Evaluation of remediation options for Elk/Beaver Lake, Victoria BC

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Executive Summary

Cyanobacteria blooms have been deteriorating Elk Lake water quality in recent years. These blooms have been linked to internal phosphorus (P) loading from the bottom sediments in several reports. The small catchment basin and limited amount of lake shore development suggest that the majority of the nuisance cyanobacteria ("bluegreen algae") blooms are likely sustained by the internal P loading. Therefore, a detailed study on potential remediation options is needed.

We here first review and update the relative importance of external versus internal load concerning cyanobacterial blooms. Next we describe several potential lake remediation techniques and provide the details and reasons for the choice of our most preferred option. Last we determine data gaps and limitations respective the chosen treatment.

Based on previously determined loads by McKean for 1988, but with revised septic input, we estimated the total external P input as 224 kg/yr, which is 91.1 mg/m^2/yr pro-rated per Elk Lake (incl. Beaver Lake) surface area. This is a very small input rate and would contribute only 12.6 µg/L to the annual average lake concentration. This input is unlikely to create the large mixed layer total P (TP) increase during fall turnover and in the winter (e.g., from 0.009 mg/L in Jul-Sep to 0.46 mg/L on Nov 25, 2014).

There are several indications that internal P loading as sediment released P is high and a potential trigger of cyanobacteria blooms in Elk Lake.

a. Anoxia: Elk Lake is extremely hypoxic compared to other lakes. The depth below which dissolved oxygen (DO) was less than 2.0 mg/L was usually at 9 m and yielded anoxic factor (AF) values of 60 d/year and 86 d/year in 1988 and 2015. The AF values specify the number of days that a sediment area equivalent to the whole lake surface area (excluding Beaver Lake) was overlain by water ≤ 2 mg/L DO. Values of 41-60 d/yr are considered to represent eutrophic conditions, values above 60 are hypereutrophic.
b. Iron and manganese: A concomitant increase of redox-dependant metals (Mn and Fe) in the anoxic hypolimnion indicate redox-related P release from the sediment.
c. Phosphorus patterns: Hypolimnetic P increases in the summer and fall during thermal stratification, but does not yet affect the phytoplankton and trophic state of Elk Lake in its surface layer. The accumulated hypolimnetic P mass then gets distributed throughout Elk Lake when it turns over, so that surface layer TP and soluble reactive P (SRP, or phosphate) are highest during mixing. The previously released and accumulated hypolimnetic P mass fertilizes surface water layers and initiates and sustains cyanobacterial blooms during the mixing period in the winter.
d. Internal load estimate: Volumetrically pro-rated TP increases in 2014 and 2015 estimated an internal load of 1,752 and 1,585 kg/yr respectively. These high values were supported by an independent estimate of internal load (from AF and a predicted release rate) of 1,333 kg/yr in 2014. This means that internal load is 6-8 times that of external load or 86-89% of the combined load in recent years. Because internal load is in a much more biologically available form, its fertilizing effect on phytoplankton is much larger.

The extremely high internal compared to external P load of Elk Lake proper, and the pattern of cyanobacterial abundance indicate that any remediation should involve internal load abatement.

We advise against commonly used techniques including destratification, mixing, aeration, and oxygenation, in part because they need to operate continuously to be effective, which requires constant energy input, management, and maintenance. More importantly, these techniques mainly

address the symptoms, but not the cause of the problem of internal P loading. If they are interrupted, water quality likely becomes worse than before. Also, there is not enough water flow to divert water from the hypolimnion in a hypolimnetic withdrawal treatment, and the addition of aluminum compounds is not acceptable to regulatory agencies.

Therefore, our preferred option is a treatment with a clay-based P-binding material, lanthanum-modified bentonite, which has been applied world-wide and proven to be non-toxic. This material, called Phoslock^{TM}, consists of bentonite clay in which naturally adsorbed cations have been replaced with the rare earth element, lanthanum (La). In the presence of orthophosphate (SRP), La forms a highly stable mineral and effectively intercepts P release from the sediment.

Elk Lake's low external compared to internal loading rate, its low summer flow rate, and high alkalinity and low dissolved organic acids of the lake water are favourable for a Phoslock treatment. We present a preliminary dose and cost analysis based on a comparably high application rate of 4.6 metric tonnes/ha to the sediment that is primarily involved in P release (at and below 9 m depth contour) and the amount of SRP in the lake water. Total costs for such a dose are estimated as Can$ 1,686,000 (based on Phoslock supplier unit cost estimates of 22 Sep, 2015). We believe that such an application would result in the inhibition of almost all internal load from escaping the bottom sediment. We can estimate a lowering of the annual average lake water concentration to 12 µg/L at equilibrium conditions, although other influences including bottom disturbing fish (carp) and decaying macrophytes may prevent reaching this theoretical level. We expect that with severely reduced internal loading during the summer and fall months, cyanobacteria will not have enough nutrients to proliferate. Because Elk Lake is relatively nutrient poor, we recommend confirming the releasable amount of P by determining the mobile P concentration in the bottom of Elk Lake sediment and the extent of decaying plant material on the bottom.

Limitations and potential problems that could limit the benefit of a Phoslock treatment include under-dosing, but any treatment would still improve water quality by decreasing internal load that is available to cyanobacteria during the mixing periods. Another potential problem would be unknown or underestimated external P input, especially that from waterfowl on and around Elk Lake during winter. We recommend further estimates and the application of best management practices with respect to waterfowl, agriculture, septic systems and direct runoff.

The following monitoring is recommended before and/or after any potential Phoslock treatment. The distribution of mobile P sediment fractions that is involved in release (Section 4.3) throughout Elk Lake would ideally be determined before the treatment to confirm the depth for application and the dose estimate. Lanthanum should be monitored before and after the treatment in the water and sediment.

There is no post-treatment management needed, but limnological monitoring should be continued for at least three post-treatment years, including temperature and DO profiles, nutrient concentrations, phytoplankton analysis and microcystin analysis to determine treatment effects. To prevent further P accumulation on the sediments, we recommend to carefully evaluate and implement any practices that decrease TP input from external sources, wherever possible.

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Acronyms and Glossary

FLNRO – Ministry of Forests, Lands and Natural Resource Operations
MOE – BC Ministry of Environment
MOECC – Ontario Ministry of Environment and Climate Change
MNRF – Ontario Ministry of Natural Resources and Forestry

Anoxic factor, AF (days/summer or days/year): active period and area that releases P and contributes to internal load

Chlorophyll a: A measure of algae biomass, the green algal pigment. This measure of algal biomass in lake water is prone to analytical errors and its standardization is difficult, so that accuracy and precision are often low.

Cyanobacteria: Often called *bluegreens* or *bluegreen algae*, although they belong to bacteria. They can produce toxins that can create health effects if ingested in quantity (life stock, pets).

Dissolved oxygen, DO: Concentration of oxygen dissolved in lake water

Epilimnion (epilimnetic): mixed surface layer during times of thermal stratification.

External load, L_{ext}: The sum of annual TP inputs from all external sources, i.e. stream, non-point and point sources, precipitation and groundwater. Units are in kg/ yr or in mg per square meter of lake surface area per year ($mg/m^2/yr$). External load is a gross estimate. Much of its phosphorus is in a chemical form that is not immediately available to algae.

Hypolimnion (hypolimnetic): stagnant deep layer during times of thermal stratification.

Internal load, L_{int}: Annual TP inputs from internal sources, i.e. the sediments. Units are in kg/ yr or in mg per square meter of lake surface area per year ($mg/m^2/yr$). Gross estimates are usually used, but net estimates, based on mass budgets, can also be calculated. Most of the TP in L_{int} is in a chemical form (phosphate) that is highly available to phytoplankton and bacteria.

Mobile sediment P fraction: Sediment P fraction that are released as internal loading: pore water P, iron bound P, and labile organic P (Reitzel et al. 2005).

Polymixis: The mixing regime in lakes and reservoirs that describes frequent (daily to weekly in the summer) mixing of the whole water column.

Sediment oxygen demand (SOD): organically enriched bottom sediment takes up oxygen from the overlaying water which creates anoxic conditions

Soluble reactive P, SRP: soluble fraction of TP that consists mostly of the biologically available phosphate

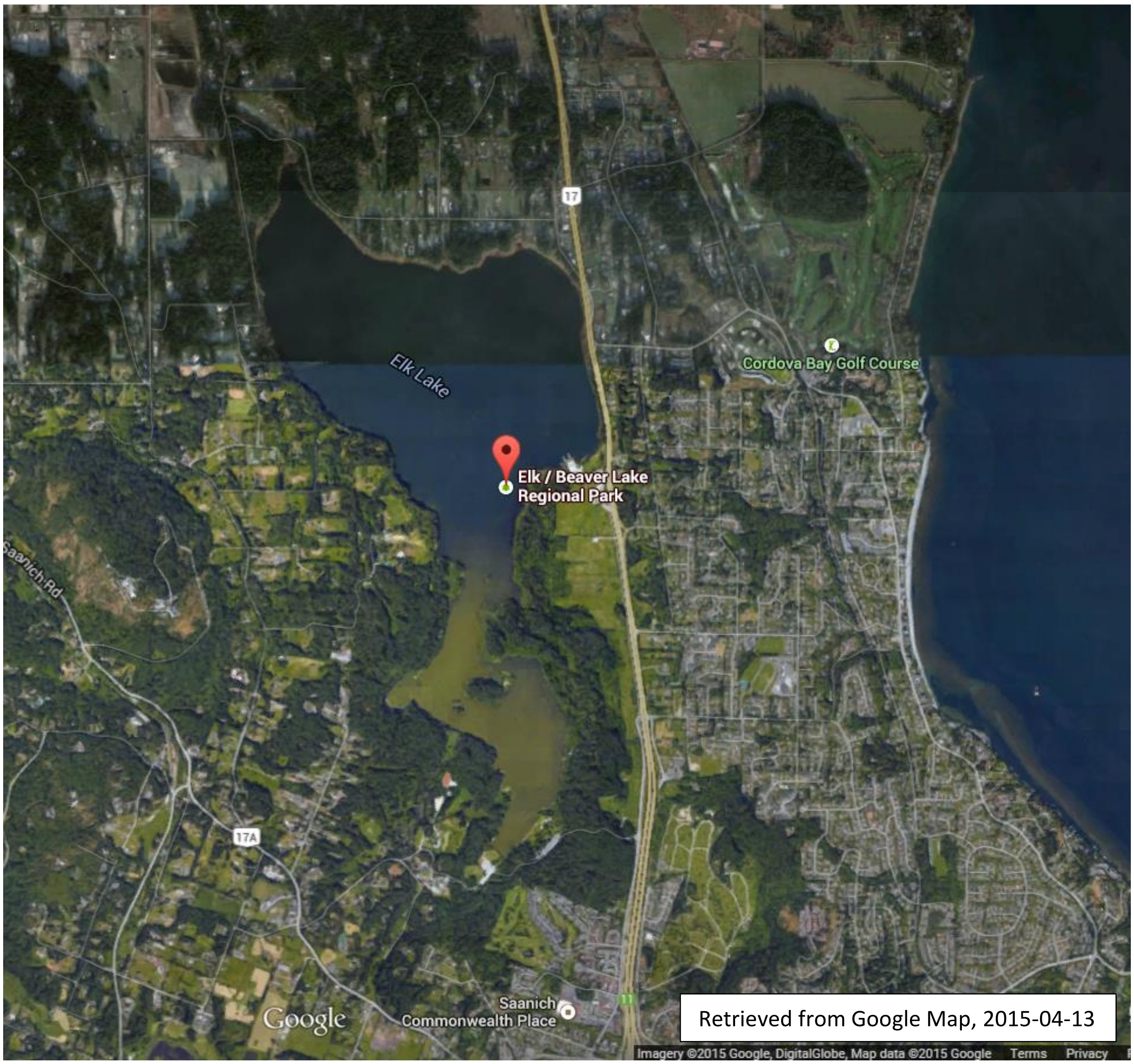
Total phosphorus, *TP*: All phosphorus (P) that can be analyzed in a water or sediment sample. It includes phosphate (highly available for algae), particulate forms (includes algae and non-living suspended particles), and forms not easily available to algae.

Elk Lake includes the Beaver Lake bay in this report, unless specified otherwise.

1 Introduction

Cyanobacteria blooms have been deteriorating Elk Lake water quality in recent years (Figure 1). These blooms have been linked to internal phosphorus loading from the bottom sediments in several reports. Even though few recent external P input values are available, the small catchment basin (watershed to lake area ratio is a low 3.8) and limited amount of lake shore development indicate that the majority of the nuisance cyanobacteria (“bluegreen algae”) blooms are likely sustained by the internal loading. Therefore, a detailed study on potential remediation options is needed.

Figure 1. Satellite graph of Elk Lake, probably in spring 2015 (exact date unknown). Note the green colour indicating a phytoplankton bloom in the southern portion, mainly Beaver Channel and Beaver Lake.



We here first review and update the relative importance of external versus internal load concerning cyanobacterial blooms. Next we describe several potential lake remediation techniques and provide the details and reasons for the choice of our most preferred option. Last we determine data gaps and limitations respective the chosen treatment.

