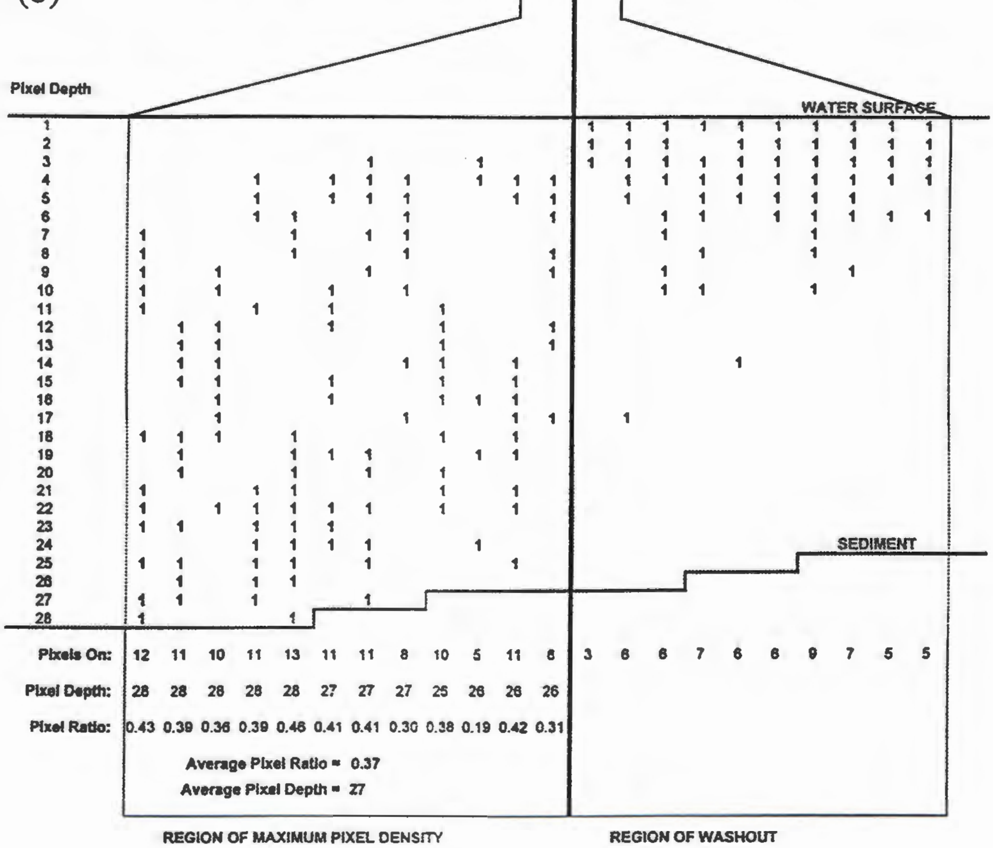


Figure A.5 (a) Echosound tracing from a Tabor Lake transect. Black line divides the uninterrupted and interrupted echosound signal. This disruption occurred as a result of ultra dense vegetation cluttering the transducer. (b) A digital representation of the area immediately prior and post disruption.

Note: Calculation of the Maximum Pixel Density (MPD) is shown at the bottom of this figure, where MPD = *pixels on* divided by *pixel depth* for each pixel column. Average MPD is calculated from regions of where maximum pixel density exists, usually found immediately prior to disruption.

three sample sites where biomass was collected and echosound disruption occurred to see if the technique improves the predictive power of the model developed in the previous section.

Methods

The areas of maximum biomass tend to be obscured in our echosound tracings, but we potentially can use the information about plant density immediately preceding the washed out tracings to provide a better estimate of biomass. The density of macrophytes which precede areas of washout are measured using a simple technique shown in Figure A.5 (b). Within these areas of maximum vegetation density, a **maximum pixel density (MPD)** is calculated by dividing the number of pixels "on" by the total number of pixels in a column (a pixel column is the number of pixels measured from the water surface to the sediment). The MPDs are then regressed against their corresponding depths and an equation is generated.

