

Characterization of Phosphorus Sources in the Mattatall Lake Watershed

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1.0 Introduction

1.1 Project Context and Background Information

This study focused on eutrophication processes in Mattatall Lake (45°41'45"N; 63°28'18"W); which is situated on the borders of Cumberland and Colchester counties in Nova Scotia. Hereafter, Mattatall Lake will be interchangeable in terminology with “the Lake.” This study was initiated in response to the observed algae blooms in Mattatall Lake that occurred in 2014, 2015, and 2016. Cyanobacteria species, in particular *Anabaena plantonica*, were the dominant group during these blooms (Nguyen-Quang, *et al.*, 2016; Hushchyna & Nguyen-Quang, 2016). This particular cyanobacteria species has the potential to produce toxins which could be harmful to humans and wildlife.

The occurrence of harmful algae blooms (HAB) and eutrophication within freshwater systems is strongly linked with phosphorus (P) concentrations (Conley *et al.*, 2009). In November 2016, the Centre for Water Resources Studies (CWRS) initiated a study to characterize sources of P in the Mattatall Lake Watershed. A P loading model was developed and used to better understand sources of phosphorus in Mattatall Lake. A lake water and sediment sampling program was conducted throughout the winter and on an approximate bi-weekly basis throughout the summer and fall of 2017. The P loading model was validated with water quality data collected during the study period of 2016-2017. In this section of the report some key limnological concepts are first reviewed.

1.1.1. Lake Thermal Regimes and Mixing Processes

Heat input to a water body is mainly from solar radiation. Other mechanisms include heat contained in surface and groundwater inputs. Nova Scotia lakes of sufficient depth are considered to be dimictic if they undergo complete thermal mixing (turnover) at the end of the summer (Figure 1) and winter stratification periods. During the summer stratification, a water body is seen to develop three distinct thermal zones (Figure 2). The upper zone, the epilimnion, circulates freely within itself, and is separated from the two zones below. Water contained in the bottom zone, the hypolimnion, also circulates within itself and resists mixing with overlying waters mainly due to differences in thermal density. The intermediate zone, the metalimnion (thermocline), is relatively thin compared to the other two zones and is characterized by temperature changes of greater than 1°C per metre. Thicknesses of each zone vary during the heating season—from lake to lake—governed by weather conditions.

One of the factors affecting growth and distribution of phytoplankton (algae) through a water column is light. The zone of suitable light intensity for photosynthetic activity ($\geq 1\%$ of the surface light intensity) is referred to as the euphotic zone. One approximation of this zone has been defined as being 2.5 times the Secchi depth (Vollenweider and Kerekes, 1982).



Figure 1. Completely mixed lake after turnover example schematic (Vertec Water Features, 2017).

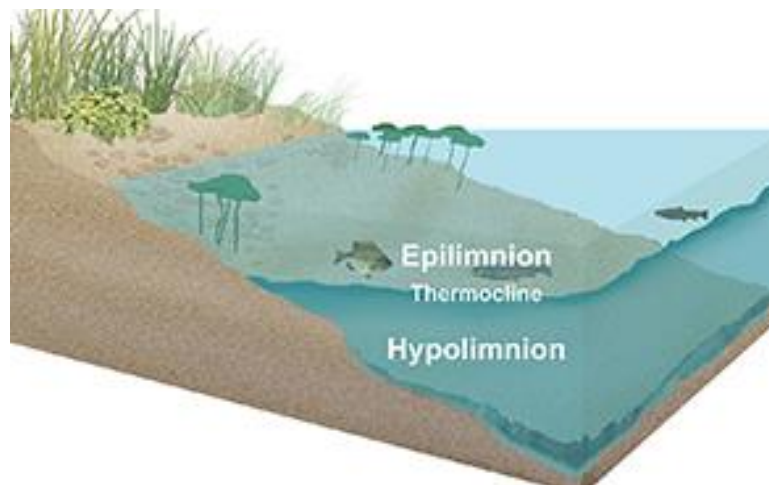


Figure 2. Stratified lake during the summer example schematic (Vertec Water Features, 2017).

1.1.2. Trophic State Definition

The trophic state of a lake is an important indicator of lake health and it can be defined traditionally by the nutrient and carbon availability for primary production (Dodds & Cole, 2007). Several systems have been developed for classification of lakes into trophic categories according to nutrient concentrations and algal biomass (Dodds *et al.*, 1998). The trophic state of water bodies are described as oligotrophic, mesotrophic, and eutrophic; in order of increasing algae and nutrients and variability in oxygen availability. Oligotrophic lakes are characterized by low nutrients, low algal biomass, high clarity, and deep photic zones. Contrastingly, eutrophic lakes may often be characterized by frequent cyanobacterial blooms, high nutrient concentrations, depletion of oxygen (anoxia), and pH elevated above neutral conditions (Dodds *et al.*, 1998). Naturally, the process of change from an oligotrophic to an eutrophic ecosystem might take thousands of years in an undisturbed lake. However, the conversion process may be greatly accelerated when inputs of nutrients increase due to anthropogenic activities. This is referred to as cultural eutrophication.

