Ecological Restoration Options for Clear Lake and South Lake (Riding Mountain National Park), Manitoba

by
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Contents

Abstract v
Dedication vi
Acknowledgements vii
List of Figures viii
List of Tables x
List of Equations xi
List of Definitions xii
List of Appendices xiii
Chapter 1 – Introduction 1
1.1 Introduction 1
1.2 First Nations 1
1.3 Riding Mountain National Park 2
1.4 Goals and Objectives 2
1.4.1 Examine options to mediate increased nutrient loading 3
1.4.2 Examine options regarding connectivity between South Lake and Clear Lake 4
1.4.3 Promote outreach in Riding Mountain National Park and the surrounding area 4
Chapter 2 – Key Concepts 4
2.1 Nutrient Monitoring 4
2.2 Climate Change 7
Chapter 3 – Clear Lake & South Lake: Historical and Current Conditions 9
3.1 Clear Lake - Physical Characteristics 9
3.2 Clear Lake - Biological Characteristics 12
3.3 Clear Lake - Nutrient and Chemical Characteristics 15
3.4 South Lake - Physical Characteristics 19
3.5 South Lake - Biological Characteristics 19
3.6 South Lake - Nutrient and Chemical Characteristics 20
Chapter 4 Ecological Stressors 21
4.1 Wastewater Treatment Facility 21
4.2 Disruption to Drainage Patterns 22
4.3 Introduced Species 24

4.4 Groundwater Pollution 24
4.5 South Lake Alternate Stable State 26
Chapter 6 – Water Quality Monitoring Study 26
6.1 Introduction 26
6.2 Methods 27
6.3 Results 28
Chapter 7 – Options Analysis 34
7.1 South Lake Restoration Options 34
7.1.1 Overview 34
7.1.2 Implementation 35
7.1.3 Budgetary Considerations 42
7.1.4 Future Management Implications 44
7.1.5 Metrics of Success 45
7.2 Clear Lake Restoration: Hypolimnetic Aeration 46
7.2.1 Overview 46
7.2.2 Installation 50
7.2.3 Budgetary Considerations 53
7.2.4 Future Management Implications 54
7.2.5 Metrics of Success 54
7.3 Clear Lake and South Lake Connectivity 55
7.3.1 Overview 55
7.3.2 Installation 58
7.3.3 Budgetary Considerations 61
7.3.4 Future Management Implications 61
7.3.5 Metrics of Success 62
7.4 Outreach & Awareness 63
References 65
Appendix 73

Abstract

Options for ecological restoration are discussed for the Clear Lake – South Lake complex of Riding Mountain National Park, Manitoba. This project consisted of a) a review of studies conducted on Clear Lake and South Lake and b) a stream water quality sampling program. The review of previous studies was to gain an in-depth understanding of historical processes which shaped Clear Lake and South Lake. Previous condition, current condition and ecological stressors are all identified based on literature from Riding Mountain National Park. The stream water quality sampling program identifies major sources of nutrients into Clear Lake. Ecological restoration options pertain specifically to the Clear Lake – South Lake complex. South Lake restoration options include supplemental planting, dredging and chemical treatments. A novel technique designed to disrupt wind driven nutrient loading is also discussed. These methods are designed to return the South Basin to a macrophyte dominated system. Addressing hypolimnetic oxygen deficiency, two forms of hypolimnetic aeration are discussed to improve water quality in Clear Lake including a 'Full lift' design as well as a Speece Cone. Three options regarding the isthmus and connectivity between Clear Lake and South Lake are examined including a fishway installation and a wattle fence installation.

Dedication

To M.
You’re my favourite book.

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List of Figures

Figure 1. Depth (m) vs Dissolved Oxygen (mg/L) for Station 2 in Clear Lake. Values are the whole summer average dissolved oxygen concentration from records since 2007 (Sampling from June-October). Average oxygen concentrations are lowest in the hypolimnion (average depth > 17 m) throughout the summer. Trendline is a two-period moving average. (Riding Mountain National Park, N.D.) 10
Figure 2. Clear Lake and its sub-basins. Note: Not all sub-basins have defined drainage courses. Sub-basin outlines and names from RMNP GIS Library (Riding Mountain National Park, N.D) 11
Figure 3. Sedimentary diatom assemblages in Clear Lake from White (2012). The Blue dashed line represents establishment of a timber reserve in the Clear Lake watershed (1895), the red dashed line represents the establishment of RMNP (1933). 13
Figure 4. Average monthly TP concentrations between 2010 and 2015. Trendlines are moving averages for each year. TP concentrations are significantly higher in October than June each year (p=0.005). 16
Figure 5. Total chlorophyll and TP concentration in a sediment core taken in Clear Lake. Both values remained relatively constant until the 1900s-1930s, this period corresponds to when activity in the park began to increase (Goldsborough & Rounds, 1992). 18
Figure 6. 1880 Surveyors map including the isthmus and breach on the westernmost side. Blue highlighter indicates historic park boundary. (Stewart, 1880) 22
Figure 7. A comparison of a historical air photo (1966; Left) and recent satellite imagery of the north basin of South Lake (right). The constructed channel connecting Ominik Marsh and South Lake is clearly visible in the recent image. 23
Figure 8. Chloride readings (mg/L) over the ten-year sampling period from (Patalas, Salki, & Stainton, 1988) Y=20.09-6.394e-9*X, R2=0.0813. There is no annual trend in chloride concentrations during the study period. Data accessed from RMNP water quality database (Riding Mountain National Park, N.D.). 25
Figure 9. Stream sampling locations and in-lake sample locations. Three in-lake sample sites are in Clear Lake (CL), two are in South Lake (SL). 27
Figure 10. DIN concentration at each station over the sampling period. Hypolimnetic DIN in Clear Lake was consistently higher than epilimnetic or metalimnetic DIN. South Lake does not stratify therefore only has an epilimnion. 29
Figure 11. Average concentration of DIN and SRP for each stream during the sampling period. Error bars indicate standard error. 30
Figure 12. Relative sub-basin inputs to Clear Lake based on estimated runoff calculated by methods outlined by Cronshey, et al. (1986). Octopus Creek and South Lake were combined based on their connectivity through the South Lake Channel. 32
Figure 13. Proposed location of emergent transplants along the shoreline of the south basin of South Lake. Groundwater flows in a north-west direction, therefor the shoreline most exposed to polluted groundwater should be targeted (Hoole et al., 2005). 37

Figure 14. Average wind direction, velocity and frequency for Wasagaming. Data from Environment Canada 1994-2005. Easterly winds are strongest, and south-westerly winds are most prevalent. Figure modified from: Hoole et al., 2005. 38
Figure 15. Proposed layout of silt fencing. This layout will disrupt mixing from all wind directions while allowing connectivity for fish movement. 40
Figure 16. Schematic of a full-lift hypolimnetic aeration system from (Ashley & Nordin, 1999). Dimensions vary depending on the size of aerator required. 48
Figure 17. Example of a Speece Cone built for Marston Reservoir, Denver, Colorado. Image from Dominik (2017) 49
Figure 18. Summer Hypolimnetic Dissolved Oxygen concentrations versus time. Oxygen decreases over the summer as it is consumed by bacterial decomposition and respiration. The slope of the trendline yields the rate of oxygen depletion (tonnes/day). Light coloured points had missing data at depths greater than 30 m which was filled in using oxygen concentrations at 29m. Data obtained from RMNP water quality database and GIS library (Riding Mountain National Park, N.D.). 50
Figure 19. Algae mats flowing into Clear Lake though the breach in the isthmus. Image taken during site visit on June 13th, 2016 by A. Butcher. 56
Figure 20. Proposed location of the water control structure in the isthmus. Longshore drift moves along the grey arrow in an easterly direction (Willis Cunliffe Tait DelCan, 1982). 59
Figure 21. Illustration of a wattle fence installation. Live cuttings are used to create a living wall for erosion control. The isthmus slope is not as steep as pictured, but wattle fences on either side of the isthmus could add enough structural integrity to prevent breaches. From Polster (1997). 60

List of Tables

Table 1. TP and alum water quality data from cell 2 and cell 3 of the Wasagaming wastewater treatment facility. 22
Table 2. Detection limit and corresponding adjusted value. Nitrate & Nitrite concentrations were reported together. DIN had a detection limit adjusted value of 0.04 mg/L. 28
Table 3. Material cost estimates for a whole basin silt fence installation and pilot project installation. Prices are estimates. 43
Table 4. Estimated project cost for dredging the south basin of South Lake. Prices are estimated based on minimum literature values. 44
Table 5. Reductions in TP, SRP, Chl-a and secchi depth during Phoslock^{TM} treatments. Modified from (Spears, et al., 2016). 46

List of Equations

Equation 1. $\text{Removal rate} = \frac{\text{Cell 2 TP}\left(\frac{mg}{L}\right) - \text{Cell 3 TP}\left(\frac{mg}{L}\right)}{\text{Alum}\left(\frac{mg}{L}\right)}$ 22

Equation 2. $Q = \frac{(P-I_a)^2}{(P-I_a)+S}$ 31

Equation 3. $I_a = 0.S$ 31

Equation 4. $S = \frac{1000}{CN} - 10$ 31

Equation 5. $L = \frac{gt^2}{2\pi}$ 39

Equation 6. $\frac{gt}{2\pi U} = 1.20 \tanh\left[0.077\left(\frac{gF}{U^2}\right)^{0.25}\right]$ 39

Equation 7. $F = \frac{28447 \times U^2 \times \tan^{-1}\left(\frac{0.332452 \times g \times \sqrt[2]{\frac{L}{g}}}{U}\right)^4}{g}$ 39

Equation 8. $\text{Tube radius} = \sqrt{\frac{\text{Water flow}}{\pi \times \text{bubble water velocity}}}$ 51

Equation 9. $\text{Entrance loss} = K_L \times \frac{8Q_w^2}{\pi^2 D^4 g}$ 51

Equation 10. $\text{Exit loss} = \frac{8Q_w^2}{\pi^2 D^4 g}$ 51

Equation 11. $\text{Friction loss} = \frac{fL \times 8 \times Q_w^2}{\pi^2 D^5 g}$ 51

Equation 12. $\text{Density of air: water mixture} = \Gamma_{aw} = \frac{L \times \Gamma_w}{L + \Delta H}$ 52

Equation 13. $\overline{Q_a} = \frac{10.4 Q_a \ln\frac{L+10.4}{10.4}}{L}$ 52

Equation 14. $Q_a = \frac{(Q_w \times \Gamma_w) - (Q_w \times \Gamma_{aw})}{(\Gamma_{aw} \times \overline{Q_a}) - \Gamma_a}$ 52

Equation 15. $\text{Pressure required} = \Delta P + h + L\left(\frac{1.0}{10.1}\right)$ 52

Equation 16. $\text{Floatation requirements} = \frac{\Delta H}{2} \times \text{Area of box} \times \Gamma_w$ 53

List of Definitions

DIN	Dissolved Inorganic Nitrogen
DO	Dissolved Oxygen
N	Nitrogen
P	Phosphorous
RM	Regional Municipality
RMNP	Riding Mountain National Park
SRP	Soluble Reactive Phosphorous
TChl	Total Chlorophyll
TP	Total Phosphorous
TN	Total Nitrogen

List of Appendices

Appendix A. DIN concentrations for in-lake samples at Clear Lake and South Lake. Bolded values indicate estimates due to results below detection limit (DL = 0.08 mg/L) 73
Appendix B. SRP concentrations for in-lake samples at Clear Lake and South Lake. Bolded values indicate estimates due to results below detection limit (DL= 0.01 mg/L). 74
Appendix C. DIN concentrations measured at inflows and outflows of Clear Lake and South Lake. . 75
Appendix D. SRP concentrations for inflows and outflows of Clear Lake and South Lake 76
Appendix E. In-lake profile data for Station 1 in Clear Lake 77
Appendix F. In-lake Profile data for Station 2 in Clear Lake 79
Appendix G. In-lake Profile data for Station 3 in Clear Lake 83
Appendix H. In-lake profile data for South Lake. S. Late Stn 1 = South Lake Station 1, S. Lake Stn 2= South Lake Station 2. 85
Appendix I. Sub-basin runoff data. See 6.3 Results for a discussion on calculation methods. South Lake's total percentage runoff was added to Octopus Creek due to the connectivity through the South Lake Channel 86
Appendix J. Average dissolved oxygen concentrations over summer DO profile sampling in 2015. Values marked in bold had incomplete profiles, values used in calculations were simply the value for the next deepest profile. Red text indicates the period of declining hypolimnetic dissolved oxygen. 87
Appendix K. Full lift hypolimnetic aerator sizing based on calculated DO consumption and the recommended doubling of DO demand. 89

Chapter 1 – Introduction

1.1 Introduction

Parks Canada has long held the objective to maintain or improve the ecological integrity of Canada's National Parks (Parks Canada, 2007). Meeting this objective extends beyond conservation practices and includes restoration of ecologically degraded sites and building engagement with Canadians. Ecological restoration can rejuvenate previously degraded ecosystems, and make them more resilient against future threats (Society for Ecological Restoration International Science and Policy Working Group, 2004). Restoring and increasing the ecological resilience of Clear Lakes will ameliorate the future effects of the continued recreation and development in the Clear Lake watershed. Further, climate change will also undoubtedly affect both biotic and abiotic factors within the watershed. Developing restoration plans which will remain effective and counteract climate changes effects is crucial to secure the long-term meso-oligotrophic status of Clear Lake. Fostering engagement with Canadians and other park visitors is an equally important step in securing the future of Clear Lake. Expressing the importance of natural ecosystems and outlining how one's actions can have consequences to those ecosystems can have profound effects on preventing future degradation of the environment. Conveying one's responsibility to maintaining the environment can develop into better management practices for future generations. As such, securing Clear Lakes future as a meso-oligotrophic lake relies on continued management, restoration actions and stewardship into the future.

1.2 First Nations

Riding Mountain National Park (RMNP) has had a long history with the First Nations peoples in the area. The Riding Mountain area has been inhabited for 6,000 years. The Ojibway peoples occupying the area were part of the larger Plains Ojibwa who migrated from the Great Lakes in the late eighteenth century. Those who would eventually come to be the Keeseekoowenin Band maintained hunting, fishing and trapping camps in the Riding Mountains (Sandlos, 2008). As European settlement began to take hold, these peoples began to witness declining wildlife populations and with it uncertain food supply. Treaty 2 was signed in 1871 by the Keeseekoowenin Band chief, Mekis, which established a reserve for the band first at Lake Dauphin. The reserve was later moved towards Elphinstone (Sandlos, 2008). A fishing reserve (61A) was established on Clear Lake for the Keeseekoowenin Band at Elphinstone in 1896 (Hoole et al., 2005). Upon establishment of the national park in 1936, this reserve was wrongfully removed and absorbed into what became Riding Mountain National Park. A combination of local and state interests (tourism promotion, elk (*Cervus canadensis*) protection and assimilation of native hunters) led to the expulsion of the

Keeseekoowenin Band (Sandlos, 2008). The inhabitants of the reserve we relocated to the Keeseekoowenin Ojibway reservation in Elphinstone (approximately 20 km away).

In 1991, following two land claims, Parks Canada recognized their past wrong-doing and returned the land to the Keeseekoowenin Ojibway First Nation (Hoole et al., 2005). The north shore of Clear Lake was included in this reservation. In 2006, Parks Canada and the Coalition of First Nations formed the Riding Mountain Forum to express concerns over the Riding Mountain ecosystem, and the relationship between the two parties has since strengthened (Sandlos, 2008).

Ecological restoration in Clear Lake and South Lake presents an opportunity to further strengthen the relationship between Parks Canada and the Keeseekoowenin Ojibway First Nation. While this report has been written, and targeted towards Parks Canada, collaboration in decisions pertaining to restoration actions is strongly advised. This is an opportunity to enhance both the working relationships between Parks Canada and the Keeseekoowenin Band and to improve the water quality and biological diversity of Clear Lake.

1.3 Riding Mountain National Park

Riding Mountain National Park was established in 1929 in an effort by the National Parks System to protect 2,969 km^2 of the southern boreal plains and plateaux system (Parks Canada, 2007). The park is in southwestern Manitoba and is the centerpiece of the 12,000 km^2 Riding Mountain Biosphere Reserve. The Riding Mountain Biosphere Reserve consists of 15 municipalities with a population of 35,500 people. Although the park encompasses many other lakes, none match the size or recreational value of Clear Lake. Riding Mountain National Park receives over 250,000 visitors annually with the majority staying in Wasagaming, a small seasonal town bordering Clear Lake (Parks Canada, 2007). Immediately south of Riding Mountain National Park is the Regional Municipality (RM) of Park (Bergman, 1987).

1.4 Goals and Objectives

Defining well planned goals and objectives is a crucial step in ensuring restoration success. Clearly stated goals and objectives allow restoration practitioners to define a clear trajectory for what results are desired for the project (goals) and steps towards achieving these (objectives) (Society for Ecological Restoration International Science and Policy Working Group, 2004). This report consists of an in-depth options analysis for Parks Canada decision makers to use for management decisions that benefit the ecological integrity and suit operational needs of the park. Ecological restoration of the ecosystem will have broad implications for aquatic habitat within the lakes, fish spawning areas and recreation. The following goals have been identified based on concerns raised by Parks Canada staff at Riding Mountain National Park.

1.4.1 Examine options to mediate increased nutrient loading

Identify sources of high nutrient concentrations (SRP and DIN) to Clear Lake.

Soluble Reactive Phosphorus (SRP) represents the inorganic phosphorus molecule (orthophosphate, PO_4^{3-}) which can be directly absorbed by phytoplankton (Nürnberg & Peters , 1984; Wetzel, 2001). Dissolved Inorganic Nitrogen (DIN) represents the sum of nitrate (NO_3^-), nitrite (NO_2^-) and ammonia (NH_3) concentrations within a water sample. These three N-containing compounds are the nitrogenous species directly available for phytoplankton uptake (Wetzel, 2001). By examining nutrient inputs of both SRP and DIN in Clear Lake and South Lake, I examined possible ecological restoration options to mediate increased nutrient inputs. This sampling program also monitored numerous water quality parameters within Clear Lake and South Lake (dissolved oxygen, temperature, pH, conductivity). Certain streams may be transporting a significant proportion of nutrients into South Lake or Clear Lake. Identifying these streams provides the basis for restoration efforts in the sub-drainage basin. Definitively identifying and addressing the underlying cause of ecological decline is the best way to ameliorate the issues at hand.

Conduct a literature review of previous studies in Riding Mountain National Park

A unique aspect of this project (as compared to many other ecological restoration projects) is that an extensive body of knowledge already exists for RMNP. This presents the opportunity to define a desirable historical condition to guide restoration efforts. Further, documentation of historical stressors for Clear Lake and RMNP can provide a deeper understanding of how the park reached its current state. These stressors primarily include: Wasagaming's wastewater treatment facilities, non-point source nutrient pollution, and historical landscape and drainage pattern modifications.

Develop ecological restoration options for Clear Lake and South Lake

Restoring the historic oligotrophic status of Clear Lake is the primary objective of this project. Options have been examined for restoration actions within Clear Lake as well as the surrounding area to ensure RMNP has all the tools required to make informed decisions. These will include tested and experimental options in lake restoration, and nutrient loading mediation. Experimental options will be developed based on scientific knowledge and are tailored for RMNP.

1.4.2 Examine options regarding connectivity between South Lake and Clear Lake

An ice push isthmus/ridge separates Clear and South Lakes. This isthmus breaks open periodically allowing connectivity between the two lakes. Fish migration has been well documented, as well as the use of South Lake as a spawning area for northern pike (*Esox lucius*) and walleye (*Sander vitreus*) (Fawcett, 2011; Maclean, 1979; Thomas, 2011). Numerous attempts have been made in the past to ensure a connection remains open between the two lakes, all of which have been unsuccessful (Rounds et al., 1992). Sand and sediment typically plug the channel following a storm event. Since South Lake has become increasingly eutrophied, the question remains whether this connection should be maintained. When the channel opens, South Lake is a significant source of nutrients into Clear Lake which is concerning given Clear Lakes meso-oligotrophic state. Conversely, if the isthmus is disrupting fish populations within Clear Lake, this could have implications in trophic interactions and further facilitate eutrophication of Clear Lake.

1.4.3 Promote outreach in Riding Mountain National Park and the surrounding area

Management options for reducing nutrient loading into Clear Lake extend beyond RMNP boundaries. Non-point source pollution from the Octopus Creek sub-basin and South Lake sub-basin will continue to decline water quality in Clear Lake regardless of ecological restoration within the park. Reducing non-point source phosphorous (P) runoff from agricultural land is a key tool in reducing overall non-point source loading. No-till farming, contour farming and terracing are farming practices which reduce runoff of fertilizer application (Daniel et al., 1994). Outreach will also enhance public appreciation and stewardship for RMNP's resources. Developing an understanding of ones own responsibility to the environment is a key tool in preserving ecological integrity in RMNP and the RMBR.

Chapter 2 – Key Concepts

2.1 Nutrient Monitoring

Eutrophication of freshwater systems is a frequent occurrence in environmental sciences. Although the term *eutrophe* and its cousins *mesotrophe* and *oligotrophe* originate as descriptors for bogs, the definition shifted to limnology in the mid-1970s (Schindler, 2006; Weber, 1907). By Webers 1907 definition, a bog with vegetation meeting the eutrophe classification had plants requiring higher concentrations of soil nutrients. Oligotrophe bogs had low concentrations of nutrients and vegetation adapted to such conditions. By 1919, Naumann was the first to apply Webers terms to lakes, using them as descriptors for water types (Hutchinson, 1973). *Eutrophic* and *oligotrophic* remained qualitative, serving to classify lakes by appearance rather than nutrient concentrations. General trends found that shallow lakes with abundant phytoplankton and algal growth were eutrophic. Conversely, deep, nutrient poor lakes typically had limited primary production. Deeper lakes have a smaller proportion of light-receiving epilimnion to volume than shallower lakes, giving the lakes a proportionally smaller volume available for primary production (Dokulil & Teubner, 2011). Deep lakes have a larger hypolimnion than shallow ones by virtue of their depth. This increased volume corresponds to decreased oxygen demand per unit volume and therefore higher dissolved oxygen at depths (Dokulil & Teubner, 2011). These were important trends that allowed early limnologists to define predictors of lake production. As lab techniques for measuring nutrients in water samples progressed, modern use of the term eutrophic began to develop quantitative aspects. Colorimetric techniques allowed early limnologists to examine minute concentrations of nutrients in waters (Hutchinson, 1973).

Early efforts to control eutrophication focused on herbicides to kill algae, rather than focusing on the source (Schindler, 2006). Eliminating excess nutrient addition to lakes has been shown to be an effective management tool (Schindler, 1974). By eliminating inputs, many lakes recover naturally by algal uptake and subsequent sedimentation. A 37-year experiment found that control of N inputs alone in eutrophic lakes is not an effective method of reducing algal growth (Schindler et al., 2008). The study concluded that continued P inputs will control algal growth, and therefore should be the focus of restoration efforts. Short-term N limitation, as has been observed in Clear Lake, is an indication that N inputs must be reduced as well (Hawryliuk, 2000; Schindler, 1977). N-fixing cyanobacteria will rapidly colonize the lake if only N inputs are removed, and N limitation occurs. Nitrogen fixing cyanobacteria often present the greatest decline to water quality and human health concerns (Schindler, 1974).

Eutrophication remains one of the most widespread water quality issues, and the problem will likely worsen in the future (Dokulil & Teubner, 2011). As the global populations increase, cities are bound to expand and agriculture will by necessity increase as well. Much of the prairies including the area surrounding RMNP, will undoubtedly see an increase in agricultural pressure. Even with advances in agricultural techniques, eutrophication is likely a problem that will persist well into the future. As such, it is important to control eutrophication in Clear Lake before the meso-oligotrophic condition is lost.