Figure 2. Approximate catchment Basin of the Elk Lake watershed, shaded areas do not belong to the catchment. **(**Source: Map for the whole Colquiz River watershed, Dale Green, 2011)

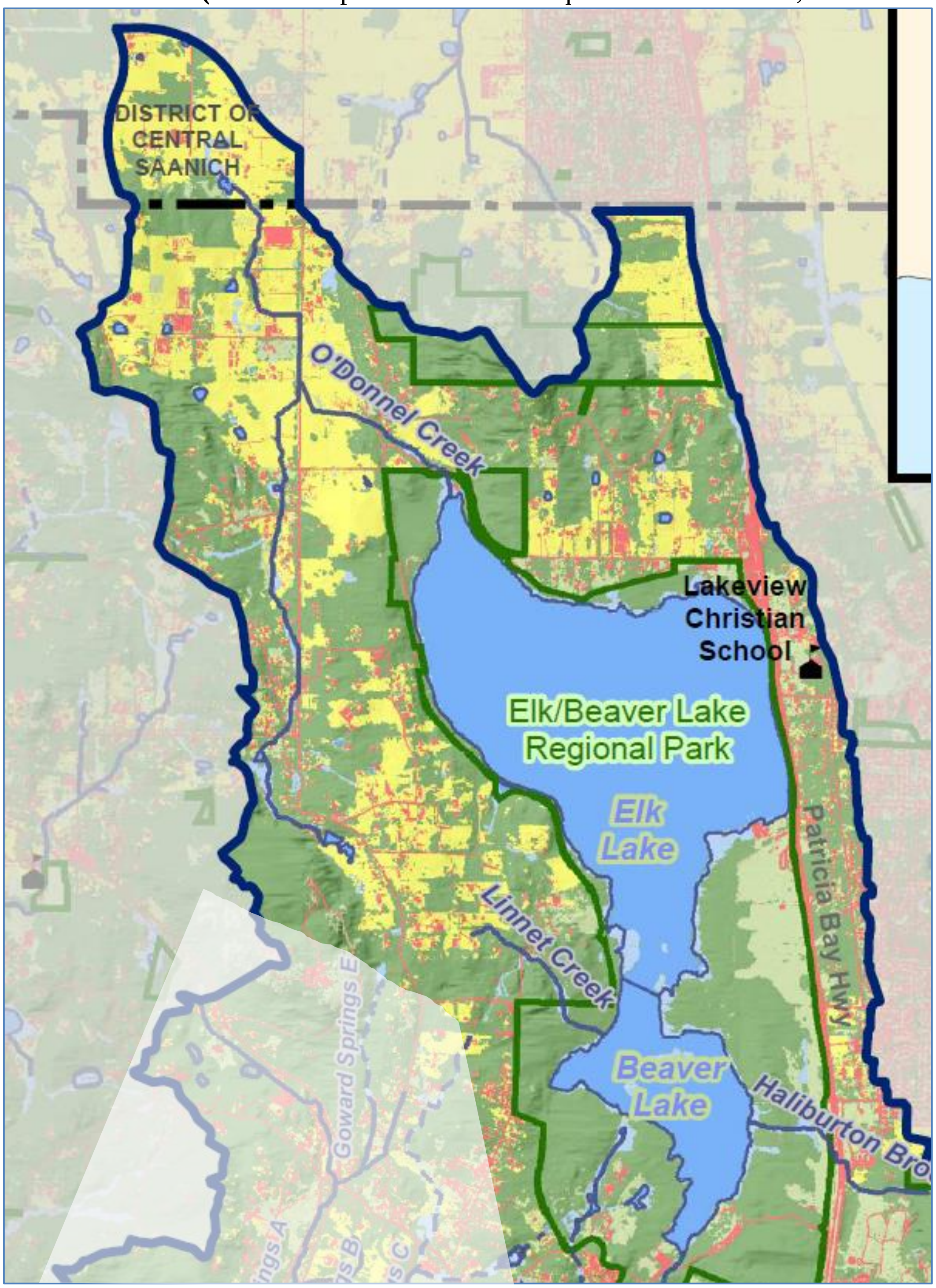
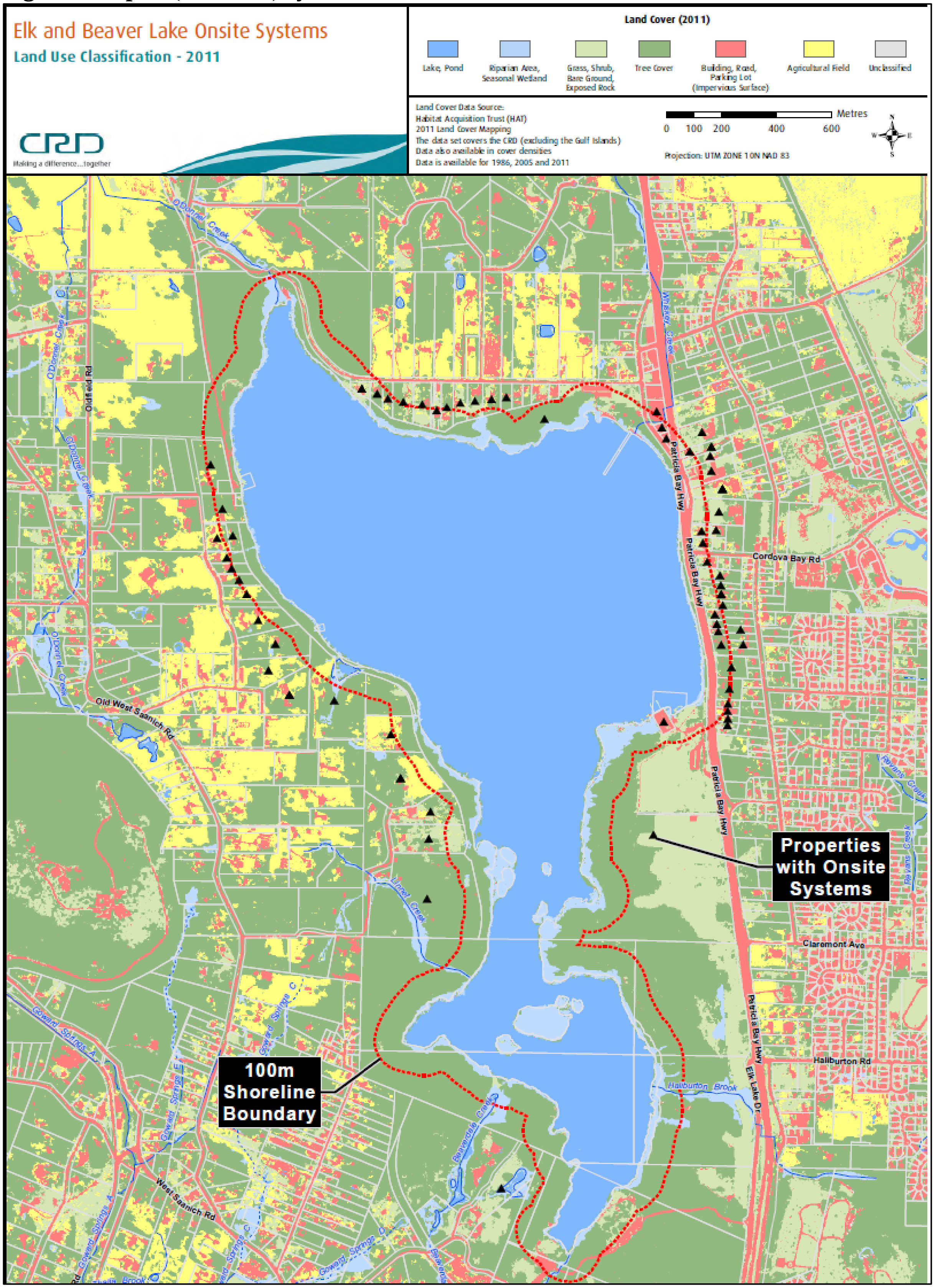


Figure 3. Septic ("Onsite") systems around Elk Lake



2 External load and its role in supporting cyanobacterial blooms

The watershed area is small compared to lake area (ratio: 3.8, Figure 2). Such a small watershed typically does not deliver a large amount of nutrients, even though a large part is agriculture. There is only limited development (63 dwellings), but the park is well used throughout the year by bikers, pedestrians, picnickers and boaters.

A previous P budget for 1988 determined an external load of 170 kg/yr from the main and ephemeral inflows, the atmosphere and birds, and another 246 kg/yr from septic systems (McKean 1992) (Table 1). Because Nordin (2015) and references therein reported the 246 kg/yr septic system input as unrealistically high, we re-estimated the load from dwellings based on export coefficients and usage experience from Muskoka cottages and the Ontario MOECC's lake shore capacity model (Nürnberg and LaZerte 2004, Paterson et al. 2006) as follows.

There are a total of 63 septic systems with many beyond the 100 m boundary of Elk Lake (Figure 3). There are none around Beaver Lake. To be conservative, we assumed that all 63 dwellings have an immediate impact on Elk Lake even if they were further away than 100 m from the lake shore. Using a usage number of 2.56 capita for each dwelling and P export of 0.3 kg/capita/yr, we calculated a total of 48.4 kg/yr from septic systems. This load is much smaller than McKean's estimate of 246 kg/yr (Table 1).

The main inflow is O'Donnel Creek, and McKean estimated 87 kg/yr in 1988. Based on "preliminary" daily flow model estimates ("preliminary" data by FLNRO's regional hydrologist) and 25 mostly biweekly TP grab samples (Figure 4), we estimated 98 kg for May 2014 – Apr 2015 (Appendix A). This estimate supports McKean's estimate and we based further calculations on the average of both values.

We estimated the total external P input as 224 kg/yr, which is 91.1 mg/m^2/yr pro-rated per Elk Lake surface area (Table 1). This is a very small input rate and would contribute only 12.6 µg/L to the average annual water column concentration (based on q_s= 1.74 m/yr, predicted retention R= 15/(18+q_s)= 0.76, (Nürnberg 1998).

Table 1. Components of external load for 1988 (McKean, 1992) and revised septic input.

External load contribution (kg/yr)	1988	2014-15	Used
O'Donnel Creek	87	98.4	93
Atmosphere	38		38
Ephemeral creeks	36		36
Birds	9		9
Septic systems	246	48.4	48.4
Sum	416		**224**

Further, the timing of external loading does not coincide with the observed lake TP increases. Elk Lake regional climate is characterized by mild wet winters and warm dry summers. There is very little inflow and precipitation in the summer months, and lake TP concentrations increase before flows and related external inputs intensify (e.g. for 2014-15, Figure 5).

In conclusion, external load cannot explain the high TP concentration in Elk Lake nor the seasonal patterns. Even so, we recommend to carefully evaluate and implement any practices to decrease TP input from these sources wherever possible.

Figure 4. Main inflow (O'Donnel) *("preliminary" data by FLNRO's regional hydrologist)*, **and TP concentration compared to surface (0.1-0.5 m) TP at Elk Lake, main station**

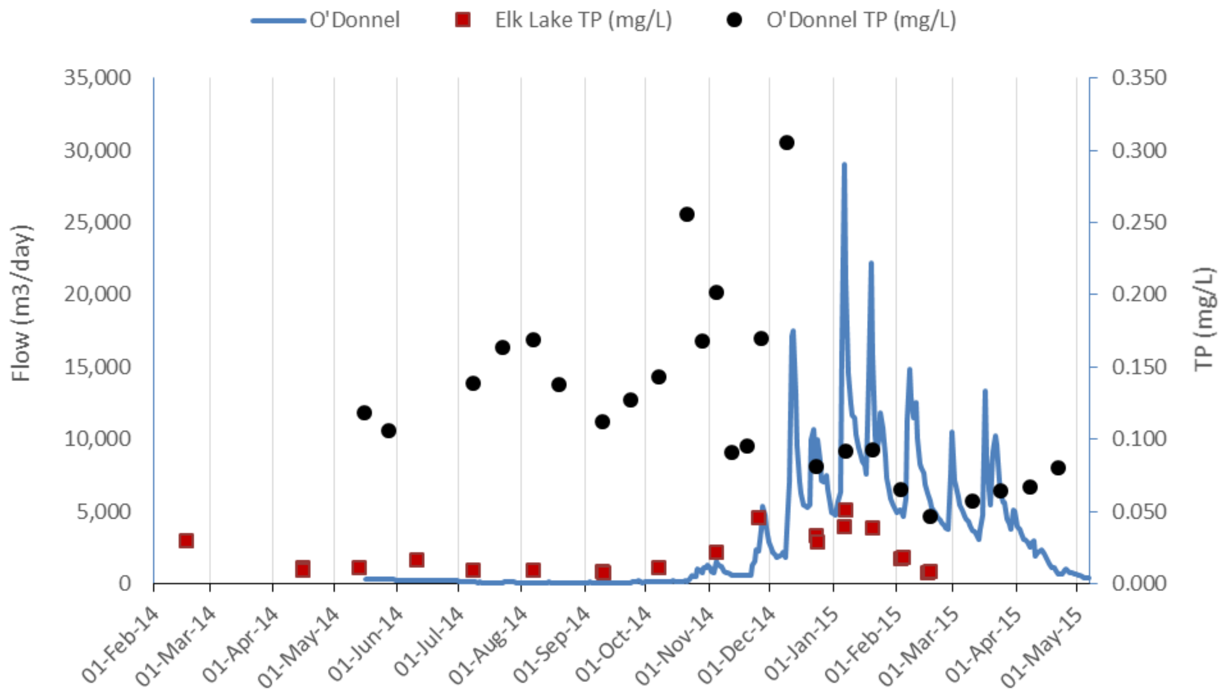
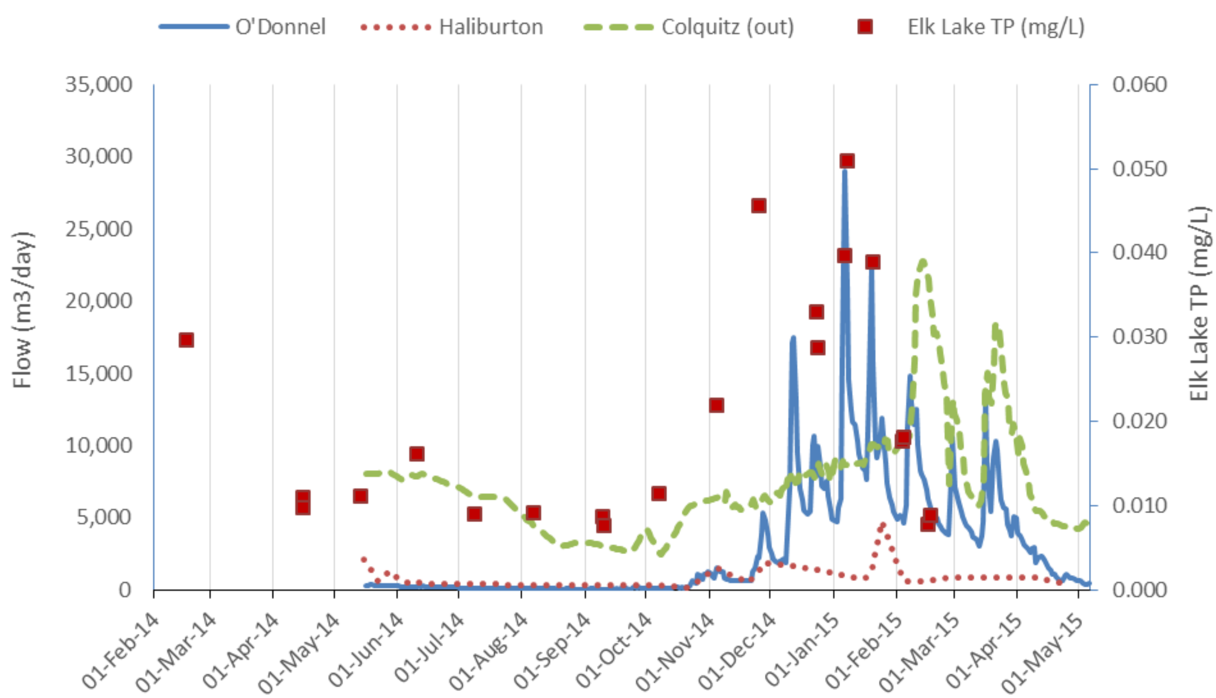