This MPD equation is applied to the three sampling stations where SCUBA obtained samples were taken and the echosound disruptions occurred. Another regression is performed on the newly revised pixel data and biomass. Both new and old regressions are compared to test if the MPD model improves the biomass prediction.

Results and Discussion

Figure 3.6 shows the relationship between MPDs and their corresponding depths. A regression analysis conducted on this data generated an R-square of 0.86.

$$\textit{Maximum Density} = 0.946 + [(\textit{Depth}) * (-0.02243)]$$

Std Err of Y Est. 0.061
Std Err of Coeff. 0.0028

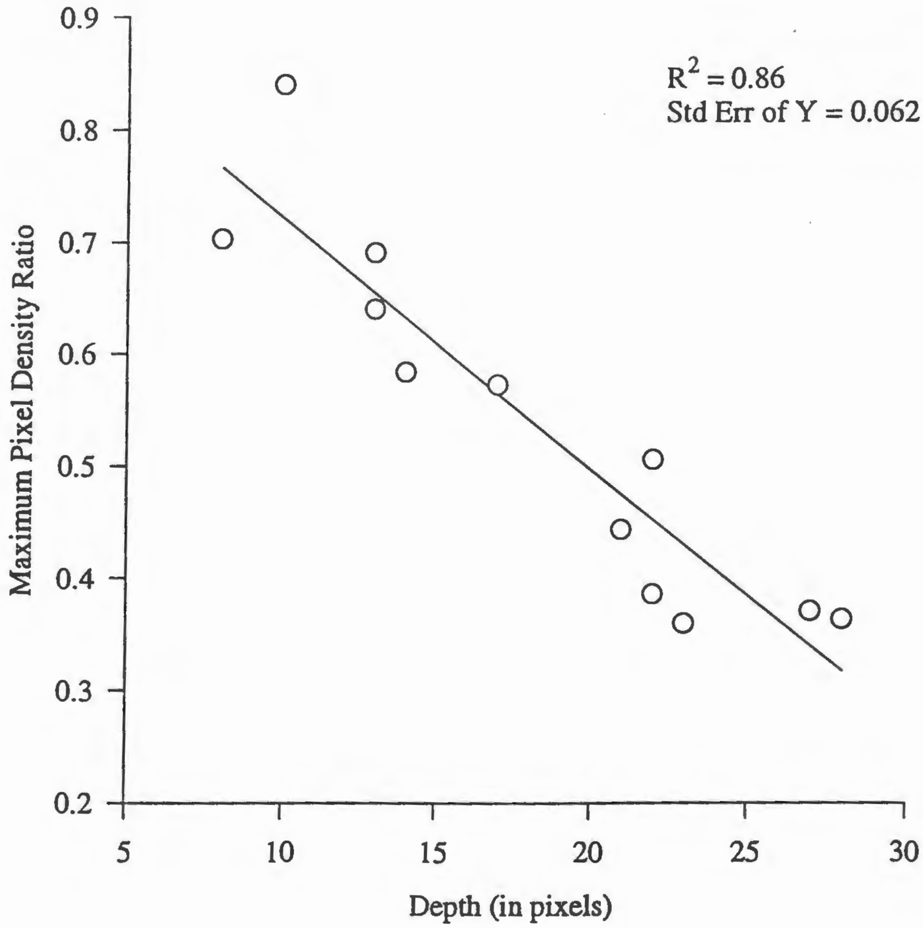


Figure A.6 Scatter plot showing relationship between *Maximum Pixel Density* and depth with regression line, R^2 value and standard error of the Y estimate.

Table A.2 (a) compares the results generated from this model to original pixel counts for the three sample sites with disrupted echosound tracings. In Table A.2 (b), a pixel count-to-biomass regression is performed on both sets of data. The new pixel counts for three disrupted sites improves the predictive power of the original equation from $R^2 = 0.49$ to $R^2 = 0.63$ and also reduces the standard error of the biomass estimate.