A key negative outcome of eutrophication is development of anoxic zones and proliferation of HAB (Conley *et al.*, 2009).

The Organization for Economic Co-operation and Development (OECD), conducted a large scale research program in the 1960s related to eutrophication, with a specific focus on the role of nutrients. The focus was on quantification of the relationships between chlorophyll *a*, total phosphorus (TP), total nitrogen (TN), and Secchi depth in lakes of varying trophic status. The Canadian supplementary report (Janus & Vollenweider, 1982) compared data collected from a suite of Canadian lakes to relationships developed in the original OECD studies. The Canadian supplementary report found that for the 58 lakes examined, the relationship between TP and chlorophyll *a* was similar to those created from the original OECD dataset of 110 lakes. This led to the development of trophic state trigger ranges based on TP concentrations. The phosphorus-based trophic state classification scheme developed by the OECD has been widely adopted in Canada (CCME, 2004). Trophic trigger ranges are summarized in Table 1.

Table 1. Summary of trophic state trigger ranges.

Trophic status	TP ($\mu\text{g L}^{-1}$)	chlorophyll <i>a</i> ($\mu\text{g L}^{-1}$)		Secchi Depth (m)	
		Mean	Max.	Mean	Max.
Source	CCME (2004)	Galvez <i>et al</i> (2007)	Galvez <i>et al</i> (2007)	Vollenweider and Kerekes (1982)	Vollenweider and Kerekes (1982)
Ultra-oligotrophic	< 4	< 1.0	< 2.5	≥ 12	≥ 6
Oligotrophic	4-10	< 2.5	< 8	≥ 6	≥ 3
Mesotrophic	10-20	2.5-8	8-25	6-3	3-1.5
Meso-eutrophic	20-35	--	--	--	--
Eutrophic	30-100	8-25	25-75	3-1.5	1.5-0.7
Hypereutrophic	> 100	> 25	> 75	≤ 1.5	≤ 0.7

Chlorophyll *a*, a primary photosynthetic pigment, is also a widely used trophic state indicator, as it has been shown that chlorophyll *a* levels can provide a reliable characterization of algal biomass (Lyche-Solheim, 2013). The primary disadvantages associated with the use of chlorophyll *a* as a trophic state indicator are the sampling and analytical requirements. Characterization of the mean chlorophyll *a* concentration in a lake requires a high sampling frequency, both temporally and spatially, as phytoplankton populations vary both in space and time. Secchi depth has also been used as a trophic state indicator. It is an empirical measurement that is based on the visual disappearance/reappearance of a physical black and white quadrant disk as it is lowered/raised in the water column. It has been shown to be well correlated to water clarity and trophic state in clear water lakes (through the increased absorption of light with increasing phytoplankton population). However, measurement of Secchi depth is operator-dependent and subjective, and it is influenced by water colour.

1.1.3. Carlson Trophic State Index

In addition to the fixed-boundary classification approach provided by the CCME (2004), the Carlson index relates TP, Secchi depth, and chlorophyll *a*, to trophic state (Equations 1-3) (Carlson, 1983). These three parameters were chosen because P is generally the limiting nutrient in freshwater systems, and Secchi depth and chlorophyll *a* are surrogates of primary production. The Trophic State Index (TSI) is a continuous scale from 0 to 100. The TSI is determined separately for each parameter; after which, the trophic state is identified by an assessment of the three TSI values (Table 2). The TSI values are not intended to be averaged. There is generally good agreement between the calculated TSI of these three parameters. However, the TSI values do not always agree, nor are they indicative of trophic state in all water bodies. Lakes with unique morphology, nutrient limitations, and/or high color may have TSI values not reflective of the trophic state. Equations used to calculate the indices are provided below (Carlson, 1983):

$$TSI(SD) = 60 - 14.41 \ln(SD, m) \quad [\text{Eq. 1}]$$

$$TSI(Chla) = 9.81 \ln(Chla, \mu g \cdot L^{-1}) + 30.6 \quad [\text{Eq. 2}]$$

$$TSI(TP) = 14.42 \ln(TP, \mu g \cdot L^{-1}) + 4.15 \quad [\text{Eq. 3}]$$

Where SD is Secchi depth, and Chla is chlorophyll *a*.

Table 2. A list of lake characteristics associated with Trophic State Index (TSI) values (Carlson 1977).