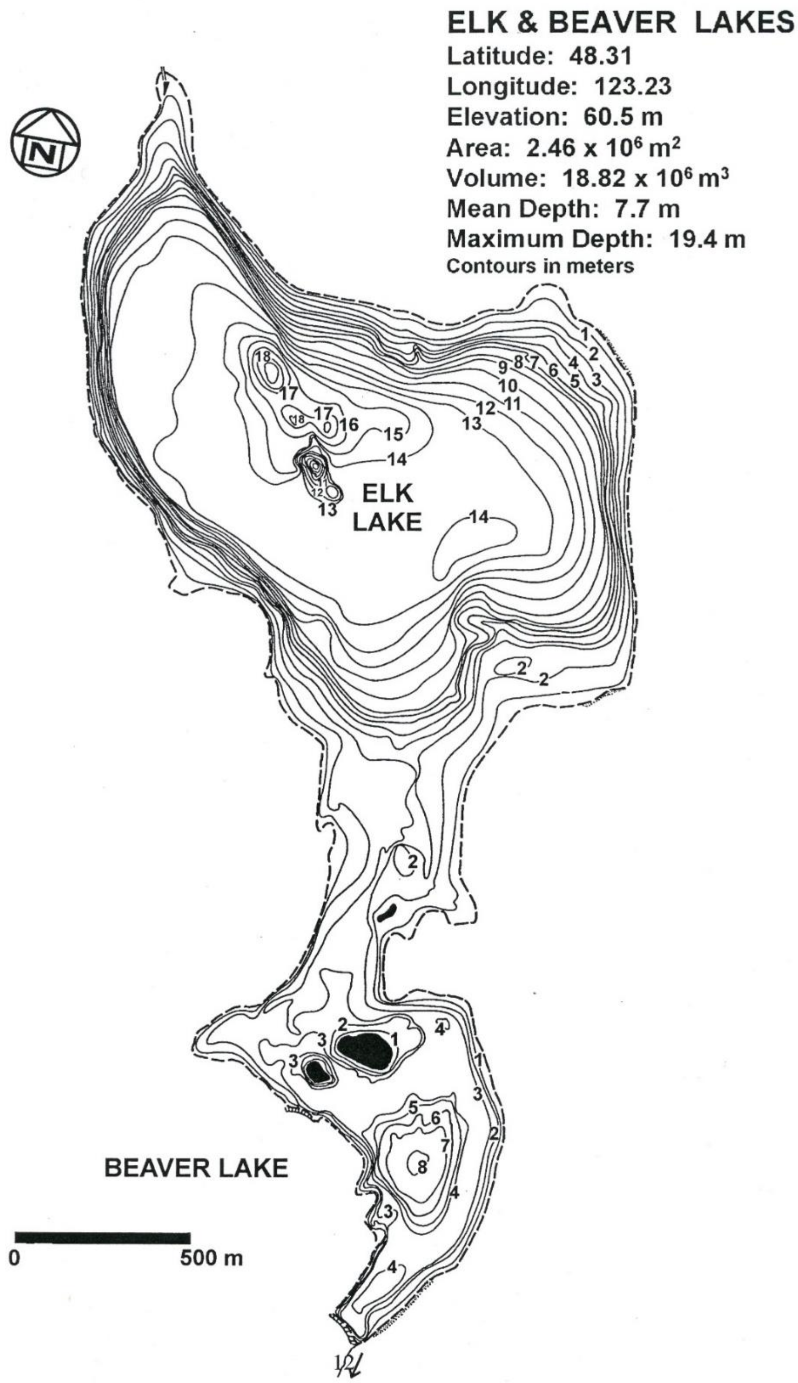
Figure 5. Main inflow (O'Donnel), small inflow (Haliburton), outflow (Colquitz) *("preliminary" data by FLNRO's regional hydrologist)*, **compared to surface (0.1-0.5 m) TP concentration at Elk Lake, main station.**



3 Characteristics that indicate the importance of internal P loading respective cyanobacteria blooms

Elk Lake consists of a deep stratified basin (Elk Lake proper) and a smaller shallow polymictic basin (Beaver Lake), (Figure 6). Long hypoxic stratification leads to a large accumulation of hypolimnetic P in the deep basin, while the shallow depth of the small basin supports nearly continuous mixing so that any sediment released P is distributed into the light infused (photogenic) zone throughout the growing period in Beaver Lake.

Figure 6. Morphometry and depth contours of Elk Lake



3.1 Anoxia

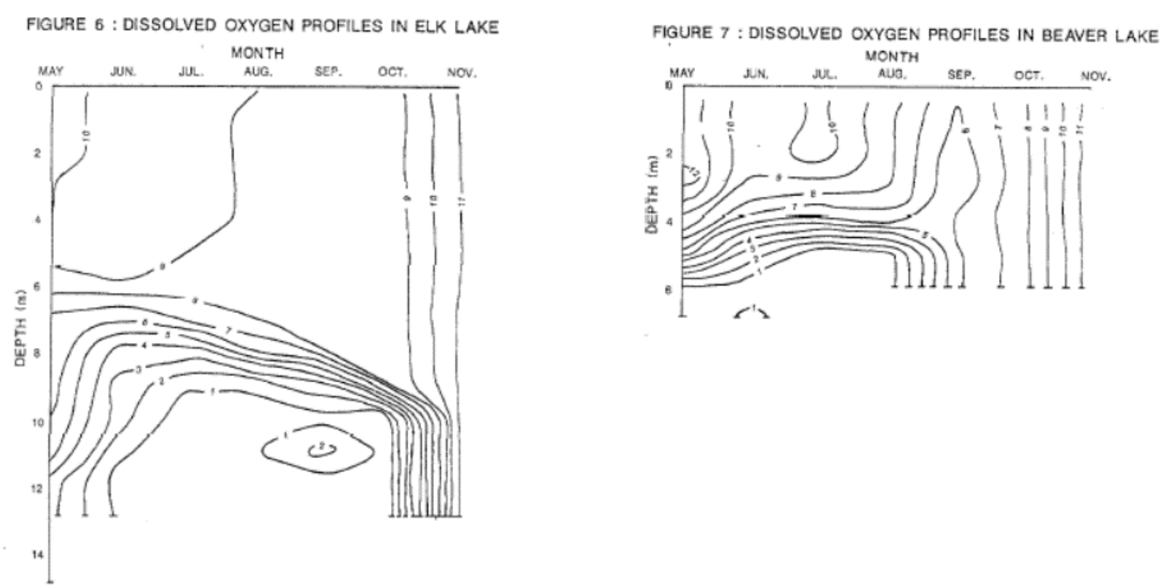
Elk Lake proper is severely hypoxic throughout the summer and fall. Typically hypoxia starts as soon as the lake stratifies indicating a high sediment oxygen demand (SOD) from accumulated organic substances.

The sediment-water interface can be assumed to be anoxic (and release P as internal loading), once dissolved oxygen (DO) concentrations fall below 2 mg/L. The depth below which DO was less than 2.0 mg/L was usually at 9 m in 2014 (measured: 10 Jun- 27 Aug., 2014, but 8 m on 23 Jul., 2014) and 2015 (measured: 9 Jun to 17 Aug 2015, MOE data). The 9 m contour coincided with low DO below 2 mg/L already in 1988 (Figure 7).

This information on the spatial and temporal extent of anoxia (≤ 2 mg/L) was summarized in one expression, the anoxic factor (AF, Appendix B). It yields a large number, 86 d/year in 2014 and 60 d/year in 1988. The AF values specify the number of days that a sediment area equivalent to the whole lake surface area (excluding Beaver Lake) was overlain by water ≤ 2 mg/L DO. Values of 41-60 d/yr are considered to represent eutrophic conditions, values above 60 are hypereutrophic (Nürnberg 1996). Obviously, Elk Lake is extremely hypoxic compared to other lakes.

But even shallow polymictic Beaver Lake has been occasionally hypoxic. DO concentration below 2 mg/L at a depth of 5 m and below were consistently measured in July and August 1988 (Figure 7) and on 13 May 13, 2014. The only other recent DO profile in Beaver Lake, of Aug 6, 2014, revealed solid anoxia at all six measured depths between 1-6 m and a low DO concentration of 2.7 mg/L at 0.5 m depth below surface.

Figure 7. 1988 DO concentration in Elk Lake proper (left) and Beaver Lake (right), (McKean 1992).



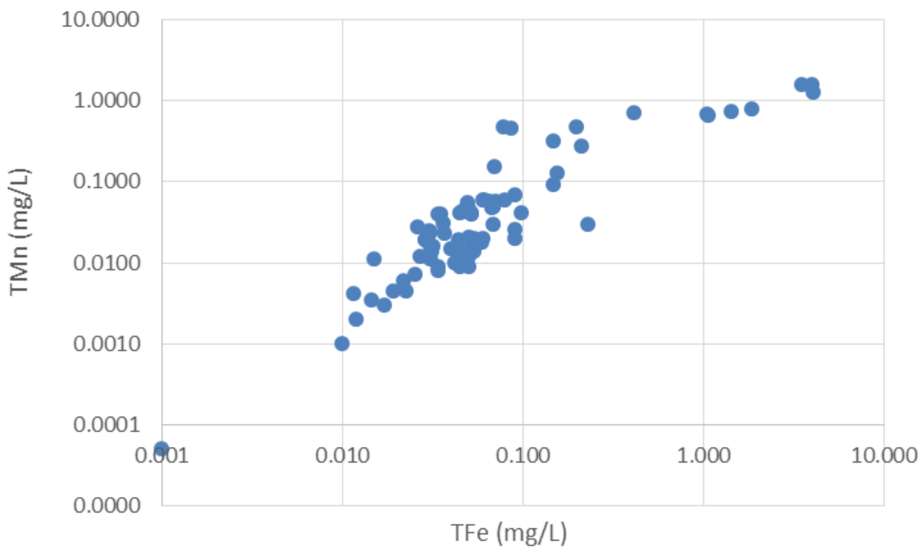
We conclude that monitored DO profiles indicate sufficient hypoxia to warrant sediment P release in both lake basins.

3.2 Iron and Manganese

Iron and manganese are released from the sediment under hypoxic (Mn) and anoxic (Fe) conditions and hence often accompany P release. Total iron concentrations Elk Lake are elevated at deeper depths during recent stratification seasons indicating redox-dependent P release (TFe >1 mg/L June-Nov 2014 at 14 m and below). Corresponding manganese concentration are elevated as well, also supporting the redox dependency of P release (Figure 8).

Figure 8. Total iron (TFe) versus total manganese (TMn) for all depths at Elk Lake main station (1988-2015).

All TFe concentration above 0.1 mg/L are from depths 10 m to bottom. Note that the axis are transformed logarithmically to the basis of 10.



3.3 Phosphorus patterns in Elk Lake proper

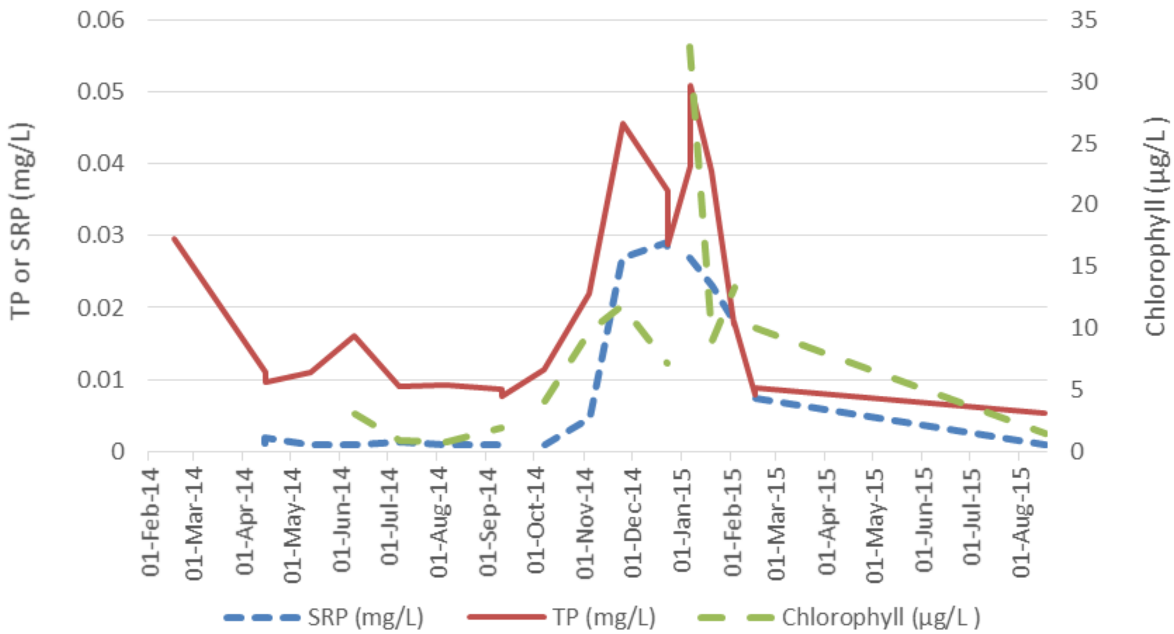
As observed by others before (Nordin 2015, McKean 1992), mixed layer (epilimnetic) TP concentrations typically exhibit a strong seasonal pattern with low concentration during the summer stratification period and elevated concentration during the late-November to mid-April winter and spring mixed period (Figure 9, Appendix D).