Although this is not a true representation of the weed densities, it is as close an estimate as can be generated with this data set. It also represents an underestimate, since the ratios calculated were from areas where the plant density had not yet disrupted the acoustic signal. However, it does provide a model to improve the accuracy to predict biomass from echosound tracings. Figure A.7 shows two scatter plots (with regression lines) of pixel count-to-biomass before and after the MPD model is applied to the three data points (arrows are used to identify the three points). In all cases, application of the MPD model shifts the data points closer to the regression line.

A.3 The use of digital echosound tracings to predict the aquatic plant biomass in Tabor Lake

The revised biomass equation generated in the previous section (*Biomass= x (0.442) + 2.42*) is used to estimate the total biomass of aquatic macrophytes in Tabor Lake. In areas with poor echosound tracings, the MPD equation is applied (*MPD=0.946 + [(Pixel Depth) * (-0.02243)]*) to generate a density estimate, and then the revised biomass equation is used to estimate biomass.

Each transect is divided into 1 metre contour intervals. Biomass for each contour is calculated by multiplying the density, estimated from equation 3.2, by the area for each contour. The area of each contour is calculated by taking the area between each adjacent

Table A.2 (a) Comparison of pixel counts before and after MPD (Max. Pixel Density) was applied to three sampling stations affected by poor echosound tracings. (b) Pixel count-to-biomass estimation model before and after MPD was applied on three sampling stations in (a).

(a)

Sampling Station	Pixel Count (prior to MPD)	Pixel Count (after MPD)
T2 - A	48	81
T2 - B	42	75
T2 - C	66	78

(b)

Biomass Estimation Model	R-squared	Std Err of Biomass *(g / m2)*
Without MPD	0.49	112
With MPD	0.63	93