SRP predominantly consists of orthophosphate which is biologically available. At SRP concentrations below 30 µg/L, SRP is highly correlated with biologically available P (Nürnberg & Peters, 1984). SRP concentrations above 30 µg/L tend to overestimate biologically available phosphorus by 8% on average. Regardless, SRP remains closely correlated to the fraction of P bioavailable to algae and is an appropriate indicator. Total phosphorous (TP) is commonly used as an indicator of P concentrations in lakes. However, TP does not accurately indicate P concentrations dictating algal growth in lakes. TP and SRP are poorly correlated and are not interchangeable measures of P (Dodds, 2003). TP includes bioavailable P compounds (SRP) as well as particulate P and organic P which are not bioavailable. Therefore, not all the P in TP is available for algal growth, meaning overestimation/underestimation of bioavailable P in lakes is possible (Wetzel, 2001). A previous study on P inputs in Clear Lake concluded that Clear Lake was not as productive as TP concentrations indicated (Hoole et al., 2005). A likely explanation of this observation is that the researchers were overestimating the potential for algal growth based on TP concentrations.

Although it has been well established that N plays less of a role in determining productivity in lakes than P, N limitation can be equally important as P limitation. Cycling of N within lakes is more complex than P since there is an additional gaseous phase. Therefore, N has several more possible inputs (atmospheric precipitation, N fixation) and outputs (volatilization) (Wetzel, 2001).

Atmospheric N deposition accounts for an average of 0.1 g $N/m^2/yr$ over continental USA (Wetzel, 2001). Since N deposition is variable across the continent, a more accurate estimate is preferable. Atmospheric nutrient deposition in Canada is monitored by the Canadian Air and Precipitation Monitoring Network, though N data is not available at this date (Environment and Climate Change Canada, 2016). The National Atmospheric Deposition Program (NADP) is an American initiative to measure atmospheric deposition trends across North America. The two nearest NADP monitoring stations to RMNP are Icelandic State Park just south of the Canada-USA border in North Dakota (approximately 265 km from Clear Lake) and Theodore Roosevelt National Park in North Dakota (approximately 485 km from Clear Lake). Average atmospheric deposition for both locations was 0.217 g $NH_4/m^2/yr$, and 0.401 g $N0_3/m^2/yr$ (1983-2015) (National Atmospheric Deposition Program, 2016). Accounting for N mass ratios and combining these, this equates to 0.242 g $N/m^2/yr$. Dry fallout adds an additional a factor of 3-4, so as much as an estimated 0.856 g N/m^2, or 25,000 kg N/year is added to Clear Lake annually (Wetzel, 2001). It should be noted that this is the high estimate and should be taken as an illustrative account of atmospheric N deposition only.

N fixation can be a significant source of N under certain conditions. Under N limitation, cyanobacteria can convert N_2 to a usable form (usually NH_3). This process however requires 12-15 adenosine tri-phosphate (ATP) per N_2 fixed, in other words, the process is energetically onerous (Wetzel, 2001). As such if biologically available N is present in substantial concentrations, the process will not occur. In Clear Lake, N limitation occurs only rarely (<5% of the time for Clear Lake) (Hawryliuk, 2000).

Like TP, total nitrogen (TN) does not represent the actual value of biologically available N. Although N is plentiful in earths atmosphere, only a small proportion (less than 2%) is available to life ie. reactive N. This includes: ammonia (NH_3-N), nitrate (NO_3-N) and nitrite (NO_2-N) (the three of which make up dissolved inorganic nitrogen) as well as numerous organic N containing compounds. For the purposes of this study, only dissolved inorganic nitrogen (DIN) was measured, which excludes organic N. Little correlation exists between organic N and lake productivity so it is convenient to only measure DIN (Wetzel, 2001).

2.2 Climate Change

Climate change remains a looming factor of uncertainty in any restoration project. To design a restoration plan without acknowledging and accounting for a changing climate is both nearsighted and irresponsible. To tailor restoration efforts to be resilient against climate change, we first must examine possible future scenarios. Although there are still many unknowns with regards to what changes will be seen where, several trends are guaranteed. Changes in water temperature, stratification depth, and hydrology are certain (Dokulil & Teubner, 2011). A major facet of climate change with respect to this project will be alterations to the water cycle. Changing hydrology will threaten wetlands, ponds and lakes worldwide. Threats of extinction and population decline associated with climate change are greatest with freshwater ecosystems (Pittock, 2005). Changes to hydrology are already being noticed in many parts of the world (Dokulil & Teubner, 2011).

Under a low carbon emissions scenario (RCP4.5), current climate change prediction models (mean projection) indicate a mean annual temperature increase of 1.8 °C between 2021-2050 (as compared to 1981-2010 baseline levels) (Prairie Climate Centre, 2016). Under this scenario, annual precipitation is projected to increase by 26 mm. Mean projection estimates for the low carbon emission scenario between 2051-2080 predict a mean annual temperature increase of 3 °C. This low carbon emission prediction model (RCP4.5) assumes dramatic reductions in emissions in approaching decades, with stable greenhouse gas emissions by the end of 2100. These low carbon emission scenarios predict a trend of warmer winters and summers, coupled with increased precipitation. Under these conditions flooding and drought are more likely as the maximum 3-day precipitation total will increase (Prairie Climate Centre, 2016). Further, warmer winters and greater winter precipitation likely means greater volumes of precipitation falling as rain, rather than snow. The high carbon emission scenario (RCP8.5) projects a further amplified warming trend. Under this scenario, mean annual temperatures will increase by 2.1°C for 2021-2050 and 4.4°C for 2051-2080. Increases in precipitation are similar to the low emission model. Both models used by the Prairie

Climate Centre are based on Pacific Climate Impacts Consortium's Global Climate Models (Pacific Climate Impacts Consortium, 2014). Changes in precipitation will undoubtedly alter water levels within RMNP. Higher than average water levels were hypothesized as the "switch" event causing a macrophyte community change in the south basin of South Lake (discussed in **Error! Reference source not found.**) (Scott & Kling, 2006). Higher than average water levels decrease light availability to submerged macrophytes.

The effects of climate change on Clear Lake will extend beyond changes to the hydrologic regime. Small changes to temperature cues have been shown to alter population dynamics within zooplankton communities in northern-temperate lakes (Dupuis & Hann, 2009). Environmental cues dictate emergence timing for diapausing zooplankton eggs, which will in turn govern species composition. This can have profound effects on trophic interactions as zooplankton communities decrease phytoplankton populations through grazing, and make up the food source for planktivorous species. In terms of physical responses to climate change in Clear Lake, stratification will be stronger and last longer. This means Clear Lakes hypolimnion will have a longer and more pronounced hypolimnetic oxygen demand, as stratification will break down later. Hypolimnetic anoxia results in increases in internal nutrient loading, which will influence trophic interactions throughout the lake (Golterman, 2001). Hypolimnetic anoxia will also undoubtedly decrease survivorship of deep-water fish species.

As concern for Climate Change grows, scientists must begin looking for solutions. While many focus efforts on developing technological answers to draw down atmospheric carbon, wetlands and small lakes provide a natural solution. Studies on carbon burial rates (the process of storing carbon through sedimentation) generally overlook small water bodies which have significantly greater rates of carbon burial (Downing, 2010). Although Clear Lake is characterized by low productivity, South Lake and Ominik Marsh are highly productive wetlands and have greater carbon burial rates. There has been some debate as to whether wetlands are net carbon sinks or greenhouse gas emitters (Kayranli et al., 2010). Over the growing season wetland plants will absorb carbon from the atmosphere to accumulate plant tissues. Due to the anoxic conditions of wetland soils, wetlands will store carbon from deposited organic matter. A complex array of methanogenic bacteria and other anaerobes work to breakdown organic matter in anaerobic conditions often forming methane (CH_4) as a bi-product. Wetlands are the greatest natural source of methane, releasing 24% of total emissions (Whalen, 2005). However, accounting for CO_2 sequestration rates and global warming potential for CO_2 versus CH_4, most wetlands are net carbon sinks (Mitsch, et al., 2013). The weighted average of global carbon sequestration by wetlands is 118 g C/m^2/year, though temperate wetlands typically have higher rates of carbon sequestration (278 g C /m^2/year) (Mitsch, et al., 2013).

As such, preserving and restoring wetlands in RMNP is an opportunity to sequester carbon from the atmosphere, and proactively combat climate change.

Chapter 3 – Clear Lake & South Lake: Historical and Current Conditions

3.1 Clear Lake - Physical Characteristics

Riding Mountain National Park is mainly comprised of the Saskatchewan Plain, a small portion of the Manitoba Plain and a section of the Manitoba Escarpment. During the Pleistocene epoch, the park was covered by glaciers and as such, current topography and soils are of glacial origin (Hutchinson, 1981). The bedrock is called the Riding Mountain Formation from the late Mesozoic/Cretaceous age. Soils in the park are predominantly grey wooded luvisols. Peat deposits are common in depressions and old stream channels (Hutchinson, 1981).

Clear Lake is moderately large (surface area of 29.37 km^2) and deep (max depth of 34.2 m) in comparison to other lakes in Manitoba (Langston et al., 2003). The lake comprises approximately 23% of the total Clear Lake watershed area (116.48 km^2), which can be divided into 10 major sub-basins (Hoole et al., 2005; Willis Cunliffe Tait DelCan, 1982; Whitehouse, 2010). Clear Lake's retention time is calculated to be 10.72 years (McGinn et al., 1998). Clear Lake generally behaves as a holomictic dimictic lake with thermal stratification occurring during summer and winter months (Rousseau, 1992). However, due to the orientation of the lakes maximum fetch (12,095 m), strong summer prevailing winds and bathymetric arrangement, the lake occasionally experiences mid-summer mixing (Hoole et al., 2005; Maclean, 1979; Rousseau, 1992). During these summer mixing events, the lake experiences a homothermal condition throughout. Early reports indicated surface water temperatures reach 20 °C with bottom temperatures as low as 10 °C (Bajikov, 1932). These temperature readings were taken in mid-September with ambient air temperature of 25 °C (Bajikov, 1932). In 1987, a water quality study reported little evidence of eutrophication in Clear Lake, though other reports have contradictory conclusions (Bergman, 1987; see White, 2012). Clear Lake has a clinograde dissolved oxygen profile, typically characteristic of more eutrophic water bodies (Figure 1; Wetzel, 2001). Clinograde profiles indicate a hypolimnetic dissolved oxygen deficiency during stratification.

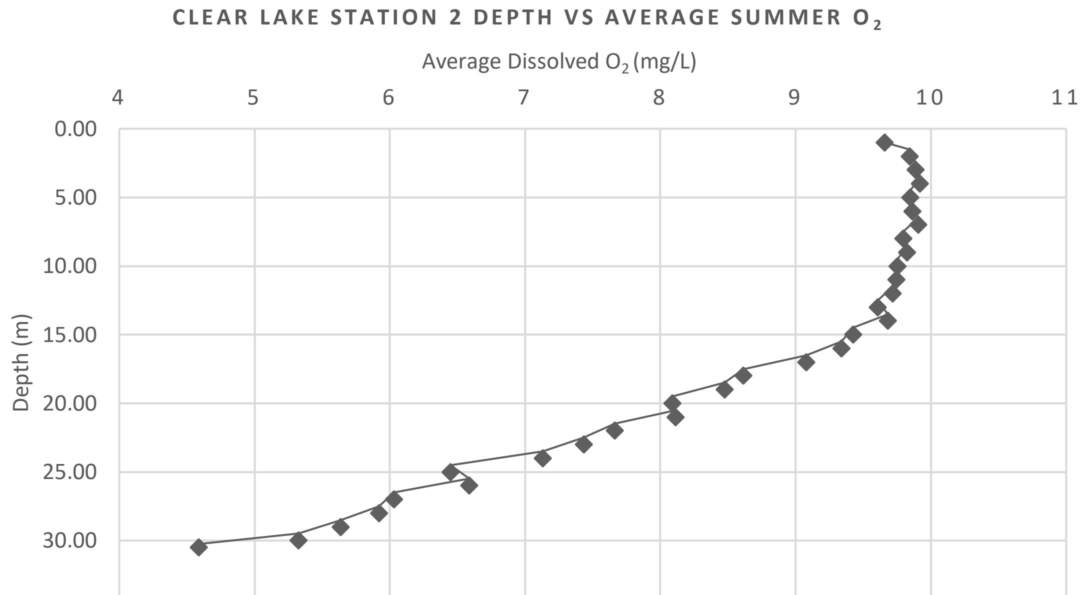

Figure 1. Depth (m) vs Dissolved Oxygen (mg/L) for Station 2 in Clear Lake. Values are the whole summer average dissolved oxygen concentration from records since 2007 (Sampling from June-October). Average oxygen concentrations are lowest in the hypolimnion (average depth > 17 m) throughout the summer. Trendline is a two-period moving average. (Riding Mountain National Park, N.D.)

Although there are 6 main inflows to Clear Lake, many small intermittent tributaries surround the lake. There is a single outflow, Wasamin (Clear) Creek, which drains the lake from the westernmost side. Clear Creek eventually joins the Little Saskatchewan River (Glufka, 1992). Connectivity and water movement between Clear Lake and its adjacent water bodies is complicated due to its size, the underlying geology of the area as well as anthropogenic forces (see 4.2 Disruption to Drainage Patterns). A preliminary water balance study of Clear Lake was conducted in 2005 to further understand Clear Lakes hydrology (Hoole et al., 2005). The investigation concluded that surface inflows are predominantly precipitation controlled rather than watershed runoff. This is understandable considering the size of Clear Lake's drainage basins. 53.2% of water losses were attributed to evaporation, with the remaining water exiting through Clear Creek. Groundwater contributions were estimated to be 36.3% of water inflow to Clear Lake (Hoole et al., 2005). A more detailed water balance was conducted by Neumann (2013), which also concluded that groundwater was a substantial source for water in Clear Lake.

Figure 2. Clear Lake and its sub-basins. Note: Not all sub-basins have defined drainage courses. Sub-basin outlines and names from RMNP GIS Library (Riding Mountain National Park, N.D)

3.2 Clear Lake - Biological Characteristics

Upland vegetation in the park is mainly mixed woody species including trembling aspen (*Populus tremuloides*), balsam poplar (*Populus balsamifera*) and white birch (*Betula papyrifera*) (Hutchinson, 1981).

Phytoplankton studies conducted in Clear Lake during 1996-1997 found that common phytoplankton species included: *Dinobryon sociale, Fragilaria crotonensis, Pediastrium duplex, Stephanodiscus niagarae, Cyclotella bodanica, Tabellaria fenestrata, and Peridinium spp.* (Hawryliuk, 2000). *Peridinium spp.* and *Fragilaria spp.* were dominant during summer months during Hawryliuk's study (2000). Phytoplankton biomass in both Clear Lake and South Lake steadily increase throughout the summer, peaking in September (Hawryliuk, 2000; Scott & Sellers, 2005).

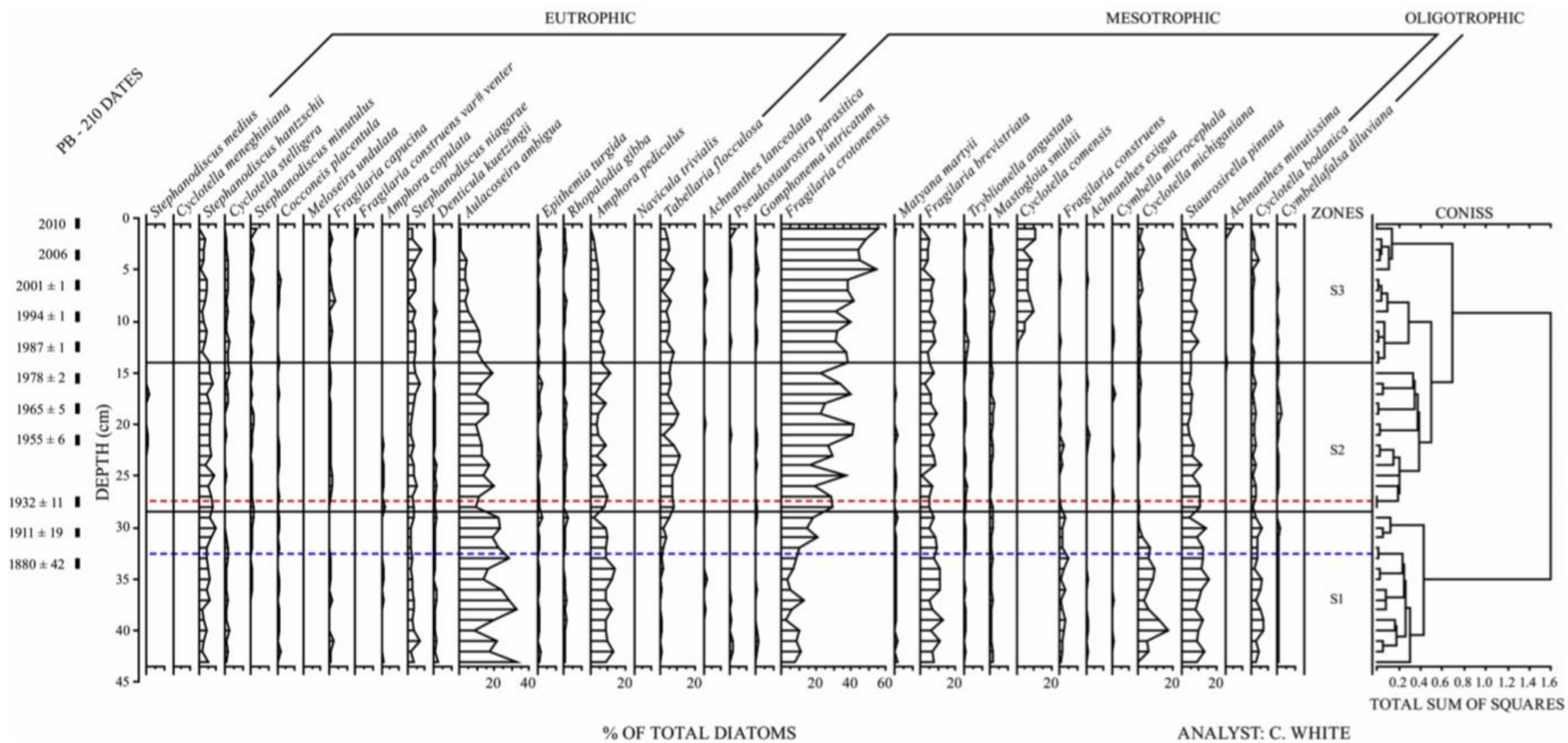

Figure 3. Sedimentary diatom assemblages in Clear Lake from White (2012). The Blue dashed line represents establishment of a timber reserve in the Clear Lake watershed (1895), the red dashed line represents the establishment of RMNP (1933).

White (2012) provides analysis of historic diatom communities which sheds light on past trophic conditions within Clear Lake. This research offers an understanding as to what the historical trophic status of Clear Lake was prior to European settlement and anthropogenic disturbances. Based on diatom community assemblages, White concluded that the oldest zone corresponded to a mesotrophic lake (Figure 3). Prior to 1895, Clear Lake was mesotrophic, during which time little European settlement had occurred. 1900-1987 corresponds to the establishment of RMNP (1929-1933) and settlement and development within Clear Lake's watershed. During this time, changes to community structure strongly suggest eutrophication of the system (White, 2012). White added the caveat that although current diatom communities are markedly different from historical, this is not specifically the result to changes in TP concentrations and may be reflective of other changes to environmental conditions within the lake.

Bajikov (1932) presents the earliest records of zooplankton plankton studies on Clear Lake. Although densities are not available, Bajikov noted several species of plankton not found in other prairie lakes. These included *Holopedium gibberum*, *Rhoicosphenia curvata, Lophocharis salpinal*. Bajikov also noted the abundance of palatable zooplankton for game fish including: *Daphnia longispina, Cyclops bicuspidatus*, *Leptodora kindii,* among others.

Clear Lake is the only deep, meso-oligotrophic lake in RMNP, as such it presents the highest species diversity and complexity in the park (Briscoe et al., 1979). There are five major native fish species in Clear Lake; northern pike, lake whitefish (*Coregonus clupeaformis*), yellow perch (*Perca flavescens*), white sucker (*Catostomus commersonii*) and cisco (*Coregonus artedi*). Of these, northern pike are the most important native fish from a fisheries perspective. In the 1950's, northern pike were the only sought after species in Clear Lake, as at the time stocking efforts of other species had been largely unsuccessful (Rounds et al., 1992). Lake whitefish have been fished in the past, as have yellow perch (Heap, 1988). There has been concern over the norther pike population for several decades in Clear Lake, although reports vary concerning historical populations. There are six species of fish in the Clear Lake-South Lake complex which are unique to the watershed. These include: lake whitefish, cisco, spottail shiner (*Notropis hudsonius*), blacknose shiner (*Notropis heterolepis*), trout perch (*Percopsis omiscomaycus*) and slimy sculpin (*Cottus cognatus*) (Briscoe et al., 1979). Clear Lake has cold, unproductive and deep water which make it a unique environment in RMNP for these species.

A total of 6 species of fish were introduced in Clear Lake in the early-mid 20th century. Walleye were stocked from 1923 to 1968 and have established a permanent population in Clear Lake. Walleye is the most popular recreationally fished species in Clear Lake (Heap, 1988). Early studies on introduced walleye populations estimated one third of walleye spawn in South Lake (Maclean, 1979).

This estimation has been called into question by later studies which found no evidence of walleye spawning, eggs or fry in South Lake (Rounds et al., 1992). Adult walleye are commonly observed moving from Clear Lake into South Lake while the channel is open, and other studies indicate walleye spawning in South Lake (Heap, 1988). Walleye age classes have been observed to be stronger in years in which the channel between South Lake and Clear Lake was open. South Lake is an optimal depth for Walleye spawning which prefer shallow waters (Hartman, 2009)

Lake Trout (*Salvelinus namaycush*), Rainbow Trout (*Oncorhynchus mykiss*), Splake (*Salvelinus fontinalis* × *Salvelinus namaycush*), Muskellunge (*Esox masquinongy*), and Brook Trout (*Salvelinus fontinalis*) were all unsuccessfully introduced into Clear Lake between 1923 and 1968 (Benson, 1984). Predation by native fish were cited as the primary cause for stocking to fail (specifically predation by northern pike, yellow perch and white suckers; Rounds et al., 1992).

3.3 Clear Lake - Nutrient and Chemical Characteristics

Clear Lakes water is slightly alkaline, 8.4-8.8 (Bajikov, 1932; Bergman, 1987). This slightly alkaline condition is characteristic of many lakes in Manitoba (Bajikov, 1932). Naturally low concentrations of N and P contribute to Clear Lakes minimal plant and algal growth and ultimately its meso-oligotrophic state (Bergman, 1987). Chlorophyll-a concentrations in Clear Lake between 1978-1988 remained below 5 ug/L (Patalas et al., 1988).

P is the limiting nutrient in Clear Lake over 95% of the summer, and severe P limitation occurs over 72% of the summer (Hawryliuk, 2000). Interestingly, a higher concentration of TP is in Clear Lake than would be expected based on water transparency, indicating either that some of this P is not bioavailable or another process is influencing algal growth (Hoole et al 2005). Since Hoole et al's (2005) study used TP, it is highly likely that algae can only utilize the bioavailable orthophosphate fraction of TP, (see 2.1 Nutrient Monitoring). Another explanation is that zooplankton predation pressure is controlling appearance of algal growth within Clear Lake.