There are many incidences when surface layer TP was highest during thermal mixing. For example, in 2014, winter mixing TP concentration (Feb 0.030 mg/L) was almost 3 times higher than during the stratified period (July-Sep/Oct, 0.009 mg/L) and high again after fall turnover (Nov-Jan <0.033 mg/L) (Figure 10). Already in 1988, May TP concentration (0.020 mg/L) was twice as high as the concentration during stratification in Aug-Oct (0.010 mg/L).

Soluble reactive P (SRP, or phosphate) concentrations follow a similar pattern and were close or below detection limits during summer stratification and increase in the mixed period in 2014/2015 (Figure 10, Appendix D). SRP, which is mostly biologically available phosphate, most likely triggered the phytoplankton growth in late January 2015 and formed the cyanobacterial blooms as reported previously (Figure 10).

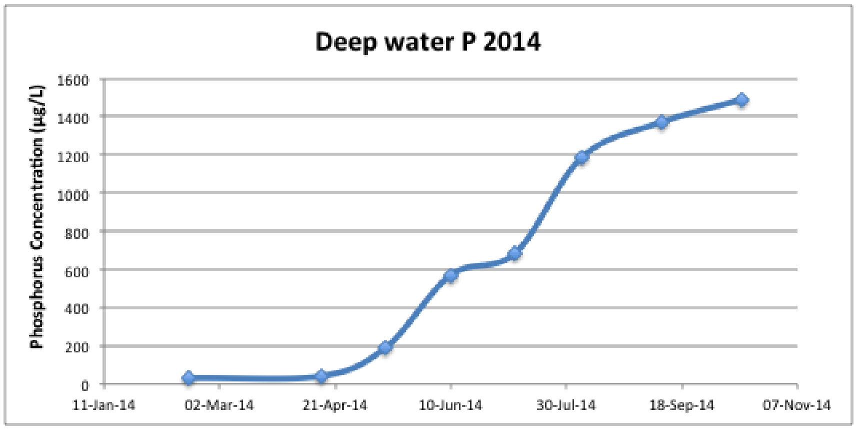
Figure 9. Surface (0.1-0.5 m) TP concentration at Elk Lake main station (1986-2015)

Figure 10. TP, SRP and chlorophyll in surface water at Elk Lake, main station (Feb 2014-Aug 2015).



This pattern of surface TP and SRP can only be explained by the high P concentration in the hypolimnion (Figure 11, Appendix D) during stratification that is then mixed into the surface water during thermocline erosion and fall mixing. External inputs are unlikely to create the large mixed layer TP increase during fall turnover and in the winter (e.g., from 0.009 mg/L in Jul-Sep to 0.46 mg/L on Nov 25, 2014, Appendix D).

Figure 11. Hypolimnetic TP in 2014, note the different concentration units 1000 µg/L = mg/L (graph from Nordin 2015)



Most of the increase of hypolimnetic P (due to P release from the bottom sediments as internal P loading) occurs in the summer and fall during thermal stratification, but does not yet affect the phytoplankton and trophic state of Elk Lake in its surface water layer. The accumulated hypolimnetic P mass then gets distributed throughout Elk Lake when it turns over, which is usually complete by the end of December. The previously released and accumulated hypolimnetic P mass then fertilizes surface water layers and initiates and sustains cyanobacterial blooms.

Different from dimictic temperate lakes, where snow cover and severe temperature prevents phytoplankton growth in the winter, Elk Lake phytoplankton biomass and cyanobacterial blooms increase in the months of January and February (Figure 10). For example, a chlorophyll maximum of 32 µg/L was measured in Jan 2015, when most of the increased pigment was due to *Aphanizomenon* at a high count of 13,000 cells/ml (Nordin 2015). High cell counts of *Anabaena, Woronichinia,* and *Rhodomonas* (26,724 cells/ml) were also observed on May 13, 2014 after spring overturn.

3.4 Quantification of internal load

Volumetrically pro-rated water-column TP concentrations were used to determine internal P loading. TP increases in the water column were high in 2014 and 2015, although initially restricted to the hypolimnion. Water-column average concentration was only 0.033 mg/L measured on Feb 17, 2014, and increased to 0.126 mg/L on Aug 6, 2014 and 0.106 mg/L on Oct 7, 2014 (computed for two TP profiles measured at 0.5 m for the mixed layer, and every 2 m for depths 8-16m, Appendix D). Similarly, water-column TP increased from 0.020 mg/L on Feb 3, 2015 to 0.104 mg/L on Aug 17, 2015. From these increases in the volume-weighted P concentration during stratification an in situ internal load can be estimated, as already pointed out by Nordin (2015). In situ internal load is much higher in recent years compared to 1988 (Table 2).

An independent estimate of internal load (AFxRR, Table 2) supports these values as it falls between the 1988 and the recent estimates. It is based on the 2014 anoxic factor (Appendix B) and a sediment release rate predicted from sediment TP of the approximate period of 1990-2001 (Appendix E).

We conclude that internal load is 6-8 times that of external load or 86-89% of the combined load in recent years. And because internal load is in a much more biologically available form, its fertilizing effect on phytoplankton is much larger (Nürnberg 2009).

Table 2. Internal load estimates for Elk Lake (w/o Beaver Lake)

Year	Internal load		Source/Method
	kg/yr	$mg/m^2/yr$*	
In situ increase:			
1988	950	509	McKean 1992
2014	1,752	939	Feb-Aug volume-weighted water column TP increase, this study
2015	1,585	849	Feb-Aug volume-weighted water column TP increase, this study
Anoxia and predicted release rate:			
2014	1,333	714	AFxRR, where AF= 85 d/yr, RR= 8.3 $mg/m^2/d$ based on sediment TP of 1.44 mg/g dry wt

*Based on Elk Lake (proper) surface area of 1,866,000 m^2 (note the areal external load in Section 2 is based on the whole lake)

4 Assessment of potential remediation techniques

The extremely high internal compared to external P load (about 8 times in 2014) of Elk Lake proper, and the pattern of cyanobacterial abundance indicate that any remediation should involve internal load abatement. Beaver Lake possibly experiences internal load as well, but the released P is distributed and available to phytoplankton immediately after release from anoxic sediment surfaces already in the summer.

Nonetheless, Elk Lake proper is the main candidate to be remediated for several reasons:

1. Limnologically: Elk Lake experiences a large internal load which dwells up to create the winter cyanobacteria blooms.
2. Elk Lake is upstream of Beaver Lake. Any water quality improvement would positively affect Beaver Lake.
3. Beaver Lake has mainly been used for family swimming, rowing, and fishing (MOE pers. comm.). There is no lake shore property around its shoreline. It used to be a shallow wetland before damming of its outflow which elevated lake level by 5 m (Nordin 2015).

4.1 Physical treatment (destratification, aeration, oxygenation)

Destratification, aeration, and oxygenation are treatment options that are sometimes applied to reduce internal sediment P loading. However, it has been shown in various studies that these techniques do not always manage to aerate the sediments sufficiently to inhibit sediment P release (Gächter and Wehrli 1998, Horppila et al. 2015) or that release has become independent of the oxygen state during severe eutrophication (Katseva and Dittrich 2012). Oxygenation did not improve mixed layer water quality in 4 out of 5 Danish lakes (Liboriussen et al. 2009), and destratification has resulted in worse eutrophication and cyanobacterial blooms (Nürnberg et al. 2003).

None of these treatments are advised in Elk Lake for the following reasons:

Destratification and **artificial mixing**: Elk Lake *proper* stratifies until late fall so that nutrients are effectively trapped below the thermocline. Summer destratification would distribute the nutrients throughout the water column at a time when light and temperature are high and therefore would benefit cyanobacteria. Consequently, artificial destratification can be expected to enhance cyanobacteria blooms throughout the summer and fall. Fall destratification has resulted in worse eutrophication and cyanobacterial blooms in a deep kettle lake (Nürnberg et al. 2003). Destratification by aeration has decreased water quality by increasing P in a study on 212 Minnesota lakes (Beduhn 1994). However, artificial mixing can shift the phytoplankton genera from cyanobacteria to green algae, if certain conditions, including permanent and deep destratification, are met (Visser et al. 2015).

Aeration and oxygenation: These treatments could be applied without destratifying the water column of Elk Lake, by only treating discrete layers, e.g., the hypolimnion. Nonetheless, neither hypolimnetic aeration nor oxygenation can be expected to effectively inhibit internal P loading from Elk Lake bottom sediment because of the high sediment oxygen demand as indicated by extended anoxia, the extremely large AF, and the presence of hydrogen sulfide in Elk Lake hypolimnetic water (rotten egg smell identified by MOE sampling staff). It would be technically difficult to contribute enough oxygen to Elk Lake with aeration equipment, and even oxygenation would unlikely seal the sediments effectively.

Although hypolimnetic aeration successfully decreased hypolimnetic TP concentration in an experimental study of a small BC lake (Ashley 1983), similar applications did not manage to aerate the sediments sufficiently to inhibit sediment P release in other lakes (Gächter and Wehrli 1998, Horppila et al. 2015) or release had become independent of the oxygen state during severe eutrophication (Katseva and Dittrich 2012). Oxygenation did not improve mixed layer water quality in 4 out of 5 Danish lakes (Liboriussen et al. 2009). Only hypolimnetic aeration or oxygenation coupled with a chemical to adsorb P, such as iron and aluminum, consistently decreased internal load and delivered positive effects on trophic state (Cooke et al. 2005, Moore et al. 2012). The increase in hypolimnetic temperature associated with this treatment would also increase sediment oxygen demand and areal P release rates (Bryant et al. 2011), potentially increasing internal P loading.

On the other hand, hypolimnetic aeration or oxygenation would likely increase DO concentration so that fish habitat would be expanded. But it is unlikely that the sediment water interface would be aerated sufficiently to improve macrobenthos habitat.

There is a long history of aeration treatment in BC lakes. In particular, Ken Ashley, Adjunct Professor, Dept. of Civil Engineering, Univ. of British Columbia, has been involved with several local applications and is one of the most recognized proponents worldwide. Based on an exchange with Ken Ashley, we discuss two recent applications in southern BC in Appendix F (in St Mary's Lake on Saltspring Island and in Langford Lake on southern Vancouver Island). Hypolimnetic aeration was discontinued in St. Mary's Lake apparently because it failed to inhibit cyanobacterial blooms. Langford Lake aeration elevated the minimum hypolimnetic oxygen concentration to 4 mg/L, but fall blooms continued to occur coinciding with the period when aerators are not in operation. (Aeration in Langford Lake targets the summer period when recreation values are high and fish struggle with high temperature and low DO.) However, no detailed study has been conducted to confirm the effect of the treatment.

An extensive literature review of about 70 references did not reveal any positive effects except that hypolimnetic anoxia was generally prevented where the systems were sized appropriately. But other water quality indicators including TP concentration and cyanobacterial biomass did not noticeably decrease in lakes treated with hypolimnetic aeration or oxygenation.

In general: The techniques mentioned above have several drawbacks in common: A. They need to operate continuously to be effective, which requires constant energy input, management, and maintenance. B. They require equipment on the lake surface that is open to vandalism and present navigational challenges. C. They mainly address the symptoms, but not the cause of the problems. D. If they are interrupted, water quality can become worse than before because oxygenation increases P retention and decreases export. This means that P accumulates on the sediment during operation and is released when aeration treatment is discontinued. This accumulated sediment P would lead to enhanced P release once the hypolimnion becomes oxygen depleted again.

4.2 Flow management (selective, i.e., hypolimnetic withdrawal)

Another treatment to reduce internal P loading in stratified lakes is **hypolimnetic withdrawal** (Nürnberg 2007). This treatment withdraws lake water preferentially from the nutrient-rich hypolimnion instead of the surface and hence increases P export. However, such treatment requires a certain amount of water flow to prevent the lake level from decreasing unacceptably. Elk Lake's low flushing rate of 0.23 per year (Zimmerman 1996)) is not sufficient (Nürnberg 2007). Furthermore, the highest nutrient accumulation in the hypolimnion occurs during summer and fall, which is also the period of extremely low in- and outflow, so that timely remediation could not be implemented. Besides, the withdrawn water would have to be treated before being released into Beaver Lake or further downstream into the salmon-carrying Colquitz Creek, which would introduce additional costs and monitoring obligations.

4.3 Preferred treatment: lanthanum-modified clay (Phoslock)

4.3.1 Background

Of all possible internal load abatement techniques we suggest that only P precipitation and capping with either aluminum-hydroxyl forming compounds or a clay-based P-binding material will have significant long-term benefits in Elk Lake. Such treatment is especially applicable because of the negligible water flow during thermal stratification and the limited external nutrient input that could interfere. This management would treat the cause, i.e., intrusion of biologically available P from internal loading into the surface, and would benefit the system, no matter the dose. The extent and duration of potential benefits would depend on the initial dosage and can be calculated (predicted) from sediment P fractions.

Of any clay-based P-binding materials, lanthanum-modified bentonite has been most researched and successfully applied in lakes. The only commercially available form of this material is called Phoslock^{TM} that was developed by the Commonwealth Scientific and Industrial Research Organization (CSIRO) of Australia. This product has been used throughout Australia (Robb et al. 2003), New Zealand (Burns et al. 2009, Hickey and Gibbs 2009), China (Liu et al. 2012), Europe (Meis et al. 2012, 2013, Lürling and Oosterhout 2013, Spears et al. 2013), and the USA (Bishop et al. 2014) to treat, lakes, rivers, stormwater ponds, and drinking water reservoirs.