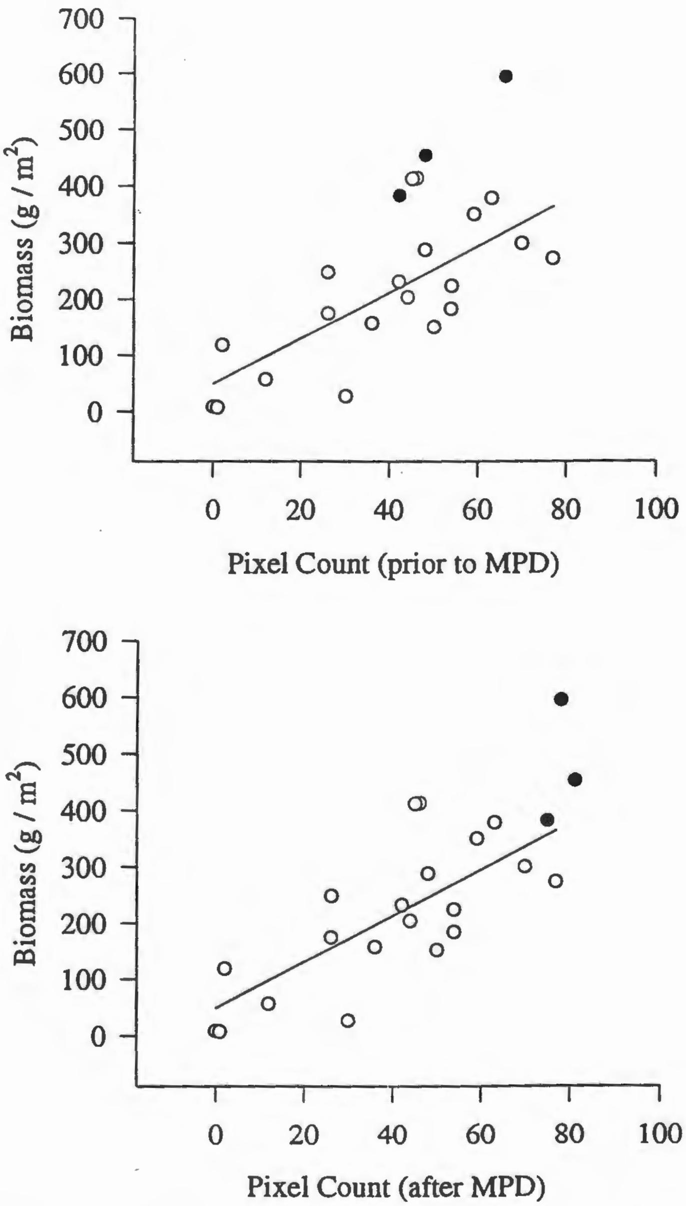


Figure A.7 Scatter plots of pixel count-to-biomass variables before and after application of the Maximum Pixel Density (MPD) model. Sampling stations with poor echosound tracings are represented as black dots in both plots.

transect and dividing it in half (see Figure A.1). Summing biomass estimates for all contours provides a total biomass estimate for Tabor Lake.

Generally, echosound tracings came to within 1 metre of the shore. To estimate biomass between the end of the echosounder and the shore, it is assumed that the biomass density between the 1 metre contour and shore is half of the biomass density between the 1 metre and 2 metre contour interval.

Error analysis and propagation in the biomass estimate

Echosound tracings which required the application of (i) a density estimate, using the MPD model, and (ii) a pixel count-to-biomass regression estimate were subjected to two sets of errors. Both errors were estimated using the standard error of the prediction multiplied by the total biomass prediction for each contour and then were added together. Addition of these two errors incorporates the influence of the MPD model and the pixel-count-to-biomass estimate into the final estimate of standard error (personal communication with B. Zumbo).

Macrophyte biomass results

Total biomass of macrophytes in Tabor Lake was estimated at 376,500 kg D.W. for the August, 1995 sampling period. Standard Error of this estimate is 124,000 kg D.W., which yields an S.E./x of 0.33. Table 3.3 compares the total biomass estimates with their standard error for Tabor Lake and Lake Baldwin. Maceina and Shireman estimated total macrophyte biomass to be 576,000 kg wet weight for Lake Baldwin. The standard error Figure was recalculated from 95% confidence intervals, and yields an S.E./x of 0.13.

Maceina and Shireman also conducted biomass sampling concurrently with the echosound technique and compared their technique to established methods of biomass estimation (also listed in Table A.3). Their results show that direct biomass sampling in Lake Baldwin predicted nearly twice as much as biomass compared to the echosounder prediction. They determined that random sampling overestimated the percent cover of

Table A.3 Comparison of whole lake biomass estimates between Tabor Lake and Lake Baldwin (Maceina and Shireman, 1980).

Method	Total biomass (kg)	Standard Error (kg)
Echosound Model from Tabor Lake	376,500	124,000
Echosound Model from Lake Baldwin	57,600*	7,500**
Direct Biomass Sampling from Lake Baldwin	94,700*	9,200**

* original values were in wet weight. Assumed 90% water content
** standard error values were recalculated from 95% confidence inter

macrophytes (86%). The echosound method revealed that only 71% of the lake was covered in macrophytes, which they assumed was more reliable due to the high number of transects (14) and the total distance traversed (11.3 km).

The high S.E./x value from the Tabor Lake study reveals significantly more variation in the total estimate than was encountered by Maceina and Shireman. However, this may be linked to the relatively large S.E./x observed at some of the sampling sites.

The high error associated with some of the quadrat samples reveal the inadequacy of following standard sampling protocol in lakes with mechanical weed harvesting. In fact, the echosounder technique has greater value in these lakes because it provides continuous sampling along two planes (vertical and horizontal). Population abnormalities require increased quadrat sampling to gain a precise estimate of the mean in these lakes, while the echosounder incorporates the abnormalities into the estimate.