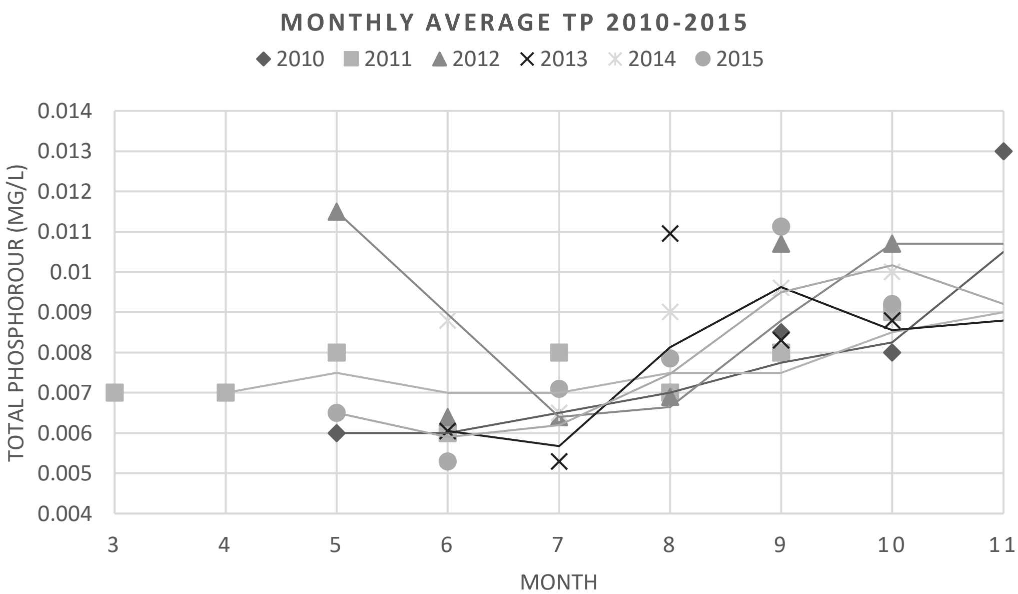

Figure 4. Average monthly TP concentrations between 2010 and 2015. Trendlines are moving averages for each year. TP concentrations are significantly higher in October than June each year (p=0.005).

A study into the P biogeochemical cycle within Clear Lake has shed light on the fate of P entering the lake. Clear Lakes epilimnion is supersaturated with calcite ($CaCO_3$). Algal blooms may result in calcite co-precipitation scavenging events (Whitehouse, 2010). During a significant algal bloom two processes will influence P concentrations within the lake. Biological assimilation by algae will decrease bioavailable P compounds within the water column. As the algal bloom continues, dramatic changes in dissolved oxygen and pH take place, altering the chemical conditions within the lake. Algae will consume carbon dioxide in the water, corresponding to an increase in pH (Whitehouse, 2010). Since chemical conditions directly surrounding an algal cell are most altered, each algal cell will act as a nucleation site for calcite precipitation. Direct co-precipitation of calcite crystals and P will scavenge more P from the water column, further decreasing P concentrations as the algal cells settle. Dissolved phosphorous concentrations within Clear Lake decreased by 60% in the hypolimnion during a 2008 algal bloom, indicating that these interactions can significantly decrease P concentrations within the lake (Whitehouse, 2010). Phosphorous concentrations increase over the summer months (Figure 4). A students t-test indicates that TP concentrations are significantly higher in October than in June each year (2010-2015; p=0.005). The trend of increasing P concentrations over the summer growing period is contrary to what is expected in natural lakes. In natural waters, P concentrations steadily decrease over the summer as macrophytes and phytoplankton grow. Over the winter, P concentrations should increase as plants die back and decompose (Wetzel, 2001). This indicates that Clear Lake is receiving excess nutrients over the summer. Further evidence for anthropogenic nutrient loading in Clear Lake comes from sediment cores in Clear Lake (Goldsborough & Rounds, 1992). Total phosphorous and total chlorophyll (TChl) analysis reveals a dramatic increase in concentrations of both analytes in sediment cores from Clear Lake (Figure 5). Both compounds remained relatively constant between 1810 and 1910, a significant upwards trend in sedimentary TP and TChl corresponds to increasing development and activity in RMNP.

Mercury contamination has been a source of concern in both Clear Lake and South Lake since the early 1970's (Kooyman, 1970). Mercury concentrations in fish tissues were detected above the maximum for human consumption (0.50 ppm) until at least 1989 (Hoole et al., 2005). Various potential sources for mercury contamination were investigated by researchers although a definitive source was never found (Glufka, 1992). Since mercury concentrations failed to subside during this time, several researchers have concluded the mercury is from natural sources (Glufka, 1992; Hoole et al., 2005). Further research should be conducted to determine if these mercury concentrations warrant concern over human health.

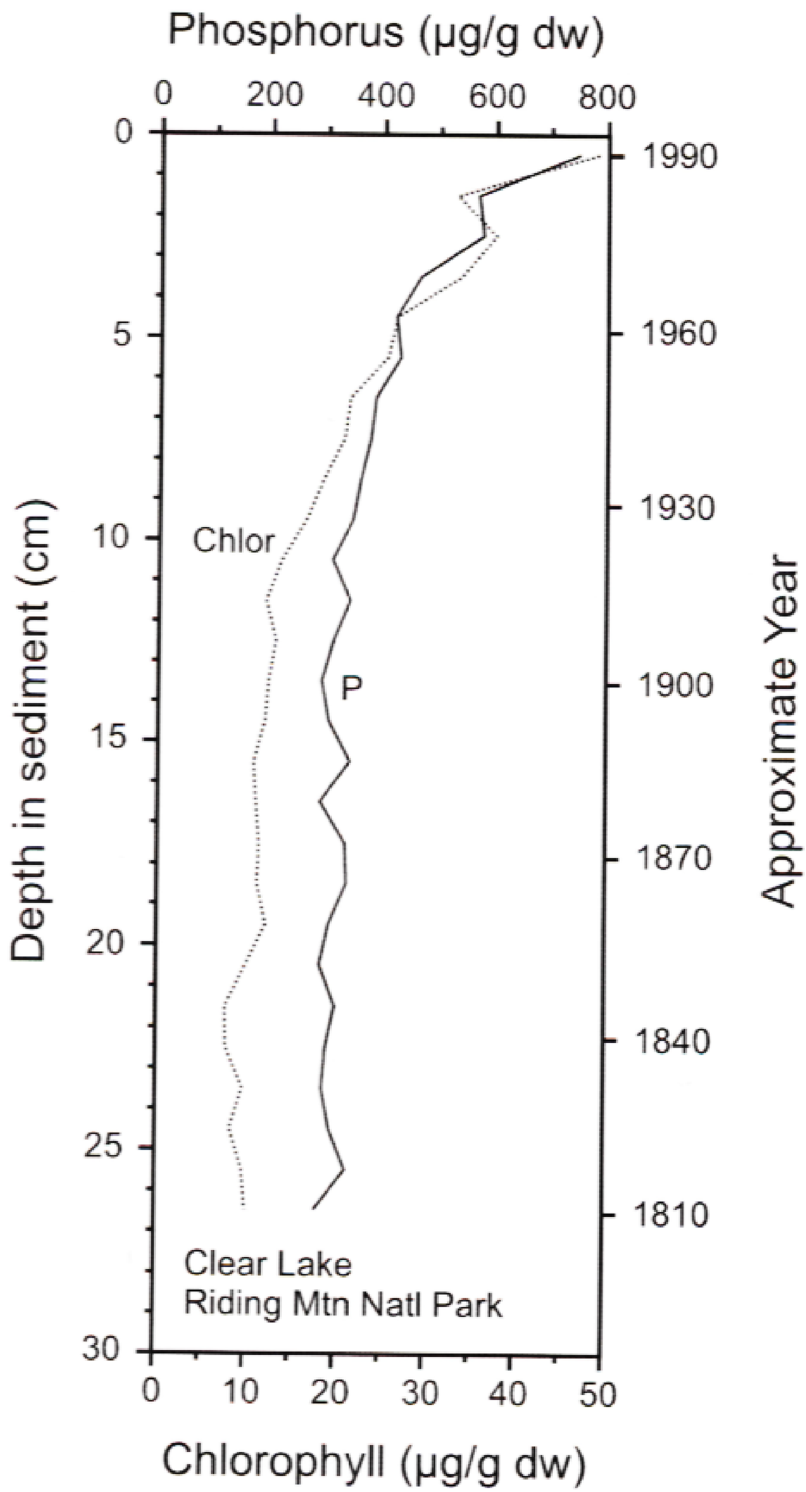

Figure 5. Total chlorophyll and TP concentration in a sediment core taken in Clear Lake. Both values remained relatively constant until the 1900s-1930s, this period corresponds to when activity in the park began to increase (Goldsborough & Rounds, 1992).

3.4 South Lake - Physical Characteristics

South Lake, the other lake examined in this study, has a maximum depth of 1.47 m, and a mean depth of 0.70 m (Robinson et al., 2003). With a surface area of 206.6 ha, South Lake is much smaller than Clear Lake. South Lake does not typically undergo thermal stratification due to its depth. South Lake has two distinct basins: a north and a south basin. Previous studies into the trophic status of South Lake concluded that the north basin is eutrophic and the south basin is hypereutrophic (Scott & Sellers, 2005). These two basins differ greatly in appearance and macrophyte community. South Lake is widely considered a recently removed bay of Clear Lake. Wave action and ice heave have been cited as potential causes of the cut-off, building up an isthmus of sediments (Briscoe et al., 1979; Glufka, 1992). This isthmus is discussed further in section 4.2 Disruption to Drainage Patterns. Heap (1988) reported overwinter dissolved oxygen concentrations in South Lake at 0 ppm and noted a strong hydrogen sulphide smell.

As discussed earlier, the largest sub-basin in the Clear Lake watershed is Octopus Creek, and is predominately located outside of the park (see 3.1 Clear Lake - Physical Characteristics; Scott & Sellers, 2005). Octopus Creek was redirected to Ominik Marsh and South Lake rather than directly entering Clear Lake in the 1960's (Neumann, 2013). The intended function of this diversion was for natural bioremediation of treated sewage effluent from the Wasagamin sewage treatment lagoons (still in use; Neumann, 2013). Prior to construction of this channel, South lake bore no surface water connection to Ominik Marsh.

3.5 South Lake - Biological Characteristics

South Lake has a dense macrophyte community along the shoreline and extensive algal growth (Bergman, 1987). Hutchinsons survey of common wetland vegetation throughout the park provides a useful inventory of characteristic vegetation in South Lake and RMNP (1981). Shallow marshes, such as sections of Ominik Marsh and South Lake, were characteristically dominated by sprangletop (*Scolochloa festucacea*), spike rush (*Eleocharis* spp.), mannagrass (*Glyceria striata*), and giant burred (*Sparganium eurycarpum*). Plants characterizing deeper marsh environments include cattail (*Typha latifolia*) and hardstem bulrush (*Schoenoplectus acutus*). Submergent plants are predominantly native water milfoil species (*Myriophyllum verticillatum* and *Myriophyllum exalbescens*) (Cody, 1988; Hutchinson, 1981). The north basin of South Lake is dominated by hardstem bulrush (Scott & Sellers, 2005).

As with Clear Lake, phytoplankton biomass peaks in September, following a steady increase throughout the summer months (Hawryliuk, 2000; Scott & Sellers, 2005). Phytoplankton communities within South Lake are representative of a transition between Ominik Marsh and Clear Lake.

Like Clear Lake, sediment cores have been analyzed to construct plankton community changes through time. Increased nutrient concentrations over the past 50 years have changed the algal community within both basins of South Lake (Scott & Kling, 2006). The cores documented a shift from mesotrophic taxa towards eutrophic characteristic cyanophytes and chlorophytes. This shift was strongest during the 1950s-1980s, during which time changes to nutrient loading and climatic shifts occurred (Scott & Kling, 2006). Rain fall during this time was above the 30-year mean, and likely lead to high water levels, interfering with light availability for submergent macrophytes (Scott & Kling, 2006).

Lake whitefish, northern pike, white sucker and yellow perch have all be observed in South Lake (Kooyman, 1970). Winterkills are common in South Lake, and as such fish species present in South Lake are akin to Clear Lake. The intermittent connectivity between Clear Lake and South Lake is a major factor in determining the fish communities of South Lake. Numerous sources speculate the importance of South Lake as spawning area for northern pike and walleye (see Briscoe et al., 1979; Maclean, 1979; Rounds et al., 1992).

3.6 South Lake - Nutrient and Chemical Characteristics

Nutrients primarily enter the north basin of South Lake through surface runoff. Due to its shallow depth, dense macrophyte community and high nutrient concentrations, South Lake is a eutrophic water body (Bergman, 1987). Sediment cores show a pronounced increase in biologically available phosphorous, suspended N and chlorophyll since 1950 (Scott & Kling, 2006). Phosphorous has been cited as the primary nutrient accelerating the eutrophication process (Bergman, 1987). The south basin is significantly more nutrient rich than the north basin, suggesting that it receives further nutrient additions from groundwater, particularly N (Robinson et al., 2003).

South Lakes pH is typically higher than Clear Lake (average 8.6 during sampling), and values exceeding 9 have been recorded (Rounds et al., 1992)

Mercury contamination has been a source of concern in both Clear Lake and South Lake since the early 1970's and is discussed in Chapter 3.3 Clear Lake - Nutrient and Chemical Characteristics (Kooyman, 1970).

Chapter 4 Ecological Stressors

4.1 Wastewater Treatment Facility

Riding Mountain National Park's wastewater treatment facility is a critical part of this restoration project. As such, it is important to understand the development of the wastewater treatment process used by Wasagaming. During the late 1950's and early 1960's, a single, unlined treatment cell (now Cell 3) was constructed out of in-situ soils. These soils were of glacial origin, and therefore porous sands, silty sands and loams (Belke & McGinn, 2003). The belief during construction was that the porous bottom of the cell would be plugged with settable solids within several years of operation. Treated greywater from the primary lagoon then passed down a 75 m long excavated ditch into the southern pond of Ominik Marsh. Baffles were installed along the length of the ditch to reduce flow velocities and slow accidental releases of effluent. Flow was then diverted from Ominik Marsh into South Lake via a 390 m drainage channel and a dam. Construction of the channel and dam redirected Octopus Creek's flow through the Ominik Marsh system and into South Lake. Historically, Octopus Creek would have drained into Clear Lake. At present, only a minimal volume of water flows through Octopus Creek's historic channel into Clear Lake. During the early 1970's, two additional treatment cells were constructed adjacent to the original cell. These additional cells (Cell 1 & 2) were also constructed with in-situ soils but were lined.

The only treatment of sewage prior to entering Cell 1 is the addition of enzymes to break down non-sewage materials (Reside B., Clear Lake Project Manager, RMNP, pers. comm. 2016). The macrophytes housed within the ponds are abundant mats of Hornwort (*Ceratophyllum demersum L*) and sparse colonies of Duckweed (*Lemna minor L*) and Horned Pondweed (*Zannichella palustrir L*). Various Typha spp and Carex spp line the borders of the lagoons.

In 2010, Wasagamin's sewage lagoons received a third upgrade. Existing cells had the organic sludge removed, and liners were installed. Aeration treatment, alum ($KAl(SO_4)_2 \cdot 12H_2O$) injection and open channel UV treatments were also added (Public Works and Government Services Canada, 2009). Alum is injected between cells 2 and 3 and removes P via flocculation. Water quality is monitored between cells 2 and 3. Average alum treatment efficiency based on available 2016 data is summarized in Table 1. TP and alum water quality data from cell 2 and cell 3 of the Wasagaming wastewater treatment facility, and calculated using Equation 1. Average TP removal efficiency was 1.28 (mg/L TP/mg/L Alum). This data assumes water in cell 3 is representative of cell 2 water + alum, and is merely to give an estimate of P removal.

Equation 1 $$Removal\ rate = \frac{Cell\ 2\ TP\ (\frac{mg}{L}) - Cell\ 3\ TP(\frac{mg}{L})}{Alum(\frac{mg}{L})}$$

Table 1. TP and alum water quality data from cell 2 and cell 3 of the Wasagaming wastewater treatment facility.

Date	Cell 2 TP (mg/L)	Cell 3 TP (mg/L)	Alum (mg/L)	Removal rate
Jul-16	3.48	0.848	0.681	3.86
Nov-16	0.506	0.610	0.850	-0.12
Dec-16	0.699	0.640	0.680	0.09

4.2 Disruption to Drainage Patterns

Surveyor notes from the 1890s show the isthmus between Clear Lake and South Lake, with a breach in the western most side (Figure 6). There have been rumors that the natural isthmus accretion has been supplemented by infilling for pathway or road construction (Reside B., Clear Lake Project Manager, RMNP, pers. comm. 2016). Evidence for construction in this case is slim, although there are many early documents which have not been digitized. The isthmus was at least present in the early 1930's as reports indicate no connection between the two lakes existed between 1930 and 1935 (Hoole et al., 2005). The isthmus, natural or otherwise, has been identified as a potential bottleneck for fish species reproducing in South Lake (Heap, 1988; Maclean, 1979).

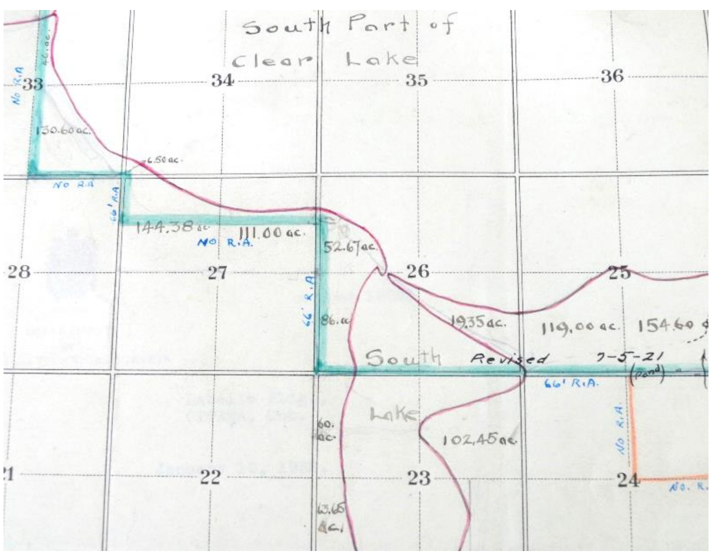

Figure 6. 1880 Surveyors map including the isthmus and breach on the westernmost side. Blue highlighter indicates historic park boundary. (Stewart, 1880)

Recommendations have been made in the past advocating Parks Canada ensure a channel through the isthmus is present during spring spawning and during fall. Opening the isthmus during fall would allow that year's fry to return to Clear Lake during the winter (Heap, 1988). Numerous attempts have been made to install a permanent opening between the two lakes including dredging and culverts, all of which have failed (Rounds et al., 1992). At present the isthmus opens and closes depending on water levels among other factors (Hoole et al., 2005).

The largest sub-basin in the Clear Lake watershed is Octopus Creek, which is predominately located outside of RMNP (Scott & Sellers, 2005). The Octopus Creek drainage system receives surface water runoff from the town of Onanole, rural and agricultural development and has exhibited eutrophic conditions due to anthropogenic inputs (Bergman, 1987; Hoole et al., 2005). Developments in the Octopus Creek sub-basin typically use septic tank systems and tile fields for wastewater disposal. As such, it is likely that seepage and contamination through problems with these systems will have effects on Octopus Creek and its drainage basin (Bergman, 1987). Nutrient inputs and pollution from these systems will ultimately have effects in Ominik Marsh and South Lake and control of these non-point contamination sources is difficult and unlikely. Historically Octopus Creek would receive additional water from Ominik Marsh and then directly enter Clear Lake. However, this natural flow pattern was disrupted by the 1960's construction of a berm redirecting much of the flow from Octopus Creek into Ominik Marsh and ultimately into South Lake (Neumann, 2013). The intended function of this diversion was for natural bioremediation of treated sewage effluent from the Wasagamin sewage treatment lagoons (still in use; Neumann, 2013).

Figure 7. A comparison of a historical air photo (1966; Left) and recent satellite imagery of the north basin of South Lake (right). The constructed channel connecting Ominik Marsh and South Lake is clearly visible in the recent image.

4.3 Introduced Species

Walleye have been the only successful fisheries introduction in Clear Lake to date. Although this population is not native to Clear Lake, eradication of the species is unlikely for several reasons. Walleye are the most sought after species in Clear Lake from a recreational fisheries perspective (Heap, 1988). As such it is likely not in the parks interest to eradicate a fisheries attraction. RMNP is within the natural range for walleye, and Clear Lakes water is conducive to their development (pH 6.0-9.0, DO >1.5, Temperature <29 °C) (Hartman, 2009). Therefore, walleye though not native to Clear Lake, are a common species in fish communities in the area. Walleye were however a new addition to the food web within Clear Lake, and as adults are full time piscivores, this likely had effects throughout Clear Lake. Young walleye, which are planktivores, more than likely altered the plankton community, especially during early stages in stocking. Competition for food may be an issue with other top predators (northern pike), though these interactions are complex and poorly studied (Hartman, 2009). Regardless, walleye have been present in Clear Lake since 1928 and have established a permanent population. No substantial disturbances to other fish species by walleye have been documented (extirpations/population depressions), and therefore walleye are not likely posing an immediate threat to the native species assemblages (Harvey, 2009).

Although RMNP is taking diligent precautions to prevent the introduction of zebra mussels (*Dreissena polymorpha*), the effects to Clear Lake and South Lake should be considered. Filter-feeding zebra mussels will usually increase water clarity through overgrazing phytoplankton (Geisler et al., 2016). As phytoplankton become depleted, shifts in the whole lake ecosystem become apparent. Increased water clarity may lead to macrophyte and benthic algae expansion. Selective feeding by zebra mussels can lead to Microcystis blooms, a toxic cyanobacteria (Vanderploeg et al., 2001). Despite enhanced water clarity, zebra mussel invasion can be accompanied by increased cyanobacterial blooms. In Western Lake Erie, SRP, ammonium and nitrate all increased following zebra mussel invasion, while TP remained constant (Holland et al., 1995).

4.4 Groundwater Pollution

The presence and effects of groundwater pollution from the Wasagaming sewage lagoons were studied by Rempel et al. in 1978. The wastewater treatment facility was found to be hydraulically and organically overloaded, with exfiltration rates estimated at 190,000 m^3 influent lost annually (Rempel, 1977; Rempel et al., 1978). Seepage from the original cell (Cell 3) was also cited as a concern during a 2003 investigation into the feasibility of lining the sewage lagoons (Dyregrov, 2003). Although Rempel at al. (1978) found that groundwater pollution of N was occurring, it was concluded that the pollution was not serious. This conclusion is problematic because groundwater samples were not tested for P concentrations. Groundwater pollution by P is often an unrecognized driver in eutrophication of freshwater systems. Since P generally doesn't leach into the groundwater from soils, anthropogenic sources are commonly the primary source of elevated groundwater P (Dokulil & Teubner, 2011). Rempel et al. (1978) identified that groundwater flows below the wastewater treatment facility in a North-West direction towards South Lake. There is also high likelihood that exfiltration from the lagoons enters directly into Ominik Marsh (Bergman, 1987). Vanderschuit (1992) speculated that exfiltration from the sewage treatment system reaches Ominik Marsh and is responsible for increasing the rate of eutrophication.

Elevated concentrations of chloride (Cl^-) have been reported in Clear Lake (Patalas et al., 1988). Palatas et al. sampled chloride values in 6 RMNP lakes between 1978-1988 and found highest concentrations in Clear Lake (mean 3-4 mg/L) (1988). High chloride concentrations were also observed at other RMNP Lakes: Moon Lake, Lake Greyling, and Lake Audy, all of which have chlorinated water wells near by. Water wells in the park are chlorinated periodically based on total coliform bacteria test results (Vanderschuit, 1992). Figure 8 shows Patalas et al. chloride concentrations at Clear Lake Station 1 (1988). The trend line (R^2=0.0813) is very weak and suggests no long term trend in chloride concentrations during the sampling period. No further sampling of chloride in Clear Lake has been done to date.