Aluminum-hydroxyl forming compounds are not acceptable to British Columbia's and other provinces' regulatory agencies (e.g., Ontario, Quebec). Instead, Phoslock treatment has been applied in Ontario because of its general lack of toxicity (Ministry of Environment 2009, Lürling and Tolman 2010). Phoslock received the US and Canadian NSF/ANSI Standard 60 Certification for use in drinking water in 2011 (Finsterle, 2014). Consequently, we considered the suitability of a Phoslock treatment to Elk Lake in detail.

In Canada, Phoslock was successfully applied to an urban hyper-eutrophic Ontario lake in 2011, where average TP concentration decreased from 0.247 mg/L to 0.099 mg/L in the first and to 0.060 mg/L in the second post-treatment year, practically eliminating internal loading (Appendix G. Case study: Phoslock application to Swan Lake, Markham, Greater Toronto, Ontario), (Nürnberg and LaZerte 2016). Phoslock was used with only limited success by the Lake Simcoe Region Conservation Authority on storm water management ponds and sections of a slow-moving river in 2008 (Lake Simcoe Region Conservation Authority 2010a). An open system and a high flushing rate was determined as the main reason for this lack of long-term water quality improvement (Moos et al. 2014).

Phoslock consists of bentonite clay with a high exchange capacity, in which naturally adsorbed cations have been replaced by the rare earth element, lanthanum (La). In the presence of orthophosphate (SRP) La forms the highly stable mineral called rhabdophane ($LaPO_4 \, nH_2O$) (Ross et al. 2008). The amount of Phoslock necessary to inactivate phosphorus is based on phosphate (SRP) in the water and releasable (mobile) P in the sediment (pore water P, iron bound P, and labile organic P; Reitzel et al. 2005). Phoslock contains 5% La by weight, and P is adsorbed by La at a molar ratio of 1:1 (Ross et al. 2008). In dosage calculations only the freely available P components are considered, i.e., SRP in the water at time of application and mobile P, mainly iron-bound P, in several centimeters of surficial sediment (Meis et al. 2012).

Phoslock is effective without changing the oxygen content of the hypolimnion. But decreases in internal load lead to decreases in productivity and organic settling that would eventually decrease sediment oxygen demand and hypolimnetic oxygen concentration. If deep water hypoxia is an immediate concern, it would have to be dealt with separately.

The immediate effect of Phoslock on lake water quality is the decrease of TP and SRP that causes a decrease in phytoplankton and cyanobacteria. Consequently, cyanobacteria blooms are less frequent or spatially and temporally more confined. Such improvement has been reported repeatedly in Phoslock treated lakes (Copetti et al. 2015, Spears et al. 2015, Nürnberg and LaZerte 2016).

4.3.2 Considerations specific to Elk Lake

In this section we discuss those Elk Lake characteristics that potentially influence and direct Phoslock application details. This information is thought to be a starting point for further elaboration by local limnologists and experienced Phoslock personnel.

The **flushing rate** of Elk Lake is low, especially in the summer (Figure 12), so that any applied chemical would settle to the sediment and not be flushed out. High flushing in a riverine environment was the most likely reason for the unsuccessful application in the Lake Simcoe watershed (Moos et al. 2014).

Chemical characteristics: A rapid disappearance of lanthanum (and avoidance of any toxicity) occurs in lake water with moderate to high **alkalinity** (>0.8 mEq/L, Spears et al. 2013). Elk Lake alkalinity was within that range at 50-60 mg/L as $CaCO_3$ (or 1-1.2 mEq/L) in the mixed layer Feb/Mar since 1993 and 139 mg/L on 25 March 2014, so that fast disappearance of La in the water column can be expected.

Dissolved organic carbon (DOC), which includes humic acids and other dissolved organic acids, can decrease P binding efficiency by competing with La. Lake DOC concentrations below 10 mg/L showed only negligible effects in laboratory experiments (Dithmer et al. 2015). Elk Lake DOC was 3.7-7.8 mg/L and averaged 5.5 mg/L since 2000 throughout the lake (including the deep hypolimnion), so that interference from organic acids is not expected.

This means that Elk Lake physical and chemical characteristics are favorable respective a Phoslock application.

Extent of bottom area to be treated: In stratified Elk Lake, the depths of the bottom sediment involved in P release can be determined from DO and TP profiles. When measured water DO is less than 2.0 mg/L during stratification, the sediment-water interface is probably anoxic and can release P (Nürnberg 2009). The depth below which DO was less than 2.0 mg/L was usually at 9 m in 2014 (measured: 10 Jun - 27 Aug., 2014, but 8 m on 23 Jul., 2014) and 2015 (measured: 9 Jun - 17 Aug 2015). The 9 m contour coincided with low DO below 2 mg/L already in 1988 (Figure 7, Appendix B).

TP concentration increased with depth during thermal stratification as discussed above (Section 3.3). When available, concentrations at 8 m depth are marginally elevated but are usually clearly elevated at 10 m (the next measured depth) (Appendix D).

Therefore, DO and TP profiles indicate that bottom sediment at and below 9 m are most likely involved in P release. Possibly, some sediments at slightly higher depths are involved as well, but because of the mechanical process of sediment focussing, settling material, which is enriched in P, accumulates preferentially at deeper locations.

Any application of Phoslock will intercept P release and thus be treating internal load even if only done to a small deep area. However, the most effective minimum dosage would be full coverage below the 9 m contour. This area extends over 1.1 km^2 (Table 6, Appendix C, Appendix D). If resources allow, the application area could be larger, perhaps just below the depth of macrophyte habitat (6 m, Section 5.3) at about 7 m depth, or even the whole lake.

Timing: To determine the most effective time of application, seasonal changes in Elk Lake limnology have to be considered. If mainly the deeper part that is not affected by macrophytes is to be treated for internal load, an application during the period of July-September, before commence of thermocline erosion and rainy season would be optimal. This period coincides with the summer-fall stratification when there is very little flushing and hypolimnetic SRP concentration is high. These conditions promise an effective treatment because:

a. External P input is low
b. Low flushing means that the chemical can accumulate on the sediment to intercept P released from sediments.
c. The high water SRP concentration at that time can be intercepted by Phoslock, so that noticeable benefits in the reduction of cyanobacterial blooms should already occur in the first mixing period following the treatment (i.e., the bloom-prone period of Jan-Mar).

However, if resources would allow for a more extensive application that includes macrophyte covered bottom areas, a time before prolific macrophyte growth would be recommendable, i.e., as soon as possible after stratification onset in the spring.

Potential resuspension and export downstream: Phoslock is unlikely to be flushed downstream into the Colquitz Creek for several reasons.

(1) Once Phoslock is distributed into the water column it quickly settle to the bottom where it is unlikely to be resuspended. Laboratory studies of artificial resuspension showed that Phoslock reduced P concentrations several days after resuspension and increased sediment stability by 265%. In comparison with alum, which was also included in the study, Phoslock was found to be five times less likely to resuspend (Egemose et al. 2010). The presence of Chironomids, which are bottom dwelling organisms that induce bioturbation of sediments, did not hinder P decrease in experiments treated with Phoslock (Reitzel et al. 2013).

(2) The morphometric conditions are such that Phoslock applied to Elk Lake proper below a depth of 6-9 m, can unlikely pass the channel into Beaver Lake, which is 3 m deep (McKean 1992). Therefore, it is even more unlikely that it would get flushed out of Beaver Lake into Colquitz Creek and from there into Portage Inlet.

Any potential sediment disturbance by bottom dwelling organisms and fish (e.g., carp) does not necessarily decrease P adsorption because such activity exposes unused P-binding sites in the clay material and therefore can enhance the P capping effect. P removed by Phoslock remains in the clay structure and therefore unavailable to phytoplankton.

Because of the rapid settling of Phoslock, the **time period for closing the lake** for public contact recreation is short and involves the period of application, which is in the order of days. Lakes are typically reopened within 24 hours of application for contact sport (Nigel Traill, pers. comm). Dissolved lanthanum (La^{3+}) concentration should be monitored in the water column to ensure concentrations are below 1 mg/L where no toxicity was determined on *Daphnia magna* (Lürling and Tolman 2010), before opening the lake to the public. We don't know of any guidelines of lanthanum concentration natural waters or sediment in provincial, federal or other organizations.

4.3.3 Preliminary estimates of dosage and cost

The dosage of Phoslock is based on the amount of free P (SRP) in the water column and mobile P in the sediment that is involved in release. We computed the volumetric SRP concentration from a profile of Oct 7, 2014 (Table 3) and assumed that the mobile sediment fraction is similar to that of previously treated eutrophic lakes. We present a preliminary dosage and cost analysis (Table 4) for two depths, based on a comparably high application rate of 4.6 metric tonnes/ha that was used in hyper-eutrophic Ontario Swan Lake (Nürnberg and LaZerte 2016). At this rate most of the releasable P should be intercepted by Phoslock. In comparison, 11 of 16 previous applications had lower application rates (Spears et al. 2015). For example, the application rate in Behlendorfer See was 4 t/ha of sediment below 7 m (Appendix H).

Table 3. Water column SRP and TP on Oct 7, 2014

Depth (m)	SRP (mg/L)	TP (mg/L)	Volume (10^6 m^3)
Surface - 8	0.001	0.011	13.465
8 - 10	0.003	0.015	2.285
10 - 12	0.096	0.104	1.787
12 - 14	0.870	1.260	1.000
14 - 16	0.920	1.200	0.268
16 - bottom	1.100	1.490	0.028
Volumetric average concentration	0.071	0.106	

Table 4. Dosage calculation and costs

	Depth <9m	Depth <7m
Sediment		
Area (ha):	110.78	126.44
Dose t/ha	4.6	5.6
Dose for sediment (tonnes)	509.6	708.1
Water		
Volumetric SRP concentration (mg/L)	0.071	0.071
Dose for water (tonnes)	1.2	1.2
Total Dose (tonnes)	510.82	709.29
Cost of Phoslock (Can$3,100/t)[a]	$1,583,542	$2,198,792
Application cost (Can$ 200/t)[b]	$102,164	$141,858
Total estimated costs	$1,685,706	$2,340,649

[a]Unit Phoslock cost as of 22 Sep 2015;
[b]Application cost includes "economies of scale" deduction.

If the dosage of Phoslock is sufficient with respect to mobile P fraction in Elk Lake sediment, this treatment will not require any maintenance, except the recommended limnological monitoring to determine treatment success. If the dosage is less than sufficient, a repeat treatment may be needed and the time frame can be determined once sediment fraction data are available (Section 5).

Sediment fractionation could also indicate whether the proposed dosage is exceedingly large compared to the mobile P fraction in Elk Lake sediment, so that a smaller dose may be sufficient. While completely decayed plant material would be considered in the fractionation as organic P within the mobile fraction, any extensive covering of the sediment by decaying macrophytes (Section 5.3), may require the dosage to be adjusted. Such complications could be determined when retrieving sediment for fractionation.

In general, we suggest a sufficient dose in an initial treatment rather than scheduled repeat applications, so that water quality improvement can commence immediately. Immediate improved conditions would lessen future settling and sediment enrichment and decrease conditions that would lead to future internal load. Consequently internal load could be expected to have no influence on P concentration any more (effectively be removed or equal zero) and only external load would approximately contribute about 0.012 mg/L to lake water TP concentration (Section 2). There may be a lag in reaching equilibrium conditions because of Elk Lake's long residence time, however, if hypolimnetic SRP can be successfully removed before being distributed into the mixed layer, such low predicted TP concentration should occur almost instantaneously, at least within a year. Predictions concerning the longevity of improved conditions or the necessity of a repeat treatment need more information about the releasable fraction in the sediment (Section 5). For adequate dosing and low external input a phoslock treatment should last in the order of decades, as sediment cores from 8 treated lakes out of 10 showed no SRP release up to 9 years after application (Dithmer et al. 2016).

4.3.4 Review of toxicity studies

We have reviewed about 60 studies on Phoslock, most of them peer-reviewed and published since 2008 which shows that Phoslock is a relatively new technique in lake restoration. Many of the following references are referred to elsewhere in this report. We here include some verbatim citations from toxicological studies to provide more detailed information. (Abstracts and sometimes the whole text of all published papers are available via Google Scholar on the internet.)

Toxicity to **humans** can be considered negligible or not-existent, because Phoslock® received US and Canadian NSF/ANSI Standard 60 *Certification for use in drinking water* in 2011. This certification ensures that Phoslock® applications to drinking water supply sources, at the maximum use rate specified on the product label, do not contribute contaminants that could cause adverse human health effects. NSF/ANSI Standard 60 is the US nationally recognized health effects standard for products which are used to treat drinking water.

A large amount of scientific literature is available on lanthanum toxicity to human health as lanthanum carbonate (trade name Fosrenol®)[1] is used orally to treat hyperphosphatemia in patients with chronic kidney disease who are undergoing dialysis. The medical doses far exceed those recommended in lake restoration.

As part of the permitting process prior to the use of Phoslock in Ontario toxicity tests were undertaken by MOECC on three types of sediment dwelling organisms (***Hyalella azteca, Hexagenia spp.*** and ***Chironomus dilutes***), **rainbow trout** and ***Daphnia magna*** (Ministry of Environment 2009).

> *Cited from executive Summary of 55 p. report*:" Standard water only 96-hour toxicity tests were performed with rainbow trout and 48-hour toxicity tests were performed with *Daphnia magna*. Standard sediment toxicity tests were performed using *Chironomus dilutus* (10-day exposure), *Hexagenia spp.* (21-day exposure) and *Hyalella azteca* (14-day exposure) in sediment and water collected from the Lake Simcoe watershed. Two application rates were assessed in the sediment test; the filtered reactive phosphorous (FRP) rate dictates the amount of Phoslock required to remove the FRP from the water column and the capping rate (3.4 mg/L) delivers enough Phoslock to create a 1 mm thick layer on the sediment surface. Nutrient and metal concentrations were monitored in the sediment and water for all tests. The 48-hour LC50 for *Daphnia magna* was 4.9 g/L and > 6.8 g/L Phoslock. The rainbow trout 96-hour LC50 was>13.6 g/L. No significant survival or growth impacts were observed in any of the sediment toxicity test species for either of the dose rates. It should be noted that application rates used in 2008 field trials in the Lake Simcoe watershed were 0.02 and 0.05 g/L."