A.4 Technique Improvements

The time and effort needed to accurately sample biomass using SCUBA divers with quadrats will continually spur on the development of alternative methods for estimating biomass. The echosound approach introduced by Maceina and Shireman, and modified by Duarte (1987) and Thomas *et al.* (1990) demonstrate the potential of this new approach to estimating biomass. The technique developed in this chapter provides an alternative to the existing techniques. The relative accuracy of this technique is comparable to Maceina and Shireman, but less accurate than Duarte's. Further modifications might improve its overall accuracy.

The problem most frequently encountered during this experiment was the loss of acoustic signals in areas of ultra dense vegetation. Typically, the echosounder would continue to work in these areas until the transducer was engulfed by macrophytes. This disruption might be reduced if the transducer is mounted flush against the boats hull, where weeds would pass underneath and not get entangled.

Thomas *et al.* (1990) found that changing the angle of the transducer beam from 10^{o} to 6^{o}, increased the efficiency of sediment detection in dense weeds. With a beam angle of 10^{o}, the sediment was recorded only about 30-40% of the time compared to a bottom detection of 55-65% for a 6^{o} beam angle. This suggests that using the lowest transmit power and the lowest receiving gain provides the greatest detection of bottom sediments.

Also, the SCUBA samples were randomly collected within 5 metres around the marker buoy, while the transducer beam width captured a smaller area (only 8 degrees from the surface). The technique employed by Duarte (1987) provided for a stricter account of vegetation biomass around each sample site than was used in the Tabor Lake study. Duarte's SCUBA divers harvested six random quadrats within 1.5 metres to either side of the buoy. A systematic sampling program to collect quadrat samples around each marker buoy could be used instead of the random sampling approach to ensure that the samples collected were within the beam width.

Using a higher resolution when digitizing the echosound tracings might also improve the accuracy of this technique. The program used to digitize the echosound tracings in this experiment was originally intended for satellite imagery. Other digital imagery software might provide opportunities to easily digitize tracings at high resolution without substantially increasing the workload.

Several problems with this technique have been just been discussed, with solutions to many of these problems being offered. More ground truthing on dives and refinements in processing are required before this technique can be easily applied elsewhere, but this digital method can potentially be used to estimate biomass with greater speed and accuracy than is currently possible using the standard SCUBA quadrat harvesting method.

Table B.1 Variable list for the regional lake survey (Chapter 2)