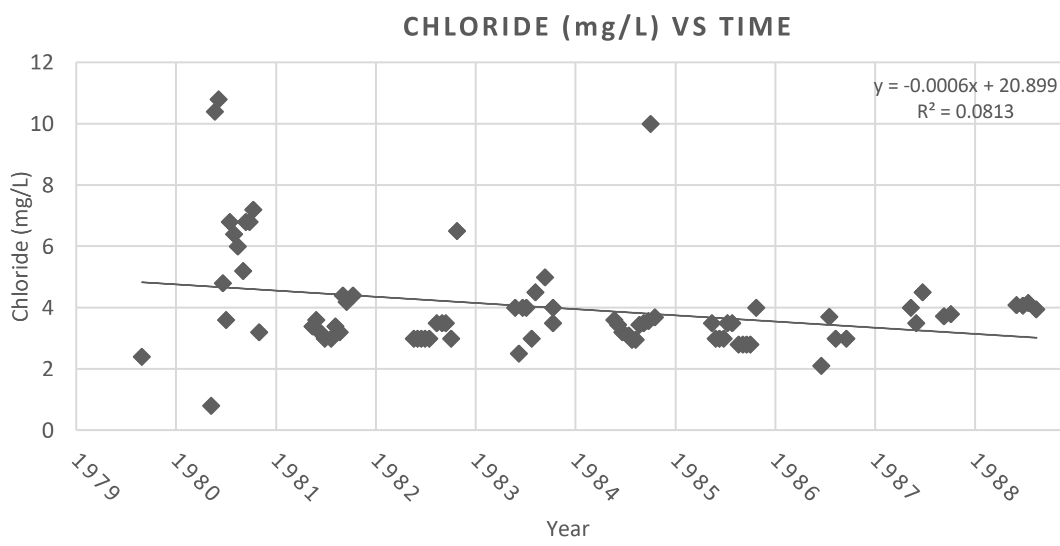

Figure 8. Chloride readings (mg/L) over the ten-year sampling period from (Patalas, Salki, & Stainton, 1988) Y=20.09-6.394e-9*X, R2=0.0813. There is no annual trend in chloride concentrations during the study period. Data accessed from RMNP water quality database (Riding Mountain National Park, N.D.).

4.5 South Lake Alternate Stable State

An investigation into food chain dynamics within South Lake and Ominik Marsh noted that provincial orthographs indicate the south basin of South Lake was less turbid than present. High concentrations of nutrients, in addition to wind mixing may have led to the decline of macrophyte communities in the basin, resulting in light limitation (Scott & Sellers, 2005). Evidence for a shift from a macrophyte dominated basin to a phytoplankton dominated basin is further exemplified by the results of sediment cores. Changes to water depth may have facilitated the transition towards the present turbid phytoplankton dominated system (Scott & Kling, 2006). The dense macrophyte communities found in Ominik Marsh and the north basin seem to outcompete phytoplankton for nutrients, resulting in clearer water despite high nutrient concentrations (Scott & Sellers, 2005; Scott & Kling, 2006). The south basin likely had similar conditions to the north basin prior to the shift towards suspended phytoplankton dominance.

Chapter 6 – Water Quality Monitoring Study

6.1 Introduction

A surface flow water quality study for Clear Lake was conducted between August 2016 – March 2017. The purpose of this study was to track concentrations of nutrients in Clear Lake and South Lake, as well as their inputs and outputs. By knowing specific sources of biologically available nutrients (SRP and DIN), specific restoration actions can be developed to target these sources. A nutrient budget was developed in 2005, but measured TP (Hoole et al., 2005). While this allows a general understanding of nutrient dynamics in the Clear Lake watershed, it is a poor indicator of bioavailable P (see 2.1 Nutrient Monitoring).

The sampling period only covered fall and winter months due to project time constraints. An entire year of sampling is strongly recommended to accurately gauge the flux of nutrients in both lakes. Previous nutrient budgets showed highest variability in nutrient inputs during June-September. Since sampling for this study began only at the end of August, a large portion of this variability was missed. Further, this nutrient budget neglects groundwater contributions regarding nutrients and water inputs. Groundwater sampling was omitted from this study due to logistical constraints. A detailed water budget was created in 2013 which found that groundwater is a major source of water to Clear Lake (Neumann, 2013). Hoole et al., (2005), also omitted groundwater and South Lake's contributions from their nutrient budget. Instead, they back-calculated contributions from these sources and found a significant contribution existed. 6.7 tonnes of TP/year in Clear Lake were attributed to groundwater and South Lake additions combined (Hoole et al., 2005). Should RMNP continue sampling SRP and DIN to complete the nutrient budget, I strongly recommend groundwater be included.

This sampling design also failed to include stream flow data, which is crucial to accurately understand the mass of nutrients flowing into Clear Lake or South Lake. This data was neglected due to a misunderstanding in the procedure for developing nutrient budgets. However, the data collected is still beneficial because it can be compared to results from previous studies. In addition, a sub-basin discharge ranking system was developed based on estimated runoff. These values were based on mean annual precipitation, and soil cover types (as outlined in Cronshey, et al., 1986).

6.2 Methods

Water samples were collected every two weeks during summer months (August & September). Come fall equinox (September 22nd, 2016), samples were collected every three weeks. Following December 22nd, 2016 samples were collected once per month. During the winter (and fall to a lesser extent), primary production is much slower than summer, so samples can be taken less often. 26 sampling locations throughout Clear Lake and South Lake were selected based on a stream survey done by RMNP Staff. Several in-lake sample site locations were selected based on previous usage

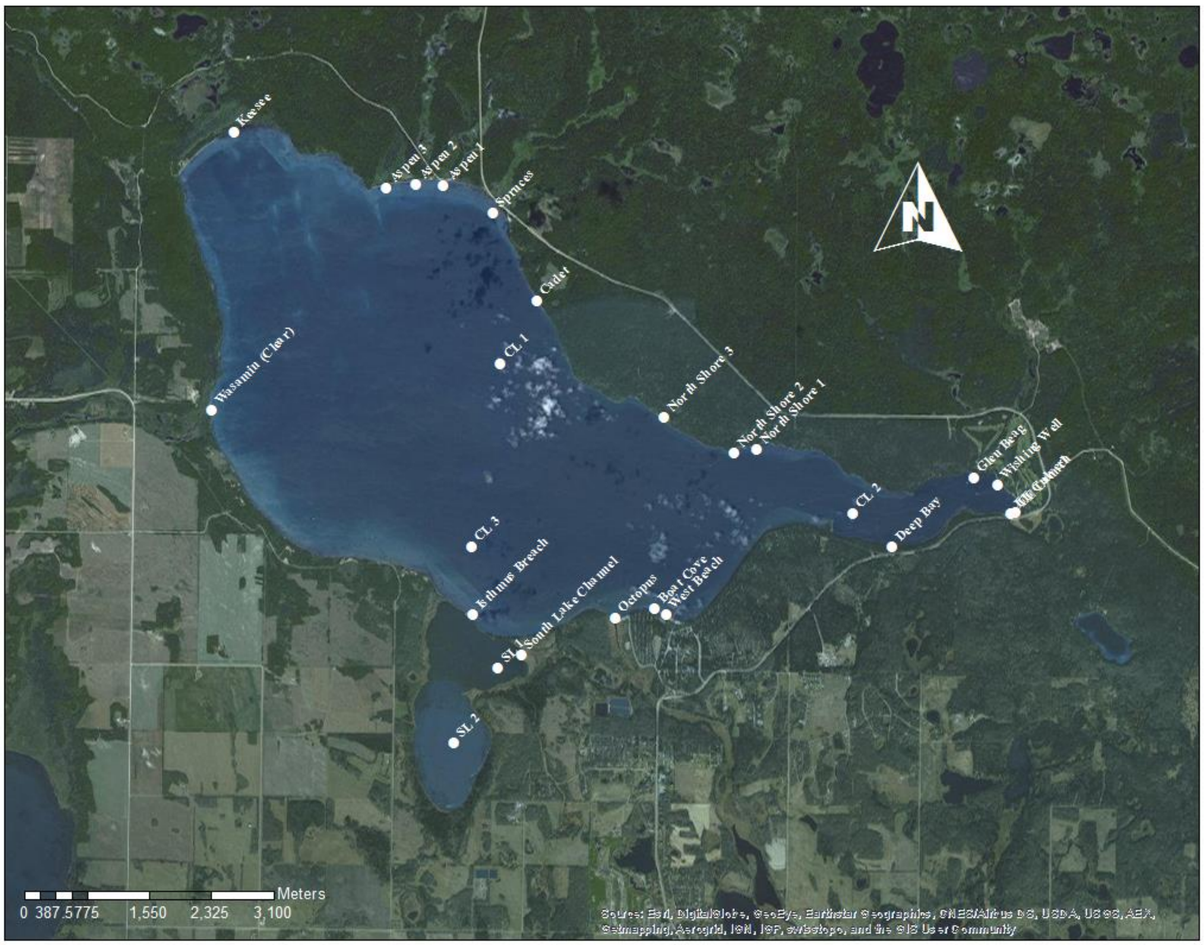

Figure 9. Stream sampling locations and in-lake sample locations. Three in-lake sample sites are in Clear Lake (CL), two are in South Lake (SL).

as water quality sampling sites by Parks Canada and previous researchers (CL 1, CL 2, SL 1, SL 2; Figure 9). Each sampling location had GPS coordinates as well as descriptions for access for use by Parks Canada employees during sampling. Samples were taken either instream or by boat when required.

Instream samples were taken mid-stream, away from the shore (safety permitting) using bottles provided by the water quality testing laboratory ALS Global. Samples for SRP and nitrate/nitrite testing were collected in 500 ml bottles, and did not need field preservation. Samples for ammonia were collected in 250 ml amber bottles, and required field preservation with sulphuric acid (H_2SO_4). Samples collected were then immediately placed in a cooler (provided by ALS Global) and mailed to the ALS Global Laboratory.

Three samples were collected at each in-lake sample site at depths of 0.5 m from the surface, at the thermocline and 0.5 m above the lake bottom using a Kemmerer sampler. All in-lake samples were accompanied by dissolved oxygen, temperature, pH and conductivity readings (using a YSI multiparameter). The YSI multiparameter was also used to determine the location of the thermocline based on temperature readings. Disruptions to sampling are noted in Appendix E. -H.

This sampling procedure was designed based on the needs of the water quality testing laboratory, ALS Global and the recommended field sampling guidelines released by the government of British Columbia (Ministry of Water, Land and Air Protection; Province of British Columbia, 2013). Water quality sampling guidelines specific to Manitoba were not available.

Samples with nutrient concentrations below detection limit were adjusted to the median of the detection limit and 0, per Table 2.

Table 2. Detection limit and corresponding adjusted value. Nitrate & Nitrite concentrations were reported together. DIN had a detection limit adjusted value of 0.04 mg/L.

Parameter	Detection Limit (mg/L)	Adjusted Value (mg/L)
Ammonia	0.010	0.005
Nitrate + Nitrite	0.070	0.035
Orthophosphate	0.010	0.005

6.3 Results

DIN in Clear Lake was significantly higher in the hypolimnion than in the epilimnion ($p=0.043$). The average epilimnetic DIN concentration was 0.045 mg/L, metalimnetic DIN was 0.047 mg/L and hypolimnetic DIN was 0.065 mg/L. Increasing DIN concentrations (particularly ammonium) with depth are indicative of eutrophic conditions (Wetzel, 2001). Ammonium (NH_4-N) accumulates in the hypolimnion of stratified lakes when large amounts of sedimentation occur and with anoxic conditions. Aerobic conditions in the hypolimnion will allow bacterial nitrification of NH_4^+ to NO_2^- and NO_3^- and absorption into the sediments. Under anoxic conditions absorbed ammonium will be released back into the water column. Since hypolimnetic oxygen deficiency was observed in Clear Lake, this process is likely responsible for high DIN concentrations in the hypolimnion.

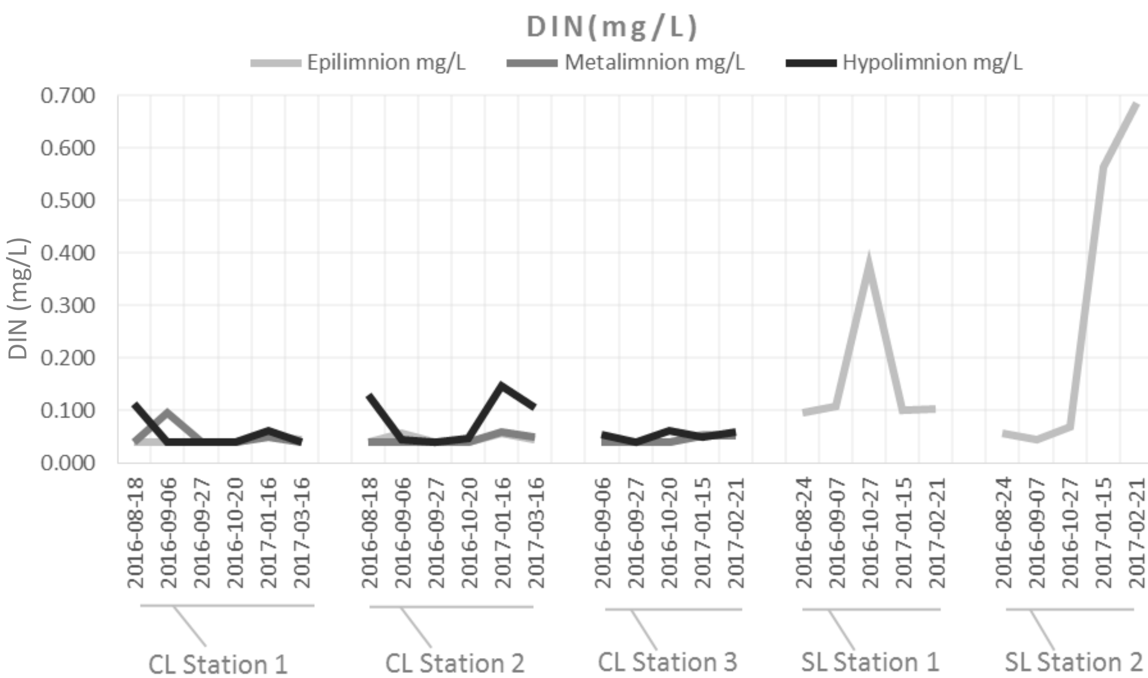

Figure 10. DIN concentration at each station over the sampling period. Hypolimnetic DIN in Clear Lake was consistently higher than epilimnetic or metalimnetic DIN. South Lake does not stratify therefore only has an epilimnion.

Four creeks did not flow during sampling: Keese, Cadet, Isthmus Channel and North Shore #2. Mean DIN concentration for all sampled rivers (0.117 mg/L) was similar to the mean literature concentration of unpolluted rivers (0.120 mg/L; Wetzel, 2001). A one-way ANOVA was conducted to compare the effect of each river on mean DIN concentrations. There was a significant difference between streams ($F(13,39)=7.3$, $p=5.49*10^{-7}$; Figure 11)c. This means that although the mean for all the streams was similar to literature values, certain streams had significantly higher or lower mean concentrations. This is expected as these streams all have different sources and land use and therefore should have different nutrient concentrations. Glen Baeg, West Beach, North Shore 3 and South Lake Channel all consistently had DIN concentrations above the mean for all streams. Glen

Baeg is very close to the golf course on the northeast corner of Clear Lake, therefore fertilizer use for the golf greens is a possible explanation for above average DIN concentrations. Glen Baeg had the highest DIN concentration measured during sampling (0.448 mg/L), although this does not mean Glen Beag was the greatest source of DIN to Clear Lake. West Beach is located very close to Wasagaming and therefore likely reflects comparatively more urbanized conditions. It is unknown why North Shore 3 had high DIN concentrations. North Shore 3 is located within the North Shore sub-basin which has private developments. The nearest development is nearly 400 m away from the stream and is close to the shoreline. Leaking septic systems is a possible explanation but given the distance from the stream is unlikely. South Channel should be expected to have high DIN concentrations because it drains Ominik Marsh into South Lake and receives wastewater from the Wasagaming wastewater treatment facility.

Aspen 1, 2, 3, and Spruces Creek were all consistently below the mean DIN concentration. These streams have sub-basins in the northwest corner of Clear Lake. This area is mostly undeveloped, so low DIN concentrations are expected.

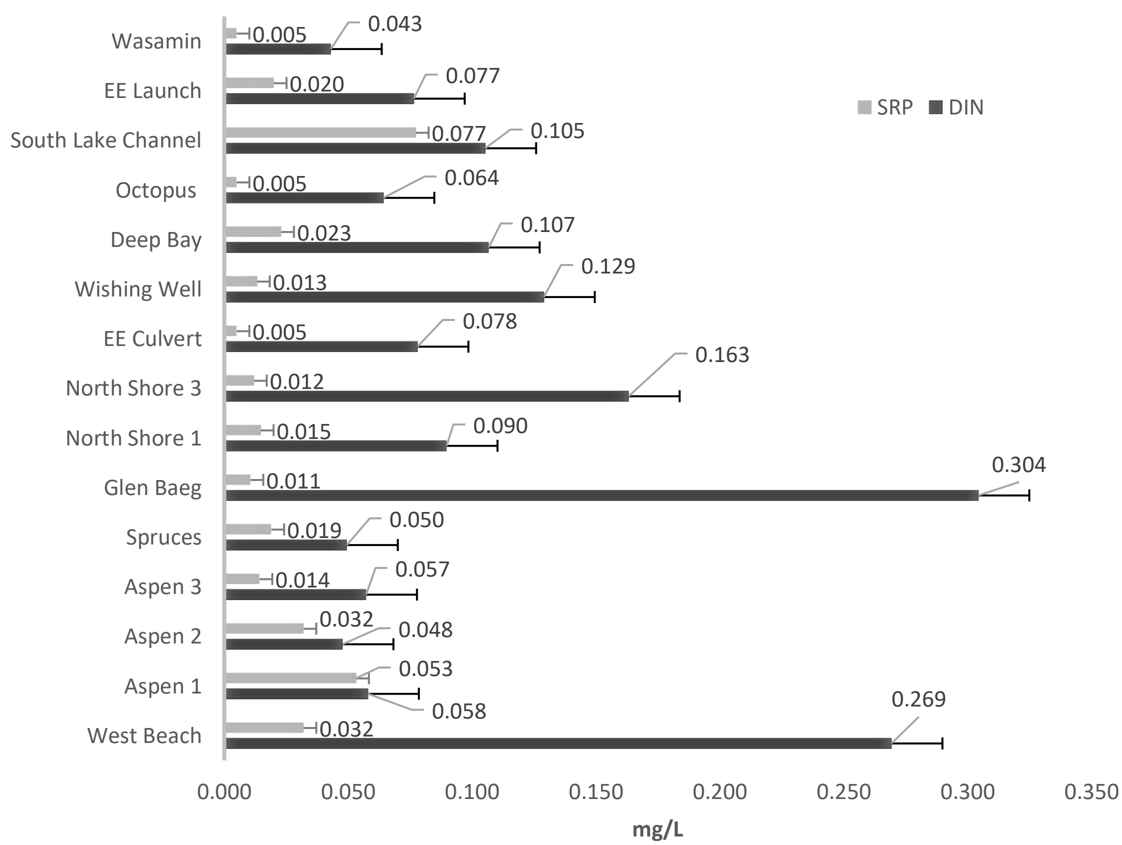

Figure 11. Average concentration of DIN and SRP for each stream during the sampling period. Error bars indicate standard error.

Since Clear Lake is P limiting 95% of the time, SRP is the most concerning analyte in this study (Hawryliuk, 2000). Mean influent stream SRP concentrations were 0.023 mg/L. Average unpolluted stream SRP concentrations worldwide are 0.01 mg/L (detection limit for this study was 0.01 mg/L) (Wetzel, 2001). As was the case for DIN, individual stream SRP concentrations varied considerably amongst the mean ($F(13,39)=6.48$, $p=2.5*10^{-6}$). South Lake Channel was the greatest source of nutrients (mean 0.062 mg/L). This was expected since the channel carries water from Ominik Marsh which receives the effluent from the Wasagaming wastewater treatment facility. This is the only channel entering the eutrophic South Lake, although surface runoff and groundwater also contribute. Hoole et al., (2005) also concluded that South Lake was a significant source of nutrients to Clear Lake. The maximum concentration of SRP sampled was 0.108 mg/L in the South Lake Channel. Although flow rates for the South Lake Channel are unknown, this is a substantial concentration of SRP, which flows into the eutrophic North Basin of South Lake. SRP concentrations associated with agricultural runoff range from 0.05 mg/L to 0.1 mg/L. Municipal waste effluent can exceed 1 mg/L, but given the small population of Wasagaming 1 mg/L is excessive. Aspen 1 had the second highest concentration of SRP (mean 0.047), though with estimated runoff, this is not a significant input to Clear Lake. Aspen 1 shared a diminishing trend in SRP concentrations over sampling which was observed in many of the streams.

Spruces Creek, Glen Baeg and Wishing Well (Bogey) creek, are the most significant water sources to Clear Lake when the isthmus is closed (Neumann, 2013). Although Glen Baeg has the highest DIN concentrations, SRP concentrations were consistently below the average for all streams. Wishing Well (Bogey) and Spruces Creek both had normal DIN and SRP concentrations. Average DIN for Wishing Well and Spruces Creek were 0.129 and 0.050 mg/L respectively. Average SRP concentrations were 0.013 and 0.018 mg/L respectively. Wasamin creek (the only outflow) was consistently below detection limit for both DIN and SRP. This should be expected as it reflects the nutrient concentrations and biological uptake within Clear Lake.

Using a sub-basin GIS file and equations 2-4, I estimated the runoff coefficient for each sub-basin draining into Clear Lake (Riding Mountain National Park, N.D). Since these formulas were developed by the U. S. Department of Agriculture, units were converted to imperial (Cronshey, et al., 1986).

Equation 2 $$Q = \frac{(P-I_a)^2}{(P-I_a)+S}$$

Equation 3 $$I_a = 0.2S$$

Equation 4 $$S = \frac{1000}{CN} - 10$$

Where *Q* is runoff (in), *P* is rainfall (in), *S* is potential maximum retention after runoff begins and *I* is the initial abstraction (in). *CN* is a ‘curve number’ which is estimated based on soil cover type, vegetation and the porosity of the soil. Vegetation and Soil cover type were approximated based on satellite imagery. It was assumed soils in RMNP had a 'moderate' porosity.

A runoff ratio was then calculated for each sub-basin based on the estimated runoff and total mean annual precipitation over RMNP. The runoff ratio was then multiplied by sub-basin area and the total mean annual precipitation to yield a total annual estimated runoff. From this I calculated the percentage contribution for each sub-basin to Clear Lake. Factors which influence the percentage contribution include: land use, vegetation cover and drainage basin size. Multiplying the percentage of total runoff by the average DIN and SRP concentration in each sub-basin output yields an estimated percentage contribution of each SRP and DIN to Clear Lake by sub-basin (Figure 12). This assumes that the sampling period was comparable to mean annual concentrations and that rainfall is evenly distributed throughout the year. Though both are inaccurate assumptions, without stream flow data these do allow for comparisons between each sub-basin. Octopus Creek was combined with South Lake due to the connectivity from the South Lake Channel. A table containing this data can be found in Appendix I.