TSI	Chla ($\mu\text{g}\cdot\text{L}^{-1}$)	SD (m)	TP ($\mu\text{g}\cdot\text{L}^{-1}$)	Attributes	Water Supply	Fisheries & Recreation
<30	<0.95	>8	<6	Oligotrophy: Clear water, oxygen throughout the year in the hypolimnion.	Water may be suitable for an unfiltered water supply.	Salmonid fisheries dominate.
30-40	0.95-2.6	8-4	6-12	Hypolimnia of shallower lakes may become anoxic.		Salmonid fisheries in deep lakes only.
40-50	2.6-7.3	4-2	12-24	Mesotrophy: Water moderately clear; increasing probability of hypolimnetic anoxia during summer.	Iron, manganese, taste, and odor problems worsen. Raw water turbidity requires filtration.	Hypolimnetic anoxia results in loss of salmonids. Walleye may predominate.
50-60	7.3-20	2-1	24-48	Eutrophy: Anoxic hypolimnia, macrophyte problems possible.		Warm-water fisheries only. Bass may dominate.
60-70	20-56	0.5-1	48-96	Blue-green algae dominate, algal scums and macrophyte problems.	Episodes of severe taste and odor possible.	Nuisance macrophytes, algal scums, and low transparency may discourage swimming and boating.

1.1.4. Internal Phosphorus Loading

Seasonal P releases from lake sediments are commonly associated with the onset of anoxia (*i.e.*, dissolved oxygen $<1.0 \text{ mg L}^{-1}$ (James *et al.*, 2015)) at the sediment-water interface. Anoxic conditions can result when the rate at which oxygen is consumed by bacterial decomposition of organic matter (respiration) is greater than the rate at which oxygen is produced through the processes of photosynthesis by aquatic plants and algae. The oxygen depletion rate is highest at the sediment-water interface where the accumulation of organic matter is greatest. The effect of the respiration-photosynthesis imbalance is most notable during periods of summer stratification with the development of anoxic zones originating at the lake bottom. The same effect can occur during periods of ice cover.

The migration of P from sediment to the overlying water column has been linked to redox conditions. These benthic redox reactions are effectively controlled by the presence of dissolved oxygen (DO) (Mortimer, 1941). With the onset of anoxic conditions and resultant decrease in redox potential, a reduction in Fe (III) occurs through microbial activity. This redox reaction releases phosphate bound in hydrous oxides and gels at the sediment surface (Carlton and Wetzel, 1988). The two primary factors which influence internal loading processes are the trophic status of the lake (Nürnberg, 1997) and the thermal regime, which is controlled by lake morphology and climate (Nürnberg *et al.*, 2012). Other factors which could affect the rate of P efflux are: pH, bioturbation, epipelagic algal (flora growing on sediments) photosynthesis, microbial metabolism (Wetzel, 2001; Carlton and Wetzel, 1988), redox-sensitive uptake, release of P by benthic communities (Gächter *et al.*, 1988), and apatite (calcium phosphate) precipitation (Golterman, 2001).

When the waters overlying lake sediments are oxidized (DO present), binding of phosphate to Fe(III) oxyhydroxides limits P release into the water column (Katsev, 2006) with a predominance of P influx occurring (*i.e.*, precipitation to sediments) (Nürnberg, 1984; Beutel *et al.*, 2008). P release has also been shown to occur in the presence of DO at the sediment-water interface in shallow hardwater lakes (*i.e.*, 125 mg L^{-1} as calcium carbonate (CaCO_3) range) with elevated pH levels (*i.e.*, 7.7 to 10.6; mean 8.8) (Hoverson, 2008). The hardness of Mattatall Lake water is approximately 12 mg L^{-1} excluding it from this category.

When lakes become thermally stratified, the diffusion of P from the hypolimnion upward into the trophogenic zone (*i.e.*, area in the water column where photosynthetic production predominates (Wetzel, 1983)) is limited. Phosphorus released from lake sediments during periods of anoxia, is restricted by the presence of a thermocline. However, with the thinning of the thermocline at spring and fall turnover, hypolimnetic P mixes throughout the water column. Nürnberg (1984) estimated that in the presence of high iron, roughly 30% of hypolimnetic P settles to lake sediments as iron precipitates, 30% is taken up by plankton, and 38% stays in solution, with the fate of the remaining 2% unknown.

Evidence that hypolimnetic P has influenced epilimnetic waters during or following turnover is reflected in P increases in either epilimnetic or thermocline water. The increase in volume-weighted water column mean concentration following turnover can also be correlated with P contained in the anoxic layer prior to mixing. Other evidence would be an increase in phytoplankton production (reflected in chlorophyll *a* levels) due to the injection of additional P into the trophogenic zone.