			Response variable		***Predictor variables***					
Lake	Date sampled	# samples	Spring P (mg/L)	St. Dev. (mg/L)	Elevation *(metres)*	Surf Area *(hectares)*	Perimeter *(metres)*	Max depth *(metres)*	Mean Depth *(metres)*	Volume *(000 m3)*
Bednesti	780524	10	0.009	0.001	787	261.0	14762	20.7	8.3	21663
Berman	780525	5	0.011	0.001	787	43.7	4365	16.5	2.6	1136
Chief	760602	11	0.032	0.004	792	729.0	21007	6.1	3.8	27702
Circle	860522	1	0.02	NA	775	74.3	5300	15.0	4.7	3492
Cluculz	830426	2	0.023	0	762	2518.4	53107	61.0	29.6	745446
Crescent	780530	3	0.027	0.006	762	59.7	4115	14.3	6.5	3881
Dahl	840610	1	0.032	NA	799	242.0	17500	23.5	4.5	10890
Dorman	840510	1	0.024	NA	760	128.0	6100	12.0	6.5	8320
Eaglet	860507	6	0.06	0.012	596	853.9	23030	9.4	5.0	42695
Eena	780517	6	0.021	0.002	763	51.0	4104	24.0	7.7	3927
Eulatezella	870507	6	0.032	0.014	792	445.2	17219	13.1	5.8	25822
Fish	840510	1	0.022	NA	914	42.9	3658	13.1	4.7	2016
Foster	840510	1	0.03	NA	823	77.3	5084	4.3	1.7	1314
Grizzly	840608	1	0.036	NA	964	72.0	3840	5.8	2.0	1440
Hallet	840510	1	0.019	NA	800	560.0	20900	31.0	17.3	96880
Karena	840510	1	0.036	NA	870	114.1	6500	5.8	2.7	3081
Henning	840510	1	0.022	NA	820	97.6	10500	15.0	4.4	4294
Leg	840510	1	0.08	NA	716	40.5	5700	5.2	1.8	729
Little Bobtail	840611	2	0.098	0.018	815	266.0	13400	14.5	4.7	12502
Marie	840510	1	0.039	NA	917	192.2	14265	11.6	4.6	8841
Murch	780530	6	0.017	0.004	762	128.3	5587	13.1	5.4	6928
Naltesby	870507	5	0.022	0.001	800	840.6	28621	20.1	11.1	93307
Ness	850515	3	0.014	0.004	710	203.6	18829	18.3	6.1	12420
Nukko	770519	4	0.019	0.002	762	457.1	17450	17.1	7.2	32911
Nulki	870513	4	0.045	0.005	720	1651.2	25359	7.6	4.5	74304
Oona	840510	1	0.026	NA	783	337.0	1372	23.2	9.7	32689
Opatcho	780601	6	0.007	0.001	720	40.5	3000	23.0	7.0	2835
Pinchi	760609	12	0.022	0.007	724	5554.2	64275	67.5	23.9	1327454
Punchaw	800508	3	0.061	0.004	762	205.2	7288	6.7	3.7	7592
Purden	780529	10	0.007	0.001	762	836.0	24705	52.4	20.2	168872
Saxton	780531	10	0.016	0.004	760	694.5	15500	16.0	9.9	68756
Tabor	850506	5	0.027	0.006	703	405.0	9100	9.2	5.4	21870
Tachick	840510	2	0.04	0.006	710	2202.0	30280	7.6	4.4	96888
Teardrop	840510	1	0.041	NA	820	38.4	3120	10.0	5.5	2112
Tezzeron	760608	13	0.014	0.005	765	7807.9	82643	43.3	11.2	874485
Verdant	780518	6	0.009	0.001	678	27.0	2310	7.6	3.0	810
Vivian	780518	6	0.007	0.001	681	45.0	2339	8.2	4.0	1800
West	770517	12	0.033	0.007	692	502.6	10944	15.2	7.9	39705
Woodcock	840606	1	0.028	NA	985	132.4	5600	15.0	6.8	9003