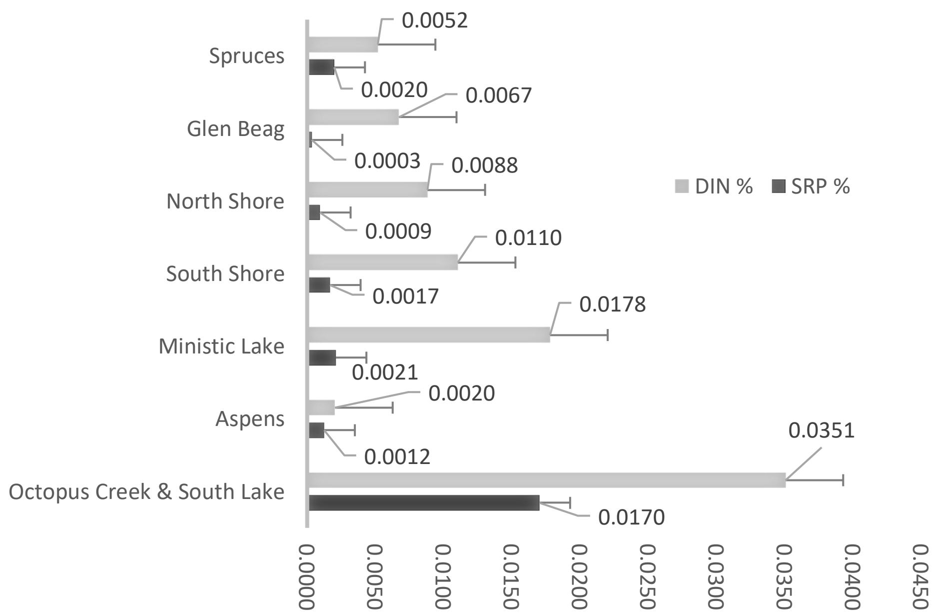

Figure 12. Relative sub-basin inputs to Clear Lake based on estimated runoff calculated by methods outlined by Cronshey, et al. (1986). Octopus Creek and South Lake were combined based on their connectivity through the South Lake Channel.

Octopus Creek and South Lake clearly deliver much more SRP and DIN to Clear Lake than the other basins, this is both a product of drainage basin size and increased nutrient loading in the basin by anthropogenic activity. Ministic Lake sub-basin corresponds to Wishing Well creek, and the drainage basin includes the golf course, which could explain the high DIN values. South Shore basin includes surface runoff from Wasagaming (West Beach) as well as Deep Bay, EE launch and EE Culvert. Although Glen Beag had the highest concentrations of DIN, factoring in the sub-basin size and runoff illustrates that nutrient inputs from Glen Beag are comparatively small. Sub-basins without defined channels were excluded from this analysis as DIN and SRP concentrations were not taken for overland flow or groundwater. Estimated percentage of total runoff can be seen for these basin in Appendix I. The total sum of runoff from these basins is estimated at 15.6% of total estimated runoff.

DIN in Clear Lake was often below detection limits, though one trend was most evident at station 2. Hypolimnetic DIN concentrations were significantly higher than epilimnetic concentrations throughout sampling ($p=0.043$). This trend is most evident when the lake was stratified both during the summer and winter. This corresponds to hypolimnetic oxygen deficiency, where ammonium (NH_3-N) is released from the sediments under anoxic conditions.

All Clear Lake samples for SRP were below detection limit. Since P is the limiting nutrient in Clear Lake low concentrations are expected. The detection limit for SRP was 0.01 mg/L, which is too high to accurately track SRP concentrations in a meso-oligotrophic lake. TP ranges from <0.001 mg/L to 200 mg/L in unpolluted waters, with less than 0.01 mg/L expected in meso-oligotrophic lakes (Wetzel, 2001). Generally, SRP is <5% of TP, so SRP values below 0.0005 mg/L can be expected if Clear Lake was meso-oligotrophic. Evidently, this expected value is far below the detection limit used in this study. The average TP concentration in Clear Lake is 0.026 mg/L, assuming SRP is 5% of TP, SRP concentrations should be expected to average 0.0013 mg/L. This number is purely speculative, and is just to illustrate the need for more accurate SRP testing. South Lake however, was above the 0.01 mg/L detection limit on October 27th 2016 with a SRP concentration of 0.011 (averaged between the north and south basins). One sample in the north basin (February 21st 2017) had a lower detection limit of 0.001 mg/L, this sample was measured to be 0.0034 mg/L SRP.

Average pH in Clear Lake during sampling was 8.36, which aligns with previous studies (Bajikov, 1932; Bergman, 1987). Dissolved oxygen followed a negative heterograde profile, as is expected based on historical data (see 3.3 Clear Lake - Nutrient and Chemical Characteristics).

Chapter 7 – Options Analysis

7.1 South Lake Restoration Options

7.1.1 Overview

Maintaining the meso-oligotrophic status of Clear Lake is a priority for RMNP. Nearly a century of manipulation has cumulatively led to degradation of water quality. To properly restore an degraded site, it is important to focus efforts not only on the symptoms but also the cause. In the case of Clear Lake, ameliorating factors negatively effecting water quality within the lake should be prioritized (ie. the cause). Symptoms in this case include algal blooms, loss of diversity and poor water quality. Excess nutrient concentrations, particularly P, are responsible for long term shifts in Clear Lakes water quality. Reducing these additional nutrient loads is a simple (in theory) approach to restore historic water quality. South Lake has long been known to be a significant source of nutrients to Clear Lake (Hoole et al., 2005). Shifts In phytoplankton and macrophyte communities within South Lake (the south basin) suggest South Lake has presented the effects of anthropogenic nutrient loading earlier than Clear Lake. In this degraded state, nutrients are cycled more rapidly in the south basin, and fewer nutrients are permanently sequestered through macrophyte uptake and sedimentation. This lack of nutrient removal limits the efficiency that wetlands and shallow lakes have in nutrient removal. In this case, restoring South Lake is of interest to mediate the effects of anthropogenic nutrient loading in Clear Lake.

As discussed in **Error! Reference source not found.**, the two basins of South Lake exemplify the d ual steady state theory of shallow lakes (Scheffer et al., 1993; Scott & Kling, 2006). Since the two basins exhibit spatial heterogeneity, but are connected, we can assume that mixing in the south basin is slow, as compared to other fully turbid lakes (Scheffer & van Nes, 2007). There is a legacy of groundwater pollution, non-point source and point source nutrient loading into South Lake. The cumulative effects of which have resulted in a shift in equilibria towards a phytoplankton dominated system. Restoration of shallow lakes with shifted equilibria is usually difficult and more complicated than simply reducing nutrient inputs (Dokulil & Teubner, 2011; Scheffer & van Nes, 2007). Lakes that are simply eutrophied by anthropogenic forces can naturally recover by reductions to the nutrient loads (specifically P) given sufficient time (Schindler, 1974). In the case of shallow lakes in phytoplankton steady states, phytoplankton communities will persist despite changes to nutrient concentrations and reductions in external nutrient loading (Huser et al., 2016; Wetzel, 2001). If restoration aims at returning the south basin to its historical macrophyte dominated steady state, an examination into what conditions favour that state is warranted. As summarized by Wetzel (2001):

1. Basins with little wind and wave action have less resuspension of sediments. This is the primary cause of the phytoplankton dominated turbid state existing in the south basin.
2. Dense macrophyte growth sequesters available nutrients, reducing available quantities for phytoplankton leading to the anoxic sediment characteristic of wetlands. When sediments

become anoxic, bacterial denitrification processes dominate and N is lost from the system through degassing of N_2. N uptake from macrophytes, epiphytic phytoplankton and bacteria in the sediments further decreases the availability of N for suspended phytoplankton.

3. Refugia created by dense macrophyte communities allows for zooplankton communities to expand. The zooplankton further reduce phytoplankton communities through grazing pressure.
4. Macrophyte communities greatly enhance surface area available for epiphytic periphyton.

To summarize, creating conditions conducive to macrophyte dominance and re-colonization is the best option restore the south basin of South Lake. I suggest a variety of restoration techniques be implemented in an experimental manner to identify effective techniques and increase chances of whole lake restoration success. Reduction of P loading is a crucial step in any re-oligotrophication restoration project. For both basins of South Lake TP concentrations are between 25-100 mg/m^3, mean TP for eutrophic is 84.4 mg/m^3 (Wetzel, 2001). As such a public awareness and outreach program for nutrient loading is still warranted (see 7.4 Outreach & Awareness).

To return the south basin to a macrophyte dominated community with clear water, several compounding factors must be compensated for. Lack of macrophyte cover is the primary reason the south basin is turbid and phytoplankton dominated. Macrophytes create a physical barrier to wind mixing and therefore reduce sediment disturbance. Below are several proposed methods to disrupt sediment mixing or P concentrations in the south basin to allow macrophyte recolonization.

7.1.2 Implementation

1. **Transplantation of floating cattail mats and hardstem bulrush to the south basin.** Common Cattail (*Typha latifolia*) is among the 500-species listed under the Manitoba Noxious Weeds Act, despite being a native species (Province of Manitoba, 2010). Under current legislation, propagation of cattails is illegal. There is however an amendment act to restructure the listings of noxious weeds into a tiered system (Legislative Assembly of Manitoba, 2015). This amendment may change regulations on cattail control and listings. In the event regulations change, cattail transplantation is included in the discussion for South Lake. *Typha latifolia* is a native species to the area and should not be confused with their invasive relative, narrow leaf cattail (*Typha angustifolia*) (also listed as noxious). Fortunately, hardstem bulrush is not listed under the Manitoba Noxious Weeds Act and is abundant in the north basin of South Lake, making it an excellent substitute for common cattail. Transplanting emergent vegetation to the south basin can have several effects. Cattails are aggressive colonizers and have been known to dominate wetlands (Grace & Wetzel, 1981, Tanaka et al., 2004). The rapid reproduction of cattails is beneficial in that it can greatly increase the vegetation cover in the area in a relatively short time frame. Cattails and bulrushes are commonly used in treatment wetlands for non-point source nutrient pollution

and are effective at nitrate removal (Gebremariam & Beutel, 2008). Further, Cattails significantly reduce sedimentary dissolved oxygen concentrations as compared to bulrush (*Schoenoplectus & Scirpus spp.*), the most abundant emergent species in South Lake. Dissolved oxygen levels facilitate colonization of anaerobic microorganisms in the sediments which convert nitrate to N gas (N_2), further reducing N concentrations (Gebremariam & Beutel, 2008). Groundwater entering the south basin has elevated concentrations of N, so additional removal pathways will benefit the basin (Robinson et al., 2003). Emergent vegetation colonization of the shoreline and floating mats in the center of the basin will improve vegetation cover and nutrient uptake/removal in the south basin. Due to the size of South Lake, transplantation is not likely to have appreciable effects in the short term. Plentiful emergent and macrophyte communities will benefit the basin in the long term through these various nutrient removal pathways. The south basin is approximately 980 m across at its widest, and 1,650 m long (measured in ArcGIS, based on turbidity plume). Emergent vegetation along the south-eastern shoreline of the south basin would be exposed to the greatest concentrations of nutrients as that is where the lake and groundwater flow intersect. Previous reports characterizing the emergent vegetation community around the shoreline of South Lake focused only on the north basin, and found that only bulrush was present (Robinson et al, 2003). Transplanting cattail mats to the division between the south and north basins would reduce mixing between the two basins, and create a physical barrier to the algae plume entering the north basin. These should be anchored to establish stationary mats and prevent movement by wind (Figure 13. Proposed location of emergent transplants along the shoreline of the south basin of South Lake. Groundwater flows in a north-west direction, therefor the shoreline most exposed to polluted groundwater should be targeted (Hoole et al., 2005).

Gebremariam et al. (2008) found that cattails removed 175 mg N/m^2/d at N concentrations of 5 mg N/L, though literature values range from 100-1,000 mg N/m^2/d. Average TN concentrations for South Lake are 1.54 mg/L, though values as high as 10.6 mg/L have been recorded (Riding Mountain National Park, N.D.). Assuming a shoreline length of 1,600 m and a cattail colony 2 m wide yields a surface area of 3,200 m^2. With removal rates at the lowest literature value of 100 mg N/m^2/d, an estimated mass of 320 kg N/day can be removed by the cattails. Using Gerbremariam et al.'s conclusion that bulrush is 25% less effective at areal N removal, we can predict 240 kg N/day would be removed if the same size colony of bulrush was established. Literature values were based on constructed wetlands for nutrient removal and microcosm experiments, therefore we can assume a lower value than estimated would likely occur in South Lake. As such, these numbers are likely an overestimation, but illustrate the scale of N removal emergent plants have nonetheless.

Figure 13. Proposed location of emergent transplants along the shoreline of the south basin of South Lake. Groundwater flows in a north-west direction, therefor the shoreline most exposed to polluted groundwater should be targeted (Hoole et al., 2005).

Transplantation of hardstem bulrush can be done using variety of methods (Tilley, 2012). This species is most abundant in the north basin of South Lake, making up 100% of the emergent vegetation (Robinson et al., 2003). Established stands of bulrush will fill in a 400 cm^2 bare patch in a single growing season (Tilley, 2012). This means that collecting transplants from the north basin will rapidly recover should a minimum 60% of each 1 m^2 area be left. Plugs (wild or from a nursery) should be spaced 30-45 cm apart along the shoreline of South lake in water no deeper than 5 cm (Figure 13). These plants can withstand depths of 1.5 m, so colony expansion into deeper waters should be expected.

Should legislation change regarding common cattails, there are many potential donor sites for transplantation. Cattails and floating cattail mats are abundant in Ominik Marsh and line the sides of Ominik Pond. Cattails may also be sourced from other locations within RMNP or in the Octopus Creek basin. Curiously, *Typha* are often viewed as a nuisance species in recreational ponds. There may be opportunities with local landowners for transplantation.

Transportation of the mats would be easiest by breaking the mats into smaller pieces and floating them down the south lake channel into the north basin. Floating detached mats of cattails were observed to move over submerged vegetation without disturbing it, meaning roots are shallow and these mats should be easily transported (Robinson et al., 2003). There may be a necessity to cut rhizome structures anchoring stationary floating mats as these mats are much larger than their mobile counterparts. Cutting the mats can be accomplished using a variety of mechanical methods (manual & gas powered cutters). Transplantation of rooted cattails can easily be done manually, using a standard shovel to cut the root system around a stalk. The entire stalk can then be removed and replanted along the shoreline of the south basin.

To enhance effectiveness of this treatment option, emergent vegetation can be mowed on an annual basis. Mowing/harvesting of emergent stalks will reduce the release of nutrients back into South Lake by reducing natural decomposition (Grosshans, 2014). This will result in permanent removal of nutrients and allow for continued effectiveness of the restoration treatment. Cattail tissue is approximately 4% N and 0.14% P (dry weight) (International Society for Ecological Modelling, 1979). Since average dry weight for emergent vegetation is 0.21% of wet weight, we can assume, cattails are 0.0084% N and 0.000294% P wet weight. This means that for every ton of cattails harvested, 8.4 kg of N and 0.29 kg of P are permanently removed from South Lake.

2. **Installation of silt fencing to disrupt wind mixing.**
   Reducing or interrupting the effective fetch of South Lake will limit the ability for wind to perturb sediments. This allows suspended particles ample time to settle and improve water clarity. To re-establish macrophyte dominance, the factors limiting their growth must be

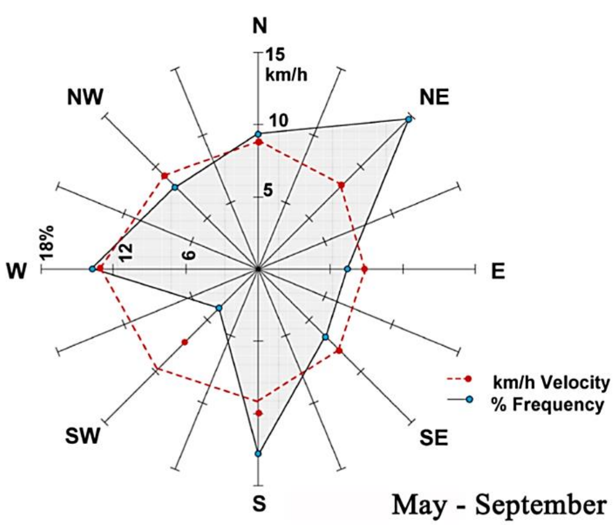

Figure 14. Average wind direction, velocity and frequency for Wasagaming. Data from Environment Canada 1994-2005. Easterly winds are strongest, and south-westerly winds are most prevalent. Figure modified from: Hoole et al., 2005.

ameliorated. Light limitation caused by the turbid waters prevents recolonization of submerged macrophytes. When wind blows across a lake, surface waves are produced. The motion within the water is complex, but the general pattern is as follows. A wave is passing through water with a depth greater than half its wavelength is known as a deepwater wave. Water molecules, and suspended particles at the surface will follow a circular path as the waves pass by. The radius of this path is equal to half the wave height, and decreases exponentially with depth (Carper & Bachmann, 1984). If the depth of water is less than half the wavelength of the deepwater wave, it will disturb sediments. The motion of the water molecules and particles becomes elliptical rather than circular and an oscillatory horizontal motion ensues, which can re-suspend sediments.

The strongest summer winds (May-September) are easterly, while most prevalent winds are south-westerly (Figure 14. Average wind direction, velocity and frequency for Wasagaming. Data from Environment Canada 1994-2005. Easterly winds are strongest, and south-westerly winds are most prevalent. Figure modified from: Hoole et al., 2005; Hoole et al., 2005). To disrupt wind circulation across the south basin, silt fencing can be installed perpendicular to these wind directions, to break up the motion of deepwater waves.

This method of restoration is a novel approach that has not been attempted in any other restoration project. As such, this method should be first attempted on a pilot project scale in South Lake. Other restoration options discussed below are costly, have had mixed success or limited case studies. Should silt fencing appreciably increase turbidity, it would be a novel and cost effective method for restoration of shallow water bodies dominated by wind mixing.

To calculate distance required between nets, we must examine what fetch is required for wind to disturb the sediments. A study into the effects of wind on a small prairie lake concluded that sediments become re-suspended when wavelength exceeds depth by a factor of 2. (Carper & Bachmann, 1984).

The fetch required to disturb sediments was calculated using the following formulae used by (Filstrup & Lind, 2010) and (Carper & Bachmann, 1984).

$$\textbf{Equation 5} \quad L = \frac{gt^2}{2\pi}$$

Where *L* is deep-water wavelength, *g* is the gravitational constant (9.8 m/s/s), *t* is the wave period (seconds).

$$\textbf{Equation 6} \quad \frac{gt}{2\pi U} = 1.20 \tanh\left[0.077\left(\frac{gF}{U^2}\right)^{0.25}\right]$$

Where *U* is the wind velocity (ms/s) and *F* is the effective fetch (m). Solving for *t* in Equation 7 and substituting yields:

$$\textbf{Equation 7} \quad F = \frac{28447 \times U^2 \times \tan^{-1}\left(\frac{0.332452 \times g \times \sqrt[2]{\frac{L}{g}}}{U}\right)^4}{g}$$

Based on Wasagaming's climate normal data, the averaged maximum sustained hourly wind speeds during the summer are 46 km/h or 12.77 m/s (Environment Canada, 2010). The maximum recorded hourly wind speed is 59 km/h or 16.4 m/s. Using the maximum recorded speed and a depth of 2.94 (twice the maximum depth), nets should be separated by 111 m. This spacing is based on the maximum recorded sustained wind speed, and is therefore closer than is required under most conditions. A spacing of 185 m apart would be adequate to disrupt deep water wave action of averaged maximum sustained hourly (46 km/h) winds.

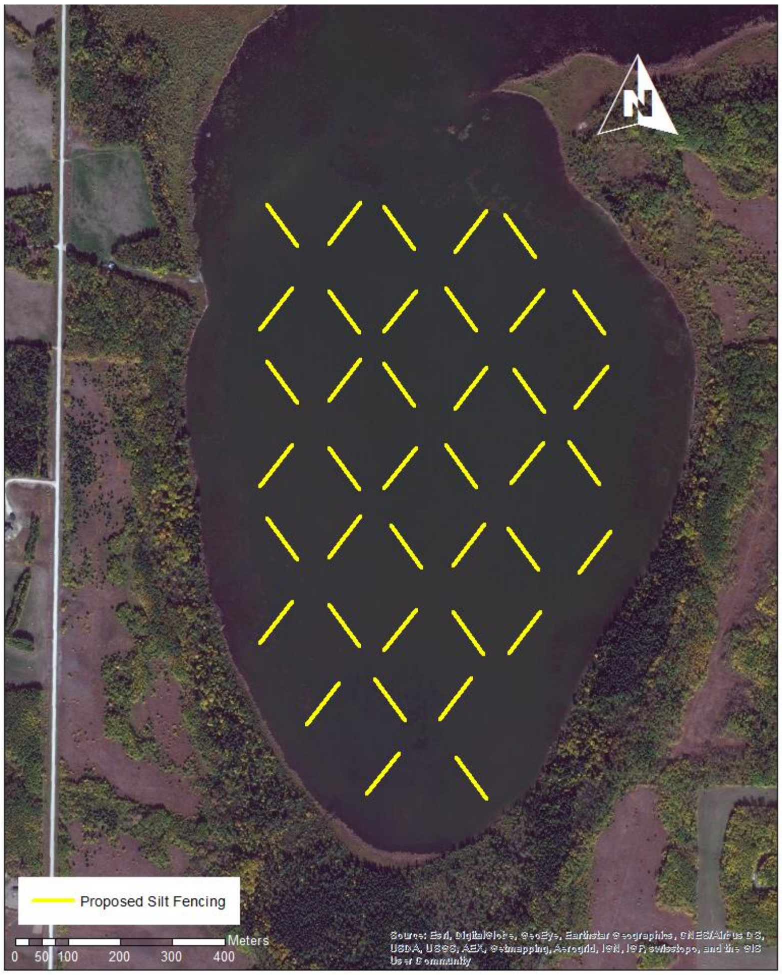

Figure 15. Proposed layout of silt fencing. This layout will disrupt mixing from all wind directions while allowing connectivity for fish movement.

100 m silt fences can be used in this approach, provided the fences are staggered. 100 m fences will not block movements of fish as much as longer ones will. These fences must be equipped with flotation devices and securely anchored to the bottom of South Lake. A potential configuration is pictured in Figure 15, fences are spaced 200 m apart. Wind from any direction will be disturbed by the fencing, making it effective in variable conditions. A pilot project should consist of only four fences in a box configuration. Monitoring for this pilot project is outlined in 7.1.4 Future Management Implications.

If this method is successful at reducing wind driven mixing but phytoplankton dominance persists, alum dosing could be used in conjunction with silt fences. Since the fences will reduce the capacity for alum flocs to re-suspend in the water column, this approach will be appropriate for P removal.

3. Chemical inactivation of Phosphorous: Options

Chemical treatments for restoration of eutrophic systems is often examined as a tool for P removal. These methods employ various chemical reactions to remove available nutrients (typically P) from the water column, rapidly reducing algal/phytoplankton communities. These methods generally are not permanent solutions and reapplication may be required depending on the treatment used (Lewtas et al., 2015).

Alum is commonly used as a P removal tool in wastewater treatment settings, as well as restoration of eutrophic water bodies (Lewtas et al., 2015). Since alum is already used in the Wasagaming wastewater treatment facility, it appears as an attractive restoration solution for the south basin. Welch and Cooke (1999) studied the reduction in TP and Chl-*a* of 7 alum treated lakes. TP concentrations were reduced by an average of 51%, with treatments remaining effective for 5-11 years. Chl-*a* showed similar trends. Unfortunately, alum is not suitable for lakes with significant wind driven mixing, such as the south basin (Egemose & Reitzel, 2013). Furthermore, alum treatments lose P sorption efficiency in high pH water bodies (Welch & Cooke, 2005).