[1] Stewart J., 2002. Administration of a novel phosphate binder, Fosrenol®, with food is associated with good tolerability and low systemic absorption. J. Am. Soc. Nephrol. (2002) 13, 386A

There are many peer-reviewed studies on Phoslock in general and its potential toxicity in particular. For example, a special issue in 2015-16 of the international journal "Water Research" is devoted to *Geo-Engineering in Lakes* by chemical treatments, with more than 50% of the contributions dealing with Phoslock. Some papers that specifically deal with toxicity are summarized here. They describe a general lack of toxicity of lanthanum based applications using analytical chemistry and biological tests.

It is important to differentiate between total and dissolved, i.e., bioavailable lanthanum (La) (the metal that could provide toxicity in Phoslock). Total La can be relatively large after a treatment, but presents no threat to the biota, only the free La^{3+} ion causes toxicity in Phoslock. A study on the persistence of La^{3+} concentration on 16 applications over up to 60 months found:

> "This modelling indicated that the **concentrations of La^{3+} ions** will be very low (<0.0004 mg L/) in lakes of moderately low to high alkalinity (>0.8 mEq/L)…" (Spears et al. 2013).

A laboratory study determined that the binding of phosphate in the water by Phoslock can lead to stunted growth of Daphnia (Lürling and Tolman 2010).

> "A life-history experiment with the zooplankton grazer ***Daphnia magna*** revealed that lanthanum, up to the 1000 mg l_1 tested, had no toxic effect on the animals, but only in medium without phosphorous. In the presence of phosphorous, rhabdophane … formation resulted in significant precipitation of the food algae and consequently affected life-history traits. With increasing amounts of lanthanum, in the presence of phosphate, animals remained smaller, matured later, and reproduced less, resulting in lower population growth rates. Growth rates were not affected at 33 mg/L La, but were 6% and 7% lower at 100 and 330 mg/L, respectively, and 20% lower at 1000 mg/L. A juvenile growth assay with Phoslock tested in the range 0–5000 mg/L, yielded EC50 (NOEC) values of 871 (100) and 1557 (500) mg/L Phoslock for weight and length based growth rates, respectively. The results of this study show that no major detrimental effects on Daphnia are to be expected from Phoslock or its active ingredient lanthanum when applied in eutrophication control."

The impact of a Phoslock treatment on algae assemblage composition and **macrozoobenthos** composition was studied in Laguna Niguel, California (Bishop et al. 2014).

> "Invertebrate communities were not significantly impacted following addition of Phoslock based on richness, diversity, and functional feeding groups …. The habitat score was similar in preapplication and postapplication as measured by the following: in stream cover, sediment deposition, water chemistry, and channel alteration."

Changes in the biota of a Scottish lake after a Phoslock application were recorded for the first post-treatment year (Meis et al. 2012).

> "… a significant decrease in dominant benthic macroinvertebrate groups (Chironomidae, Oligochaeta and Sphaeriidae) in summer and autumn in the first year post-application, whereas abundance of zooplankton groups did not change significantly in the first year following the application of Phoslock®. Similarly, abundance of fish (three-spined stickleback, Gasterosteus aculeatus L.) did not change significantly postapplication."

The author further speculated that the decrease in macrobenthos may have been due to the reduction in the trophic state (lower TP concentration) or changes in the sediment habitat (Phoslock's bentonite material) rather than La toxicity.

"… studies assessing the response of macroinvertebrate abundance after external load reductions find similar declines in abundance (…), so that any potential toxic effect cannot be easily separated from effects occurring during the reduction of nutrient concentrations. However, the application of Phoslock® may be comparable to loading scenarios of fine inorganic sediment which can detrimentally affect benthic macroinvertebrate abundance (…).

In a small Dutch application that included iron chloride in addition to Phoslock, **fish biomass** and therefore indirectly macrobenthos increased after treatment (Waajen et al. 2015). The biomass of most fish species remained constant with small increases in Pike. The main increase was due to increased weight in carp, a fish that was appreciated by local fishermen.

"The fish stock increased from a low 50 kg/ha before to more than 130/ha after the treatment. This increase was mostly due to adult carp reflecting the greater appreciation of fishermen (and consequently uncontrolled stocking) of the improved water quality."

In summary, toxicity concerns are low for humans and the biota. While there may be some decrease in the abundance of benthic macroinvertebrate within the first months after treatment (possibly due to smothering by the clay material), studies showed no long-term detrimental effect on fish, macrobenthos, and zooplankton. The apparent lack of toxicity in the field is supported by numerous laboratory toxicity tests. Fish kills during or after a proper (and pure) Phoslock application have not been observed in lakes with appropriate alkalinity.

4.3.5 Standards and policies by governmental agencies world-wide

Much of the information in this section is based on Finsterle (2014) and personal communication with Nigel Traill, Regional Manager - Europe, North and South America, Phoslock Water Solutions Ltd, Digital World Centre, 1 Lowry Plaza, Salford Quays, M50 3UB, Great Britain, ntraill@phoslock.com.au. Mr. Traill should be contacted directly for further details and confirmation.

North America

Phoslock® received US and Canadian NSF/ANSI Standard 60 **Certification for use in drinking water** as discussed in Section 4.3.4. There were several applications done in the **US** with permitting requirements differing by State. Phoslock conforms to ANSI Z400.1-2004 Standard (http://www.sepro.com/documents/Phoslock_MSDS.pdf).

In **Canada**, provinces have the jurisdiction and permitting agency except for lakes and water bodies on Federal Crown Land or Federal Crown Sea Bed. For these water bodies an approval from the federal Department of Fisheries and Oceans (DFO) is needed.

British Columbia: not used up to now.

Ontario: The MOECC (Ontario Ministry of Environment and Climate Change) allows Phoslock to be used without further approval, provided the attached "Standard Operating Procedures" (2010) are followed and provided the receiving water body is not located on "Crown Sea Bed". In that case, the provincial Ministry of Natural Resources and Forestry (MNRF) still needs to give a "work permit". Phoslock has been used in the Lake Simcoe watershed and in Swan Lake, Markham

(Section 4.3.1). Contact: Dan Orr, Manager, Technical Support, Central Region, MOECC <Dan.Orr@ontario.ca>

Alberta: Alberta Environment and Sustainable Resource Development (AESRD) provided a permit to one or multiple applications on Henderson Lake in Lethbridge. The City of Lethbridge intends to treat Henderson Lake in early 2016. Contact: Lethbridge Parks Manager Dave Ellis (tel: 403-320-3848), AESRD, Ron Zurawell <Ron.Zurawell@gov.ab.ca>. Besides a lake treatment, the Cities of Edmonton and Lethbridge have been using Phoslock for two years in their stormwater ponds, where no permit is required.

Manitoba: It is not certain what permit is required by Manitoba Conservation and Water Stewardship. MCWS has been interested in a potential application on Killarney Lake and contracted Phoslock for sediment fractionations. Contact: Elaine Page (Manager, Water Quality Management Section, Water Science and Management Branch, <Elaine.Page@gov.mb.ca>) and Cassie McLaine.

Quebec: It is not certain what permit is required by the Ministère du Développement durable, de l'Environnement de la Faune et des Parcs (MDDEFP). The City of Bromont is interested in a Phoslock application of Lac Bromont.
Contacts: Anne Joncas of the Lac Bromont Restoration Group, <anne_joncas@hotmail.com>; Louis Roy of MDDEFP Louis.Roy@mddefp.gouv.qc.ca

Other continents

In **Brazil**, Phoslock® has been certified by IBAMA (the Brazilian Ministry for the Environment) for import, sale and use in Brazil.

In **Australia**, it is certified by NICNAS (National Industrial Chemicals Notification and Assessment Scheme).

In **Europe**, the product can be legally imported and sold under REACH (Registration, Evaluation, Authorisation and Restriction of Chemicals) regulations.

In **Asia**, Phoslock has been used in pilot experimental studies in China, permitting information is not available to us.

5 Limitations, data gaps, and recommendations

5.1 Under-dosing

While any amount of Phoslock would have a beneficial effect by intercepting and adsorbing P released from the sediments, under-dosing would not intercept all P and cyanobacterial blooms could still be triggered during mixing events. The probability of these blooms would be expected to be lower than in untreated conditions, because they are dependent on P concentration (Downing et al. 2001). Consequently, the effect of under-dosage could be remediated by a further dose.

Under-dosing is sometimes deliberately conducted when limited resources do not allow a full application, or when a planned secondary application is targeted to those sediment areas that are still actively releasing P. In such an application the concentration of sediment fractions that are involved in internal loading are mapped and the dosage in a second application adjusted accordingly (Yasseri and Epe 2015).

By targeting the anoxic area at and below the 9 m contour, a selective treatment is proposed. It is possible that small quantities of P are released occasionally from areas at shallower depths that could contribute to internal load. Sediment P fraction analysis in these areas would help determine the potential of internal loading from such areas. But even if releasable P is present, the generally aerated conditions in the overlying water (Appendix B) should prevent P release.

5.2 Unknown or underestimated external P input

If certain external P sources were overlooked or large amounts underestimated, cyanobacteria blooms may still be supported despite the successful cessation of internal loading.

Waterfowl: Input during the critical winter mixing period could occur due to an unpredictably large abundance of waterfowl on and around Elk Lake. We did not have any information on waterfowl patterns and census data so we relied on the 1988 estimate of 9 kg/yr (Table 1). Using an egestion rate of 0.61 g/day/goose (Moore et al. 1998), 9 kg would be added by 14,754 goose days/year. In a much smaller urban lake (5.5 ha Swan Lake, Greater Toronto Area) a biweekly census between July and December determined almost 23,000 geese/year and related P input of 14 kg/yr. While the Phoslock treatment in Swan Lake was successful in decreasing TP concentration by 60% and 76 % in 2 subsequent years, beneficial effects were likely curtailed by this unexpected P input by the waterfowl (Appendix G). Although Elk Lake's size and large volume would make a major effect on its P budget unlikely, we recommend an evaluation of this potential P input including geese census especially during the critical winter period.

Agriculture: There is a fair amount of agricultural area to the north and west of Elk Lake that is mainly drained by O'Donnel Creek and Linnet Creek, respectively (Figure 2). These creeks do not have large flow rates and their TP loads are estimated to be low (Table 1). However, we recommend implementing best management practices (BMPs) throughout, so that the generally elevated P concentrations are decreased (Appendix A). Continued monitoring of O'Donnel Creek's TP and SRP concentration, especially during higher flow period Nov-Mar is recommended. A flow gauge would help determining the actual load. Such information would be important when evaluating any effects of a potential lake treatment.

Septic systems: Evaluation of the existing septic systems and continued inspection is recommended. But because of the relative low density of development, their influences should be minor as evidenced by the revised septic input calculations in Section 2.

Direct runoff: There are several developed areas especially on the east side, including the Patricia Bay Highway, the Lakeview Christian School and other roads and walkways (Figure 2). BMPs including slope and turf management are recommended to minimize the influence of nutrients and pollutants.

To summarize, we are most concerned about waterfowl input that might delay treatment effects. While other input from external sources should be minimized, they appear to be limited because of the small catchment basin. Nonetheless, we recommend to carefully evaluate and implement any practices that would decrease TP input from external sources.

5.3 Macrophytes

The mandate and intention of this study has been to address concerns over cyanobacteria blooms in Elk Lake (proper), but lake users also felt discomfort with the amount of macrophytes (water weeds) that grow in the Elk/Beaver Lake system, so that we want to address this concern here. Beaver Lake by itself is apparently close to 100% covered by water plants during the growing season (May–Oct, MOE Staff, pers. comm.). Since the study goal here is not the remediation of Beaver Lake we are not discussing this problem further. In comparison, it seems that less than 35% of Elk Lake proper offers habitat for plants. Macrophytes thrive in Elk Lake at a depth of 0-6 m, which presents 35% of the Elk Lake proper surface area (Nordin 2015) [2].

Generally the occurrence of macrophytes is necessary to support the fisheries as water plants provide protected regions for recruitment or "nurseries", and 25% coverage has been named as ideal. Further, the establishment and persistence of macrophytes in freshwater environments provide important ecosystem services, including (1) improving water quality by decreasing TP concentration and turbidity; and (2) stabilizing sediments, reducing sediment resuspension, and erosion (Madsen et al. 2001). Especially in shallow lakes, the prevention of elevated lake TP concentration associated with sediment resuspension and turbidity has been described repeatedly. Consequently in eutrophic and hypereutrophic lakes that are phytoplankton-dominated, restoration efforts include the re-establishment of macrophytes to decrease the frequency of cyanobacterial blooms (Cooke et al. 2005).

It has sometimes been suggested that macrophyte contribute to internal P loading when they decay. While this mechanism occurs, there is no consistent evidence that there is a net release of P when considering the growth circle throughout the year. It is generally accepted that macrophytes remove inorganic P from the sediment when growing but return P to the organic sediment pool when decaying, which then eventually can be released again after diagenesis (Kim et al. 2013).

Because of the limnological benefits of macrophytes, we do not deem it necessary nor advisable to address them in the overall restoration effort of Elk Lake proper at this time. Instead we recommend for now, the provision of important access routes and boat channels, even swimming beach creation, by using geodesic membranes in limited areas. Trophic state changes may also decrease macrophyte growth, and we don't expect macrophyte habitat to increase below the current 6 m depth contour.

With respect to a potential Phoslock treatment, the existing macrophytes should not create any interference. An application during the low growth period is probably preferable so that the material can settle onto the decayed plant remnants on the sediment and intercept any P liberation. If sediment monitoring (for example, by taking cores for sediment fractionation, Section 5.5) should reveal large amount of decaying plant material at locations below 8 m, possible additional P release from decaying plant material should be considered in dosage calculations.