Table B.1 Variable list for the regional lake survey (Chapter 2) *(continued)*

	Wetlands		**Agriculture**			**Inhabitants**		
	Along Streams *(0-2)*	Watershed Total *(0-2)*	Along Streams *(0-2)*	Lake Perimeter *(0-5)*	Watershed Total *(0-2)*	Along Stream *(0-3)*	Lake Perimeter *(0-3)*	Watershed Total *(0-3)*
Bednesti	2	1	0	0	0	0	1	1
Berman	2	1	0	0	0	0	1	1
Chief	2	2	1	0	1	3	1	3
Circle	1	1	2	1	1	1	0	1
Cluculz	1	1	1	0	1	1	3	3
Crescent	2	2	2	2	2	1	1	1
Dahl	1	1	0	0	0	0	0	1
Dorman	0	1	0	0	0	0	0	0
Eaglet	0	0	2	4	1	1	1	1
Eena	0	1	0	0	1	0	1	1
Eulatezella	1	1	0	0	0	0	0	0
Fish	2	1	0	0	0	0	0	0
Foster	2	1	0	0	0	0	0	0
Grizzly	0	1	0	0	0	0	0	0
Hallet	1	1	0	0	0	0	0	0
Karena	1	0	0	0	0	0	0	0
Henning	2	1	0	0	0	0	0	0
Leg	1	1	1	0	1	0	0	1
Little Bobtail	0	1	0	0	0	0	0	0
Marie	2	1	0	0	0	0	0	0
Murch	0	1	1	2	1	0	1	1
Naltesby	1	0	0	0	0	0	1	1
Ness	2	1	2	1	1	1	2	2
Nukko	2	2	2	1	1	0	3	3
Nulki	1	1	1	4	1	0	1	1
Oona	1	1	0	0	0	0	0	0
Opatcho	0	0	0	0	0	0	0	0
Pinchi	2	1	0	0	0	1	1	1
Punchaw	0	1	0	0	0	0	0	0
Purden	1	1	0	0	0	1	0	1
Saxton	1	2	1	1	1	1	1	1
Tabor	1	1	0	1	1	1	1	2
Tachick	1	2	2	2	1	3	1	3
Teardrop	0	0	0	0	0	0	0	0
Tezzeron	2	2	0	0	0	0	1	1
Verdant	0	0	0	1	1	0	0	0
Vivian	2	2	0	2	1	0	0	1
West	0	0	1	1	1	1	1	2
Woodcock	1	1	0	0	0	0	0	0

Table B.1 Variable list for the regional lake survey (Chapter 2) *(continued)*

	Soils		Forest Age Classes			Watershed	Sub-basins	
	Soil Texture *(1-5)*	Slope *(4, 10, 16)*	Newfor	Midfor *(ranked 0-4)*	Oldfor	Area *(km2)*	Avg. area *(km2)*	Number
Bednesti	3	10	1	3	2	70.7	17.5	4
Berman	3	10	1	3	2	6.8	70.7	5
Chief	5	16	1	3	2	152.2	37.9	12
Circle	4	4	0	3	2	4.2	0.6	3
Cluculz	3	10	2	2	3	642.2	38.2	17
Crescent	5	16	0	4	1	6.4	78	3
Dahl	3	10	1	4	2	11.8	3.2	2
Dorman	1	10	0	3	2	9.7		1
Eaglet	2	4	1	2	3	228.2	2.2	6
Eena	4	4	2	2	2	4.6		1
Eulatezella	3	10	2	2	2	233.1	6.3	7
Fish	1	10	2	0	3	8.5		1
Foster	1	10	2	2	2	17.8		1
Grizzly	5	10	4	1	0	3.2		1
Hallet	5	16	1	2	3	313.7	18.6	18
Karena	1	10	3	3	1	23.1	12.8	3
Henning	5	16	2	3	2	62.8	9.9	3
Leg	2	4	2	3	3	24.3		1
Little Bobtail	3	10	1	2	2	36.9		1
Marie	5	16	2	1	3	34.5	23.9	2
Murch	4	4	0	4	1	12.2	1.3	2
Naltesby	5	16	2	1	3	159.8	17	14
Ness	4	4	0	2	3	22.1	5.5	4
Nukko	4	4	1	3	2	89.3	2.2	3
Nulki	1	10	2	2	2	321.6	24.2	7
Oona	1	10	1	3	3	106.6	9.1	5
Opatcho	5	10	0	4	0	2.5		1
Pinchi	2	4	1	1	4	1004.9	33.8	36
Punchaw	5	16	1	2	4	60.1	2.3	4
Purden	5	10	1	2	1	51.5	2	4
Saxton	5	16	1	4	1	93.1	10.5	10
Tabor	5	10	0	2	3	29.0		1
Tachick	2	4	2	2	3	124.8	321.6	4
Teardrop	5	16	3	2	0	3.8		1
Tezzeron	1	4	1	1	3	1298.1	14.9	35
Verdant	5	16	0	3	2	0.8		1
Vivian	5	16	0	3	2	1.8	0.8	2
West	2	10	1	3	2	88.5	4	3
Woodcock	5	16	2	1	1	6.1		1

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