Other P inactivation treatments such as modified zeolite clays have been shown to be effective sediment caps, eliminating sediment wind disturbance (Gibbs & Ozkundakci, 2011; Lewtas et al., 2015). The clay (marketed as Z2G1) binds with P and N in the water, settles to the sediment surface and forms a cap, preventing resuspension and nutrient loading. Since N concentrations are particularly high in the south basin, this method could ameliorate elevated P and N concentrations (Robinson et al., 2003). A pilot project concluded that a thin layer (1-3mm) of Z2G1 completely blocked release of sedimentary P. Further a small grain size was 50% more effective at P binding (<1 mm vs 1-3 mm) (Gibbs & Ozkundakci, 2011). Modified zeolite was used to treat the eutrophic Lake Okaro in New Zealand, though only marginally successful. Although the Lake did not fully recover to its historic trophic state, the Z2G1 treatment was more effective than other restoration treatments including alum, local farm management and constructed wetlands (Ozkundakci et al., 2010). Inadequate dosing and application methods were blamed for the failure (Ozkundakci et al., 2010). Lake Okaro was dosed using the less efficient grain size (1-3 mm) and at 350 g/m^2, which lead to incomplete coverage of lake sediments. Other modified clays include the proprietary Phoslock^{TM}, a modified bentonite clay with lanthanum (La) used to treat P release from sediments. Although this product is effective at inactivating P at a higher pH than alum, lanthanum is a toxin and addition of Phoslock^{TM} is not recommended for reservoirs providing drinking water (Lewtas et al., 2015). Although Wasagaming's drinking water is taken from Clear Lake, the connectivity to South Lake is cause for concern.

4. **Dredging and removal of sediment**
South lake is shallow and does not thermally stratify, during which time internal loading from sediments can be significant sources of nutrients (Scott & Kling, 2006). Dredging is a solution used in similar lakes where nutrient loading is significant. Removing nutrient rich surface sediments decreases internal nutrient loading, which will decrease P concentrations within the lake. Several concerns arise when considering dredging as a restoration treatment. First, external nutrient loading must be significantly reduced prior to dredging, otherwise nutrient rich sediments will be redeposited and the problem will perpetuate (Kleeberg & Kohl, 1999). Disposal of dredged material adds complexity to these projects. If dredged material is to be deposited on land nearby, alum injection to reduce P concentrations will be required, introducing further cost to the project (Lewtas et al., 2015). Silt fencing would likely be required to separate the south and north basins during dredging. Many aquatic organisms are negatively effected by highly turbid waters and the dredging process will undoubtedly increase turbidity. Hydraulic dredges will however generate less re-suspended sediment as opposed to mechanical dredges (Lewtas et al., 2015). Further, deepening the basin decreases available macrophyte habitat and destroys benthic fish-food habitat (Lewtas et al., 2015).

7.1.3 Budgetary Considerations

1. **Emergent vegetation transplantation**
This restoration technique is the cheapest option for reducing nutrient additions to South Lake, though it will not restore South Lake alone. Costs would be associated with labour and purchasing of nursery plants and transportation. Costs can be reduced greatly by incorporating this project into an outreach program. Hosting an event with volunteers and students creates an opportunity to engage community involvement, increase awareness of the problem of eutrophication and reduce labour costs. Transplantation of emergent vegetation is cheaper than purchasing plugs from a nursery, but will disturb donor sites. Since transplants will have greater establishment success, a mixture of transplantation and nursery plugs will maximize cost-coverage ratios. Assuming plants are spaced 0.45 m apart, along the 1,600 m shoreline of the south basin, over 7,000 plants would be required to create a 1 m^2 band. The shoreline likely contains sections unsuitable for hardstem bulrush, or may already have colonies, so this number is likely an overestimate.

   Mowing/harvesting of emergent vegetation should be done on an annual basis (late summer/early fall) to maximize nutrient removal. Although harvesting adds maintenance costs to the project, there are opportunities for the creation of biochar or bioenergy products (Grosshans, 2014). Sale of harvested stems could offset some operational and maintenance costs of this treatment option.

2. **Installation of silt fencing to disrupt wind mixing**

This method requires a pilot project prior to full installation. The pilot project should be a small cost compared to P inactivation treatments and dredging. Four 100 X 1.5 m silt barriers would be required for the pilot project. Thirty-nine 100 m fences are required for the full installation pictured in Figure 15. Other costs would entail labour for installation and monitoring.

Table 3. Material cost estimates for a whole basin silt fence installation and pilot project installation. Prices are estimates.

Item	Price	Quantity	Total
Silt Fences	$500/100 m	39	$19,500
Anchors	$50 ea	100	$5,000
		Whole Project Materials Total	$24,500
Silt Fences	$500/100 m	4	$2,000
Anchors	$50 ea	12	$600
		Pilot Project Materials Total	$2,600

3. **Phosphorous Inactivation treatments**
   Costs associated with chemical inactivation treatments are dependant on chemical used and application method. Costs were not available for modified zeolite, but a similar proprietary product Phoslock^{TM} costs between $0.75 to $6.02 per m^2 depending on dosage rate. Other modified clay products range in price from $0.13 - $4.15. The south basin has an estimated surface area of 1,158,000 m^2, so costs associated with chemical inactivation of P will range from $ 150,540 to $6,971,160 (surface area based on plume extent and measured in ArcGIS). It should be noted that Phoslock^{TM} has never been applied to a basin of similar size. Evidently cost is widely variable dependant on materials and dosage. Given the size of the south basin, and that re-application will be required, chemical treatments rapidly become cost prohibitive.

4. **Dredging**
   Dredging is potentially the most expensive option. Per a literature review, Lewtas et al. (2015) found costs for dredging ranged between $3,200 and $60,000 per hectare. Variation in costs depend on the volume of sediment removed and removal technique. Draining and bulldozing of the lake accounts for the highest cost per hectare and is not feasible in South Lake, so actual dredging costs using a barge would be well below $60,000 per hectare. Regardless assuming the minimum cost of $3,200 per hectare, dredging 100% of the south basin would cost $370,560 excluding costs associated with designing and coordinating such a project. Silt fencing to divide the north and south basin requires approximately 450 m of silt fencing, creating an additional cost of $2,200 (at approximately $500/100 m).

Table 4. Estimated project cost for dredging the south basin of South Lake. Prices are estimated based on minimum literature values.

Item	Price	Quantity	Total
Dredging	$3,200/ha (minimum)	115.8 ha	$370,560
Silt fencing	$500/100 m	440 m	$2,200
		Total:	$372,760

7.1.4 Future Management Implications

1. **Emergent vegetation transplantation**
   Increasing the emergent vegetation cover alone in the south basin will not have an appreciable effect on water quality. The rational behind this action is to reduce the quantity of nutrients entering the basin and perpetuating the algal dominated steady state. Maintenance is limited to estimating planting survival following the growing season, with replanting if necessary.

   Harvesting should be done following the end of the growing season (late summer/early fall) to maximize biomass removal. Monitoring at the end of each growing season should include survivorship following mowing.

2. **Installation of silt fencing to disrupt wind mixing**
   Should silt fencing cause an appreciable decrease in turbidity, aquatic macrophytes will be able to recolonize the south basin. The silt fencing can be removed once macrophyte dominance is re-established. If this method is successful, it would require minimal maintenance until removal. Following removal, no further maintenance or re-application is necessary, unlike chemical treatments.

3. **Phosphorous Inactivation treatments**
   Chemical treatments will often need reapplication over time, especially if nutrient inputs are not controlled. The barrier created by modified clays is ineffective at controlling P release from sediment that is deposited following treatment (Gibbs, et al., 2007). As such, treatments will need to be re-applied as sediment accumulates. These treatment options would therefore require post-treatment monitoring of water quality to determine if sediments have accumulated enough to allow internal nutrient loading again. If the treatments are successful, sedimentation rates in the south basin should diminish as primary productivity decreases. Monitoring sedimentation rates post treatment could give a prediction metric for longevity of the treatments.

4. **Dredging**
   Dredging has better longevity compared to chemical treatments. Since sediment is physically removed, the time required to infill the dredged section is related to the sedimentation rate. The pre-dredging condition will eventually return if external nutrient

loading is not ameliorated (Kleeberg & Kohl, 1999). Like chemical treatments, a pre- and post-treatment monitoring program is required to track the long-term efficacy of the project.

7.1.5 Metrics of Success

1. **Emergent vegetation transplantation**
   Quantifying the direct effects of increased emergent vegetation cover requires an understanding of concentrations of nutrients entering the basin and concentrations in the water column. This requires further water quality monitoring and more accurate measures of surface runoff and groundwater inputs. It is therefore unnecessarily complicated if we accept the literature consensus that emergent vegetation take up nutrients. A far more accessible metric to quantify transplant success is vegetation cover and transplant survivorship along the transplanted areas. Prior to supplemental planting, vegetation cover can be measured using 1 m^2 quadrats, located at random intervals long the shoreline. Post-monitoring of planting success should be conducted at the end of the growing season for 3 years. This post-monitoring should include transplant locations in the north basin to ensure regrowth.

2. **Installation of silt fencing to disrupt wind mixing**
   Installing silt fencing is designed to diminish turbidity in the south basin through eliminating wind driven sediment mixing. An obvious metric of success for this pilot project would be to monitor water clarity on either side of the silt barrier. Secchi depth readings are an easy and cost effective method to measure water clarity. However, Secchi disks have limitations including: operator error, water surface roughness and water composition (Larson et al., 2007). More accurate measures such as sediment traps or turbidity may be more appropriate. Methods for sediment trap design are outlined in Filstrup & Lind (2010). Measurements of wind speed and current on either side of the barrier would also assist in determining the effectiveness of this technique.
   This monitoring program should continue following full installation for 5 years to ensure the treatment is effective. Macrophyte cover monitoring will allow RMNP to track the macrophyte response to a fully installed system and should begin once appreciable improvements in water clarity are made. Macrophyte cover monitoring should continue until plant cover estimates match the north basin.

3. **Phosphorous Inactivation treatments**
   Numerous water quality metrics have been employed to monitor effectiveness of in-lake chemical treatments including TP, SRP and Chl-*a*, Secchi depth (Spears, et al., 2016). To accurately determine if the treatments were effective, several of these parameters should be sampled at multiple locations throughout the basin, and compared on a 24 months pre/post treatment scale. Pre/post treatment comparisons are crucial in determining the immediate effectiveness of the chemical treatments. Spears et al., (2016) conducted a meta-analysis of 18 lakes treated with Phoslock™ and observed significant changes in all four parameters with strongest significance in TP and SRP reductions (Table 5). Monitoring should be

conducted at random stations in the south basin and compared to pre-treatment data. 3 in lake samples should be taken monthly over the summer to account for seasonal variability in P concentrations.

4. **Dredging**
The effectiveness of dredging can me measured in similar ways to chemical inactivation treatments. This treatment is designed to eliminate internal loading of nutrients, therefore measuring water quality parameters pre/post treatments is a viable method to determine effectiveness. Significant reductions in P concentrations should be observed following dredging. Turbidity should be expected to increase immediately following dredging, but should subside.

Table 5. Reductions in TP, SRP, Chl-*a* and secchi depth during Phoslock^{TM} treatments. Modified from (Spears, et al., 2016).

Parameter	24 months Pre-treatment	24 months Post-treatment	Difference
TP (mg/L) (annual median)	0.08	0.03	-0.05
SRP (mg/L) (annual median)	0.019	0.005	-0.014
Summer Chl-*a* (μg/L) (Summer)	119	74	-45
Secchi disk depth (cm) (Summer)	398	506	108

7.2 Clear Lake Restoration: Hypolimnetic Aeration

7.2.1 Overview

Decreasing hypolimnetic oxygen concentrations is an obvious trend observed in dissolved oxygen profiles in Clear Lake (Figure 1). This trend is strongest over the summer stratification period as compared to winter stratification (Appendix E-H). During summer stratification, Clear Lake becomes divided into distinct zones. The hypolimnion is home to oxidative processes, namely decomposition of organic seston by bacteria. Bacterial respiration takes place throughout the water column, but the epilimnion is constantly replenished with oxygen, making it resilient against oxygen depletion (Wetzel, 2001). Further, bacterial respiration is greatest at the sediment-water interface. Prior to the onset of stratification, oxygen concentrations will increase over the spring while the lake mixes.

Once stratification sets in, no more oxygen is added to the hypolimnion, and dissolved oxygen steadily declines over the summer. Low dissolved oxygen concentrations have numerous effects both on biotic life and chemical reactions. Oxygen is a required metabolite for most aquatic species, and an anoxic hypolimnion limits where in Clear Lake these species can survive. The species which are most sensitive to hypolimnetic oxygen depletion are ones which are adapted to live in the colder deep water. Lake whitefish and slimy sculpin are both deep-water fish species which require a well oxygenated hypolimnion and have been identified in the past as indicator species for Clear Lake (Hoole et al., 2005). Although lake whitefish are abundant and not highly sought after by anglers, they require a well oxygenated hypolimnion over the summer. Chemical interactions caused by aeration are an equally important feature of hypolimnetic aeration for Clear Lake. Although many aeration systems have been developed, this discussion focuses on hypolimnetic aeration systems out of interest in preserving stratification and a cool hypolimnion in Clear Lake (Singleton & Little, 2006).

The earliest record of hypolimnetic dissolved oxygen concentrations were documented by Bajikov (1932). Although the exact location of sampling is unclear, dissolved oxygen concentrations were 6.1 mg/L at 30 m on September 10th. For comparison dissolved oxygen concentrations on September 23rd 2015 were 0.2 mg/L at 30 m. Surface DO concentrations were comparable for 1932 and 2015 at 9.4 mg/L and 9.0 respectively. Although it unclear if mid-summer whole lake mixing occurred in 1932, the differences in hypolimnetic DO are striking. Since hypolimnetic DO demand increases with eutrophication, a possible explanation is that Clear Lake's current hypolimnetic DO demand reflects a shift towards mesotrophic conditions (Wetzel, 2001).

Given the great concern over long term trophic shifts in Clear Lake, increasing the lakes buffering capacity to nutrient inputs will increase the lakes resilience against eutrophication. Aeration has been shown to decrease hypolimnetic ammonia concentrations and reduce internal P loading (Ashley, 1983). Reductions in hypolimnetic P (namely orthophosphate) concentrations are likely due to co-precipitation with calcium carbonate, as has been observed in Clear Lake during previous studies (Ashley, 1983; Whitehouse, 2010). P reductions have also been observed following aeration in lakes with iron redox sensitive sediments. Anoxic conditions at the sediment-water interface results in the release of orthophosphate stored by the sediments, though the exact mechanisms are poorly understood (Golterman, 2001). Although N is rarely a limiting nutrient in Clear Lake, the N cycle is also affected by aeration (Ashley, 1983; Hawryliuk, 2000). Ammonia is directly volatilized into the atmosphere in the separation box of the full-lift design (discussed below). Increased oxygen concentrations also allow nitrifying bacteria to reduce ammonia levels through oxidation of nitrate and nitrite. Although the hypolimnion is separated from the photic epilimnion during stratification, once lake mixing begins nutrients are circulated in the lake. Thus, hypolimnetic nutrient concentrations will have effects throughout the lake following circulation. Ashley (1983) also observed increased ammonia volatilization, and decreased concentrations of bicarbonate, manganese and magnesium during the study period.

Considering the discussion on climate change, aeration will eliminate the negative effects of temperature increases and strengthened stratification in Clear Lake. With longer periods of stratification, there is a far greater chance of the hypolimnion becoming anoxic, as there will be more time for oxygen to be consumed. If the hypolimnion becomes anoxic over the summer, deep-water species must move to warmer waters in the metalimnion and epilimnion which are not conducive to these species that are specifically adapted to cold water.

There are several designs for hypolimnetic aerators. Most common is the ‘full lift’ design, which transports hypolimnetic water to the lake surface and back again. This design is most recommended for this project as it is comparatively simple from a maintenance and operational perspective. These aerators consist of a riser tube, a diffuser, and air-water separation chamber and one or two return pipes. Compressed air is injected through diffusers at the base of the riser tube. Water is then transported surface level by the buoyancy of the air bubbles. Water is then forced down the exit tube by new water coming through the riser tube. This method draws in a large volume of water at low velocity, which is an important consideration in aerator design (Ashley, 1983). An obvious drawback with this design is that the aerator protrudes from the water surface, creating a hazard for boats. Although this can be mitigated by installing signage and proper marking, given the recreational activity on Clear Lake, this may be an undesirable trait.

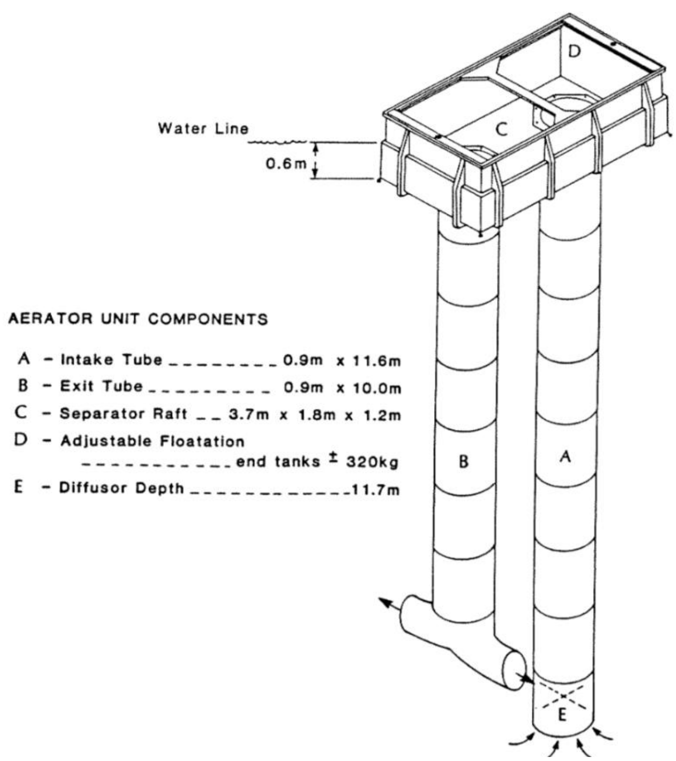

Figure 16. Schematic of a full-lift hypolimnetic aeration system from (Ashley & Nordin, 1999). Dimensions vary depending on the size of aerator required.

Alternatively, the Speece Cone design is highly efficient and compact compared to full lift aerators (Ashley, 2000). This design uses compressed oxygen and a conical chamber for oxygen transfer into the water. Oxygen bubbles are mixed with water at the top of the cone, and while water flows downward towards the base of the cone, the bubbles diffuse into the water. The conical design slows water velocity as the cross-sectional area of the cone increases. At the base of the cone, the water velocity is less than the rise velocity of the air bubbles. Cost associated with this design is greater than the full-lift design, but oxygen transfer is superior to the full lift design (Ashley, 2000). The Speece cone design also remains fully submerged, so it will not disturb recreational boating and fishing activity.

Figure 17. Example of a Speece Cone built for Marston Reservoir, Denver, Colorado. Image from Dominik (2017)

7.2.2 Installation

To calculate the hypolimnetic dissolved oxygen demand in Clear Lake, I used 2015 oxygen profiles at Station 2 (Riding Mountain National Park, N.D.). 2016 data was not available at the time of preparing the calculations. Using volumetric data from the RMNP GIS library, I calculated the total mass of oxygen in the hypolimnion during each sampling period over the summer of 2015 (Riding Mountain National Park, N.D). Plotting the total mass of oxygen contained in the hypolimnion over the summer yields a downward trend (Figure 18). As oxygen is consumed, the mass of oxygen in the hypolimnion will decrease. Only data following the onset of stratification was used in this calculation, since turnover will result in added oxygen (18-Jun-15). At low oxygen concentrations, aerobic bacterial respiration will slow resulting in an artificially low depletion rate (Wetzel, 2001). As such, profiles beyond 17-Aug-15 were excluded (<3.0 mg/L). The slope of this graph yields the mass of oxygen removed over time: 14.549 tonnes/day or a loss of 0.0085 mg/L/day over the entire hypolimnion. Two profiles (23-Jun-15, 04-Aug-15) used in the graph did not extend the full depth of the basin (>30 m), in these cases the dissolved oxygen concentration in the 25-29 m range was used.

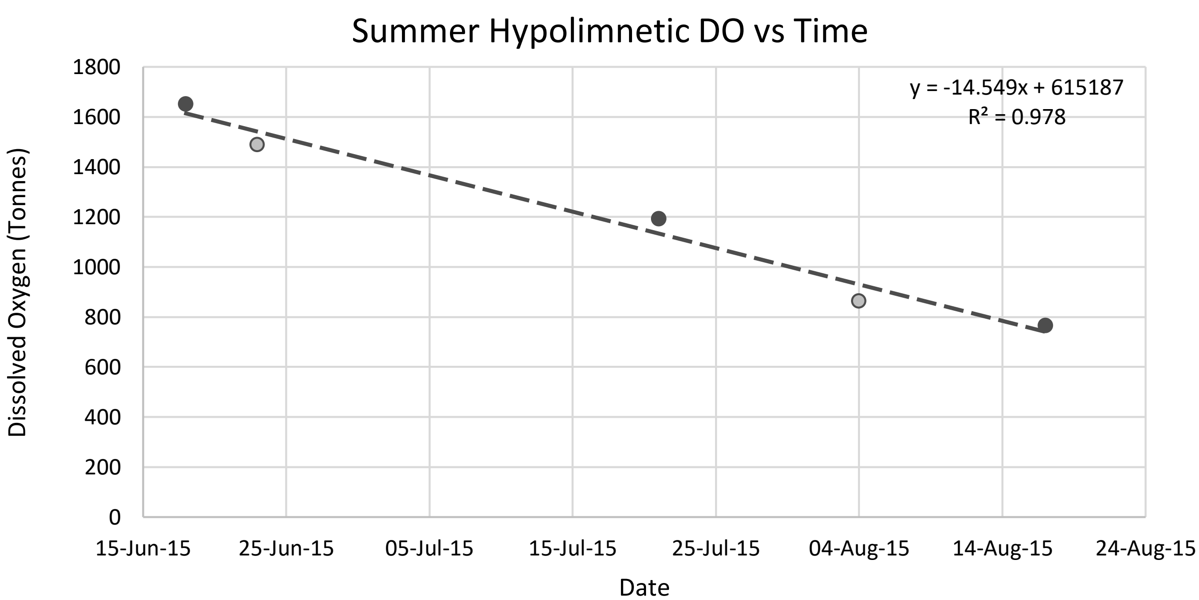

Figure 18. Summer Hypolimnetic Dissolved Oxygen concentrations versus time. Oxygen decreases over the summer as it is consumed by bacterial decomposition and respiration. The slope of the trendline yields the rate of oxygen depletion (tonnes/day). Light coloured points had missing data at depths greater than 30 m which was filled in using oxygen concentrations at 29m. Data obtained from RMNP water quality database and GIS library (Riding Mountain National Park, N.D.).

This means that if stratification sets in with 10 mg/L dissolved oxygen in the hypolimnion on June 15th, by October 10th, the entire hypolimnion would be fully anoxic at 0 mg/L (assuming calculated loss of 0.0085 mg/L/day).

The following discussion outlines sizing of a full lift type aerator for Clear Lake. Aeration system efficiency is largely dependant on the air supply which can be divided into: compressors, power source, power transmission and air lines (Ashley, 1985). Air compressors most applicable to lake aeration systems are rotary screw style compressors and rotary vane compressors. These are designed for continuous and high power operation. Both these styles of compressors operate using either gas or electrical power. Gas powered compressors are not recommended for this application because a) they require much more frequent maintenance, b) will have increasing costs associated with gas prices, c) rely on combustion of fossil fuels which release green house gasses. Electrical motors are the best option in RMNP given the lakes proximity to Wasagaming. With regards to power transmission, Ashley (1985) recommends flange-mounted motors which has no transmission power loss and reduced maintenance requirements. Choosing an appropriate air line is also an important consideration. Air line material varies in both installation ease and cost. Rubber air hoses are considerably more expensive than other types but are more durable and easily installed. Should the internal diameter of airline be too small, air flow will be restricted and aeration efficiency will be lost due to pressure loss. Conversely, a wide diameter air line will be unnecessarily expensive.