It is not expected that the macrophyte abundance would drastically increase in Elk Lake after a Phoslock treatment as has been observed in more enriched lakes (Spears et al. 2015). These lakes were shallow (mean depth 0.8-2.8 m) and/or were eutrophic with much larger TP concentrations

[2]Rick Nordin, pers. comm. 21 Nov 2015: The littoral zone of the lake (depth less than 6 m) comprises a large proportion of the lake - about 45% of the lake surface area of Elk Lake and Beaver Lake combined and about 35% if the Elk Lake itself (the main basin) is considered. Beaver Lake / Bay is virtually all littoral (less than 6 m). If the channel is included and dependent where the boundary between the two water bodies is drawn, the littoral for Elk could be less than 35%.

and turbidity than Elk Lake so that they were light-limited before the treatment. Phoslock treatment in these enriched lakes improved general water quality (decreased TP, algal biomass, and turbidity) so that the changed conditions allowed the desired growth of submergent macrophytes (see also hyper-eutrophic Behlendorffersee, Appendix H), instead of cyanobacteria.

5.4 Influence on downstream Beaver Lake

Because of no direct contact of Phoslock with downstream waters (Section 4.3.2), there is no fear of potential toxicity (Section 4.3.4) transfer. On the other hand, the reduction of TP and especially the biologically available SRP fraction, and the possible reduction of cyanobacterial cells would positively affect downstream waters.

Therefore, when treating Elk Lake proper, downstream Beaver Lake would be beneficially effected as well. Beaver Lake experiences blooms when Elk Lake does not (Figure 1) and we assume that there is no “back flow” from downstream Beaver to Elk Lake. This assumption seems plausible, because the interconnecting channel is only about 3 m deep and, more convincingly, epilimnetic TP concentration at the Elk Lake main station is much lower than that in Beaver Lake. Nonetheless, we suggest this pattern to be confirmed by a hydrologist.

5.5 Monitoring for preferred treatment

As evident from this report, most of the necessary information for an informed decision about treatment choice in general and a Phoslock treatment in particular, is available. Furthermore, most of the pre-monitoring suggestions in the *Standard Operating Procedures for Phoslock Applications* (SOP) in Ontario (Lake Simcoe Region Conservation Authority 2010b) are fulfilled (Appendix I).

However, there is an important lack of information involving the P fractions in the bottom sediments that can potentially be released under anoxic conditions. The distribution of the mobile P fraction that is involved in release (Section 4.3) throughout Elk Lake would ideally be determined in a quantitative matter[3]. A map of mobile sediment P fractions including several locations on at least 2 trajectories between 7 m and the deep main sampling location for (at least one depth) 0-5 cm bottom sediment would help determine the most efficient application. It would help refine the recommendation for the exact water depth of the application and would improve the determination of an exact dose.

When retrieving sediment for fractionation we recommend noting any extensive covering of the sediment by decaying macrophytes (Section 5.3), for potential dosage adjustment.

Lanthanum concentration before and after the treatment should be determined. The occasional measurement of total and dissolved lanthanum (TLa and DLa) after application is recommended to trace the applied chemical. TLa could serve as tracer of the Phoslock^{TM} material, and DLa can serve as approximation of free lanthanum. DLa may overestimate “free” La, however, because it is possible that LaP rhabdophane particles evade separation by the standard 0.45 µ pore size filters (Davies 2011).

[3] For this analysis the City of Lethbridge, AB and Manitoba Conservation and Water Stewardship hired the Institut Dr. Nowak in Germany http://www.limnowak.com. See also: (Yasseri and Epe 2015)

The post-treatment sampling program should include the monthly analysis of TP, TN, chlorophyll, Secchi transparency and phytoplankton species and, if toxin-producing algae are identified, microcystin analysis for samples of the mixed water layer (approximately 0.5 - 2 m). To determine changes in internal P load, monthly profiles of TP (perhaps at 5 depths intervals) are suggested. Temperature and dissolved oxygen profiles should be established every month. A post-treatment evaluation program typically includes at least 3 years of monthly monitoring. Throughout the post-application period it may be useful to have limnologists evaluate treatment success with a final report after three years. At this point any needs for other or repeated measures could be evaluated.

Costs could be minimized by using local, trained volunteers in combination with licensed analytical laboratories. For example, Secchi transparencies, temperature and oxygen profiles, and water sampling could be accomplished that way as long as supervision by limnological professionals (MOE or consultants) is warranted. Otherwise, local limnological consulting firms could be hired.

6 Conclusion

The high internal P loading from bottom sediments compared to loading from external sources most likely causes cyanobacteria blooms in Elk Lake. Because of various shortcomings of commonly used restoration techniques (destratification, mixing, aeration, and oxygenation), we recommend a sediment treatment with a clay-based P-binding material, lanthanum-modified bentonite. This material, called Phoslock, has been applied world-wide and proven to be non-toxic.

Elk Lake’s low external compared to internal loading rate, its low summer flow rate, and high alkalinity and low dissolved organic acids of the lake water are favourable for a Phoslock treatment. We present a preliminary dose and cost analysis based on a comparably high application rate. Total costs for such a dose are estimated as Can$ 1,686,000 (based on Phoslock supplier unit cost estimates of 22 Sep, 2015). We believe that such an application would result in the inhibition of almost all internal load from escaping the bottom sediment. We can estimate a lowering of the annual average lake water concentration to 12 µg/L at equilibrium conditions although other influences including bottom disturbing fish (carp) and decaying macrophytes may prevent reaching this theoretical level. We expect that without the internal loading during the summer and fall months, cyanobacteria will not have enough nutrients to proliferate. Because Elk Lake is relatively nutrient poor, we recommend confirming the releasable amount of P by determining the mobile P concentration in the bottom of Elk Lake sediment and the extent of decaying plant material on the bottom.

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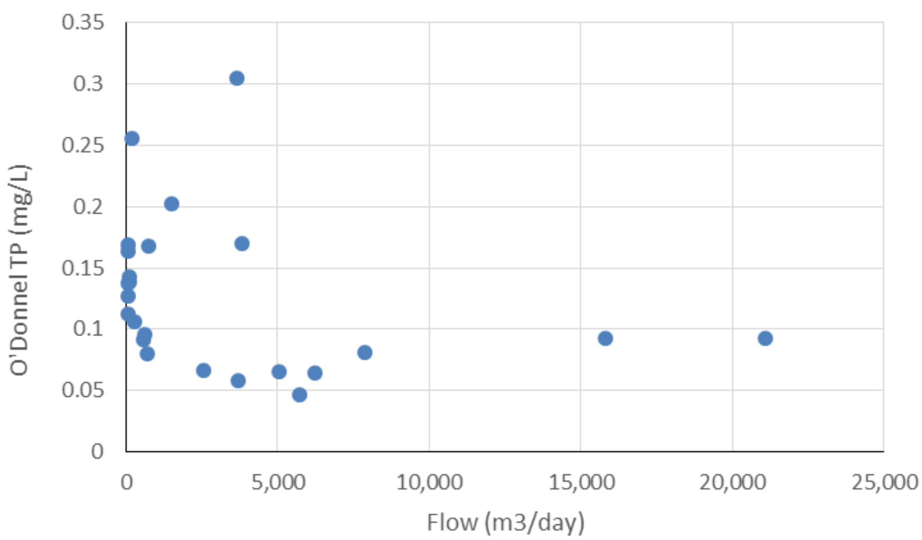
Appendix A. O'Donnel Creek

Load from the main inflow, O'Donnel Creek, was computed as the sum of concentration averages for two periods with distinct concentrations and flows (Table 5, see pattern in Figure 4). It is also obvious that high TP concentration occur predominately at low flows and vice versa (Figure 12). This means that winter concentrations in the lake are less likely to be severely affected by input from O'Donnel Creek.

Table 5. Computation of O'Donnel Creek input 2014-15.

Period	Days/Period	Flow		TP	
		(m^3/day)	(10^6 m^3/period)	Concentration (mg/L)	Mass (kg/period)
May-Dec 2014	244	1,221	0.298	0.154	46
Jan-Apr 2015	121	6,130	0.742	0.071	52
May 2014 - Apr 2015 (Sum)	365		1.040		98.4

Figure 12. O'Donnel Creek TP concentration versus daily flows 2014-15.



Appendix B. Anoxic Factor

The anoxic factor was computed from the following equation (Nürnberg 1995):

$$AF = \sum_{i=1}^{n} \frac{t_i \times a_i}{A_o}$$

where t_i, the period of anoxia (days), a_i, the corresponding area (m^2), A_o, lake surface area (m^2), and n, numbers of periods with different oxycline depths. (A threshold of 2.1 mg/L was used for the DO profiles determined by *in situ* probe measurements, considering that the sediment surface likely is anoxic when the oxygen probe measures a small amount of DO in the overlaying water.)

	Date-beg	Date-end	<2mg/L Depth (m)	Days	Area (m^2)	A-Factor* (d/period)	AF (d/year)
1988	21-Jun	01-Jul	10	10	1,008,938	5.41	
	01-Jul	12-Aug	8	42	1,188,966	26.76	
	12-Aug	15-Sep	9	34	1,107,821	20.19	
	15-Sep	30-Sep	10	15	1,008,938	8.11	60.5
2014	05-Jun	05-Jul	13	30	586,592	9.43	
	05-Jul	20-Jul	9	15	1,107,821	8.91	
	20-Jul	20-Aug	8	31	1,188,966	19.75	
	20-Aug	10-Sep	9	21	1,107,821	12.47	
	10-Sep	01-Oct	8	21	1,188,966	13.38	
	01-Oct	14-Oct	9	13	1,107,821	7.72	
	14-Oct	25-Oct	12	11	803,352	4.74	
	25-Oct	11-Nov	10	17	1,008,938	9.19	
	11-Nov	20-Nov	16	9	40,223	0.19	85.8

*Computed as: ***Days x Area/Elk Lake (proper) surface area*** of 1,866,000 m^2

Appendix C. Morphometry

Table 6. Morphometric information of the Elk-Beaver Lakes combined (Source: MOE)

Depth (m)	Area (m^2)	Interval Volume (10^6 m^3)
0	2,458,543.60	2.4046
1	2,291,510.40	2.1945
2	2,077,010.40	1.9112
3	1,783,642.20	1.6521
4	1,505,536.40	1.4598
5	1,404,973.60	1.3395
6	1,332,603.50	1.2728
7	1,264,402.50	1.2306
8	1,188,966.20	1.1821
9	1,107,821.20	1.1027
10	1,008,938.50	0.9784
11	901,676.00	0.8085
12	803,352.00	0.6058
13	586,592.20	0.3942
14	168,715.10	0.2039
15	83,798.90	0.0645
16	40,223.50	0
17	18,435.80	0
18	2,793.30	0.0283
19.4	0	0

Appendix D. 2014-15 TP and SRP concentrations, *used for in situ internal load

Date	Depth (m)	SRP (mg/L)	TP (mg/L)	V (10^6 m^3)	TP avg water column
17-Feb-14	0.5		**0.030**	18.834	0.033*
17-Feb-14	10		**0.037**		
17-Feb-14	16		**0.032**		
25-Mar-14			**0.022**		
15-Apr-14	0.5	0.001	**0.011**		
15-Apr-14	10	0.029	**0.041**		
15-Apr-14	17.5	0.038	**0.048**		
13-May-14	0.5	0.001	**0.011**		
13-May-14	10	0.035	**0.055**		
13-May-14	17	0.170	**0.192**		
10-Jun-14	0.5	0.001	**0.016**		
10-Jun-14	10	0.026	**0.047**		
10-Jun-14	16	0.490	**0.570**		
08-Jul-14	0.5	0.001	**0.009**		
08-Jul-14	8	0.002	**0.019**		
08-Jul-14	10	0.089	**0.136**		
08-Jul-14	12	0.410	**0.716**		
08-Jul-14	14	0.410	**0.687**		
06-Aug-14	0.5	0.001	**0.009**	12.235	
06-Aug-14	7		**0.030**	1.231	*(estimated)*
06-Aug-14	8	0.004	**0.052**	2.285	
06-Aug-14	10	0.290	**0.468**	1.787	
06-Aug-14	12	0.440	**0.931**	1.000	
06-Aug-14	14	0.460	**1.120**	0.268	
06-Aug-14	16	0.430	**1.190**	0.028	0.126*
10-Sep-14	0.5	0.001	**0.008**		
10-Sep-14	10	0.170	**0.256**		
10-Sep-14	13.5	0.430	**1.370**		
07-Oct-14	0.5	0.001	**0.011**	13.465	
07-Oct-14	8	0.003	**0.015**	2.285	
07-Oct-14	10	0.096	**0.104**	1.787	
07-Oct-14	12	0.870	**1.260**	1.000	
07-Oct-14	14	0.920	**1.200**	0.268	
07-Oct-14	16	1.100	**1.490**	0.028	0.106
04-Nov-14	0.5	0.005	**0.022**		
04-Nov-14	10	0.009	**0.025**		
04-Nov-14	15	0.170	**1.450**		
25-Nov-14	0.5	0.027	**0.046**		
25-Nov-14	16	0.027	**0.071**		
25-Nov-14	10	0.028	**0.047**		
23-Dec-14	0.5	0.029	**0.036**		
23-Dec-14	16	0.029	**0.030**		
23-Dec-14	0.1	0.028	**0.029**		
06-Jan-15	0.5	0.027	**0.045**		
06-Jan-15	10	0.027	**0.041**		
06-Jan-15	16	0.028	**0.041**		
20-Jan-15	0.5	0.023	**0.039**		
20-Jan-15	10	0.024	**0.041**		
20-Jan-15	17	0.029	**0.053**		
03-Feb-15	0.5	0.018	**0.018**	13.465	
03-Feb-15	10	0.020	**0.021**	4.072	
03-Feb-15	15	0.034	**0.038**	1.297	0.020*
16-Feb-15	0.5	0.007	**0.008**	13.465	
16-Feb-15	10	0.025	**0.028**	5.340	
16-Feb-15	17	0.071	**0.072**	0.028	0.014
17-Aug-15	0.5	<0.0010	**0.005**	6.510	
17-Aug-15	3	<0.0010	**0.006**	4.451	
17-Aug-15	6	0.001	**0.010**	2.503	
17-Aug-15	8	0.001	**0.013**	2.285	
17-Aug-15	10	0.059	**0.091**	1.787	
17-Aug-15	12	0.541	**1.320**	1.268	
17-Aug-15	16	0.560	**1.560**	0.028	0.104*

Appendix E: Calculations for internal loads (RR x AF)

In this approach, summer internal load was determined as the product of release rate (RR) and anoxic factor according to Equation 1 (Nürnberg 2009).

$$L_{int} = RR \times AF \quad (1)$$

AF was determined according to Appendix B.