1.1.5. Climate Change Effects on Lakes

Climate change is slowly increasing global air temperatures and water temperature of freshwater lakes are also increasing. Surface water temperatures in seasonally ice-covered lakes are increasing by 0.72°C per decade (O'Reilly *et al.*, 2015). Several researchers have identified climate change as a key factor driving the increase in cyanobacterial blooms through several mechanisms. Nürnberg *et al.* (2012) focused on the role that climate (temperature and precipitation) plays in accelerating internal P loading within a large shallow lake in Finland, showing that internal loading increased as temperature increased and flushing rate decreased. There has been considerable interest in assessment of the response of algae populations—in particular harmful cyanobacteria—to climate change. Increasing temperatures can directly influence the growth of algae, and can alter the strength and duration of stratification phenomena, which also affects the population size and species distribution of algae. Several researchers have found that increasing water temperatures favors the dominance of cyanobacteria (Paerl & Husiman, 2008; Elliott, 2010). Rigosi *et al.* (2015) demonstrated through a modeling study that a small water temperature increase (by 0.08°C from 24°C) can increase the risk of harmful cyanobacteria blooms by 5%; the same increase in bloom risk was found for a P increase from 10 µg L⁻¹ to 20 µg L⁻¹. In general, their study indicated that rising temperatures may be a more important factor influencing lake trophic state than P levels. Specifically in Nova Scotia, a paleolimnological study conducted by Ginn *et al.* (2015) which assessed pre- and post-disturbance conditions based on diatoms assemblages in lake sediment cores showed evidence that 19 out of the 51 lakes studied in Halifax Regional Municipality were experiencing biological changes linked to climate change.

1.1.6. Flushing Rates

The flushing rate of a lake, which is related to water retention time (WRT), has a strong influence on both nutrient levels and growth of algae (Jones and Elliott, 2007). There is a general consensus in the literature that as flushing rates increase (and WRTs decrease) lakes are less vulnerable to trophic state changes as a result of nutrient loading. Early work conducted by Kerekes (1975) on a set of lakes in southwestern Nova Scotia demonstrated the influence of flushing rates on nutrient levels, showing that lakes with high flushing rates (>7 yr⁻¹) were less vulnerable to pollution than lakes with low flushing rates. Higher flushing rates also shorten the time that P is available to be assimilated by algae, and the time that algae have to establish communities. Therefore, high flushing rates (shorter WRTs) are also negatively correlated with algal blooms (Kalff, 2002; Londe *et al.*, 2016).

Several researchers have empirically observed a relationship between decreased algal growth and high flushing rates (Dickman, 1969; Reynolds & Lund, 1988; Maberly *et al.*, 2002). Jones & Elliott (2007) specifically examined the influence of WRT on phytoplankton growth and mean chlorophyll *a* levels using a calibrated process based modeling approach. They observed a four-fold decrease in mean chlorophyll *a* concentrations moving from a WRT of 338 to 8 days.

1.2 Objectives

The primary objective of this study was to characterize sources of P in the Mattatall Lake Watershed, and use this information to help understand causes of recurrent algae blooms in the lake. Specific objectives of the study were to:

- (i) Construct and validate a P loading model to predict TP concentrations in the lake;
- (ii) Characterize land use change in the watershed over the past 32 years and how these changes have influenced TP levels in the lake;
- (iii) Characterize the current trophic state of the lake using both monitoring and modeling results;
- (iv) Assess the potential for internal loading of P within the lake; and
- (v) Provide recommendations for monitoring and managing P loading processes in the future.

1.3 Project Scope and Limitations

The project scope included an assessment of P sources within the Mattatall Lake watershed. Based on this assessment potential targeted strategies for reduction of P loading into Mattatall Lake were identified. The scope of the study did not include an evaluation of other algae bloom mitigation approaches (*e.g.*, algaecides, food web manipulation, *etc.*). The P loading model applied to the Mattatall Lake watershed is a mass balance steady-state model designed to predict average annual concentrations of TP in lakes. The model has its limitations and relies on several assumptions. The assumptions and limitations of the model used in this study are detailed in the following list (in no particular order):

- Individual export coefficients incorporated in the model were assumed to be a close representation of the various land uses found within the drainage basin. They were selected as mean values developed from a series of data sets representing a specific land use category;
- The main function of the model is to predict steady-state conditions (*i.e.*, what TP levels will be once the system has reached equilibrium following a change in land use);

- The model predicts average annual lake TP concentrations and is not intended to address temporal scales on less than a yearly basis;
- The model predicts the spatially average lake TP concentration and therefore model estimates are spatially limited to the lake, or basin, scale;
- A period of five-years was assumed to be the duration of time over which the P export would return to pre-cutting conditions after clearcutting events; and
- Examples of potential P sources/sinks which are not accommodated by the model include waterfowl, and recycling of nutrients from aquatic plants.

The P modeling produced for this study was based on the best available information at the time of the reporting. Should new information pertinent to the study become available, CWRS requests notification in order to update the conceptual model.