Ashley (1985) outlines how to correctly size an aeration system. Using a loss of 14.55 tonne/day, the aerator must input a comparable amount of oxygen per day. It should be noted that the aerator should be sized to accommodate double the hypolimnetic oxygen demand, these values have also been calculated and are summarized in Appendix J. Water flow must therefore equal 1.455 x 10^8. Assuming an aerator input rate of 2 mg/l, and a bubble-water velocity of 1.2 m/s the riser tub requires a radius of 1.5 m or a diameter or 3 m (Ashley, 1985).

Equation 8 $$\boldsymbol{Tube\ radius = \sqrt{\frac{Water\ flow}{\pi \times bubble\ water\ velocity}}}$$

Equation 9 $$\boldsymbol{Entrance\ loss = K_L \times \frac{8Q_w^2}{\pi^2 D^4 g}}$$

Equation 10 $$\boldsymbol{Exit\ loss = \frac{8Q_w^2}{\pi^2 D^4 g}}$$

Equation 11 $$\boldsymbol{Friction\ loss = \frac{fL \times 8 \times Q_w^2}{\pi^2 D^5 g}}$$

Where K_L (entrance constant) = 0.5, f (friction factor) = 0.02, Q_w = water flow (m^3/s), *D* = diameter (m), *g* = acceleration of gravity (m/s), *L* = depth of air release (m).

I used a release depth of 25 m, which is well into the hypolimnion, but no so deep as to directly disturb sediments. These yield an entrance loss of 0.0367 m, exit losses of 0.0735, and friction losses of 0.0123. Total head loss is therefor 0.122.

Next, we must determine the density of the air-water mixture. The density difference between the air-water mixture in the riser tube and the outside water results in the theoretical head. Half of this theoretical head is used to pump water up the rise tube, and the other to push water down the outflow tube.

Equation 12 $$\textit{Density of air}: \textit{water mixture} = \Gamma_{aw} = \frac{L \times \Gamma_w}{L + \Delta H}$$

Where: Γ_w = density of water (kg/m^3), L = depth of air release (m) and ΔH = entrance, exit and friction losses (m).

Using the release depth of 25 m, the density of the air-water mixture is 995.12 kg/m^3. Now that the density of the air-water mixture in the riser tube have been calculated, the actual air flow through the riser tube needs to be calculated. Ashley (1985) includes a design check, using the theoretical air volume requirement calculated earlier. This theoretical value must be less than the calculated airflow requirement.

Equation 13 $$\bar{Q_a} = \frac{10.4 Q_a \ln\frac{L+10.4}{10.4}}{L}$$

Equation 14 $$Q_a = \frac{(Q_w \times \Gamma_w) - (Q_w \times \Gamma_{aw})}{(\Gamma_{aw} \times \bar{Q_a}) - \Gamma_a}$$

Where $\bar{Q_a}$ = mean volumetric air flow rate (m^3/s), Q_a= volumetric air flow rate (m^3/s), Q_w = water flow rate (m^3/s), Γ_a= density of air (kg/m^3), Γ_w = density of water (kg/m^3), and Γ_{aw} = density of air:water mixture (kg/m^3).

Without knowing Q_a, Equation 13 can be simplified to 1.86Q_a, which can be substituted into Equation 14. This yields a volumetric air flow rate of 0.022 m^3/s or 1.33 m^3/min.

Now we need to estimate the compressor pressure and power requirements.

Equation 15 $$\textit{Pressure required} = \Delta P + h + L\left(\frac{1.0}{10.1}\right)$$

Where ΔP = Pressure drop across diffuser orifice (kg/cm^2), h = pressure drop in supply system (kg/cm^2), and L = depth of air release (m). Both ΔP and h are estimations.

Again, using the 25-m riser tube length, we can estimate pressure required using Equation 15. Pressure drop in the air supply system (*h*) is dependant on any valves, elbows, and the air line. In these calculations *h* is estimated to be 0.35 and 1.4 kg/cm^2, as these were the values used by Ashley (1985). ΔP must also be estimated (approximately 0.4-0.7 kg/cm^2). Using the low estimates, 3.23 kg/cm^2 pressure is required to operate the aerator. Using high estimates (higher friction and pressure losses), 4.58 kg/cm^2 of pressure is required. These values are independent of air flow rate, and instead vary with rise tube length. Actual compressor power depends on the model chosen, and has not been calculated. Ashley (1985) recommends selecting a compressor operating at 0.7-1.4 kg/cm^2 above what is estimated from the formula as a contingency.

Finally, dimensions the box of the aerator can be determined. The box holds the riser and outflow tubes and prevents the hypolimnetic water from mixing with the epilimnion. This also provides an arena for the aerated water to de-gas. It is crucial that air bubbles are not released though the outflow tube as this may disrupt stratification. Per Ashley (1985), dimensions of the box typically have a length:width:height:tube diameter ratio of 3.6:1.58:1.18:1. Using this ratio, the box will need to be a length of 10.8 m, a width of 4.7 m and 3.5 m tall. In practice, the box dimensions do not need to follow this ratio explicitly and need only accommodate the diameter of the riser and exit tubes. The height required for the box must be able to contain the head generated from the water moving through the aerator, otherwise hypolimnetic waters will escape and risk de-stratifying the lake. Riser tubes and the separation box will need to be insulated to limit hypolimnetic warming, a common concern with aeration systems.

Floatation is the final element in design of the aerator system. The floatation requirements include the weight of the aerator and the water head. The water head can be substantial while the system is operating. Per Ashley (1985), waterproof foam supports approximately 27 kg per 0.028 m^3 of foam. The weight generated by the water head can be calculated using Equation 16.

Equation 16 $$\boldsymbol{Floatation\ requirements} = \frac{\boldsymbol{\Delta H}}{\boldsymbol{2}} \times \boldsymbol{Area\ of\ box} \times \Gamma_{\boldsymbol{w}}$$

Where: ΔH is the total losses, and Γ_w is the density of water (kg/m^3).

This yields a total weight of 3112 kg. Importantly, this weight only accounts for the water in the aerator and not the weight of the aerator itself. Floatation foam should be secured around the perimeter of the aerator.

One method to greatly increase oxygenation transfer efficiency is to use pure oxygen rather than air. Given the size of Clear Lake, this option would be greatly beneficial in scaling down the size requirement of the aerator. Pure oxygen comes with additional logistical and operational complexity, and a higher cost. On site generation of pure oxygen using pressure swing adsorption (PSA) technologies is likely the best option. Using externally produced pure oxygen would require delivery and on site storage which is likely infeasible given the rurality of Clear Lake (Ashley & Nordin, 1999).

7.2.3 Budgetary Considerations

Installation of an aerator system is a considerable expense, and cost varies widely between projects (Lewtas et al., 2015). Unfortunately, these projects incur large operational costs since they must remain operational for substantial portions of the year. Costs for aeration systems are largely a function of lake size, in larger lakes (such as Clear Lake) installation cost is much greater than smaller systems. Fortunately, total costs per hectare decreases as lake size increases (Ashley & Nordin, 1999). This economy of scale is the result of fixed costs such as the compressor shed, power line costs, and electrical equipment.

Cooke et al. (2005) examined project costs associated with 7 aeration projects installed during the 1970s to 1980s and determined average cost was \$0.39/kg/O_2/day. This estimate assumes a 10-year longevity and includes installation, and operational costs over a 160-day operational window. Using this estimate, the hypolimnetic aeration system designed above would cost \$5674 per day. Actual costs for aeration should be less than this estimate for several reasons. First the 7 aeration projects outlined by Cooke et al (2005) were all installed during the 1970s and 1980s, modern air compressors are likely more operationally efficient, which would decrease power consumption costs.

Secondly, the aerator need only be operational during stratification, which for Clear Lake was approximately 80 days in 2015, as compared to the 160-day estimate used by Cooke et al (2005). A twin aerator installation in Lake Stevens (two units) input 15.5 tonnes O_2 per day. For comparison, Clear Lake requires at minimum 14.5 tonnes O_2 per day. The Lake Stevens aerators operated for 160 days at \$0.27/kg/O_2/day, with a total operational cost of \$100,000 annually (Snohomish County Surface Water Management, 2016).

The Speece cone design is much more efficient with regards to oxygenation. As such, a comparatively smaller unit would be required for Clear Lake. Capital costs for this design are likely much larger than the full lift system. However, higher oxygen transfer efficiency could appreciably decrease operational costs. ECO_2, a company specializing in Speece Cone installations was contacted for approximate costs and installation design. Two 3.7 m diameter cones and a supplementary 2.1 m diameter cone is required to add 14.55 tonnes O_2/day. This system will recirculate 75,700 m^3 of hypolimnetic water daily. Excluding the oxygen supply system, the cones will cost under \$2 million.

7.2.4 Future Management Implications

One of the most significant drawbacks to hypolimnetic aeration is the annual cost associated with operation. Fortunately, long term benefits associated with lake aeration will preserve fish survivorship and eliminate internal nutrient loading considering future non-point source nutrient additions and climate change.

Installing a hypolimnetic aerator will have two major effects on Clear Lake in the future. Climate change is projected to lengthen the summer stratification period, which increases the likelihood of the entire hypolimnion becoming anoxic. Fish kills of deep-water species will become commonplace. Even under the current length of summer stratification, the bottom 10 m of the hypolimnion was below 1 mg/L O_2. The effects of low dissolved oxygen concentrations extend beyond fish respiration. Feeding, reproduction and predator avoidance activities all decrease with decreasing DO in fish (Kramer, 1987). In addition to DO, many fish species are highly dependant on temperature. For this reason, it is imperative that the aeration system does not disturb stratification, or hypolimnetic temperature.

Beyond adding oxygen for deep-water fish, aerators will improve water quality into the future. While limiting external nutrient inputs to Clear Lake should be imperative, protecting the lake from internal nutrient loading is equally important. Increasing hypolimnetic oxygen concentrations will reduce hypolimnetic orthophosphate and ammonia concentrations by preventing redox driven reactions at the sediment water interface (Golterman, 2001).

7.2.5 Metrics of Success

Aerator success is easily measured. Tracking DO concentrations with depth profiles is already done over the summer stratification period. The full lift aerator described above is designed to match the hypolimnetic oxygen demand. In theory, hypolimnetic DO should remain near constant over the course of summer stratification. Bacterial respiration has been known respond to increased DO and subsequently increase the respiration rate (Ashley & Nordin, 1999). As such, DO concentrations should still be expected to decrease over the summer, but at a substantially lower rate.

7.3 Clear Lake and South Lake Connectivity

7.3.1 Overview

The isthmus separating South Lake and Clear Lake has been the subject of much debate in the past. Based on the literature review the isthmus has naturally accreted, but concern over spawning connectivity has been identified on multiple occasions (Briscoe et al., 1979; Maclean, 1979; Rounds et al., 1992). Northern pike and walleye in specific appear to use South Lake as a spawning area (Fawcett, 2011; Rounds et al., 1992; Thomas, 2011). Both Thomas (2011) and Fawcett (2011) used VHF telemetry tags in northern pike to determine spawning areas. Their results indicate that South Lake is a 'critical' spawning area for this species. However, given the inconsistency of breaches in the isthmus spawning area must exist in Clear Lake or other sub-basins as well. Current population sizes are unknown so it is unclear whether the populations are declining or increasing. Northern pike populations have been variable in the past, with periods of significant decline (Briscoe et al., 1979; Heap, 1988). Walleye populations are also not actively monitored and their populations have too been variable (Heap, 1988). Young northern pike are dependant on macrophyte cover and access to vegetation cover is critical to their development (Craig, 2008). The dense macrophyte cover in the north basin is a perfect example of this habitat requirement.

The isthmus does not breach annually and is therefore a potential bottleneck for spawning. Attempts were made to secure a permanent opening between 1962 and 1972 with the installation of a culvert, however sediments from Clear Lake would infill the culvert. The culvert was removed in 1982 at which point a dredging program began (Rounds et al., 1992). Again, sediment would regularly infill the dredged portion. High water levels in South Lake appear to be the natural mechanism for breaching the isthmus (Neumann, 2013). Given the alterations to the natural drainage system in the Octopus Creek & South Lake sub-basins, an increased volume of water is entering South Lake. As such, it is possible that these events have increased the frequency of isthmus breaching conditions (Neumann, 2013). Since climate change is projected to change water levels in RMNP, the frequency of breaches in the future may be more variable. With drier conditions, evaporation increases and water levels should drop. Conversely, climate change is also projected to increase three-day precipitation totals (Prairie Climate Centre, 2016). This means South Lake (and Clear Lake) will be 'flashier', with periods of rapidly changing water levels. This could further increase the frequency at which the isthmus breaches.

While the isthmus may be beneficial to the walleye and pike populations, it can be a significant source of nutrients to Clear Lake. Hoole et al., (2005) concluded that South Lake and groundwater nutrient contributions to Clear Lake were five times greater than surface flow sources. For this reason, it is likely in RMNP's interest to limit connectivity between the two lakes. Since the isthmus is made up of ice pushed sediments (mainly sand), groundwater flux occurs regardless of a breach.

Neumann (2013), observed between -8 and 173 m^3/day of groundwater flow through the isthmus. The negative value indicates flow out of Clear Lake and into South Lake. Although this range is small compared to other sites in clear lake (max flow 7232 m^3/day), differences in nutrient concentrations could constitute disproportionately large inputs of nutrients via groundwater through the isthmus. Comparing data collected during this study, Clear Lake's epilimnetic SRP concentrations were not significantly different than South Lake, although this is very likely a product of a high minimum detection limit ($p=0.25$). Of the 25 samples collected, 2 were above the 0.01 mg/L detection limit (both in South Lake). Drawing conclusions from data with so many artificial values is ill advised (see Chapter 6 – for a discussion on detection limits). Comparing epilimnetic DIN between the two lakes, a significant difference exists ($p=0.042$), in this case 15 of the 25 samples were above detection limit. All samples for DIN in South Lake were above the DIN detection limit. During sampling the isthmus remained closed and therefore was not sampled, though water flowing from south lake should match that of the water in the north basin. A students t-test comparing TP concentrations between the north basin of South Lake and Station 2 in Clear Lake suggests the two bodies have significantly different TP concentrations ($p<0.005$). The isthmus was breached during the summer field visit, at which time large clumps of algae could be seen flowing into Clear Lake (Figure 19).

Figure 19. Algae mats flowing into Clear Lake though the breach in the isthmus. Image taken during site visit on June 13th, 2016 by A. Butcher.

There are still many unknowns regarding the effect the isthmus on Clear Lake both present and future. Based on the projected effects of climate change it is unclear if the isthmus will breach more or less frequently. If the isthmus were to remain open, fish species would have access to spawning area but at the expense of water quality in Clear Lake. There does not appear to be an immediate danger to fish populations. Given the lack of data available regarding northern pike and walleye populations, it is assumed that these populations are at least stable. Securing a permanent opening through the isthmus has been challenging in the past and mostly futile. A groyne or similar structure would likely be required to prevent long-shore sediments from infilling the channel. These structures by design limit sediment movement which can have effects elsewhere along the shoreline, as happened at the Clear Lake Beach (Willis Cunliffe Tait DelCan, 1982).

Three options arise regarding the isthmus.

1. **Allow natural processes to continue**
   The isthmus will continue to breach on years with high water levels in South Lake. Fish access will be intermittent and access to spawning ground will remain dependant on climactic factors. This is evidently the cheapest option and has the lowest maintenance requirements. There will be a periodic influx of nutrients to Clear Lake from South Lake. Groundwater will continue to flow regardless. Climate change will potentially increase the frequency of breaches, where high levels in South Lake will break the isthmus, and a subsequent storm will close it. This process could repeat over the spring and fall when precipitation is projected to increase. Summer conditions will likely be drier and hotter with an increasing number of days above 30 °C (Prairie Climate Centre, 2016). With warmer summers comes greater evaporation and normal water levels will be lower, breaches in the isthmus will then be dependant on storms. Summer storms are projected to increase in frequency, sudden influxes of precipitation will undoubtedly raise water levels in South Lake. At the same time storms are also the typical cause for closing the isthmus.

2. **Parks Canada controls breaches**
   Given the uncertainty in frequency of breaches associated with natural processes, Parks Canada can attempt to control the frequency of breaching. A permanent opening is undesirable as it would result in a constant influx of South Lakes eutrophic waters to Clear Lake. Further, maintaining a constant opening will require specialized structures or excessive maintenance. A semi-permanent opening would allow Parks Canada to control fish access while limiting water connectivity. Northern pike spawn in the spring and their young migrate to deeper waters as they mature (Harvey, 2009). Walleye have a similar pattern of spring spawning and young migrating to deeper waters with maturation (Hartman, 2009). Since South Lake is prone to winter kill, opening the isthmus only in the spring would have limited effect on the spawning success of northern pike and walleye. Fry will need to be able to return to Clear Lake in mid-summer and early fall (Hartman, 2009; Harvey, 2009). By directly managing the breach, Parks Canada can allow the continued spawning and rearing benefits of both species in South Lake. Walleye require shallow water with temperatures below 11.1 °C to spawn (Hartman, 2009). With the projected increases in air temperatures associated with climate change, South Lake may not be appropriate spawning area for walleye in the future. Spawning pike can tolerate higher water temperatures and temperature cues for spawning range from 8-12 °C (Harvey, 2009).

To prevent unintentional breaches in the isthmus, reinforcing the isthmus will be necessary. The site of breaching has varied over the years, so it can be expected that if the current location of the breach is blocked another section of the isthmus may become compromised. Specific methods to prevent breaches will be discussed in #3 of this section. However, since some water will be released while during spring and mid-summer/fall, this may be less of an issue. In addition to blocking breaches in the isthmus, a location of a controlling structure may also be required.

3. **Permanently close the isthmus**

   Permanently closing the isthmus is another management option. No further breaches in the isthmus will prevent direct movement of water from South Lake into Clear Lake. Actively preventing groundwater flow through the isthmus would require sheet piling or the installation of a dam. These are not recommended due to the high cost, flooding potential and maintenance requirements. Since ice push and sediment movement is a major force acting on the isthmus, hard structures such as sheet piling will more than likely be displaced and rendered ineffective. Since South Lake lacks an outflow (in the absence of the breach), groundwater must be the only constant drainage pathway. Allowing groundwater to continue its natural path through the isthmus is important to prevent disturbances such as flooding. To reinforce the isthmus without hard engineering requires the establishment of vegetation. It has been widely documented that the root systems of established vegetation anchor their substrate. Trees and other vegetation has colonized many sections of the isthmus apart from the current location of the breach. Since vegetation has not secured sediments in the breached section, it is prone to erosion and channel formation.

7.3.2 Installation

1. **Allow natural processes to continue**

   This method requires no installation or maintenance. Monitoring should be implemented regardless of intervention actions to determine frequency of breaching and flow rate. If breaches become very common and substantial volumes of water are regularly entering Clear Lake, RMNP should consider restoration measures. Restoration of South Lake as discussed in Chapter 7.1 South Lake Restoration Options, is a mechanism for improved quality of water entering Clear Lake through a breach in the isthmus.

2. **Parks Canada controls breaches**

   To control breaches in the isthmus, vegetation should be planted as discussed in #3. A water control structure should be added to allow for manual opening and closing of connectivity between Clear Lake and South Lake. These structures should be designed by a qualified engineer so specifics regarding the water control structure will be omitted from this discussion. Since this structure will increase water storage in South Lake, water storage permits may be required. Challenges are associated with deposition of longshore sediments which can block the outlet and impair function of the water control structure. Prevailing winds are north-westerlies, which would cause longshore drift to occur in an easterly direction along the southern shore of Clear Lake (Figure 20) (Willis Cunliffe Tait DelCan, 1982). The proposed location for the water control structure is based on potential protection from the land outcrop immediately west of the location. This was also the last location to infill before South Lake was cut off from Clear Lake.

Figure 20. Proposed location of the water control structure in the isthmus. Longshore drift moves along the grey arrow in an easterly direction (Willis Cunliffe Tait DelCan, 1982).

The water control structure should only be open twice a year corresponding with spawning requirements of both northern pike and walleye. Spawning walleye have a water temperature range of 1.1-11.1 °C, and will typically spawn before June (Hartman, 2009). Northern pike spawn following ice breakup when water temperature range from 8-12 °C (Harvey, 2009). Since temperature appears to be the signal for spawning migrations, the water control structure should remain open while water temperatures fall within this range, though to limit surface flow only for a limited time. Defining a specific period is difficult since balance must be met between fish migration and minimizing transfer of water. Closing the control structure at night would effectively cut the open time in half. Unfortunately, while pike are inactive at night, walleye spawn overnight and adults will out migrate before dawn (Hartman, 2009; Harvey, 2009). Monitoring fish movement though the control structure could be an effective means to determine optimal open time. If fish passage was actively monitored, once migration begins to slow the gate could be closed. Timing the re-opening the control structure is an equally difficult task, but similar methods could be employed.

3. **Permanently close the isthmus**

The current breach location is unvegetated due to the frequent disturbance to sediments associated with breaches. Due to the dynamic nature of the isthmus, hard engineering techniques are undesirable. Sediment movements, infilling and erosion have all been the downfall of previous attempts to control the isthmus. Planting and implementing bioengineering techniques can be more adaptive and flexible than installing hard structures. Additionally, this approach adds a natural looking element, a desirable trait in a park. Plants used should meet several criteria: pioneer plants which grow rapidly, deep rooting systems, and easy propagation (Norris et al., 2008). Due to proximity to both Clear Lake and South Lake, planted species must be tolerant to a high water table. For this reason, willow (*Salix spp.*) is an excellent option. Willow has been used as an erosion control method for centuries (Evette et al., 2009). Willow is also common along the isthmus already and is a characteristic plant in wetlands throughout RMNP (Hutchinson, 1981).

Installing a live wattle fence or similar would create a strong barrier to erosion of the isthmus. Wattle fences are essentially a wall built from live cuttings which root into the substrate and create a strong living barrier (Polster, 1997). This technique is often used on steep hillslopes prone to landslides, where wattle fences protect other establishing vegetation from shifting substrates (Polster, 1997). Long willow stakes are driven into the ground perpendicular to the substrate. Willow stakes are then woven between the vertical stakes parallel to the ground.

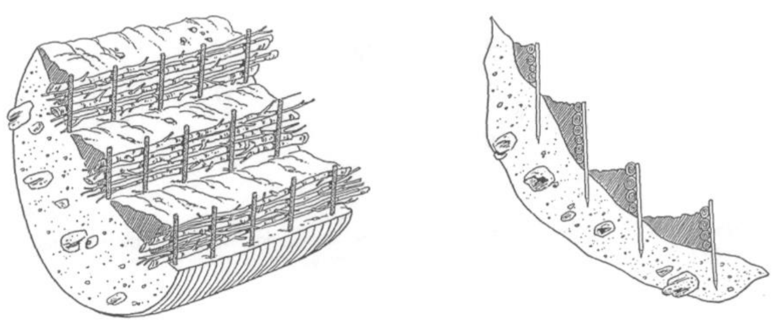

Figure 21. Illustration of a wattle fence installation. Live cuttings are used to create a living wall for erosion control. The isthmus slope is not as steep as pictured, but wattle fences on either side of the isthmus could add enough structural integrity to prevent breaches. From Polster (1997).

Additional live staking of willow should be done along the breach area. Per Polster (1997), willow stakes should be inserted top end up, three-quarters of their length into the ground and spaced at least 10 cm apart. Willow stakes can easily be harvested from established plants along the isthmus.

7.3.3 Budgetary Considerations

1. **Allow natural processes to continue**
   This method requires no installation or maintenance. The only budgetary consideration is labour for the continued monitoring of the isthmus. SRP concentrations should be regularly monitored in South Lake and Clear Lake to determine changes to water quality.