RR was predicted from 0-5 cm sediment TP concentration (TP_{sed}) of the main deep location according to Equation 2 (log, logarithm to base of 10) (Nürnberg 1988).

$$\text{Log}\,(RR) = 0.8 + 0.76 \log\,(TP_{sed}), \quad (2)$$

where TP_{sed} 1.44 mg/g dry weight sampled on Nov 24, 2001, average of the 0-5 cm average sediment depth.

Assuming a settling rate of 0.5 cm/yr, this RR therefore represents sediment accumulated during approximately 10 years before sampling.

Appendix F: Hypolimnetic aeration

Because of the long history of aeration treatment in BC lakes, a more in depth account for our reasoning follows. Much of this information is based on an exchange with Ken Ashley, who has been involved with several local applications and is one of the most recognized proponents worldwide.

We here discuss two recent applications of hypolimnetic aeration in the vicinity, St Mary's Lake on Saltspring Island (2009-20012) and Langford Lake on southern Vancouver Island (2012-2014). Theses lakes have similar limnological characteristics as Elk Lake (Table 7) and experience collected from those applications could inform possible treatment of Elk Lake. In both cases full lift units (as opposed to partial lift or a Speece Cone) were installed. A detailed list of costs provided by staff members of the operators describes a total of $282,251 for Langford Lake and of $640,745 for St. Mary's Lake, which required 2 units. Because of the larger size, about 3 units would be needed in Elk Lake for a total cost of about $900,000 (pro-rated costs, Table 8). In addition, maintenance and energy costs would have to be added for each year of operation. Pro-rated costs would be about $63,000 per year, so that a total pro-rated cost for 10 years of treatment (at earlier rates and without repair costs) would be about $1,530,000 (Table 8).

Observations by stakeholders at St. Mary's Lake include sediment disturbances during aeration and cormorant nesting (and fouling) on the aeration units creating a general dissatisfaction of the efficiency of the treatment (personal communication of staff of North Salt Spring Waterworks District and St Mary Lake Integrated Watershed Management Plan, 2015).

In both lakes the aeration treatment was conducted for at least 4 years. While we don't have access to detailed limnological records, it appears that the goal of elevated hypolimnetic DO had been attained and the DO concentration decreased only to 4-6 mg/L and rarely dropped below 2 mg/L, the threshold when in untreated stratified lakes, sediment surfaces can be expected to be anoxic and release P (Nürnberg 2009). Before aeration these lakes exhibited anoxic conditions (down to 0 mg/L DO) so that internal loading from anoxic sediment surfaces was large, as determined by previous studies (Table 7).

Despite elevated DO concentrations apparently cyanobacterial blooms did not permanently disappear. It is possible that the lake bottom was still releasing phosphate as was the case in hyper-eutrophic Swiss Lake during oxygenation treatment (Gächter and Wehrli 1998). Some available measurements in St. Mary's Lake indicate a general decrease of TP concentration in the mixed upper layer and the bottom water in the initial years of treatment, but there are no volume-weighted water column average TP concentrations available to us to confirm a continued decrease in internal loading. Decreased hypolimnetic TP concentration during aeration could be an artifact due to relocation (mixing throughout the aerated hypolimnion) of sediment released P, rather than of ceased sediment release.

Whatever the mechanism, the water quality development during these applications do not provide an incentive for using the treatment in Elk Lake.

Table 7. Comparison of lake characteristics between Elk Lake and aeration treated Langford and St Mary's Lake.

Characteristics	Elk proper	Langford	St. Mary's
Study years before treatment		2008-09	1979-81
Watershed Area, Ad (km²)	8.59	3.3	5.25
Surface Area, Ao (km²)	1.87	0.612	1.82
Maximum Depth, zmax (m)	18.0	17	17
Mean Depth, z (m)	9.20	6.5	8.8
Volume (10^6 m³)	17.17	3.98	16.02
Phytoplankton	Cyanobacteria	Cyanobacteria	Cyanobacteria
Internal load (% of total load)	86-89%	69%	71%
Bottom TP before	Elevated	Elevated	Elevated
Bottom DO before	Low	Low	Low
Hypolimnetic aeration	-	2012-2015	2009-2012
Source	*Nordin 2015, This study*	*Murray 2010*	*Nordin et al. 1983*

Table 8. Costs (minimum) of aeration treatment in Langford and St Mary's Lake and pro-rated probable costs for Elk Lake.

Aeration related costs	Langford	St. Mary's	Elk proper (prorated)	
Number of units	1	2	3 x Langford	1.5 x St. Mary's
Year installed	*2012*	*2013*		
Capital costs*	$282,251	$640,745	$846,753	$961,117
Maintenance costs per year				
not included: Repair, limnological analysis, boat storage or transportation, extra staff time				
Power**	$12,500	$15,000		
Operator (required daily checks)		$11,250		
Compressor maintenance		$4,500		
Inspection (dive team)	$1,890	$8,000		
other	$5,000	$6,136		
Total	$19,390	$44,886	$58,170	$67,329
10 year lifespan	$193,900	$448,860	$581,700	$673,290
Total minimum costs for 10 years of operation	**$476,151**	**$1,089,605**	**$1,428,453**	**$1,634,407**

* Capital cost includes the replacement of the riser pipe insulation in the St. Mary's Lake application

**Power costs: Langford at 2015 rates; St. Mary's at 2013 rates

Source: personal communication of staff City of Langford and North Salt Spring Waterworks District (NSSWD).

Appendix G. Case study: Phoslock application to Swan Lake, Markham, Greater Toronto, Ontario

Abstract (Nürnberg and LaZerte 2016):

Urban lakes are important assets to highly populated regions. However, extensive usage and other influences degrade their water quality, which then requires rehabilitation and maintenance. Hyper-eutrophic Swan Lake, Greater Toronto, Canada (5.5 ha, 4.4 m maximum depth) was a gravel pit that became degraded by elevated total phosphorus (TP) concentrations mostly from internal P sources. Because Swan is a terminal lake with limited flushing and small external load, a phosphate adsorbing and sediment capping agent, lanthanum-modified bentonite (Phoslock), was applied in spring 2013 to intercept the internal load. Average TP concentration decreased from 0.247 mg/L to 0.099 mg/L in the first and 0.060 mg/L in the second post-treatment year. A TP mass balance model adequately predicted post-treatment annual average TP concentration by not including the pre-treatment internal load estimate of 650 to 1,100 mg/m^2/yr. Phytoplankton biomass decreased only in the second post-treatment year, when Secchi transparency (highly correlated with chlorophyll concentration) increased to a growing season average of 1.4 m (range 0.7-2.7) compared to 0.5 m (0.37-0.63) before treatment. We explain the lack of response in the first treatment year with a relatively late application (29 April - 1 May 2013), when P released from the winter bottom sediments had already been taken up by phytoplankton. Recently, a growing population of waterfowl (mostly Canada geese, *Branta canadensis*) were the highest contributors of nutrients (75%), as indicated by a mass balance based on literature-derived goose P export and bi-weekly bird census. We recommend waterfowl management or repeated treatment to prevent further deterioration.

Appendix H. Case study: Phoslock application to Behlendorffersee

Extracted from Overview of Phoslock Properties and its Use in Aquatic Environments (compiled by K. Finsterle, October 2014, data courtesy of Institute Dr. Nowak, Germany).

5.1 BEHLENDORFER SEE, GERMANY

The largest application of Phoslock® (by volume of product) to date took place in December 2009 on the Behlendorfer See near the town of Ratzeburg in Northern Germany. The Behlendorfer See is a 63 ha lake with a stable thermal stratification during summer months, a maximum depth of 15 m and an average depth of 6.20 m. The lake is frequently used for recreational purposes and is situated in an intensive farmed agricultural area. For many years the Behlendorfer See had received high inputs of nutrients. The naturally clear water conditions, typical for an oligotrophic water body, had switched to a turbid water state as a result of the regular summer blue green algal blooms and submerged macrophytes were poorly developed. As a result, the lake did not meet the ecological criteria outlined under the European Water Framework Directive. An application of 214 tonnes of Phoslock® (marketed as Bentophos® in Germany) was undertaken on the lake in December 2009 in order to permanently bind phosphate released from the sediment and break the lake's phytoplankton cycle. The material was applied to a surface area of the lake of only approximately 40 ha, which corresponded to the area of the lake which is deeper than 7m. The aim of the application was to remove 550 kg of phosphorus from the water column and 1590 kg of immediately and potentially releasable phosphorus from the upper 5 cm layer of the lake sediments.

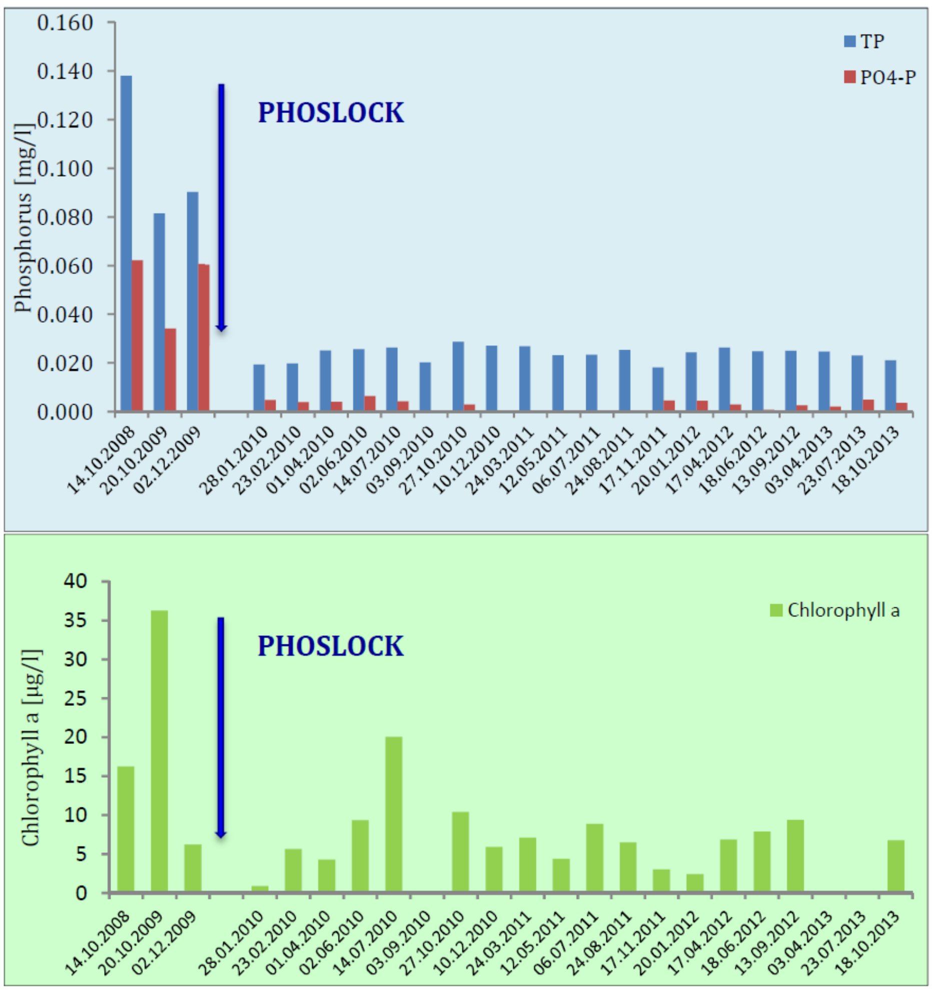


Figure 9. Phosphorus, phosphate and chlorophyll a concentrations measured in the Behlendorfer See before and after Phoslock® application in December 2009.

Since the application, the lake has been regularly monitored by the regional environmental agency, LLUR (Landesamt für Landwirtschaft, Umwelt und Ländliche Räume, the Schleswig-Holstein - Agency for Agriculture, Environment and Forestry) and the limnological institute IDN (Insitute Dr Nowak, Ottersberg). The monitoring data collected since the application indicate that phosphorus and phosphate concentrations have dropped substantially (Figure 9). Phosphorus release from the sediment has ceased and in the summer seasons following the application the summer algal biomass has been considerably lower and the Secchi depth higher. The improved light penetration in the lake has resulted in increased growth of submerged macrophytes.

Appendix I. *Standard Operating Procedures for Phoslock Applications* (SOP) in Ontario (Lake Simcoe Region Conservation Authority 2010b)

Click here for entire SOP report (pdf file)

This document was put together by Phoslock Staff and the "Phoslock Steering Committee" in 2010. While it provides many useful suggestions and guidelines, more rigorous scientific involvement since then suggests to include methods that were not mentioned. In particular, the sediment fractions of “mobile” P are now suggested to be analyzed and used for the determination of the dose and most effective areal distribution of Phoslock.

Example of suggested needed information before the treatment (Section 3.3.1 of SOP)

- Chemical data (for water and sediment):
    a. FRP
    b. TP
    c. Alkalinity
    d. Nitrogen
    e. Total and dissolved lanthanum
- Physical water data:
    a. pH
    b. DO
    c. salinity
    d. redox
    e. water temp
    f. turbidity
- Biological data:
    Phytoplankton data
- Physical data:
    a. Volume of water body
    b. Inflows
    c. Depth
    d. Hydrology
    e. Catchment information