2. **Parks Canada controls breaches**
   Installing a water control structure will require an engineers input and design. This approach will likely have the highest cost associated with it of any restoration approach on the isthmus. The structure will require maintenance and inspections to ensure it remains functional. Monitoring will be required to ensure the isthmus does not breach. Ongoing water quality monitoring for Clear Lake and South Lake should also be included in this plan. Determination of optimal open/close times would also involve monitoring of the water control structure to track fish migrations. Costs associated with this monitoring is primarily dependant on labour costs.

3. **Permanently close the isthmus**
   Harvesting and planting willow stakes and wattle fences are easy and have minimal cost associated with them. If stakes are harvested from established plants along the isthmus, labour costs are the only expense. Continued monitoring of the integrity of the isthmus is also required to ensure a new breach does not open.

7.3.4 Future Management Implications

1. **Allow natural processes to continue**
   This method will allow continued access to spawning and rearing habitat for both northern pike and walleye. Importantly, these species will only be able to successfully utilize this spawning area if access continues throughout the season or a breach occurs during mid-summer or fall. The trade-off for fish access is potentially compromising water quality in Clear Lake. Future shifts in the climate may enhance or diminish the frequency at which breaches in the isthmus occur. Breaches may cease to occur for several years as was the case from 1930-1935 (Rounds et al., 1992). Many unknowns remain for this approach and monitoring is therefore strongly recommended.

2. **Parks Canada controls breaches**

   This method ensures continued fish access between Clear Lake and South Lake, while limiting direct surface water transfer. This should have benefits to northern pike and walleye populations in Clear Lake while limiting additional nutrient inputs.

   Tracking water quality on either side of the isthmus is important to ensure water entering Clear Lake does not pose a direct threat to eutrophication. Canadian water quality guidelines suggest TP remain below 0.020 mg/L to maintain mesotrophic conditions (Canadian Council of Ministers of the Environment, 2004). Suggested SRP concentrations are not available, but SRP is generally <5% of TP, so we can assume 0.001 mg/L or less should be maintained (Wetzel, 2001). This value is well below the detection limits of this study, so more accurate analysis will be required for monitoring.

3. **Permanently close the isthmus**

   Permanently closing the isthmus will prevent direct transfer of eutrophic water into Clear Lake. This should appreciably reduce external nutrient loading into Clear Lake. Although groundwater flow from South Lake into Clear Lake will continue, direct transfer of particulate matter (algae clumps, vegetation etc.) will cease. There is the possibility that breaches may occur elsewhere along the isthmus. Breaches have occurred in at least 3 locations along the isthmus: at the west end, 250 m east from that and in the current location (Neumann, 2013). These historic breach locations have since been colonized by vegetation and are likely supported by root structures. However, monitoring and additional planting may be required should high water levels occur in South Lake.

   Since access to spawning area will be cut off for both northern pike and walleye, monitoring of populations should be conducted. If a noticeable decline is observed, installation of a fish-way should be considered. Both northern pike and walleye have been documented using fish-ways for migration (Katopodis, 1992).

7.3.5 Metrics of Success

1. **Allow natural processes to continue**

   Since this method does not involve any action on the part of RMNP, metrics of success cannot be strictly defined. This approach could see increased connectivity between the two lakes, or decreased connectivity in the future. Regardless, two conditions should be expected to remain moderately constant. Water quality in Clear Lake should not deteriorate further and observable declines in pike or walleye populations should not occur. Of course, other forces could be cause for future decline in either species (resource depletion, disease, changes to water quality etc). If no action is taken, monitoring should be conducted to determine if/when intervention is necessary.

2. **Parks Canada controls breaches**

   Northern pike and walleye in-migration to South Lake and out-migration to Clear Lake can give estimates of spawning use by these species. Maintenance of water quality in Clear

Lake is an imperative for this approach. Current mean TP concentration in Clear Lake is 0.026 mg/L, maintaining or decreasing this value is imperative to maintain the meso-oligotrophic status of Clear Lake.

3. **Permanently close the isthmus**
   The most evident metric of success for this method entails ensuring no further breaches occur in the isthmus. Wattle fences and live stakes should have visible growth during the growing season following installation.
   Secondly, no noticeable decline in northern pike or walleye populations should be observed. Spawning ground exists elsewhere in Clear Lake for both species since the isthmus is not breached annually. Should observable decline in these populations occur, alternate action should be implemented. Installation of a fishway could allow access to spawning ground and be designed to limit transfer of water.

7.4 Outreach & Awareness

The importance of reducing excess nutrient inputs to the Clear Lake watershed has been stressed throughout this report and others (Hoole et al., 2005). With a portion of the Clear Lake watershed located outside of the park, any restoration efforts must be complimented with additional outreach programs. The two sub-basins for Clear Lake which deliver the greatest nutrient concentrations both have portions of the basin outside RMNP boundaries (Octopus Creek and South Lake). This demonstrates the importance of community outreach, and expanding efforts beyond RMNP boundaries. Parks Canada demonstrating the importance of responsibility towards the landscape is only the first step to garner public interest. RMNP has actively shown concern over the trophic status of Clear Lake, and has invested resources and effort into restoration/maintaining said trophic status. This creates a positive image of RMNP for visitors and those in neighbouring towns. To limit excess nutrient inputs to Clear Lake, the entire watershed must limit nutrient inputs. While enforcing reduction to nutrient inputs outside of RMNP is impractical, steps can be made to demonstrate the benefits of limiting nutrient inputs. A community outreach program can work towards convincing people to follow water course buffer strip regulations, limit fertilizer usage on their agricultural lands, and to replace leaky septic systems which pollute the groundwater. Further, community outreach can convince land developers to install more efficient septic systems and minimize nutrient release.

As discussed in Chapter 1.2 First Nations, ecological restoration should be approached in collaboration with the Keeseekoowenin Ojibway Band. Maintaining the meso-oligotrophic status of Clear Lake is a shared goal between both parties. Collaboration and consultation should start from the planning phase of restoration and continue throughout the restoration process. Ecological restoration is an opportunity to not only improve water quality but also to improve the working relationship between Parks Canada and the Keeseekoowenin Ojibway Band.

An interpretive trail already exists through Ominik Marsh. Expanding this signage to significant areas around Clear Lake and South Lake will enhance understanding and appreciation of the natural beauty of the park. Installing signage along the isthmus trail explaining what restoration activities are being conducted is a low effort way to enhance public understanding and awareness of environmental impacts. These signs should include the cause of decline to water quality and why restoration is needed. Exact signage locations depend on what restoration activities are conducted by RMNP. Costs associated with this method entail signage design, construction, printing and installation. Beyond maintaining these signs, no more effort or labour is required.

Restoration activities such as planting can greatly benefit from outreach and volunteer programs. These activities can be accomplished much quicker with volunteers and do not require extensive training or experience. Hosting volunteer days not only improves efficiency of restoration projects by decreasing labour costs, but instills a great sense of accomplishment and appreciation for ecology. Youth outreach programs have been shown to increase knowledge and retention of knowledge learned over the program (Dieser & Bogner, 2016). This not only encourages learning about the environment but can also encourage environmentally responsible behaviour in the future. Outreach events such as these require some organization and planning, but can be developed alongside a restoration plan to maximize the success of the project.

References

Ashley, K. (1985). Hypolimnetic Aeration: Practical Design and Application. *Water Research, 19*(6), 735-740. Retrieved from http://dx.doi.org/10.1016/0043-1354(85)90120-4

Ashley, K. I. (1983). Hypolimnetic Aeration of a Naturally Eutrophic Lake: Physical and Chemical Effects. *Canadian Journal of Fisheries and Aquatic Sciences, 40*, 1343-1359.

Ashley, K. I. (2000). Recent advances in aerator design. *Verhandlungen des Internationalen Verein Limnologie, 27*, 2256-260.

Ashley, K., & Nordin, R. (1999). Lake aeration in British Columbia: Applications and experiences. *Aquatic Restoration in Canada*, 87-108.

Ashley, K., Fattah, K., Mavinic, D., & Kosari, S. (2013). Analysis of Design Factors Influencing the Oxygen Tranfer of a Pilot-Scale Speece Cone Hypolimnetic Operator. *Journal of Environmental Engineering, 140*(3), 1-9. doi:10.1061/(ASCE)EE.1943-7870.0000789

Bajikov, A. (1932). *Report of Clear Lake Riding Mountain Natonal Park.* Calgary AB: Department of Indian Affairs and Northern Development.

Belke, S., & McGinn, R. (2003). *Monitoring Anthropogenic Nutrients in a Modified Natural Wetland.* Brandon University, Department of Geography. Brandon: Department of Geography Brandon University.

Benson, D. (1984). *Discussion Paper on Trout Stocking: RMNP.* Wasagaming: Canadian Park Service.

Bergman, D. (1987). *Water Quality Study for the Wasagaming Region of Riding Mountain National Park.* Environment Canada Environmental Protection Manitoba District.

Briscoe, B., Lee, B., Allan, C., & Tempay, I. (1979). *Riding Mountain National Park Resource Description and Analysis .* Wasagaming: Riding Mountain National Park.

Canadian Council of Ministers of the Environment. (2004). *Canadian water quality Guidelines for the protection of aquatic life: Phosphorous: Canadian Guidance Framework for the Management of Freshwater Systems.* Winnipeg: Canadian Council of Ministers of the Environment. Retrieved from http://ceqg-rcqe.ccme.ca/download/en/205

Carper, G., & Bachmann, R. (1984). Wind Resuspension of Sediments in a Prarie Lake. *Canadian Journal of Fisheries and Auatic Sciences, 41*, 1763-1767.

Cody, W. (1988). *Plants of Riding Mountain National Park.* Ottawa: Agriculture Canada.

Cooke, G. D., Welch, E. B., Peterson, S. A., & Nichols, S. A. (2005). *Restoration and management of lakes and reservoirs* (Third ed.). Boca Raton: Taylor & Francis.

Craig, J. F. (2008). A short review of pike ecology. *Hydrobiologia, 601*, 5-16. doi:10.1007/s10750-007-9262-3

Cronshey, R., McCuen, R., Miller, N., Rawls, W., Robbins, S., & Woodward, D. (1986). *Urban Hydrology for Small Watersheds (Technical Report 55).* Natural Resource Conservation Service, USDA. Washington: U. S. Department of Agriculture.

Daniel, T., Sharpley, A., Edwards, D., Wedepohl, R., & Lemunyon, J. (1994). Minimizing Surface Water Eutrophication from Agricuture by Phosphourous Management. *Journal of Soil and Water Conservation*, 30-38.

Dodds, W. (2003). Misuse of Inorganic N and Soluble Reactive Phosphorous to Indicate Nutrient Status of Surface Waters. *Journal of North American Benthological Socienty, 22*(2), 171-181.

Dokulil, M., & Teubner, K. (2011). Eutrophication and Climate Change: Present and Future Scenarios. In A. Ansari, S. S. Gill, G. R. Lanza, & W. Rast, *Eutrophication: Causes, Consequenses and Control* (pp. 1-16). New York: Springer.

Dominik, S. (2017). *Marston Reservoir, CO*. Retrieved from ECO Oxygen Technologies: http://www.eco2tech.com/installation/marston-reservoir/

Downing, J. (2010). Emerging global role of small lakes and ponds: litle things mean a lot. *Limnetica, 29*(1), 9-24.

Dupuis, A., & Hann, B. (2009). Climate chane, diapause termination and zooplankton population dynamics: an experimental and modelling approach. *freshwater Biology, 54*, 221-235. doi:10.1111/j.1365-2427.2008.02103.x

Dyregrov, A. (2003). *Additional Enginering Studies Wastewater Treatment Pond.* DyreGrov Consultants and Stantec Consulting LTD.

Egemose, S., & Reitzel, K. (2013). Resuspension-mediated aluminium and phosphorous distribution in lake sediments after aluminium treatment. *Hydrobiologia, 701*, 79-88. doi:10.1007/s10750-012-1258-y

Environment and Climate Change Canada. (2016, 08 04). *The Canadian Air and Precipitation Monitoring Network (CAPMoN)*. Retrieved 02 10, 2017, from Environment Canada: https://www.ec.gc.ca/rs-mn/default.asp?lang=En&n=752CE271-1

Environment Canada. (2010). *Climate Normals 1981-2010 Wasagaming MB Station Data.* Ottawa: Environment Canada.

Evette, A., Labonne, S., Rey, F., Liebault, F., Janke, O., & Girel, J. (2009). History of Bioengineering Techniques for Erosion Control in Rivers in Western Europe. *Environmental Management*(43), 972-984. doi:10.1007/s00267-009-9275-y

Fawcett. (2011). *Identification of Northern Pike (Esix lucius) Spawning Sites in South Lake, Riding Mountain National Park, Using VHF Telemetry.* Brandon University, Geography. Brandon: Brandon University.

Filstrup, C., & Lind, O. (2010). sediment transport mechanisms influencing resuspension patterns in a shallow, polymictic reservoir. *Lake and Reservoir Management, 26*(2), 85-94. doi:10.1080/07438141.2010.49077

Gebremariam, S., & Beutel, M. (2008). Nitrate removal and DO levels in batch wetland mesocosms: Cattail (Typha spp.) versus bulrush (Scripus spp.). *Ecological Engineering, 34*, 1-6. doi:10.1016/j.ecoleng.2008.06.00

Geisler, M., Rennie, M., Gillis, D., & Higgins, S. (2016). A predictive model for water clarity following dreissenid invasion. *Biological Invasions*, 1989-2006. doi:10.1007/s10530-016-1146-x

Gibbs, M., & Ozkundakci. (2011). Effects of modified zeolite on P and N processes and fluxes across the lake sediment-water interface using core incubations. *Hydrobiologia, 661*, 21-35. doi:10.1007/s10750-009-0071-8

Gibbs, M., Dudli, S., Vopel, K., Hickey, C., Wilson, P., & Ozkundakci, D. (2007). *P-inactivation efficacy of Z2G1 as a capping agent on Lake Okaro Sediment.* Hamilton, NZ: National Institute of Waer and Atmospheric Research.

Glufka, R. (1992). *Water Quality Issues in Clear Lake Basin Riding Mountain National Park.* Brandon MB: The Rurual Development Institute.

Goldsborough, G., & Rounds, R. (1992). *Analysis of historical changes in the primary production of Clear Lake (Riding Mountain National Park).* Wasagaming: Riding Mountain National Park.

Golterman, H. L. (2001). Phosphate release form anoxic sediments or "What did Mortimer really write?". *Hydrobiologica, 450*, 99-106.

Grace, J., & Wetzel, R. (1981). Habitat Partitioning and Competitive Displacement in Cattails (Typha); Experimental Field Studies. *The American Naturalist, 118*(4), 463-474. Retrieved from http://www.jstor.org/stable/2460778

Hartman, G. (2009). *A Biological Synopsis of Walleye (Sander Vitreus).* Nanimo: Fisheries and Oceans Canada.

Harvey, B. (2009). *A Biological Synopsis of Northern Pike (Esox Lucius).* Nanaimo: Department of Fisheries and Oceans Canada.

Hawryliuk, Y. (1995). *Clear Lake Basin Water Quality Report.*

Hawryliuk, Y. (2000). *Assesment of Phytoplankton Productivity in Clear Lake (Riding Mountain National Park), Manitoba.* Winnipig, Manitoba: University of Manitoba.

Heap, M. (1988). *Review of the South Lake/Clear Lake Fish Study for 1987 an 1988.* Wasagaming: Riding Mountain National Park.

Holland, R., Johengen, T., & Beeton, A. (1995). Trends in nutrient concentrations in Hatchery Bay, western Lake Erie before and after Dreissena polymorpha. *Canadian Journal of Fisheries and Aquatic Sciences*, 1202-1209. doi:10.1139/f95-117

Hoole, A., Nedotiafko, R., Ramsey, D., & Werner, C. (2005). *Securing the Integrity of Clear Lake and Area.* Report, Winnipeg.

Huser, B., Futter, M., Lee, J., & Perniel, M. (2016). In-lake measures of phosphorous control; the most feasible and cost-effective solution for long-term management of water quality in urban lakes. *Water Research, 97*, 142-152. Retrieved from http://dx.doi.org/10.1016/j.watres.2015.07.036

Hutchinson, G. E. (1973). Eutrophication: the Scientific Background of a Contemporary Practcal Problem. *Marginalia, 61*(3), 269-279.

Hutchinson, R. (1981). *Aquatic Resources of Riding Mountain National Park Vol 6.* Winnipeg: Canadian Wildlife Service.

Katopodis, C. (1992). *Introduction to Fishway Design.* Department of Fisheries and Oceans. Winnipeg: Freshwater Institute.

Kayranli, B., Scholz, M., Mustafa, A., & Hedmark, A. (2010). Carbon Storage and Fluxes within Freshwater Wetlands: a Critical Review. *Wetlands, 30*, 111-124. doi:10.1007/s13157-009-0003-4

Kleeberg, A., & Kohl, J.-G. (1999). Assesment of the long-term effectiveness of sediment dredging to reduce benthic phosphorous release in shallow lake Muggelsee (Germany). *Hydrobiologia, 394*, 153-161.

Kooyman, A. H. (1970). *Sport Fishery Management in Riding Mountain National Park.* Wasagaming: Canadian Wildlife Service, Limnology Section.

Kramer, D. (1987). Dissolved Oxygen and Fish Behavior. *Environmental Biology of Fishes, 18*(2), 81-92.

Langston, R., McGinn, R., & Rousseau, P. (2003). *Outlet Flow for Clear Lake.*

Larson, G., Hoffman, R., Hargreaves, B., & Collier, R. (2007). Predicting Secchi disk depth from average beam attenuaion in a deep, ultra-clear lake. *Hydrobiologia, 574*, 141-148. doi:10.1007/s10750-006-0349-

Legislative Assembly of Manitoba. (2015). The Noxious Weeds Amendment Act. Manitoba. Retrieved from https://web2.gov.mb.ca/bills/40-4/b032e.php

Lewtas, K., Paterson, M., Venema, H. D., & Roy, D. (2015). *Manitoba Prarie Lakes: Eutrophication and In-Lake Remediation Treatments.* Winnipeg: International Institute for Sustainable Development.

Maclean, A. (1979). *Relationship of South and Clear Lakes for Walleye Reproduction Progress Report No.1.* Riding Mountain National Park. Wasagaming: Environment Canada.

McGinn, R., Parsons, F., & Rousseau, P. (1998). *Snowmelt and Lake Levels in the Clear Lake Watershed.* Wasagaming: Riding Mountain National Park.

Ministry of Water, Land and Air Protection; Province of British Columbia. (2013). *British Columbia Field Sampling Manual.* (C. M, Ed.) Victoria, British Columbia, Canada: Her Majesty the Queen in Right of the Province of British Columbia. Retrieved from http://www2.gov.bc.ca/assets/gov/environment/research-monitoring-and-reporting/monitoring/emre/field_sample_man2013.pdf

Mitsch, W., Bernal, B., Nahlik, A., Mander, U., Zhang, L., Anderson, C., . . . Brix, H. (2013). Wetlands, Carbon, and climate change. *Landscape Ecology, 28*(4), 583-597. doi:10.1007/s10980-012-9758

National Atmospheric Deposition Program. (2016). *NTN Site ND08 Seasonal Precipitation-Weighted Mean Concentrations.* University of Illinoise, Illinoise State Water Survey. Champaign, IL: National Atmospheric Deposition Program.

Neumann, N. (2013). *Ecohydrology of Coupled Surface Water - Groundwater Systems.*

Norris, J., Stokes, A., Mickovski, S., Cammeraat, E., van Beek, R., Nicoll, B., & Achim, A. (2008). *Slope Stability and Erosion Control: Ecotechnological Solutions.* Dordrecht, The Netherlands: Springer Science + Business Media B.V.

Nürnberg , G., & Peters , R. (1984). Biological Availability of Soluble Reactive Phosphorous in Anoxic and Oxic Freshwaters. *Canadian Journal of Fisheries and Auatic Sciences*, 757-765.

Ozkundakci, D., Hamilton, D., & Scholes, P. (2010). Effect of intensive catchment and in-lake restoration procedures on phosphorous concentrations in a eutrophic lake. *Ecological Engineering, 36*, 396-405. doi:10.1016/j.ecoleng.2009.11.00

Pacific Climate Impacts Consortium. (2014). *High Resolution Climatology.* Victoria : University of Victoria .

Parks Canada. (2007). *Riding Mountain National Park of Canada Management and Riding Mountain Park East Gate Registration Complex National Historic Site of Canada: Management Plan.* Wasagaming: Parks Canada.

Patalas, K., Salki, A., & Stainton, M. (1988). *Chemical Composition of Water in Six Riding Moutains Lakes (Preliminary draft).* Wasagaming,MB.

Pittock, A. B. (2005). *Climate Change: Turning up the Heat.* London: Earthscan.

Polster, D. (1997). Retsoration of Landslides and Unstable Slopes: Considerations for Bioengineering Interior Locations. *British Columbia Mine Reclamation Symposium* (pp. 1-14). Cranbrook: Technical and Research Committee on Reclamation .

Prairie Climate Centre. (2016). *Manitoba Boreal Plains Report Card.* Winnipeg: Prarie Climate Centre.

Province of Manitoba. (2010). the Noxious Weeds Act. Manitoba: Province of Manitoba. Retrieved 2017, from http://web2.gov.mb.ca/laws/statutes/ccsm/n110e.php

Public Works and Government Services Canada. (2009). *Wasagaming, Manitoba Riding Mountain National Park Sewage Lagoon Upgrade, Phase III.* Wasagaming: Public Works and Government Services Canada.

Rempel, G. (1977). *Engineering Study of Sewage Treatment and Disposal Requirements for Riding Mountain National Park Manitoba.* Winnipeg: James F. MacLaren Limited.

Rempel, G., Macbride, B., Knapp, R., Albiani, C., & Benoit, E. (1978). *Groundwater Investigation at the Wasagaming Lagoon Site.* Winnipeg: James F. Maclaren Limited.

Reside, B. C. (2016, 06 13). Clear Lake. (A. Butcher, Interviewer) Wasagaming, MB, CA.

Riding Mountain National Park. (N.D). RMNP GIS database. (S. Frey, Ed.) Wasagaming, Manitoba, Canada. Retrieved February 20, 2017

Riding Mountain National Park. (N.D.). *Water Quality Analysis Database.* Wasagaming: Riding Mountain National Park. Retrieved 2016

Robinson, G., Scott, K. J., & Sellers, P. (2003). *A Study on theImpact of Sewage Lagoon Effluent on a Receiving Wetland and Adjoining Shallow Lake in the Riding Mountain National Park.* Department of Botany. Winnipeg: University of Manitoba.

Rounds, R., Bazillion, H., & Braun, C. (1992). *Human Intervention in the Fishery of Clear Lake Basin in Riding Mountain National Park.* Brandon MB: The Rural Development Institute.

Rousseau, P. (1992). *Water Quality Monitoring Plan - Clear Lake.* Wasagaming, Manitoba: Parks Canada.

Sandlos, J. (2008). Not Wanted in the Boundary: The Expulsion of the Keeseekoowenin Ojibway Band from Riding Mountain National Park. *The Canadian Historical Review, 89*(2), 189-221. doi:10.3138/chr.89.2.189

Scheffer, M., & E, J. (2007). Regime Shifts in Shallow Lakes. *Ecosystems*(10), 1-3. doi:10.1007/s10021-006-9002-y

Scheffer, M., & van Nes, E. (2007). Shallow lakes theory revisited: various alternative states driven by climate, nutrients, depth and lake size. *Hydrobiologia*(584), 455-466. doi:10.1007/s10750-007-0616-7

Scheffer, M., Hosper, S., Meijer, M., Moss, B., & Jeppesen, E. (1993). Alternative equilibria in shallow lakes. *Trends in Ecology and Evolution, 8*, 275-279.

Scheffer, M., Rinaldi, S., Gragnani, A., Mur, L., & van Nes, E. (1997). On the Dominance of Filamentous Cyanobacteria in Shallow, Turbid Lakes. *Ecology, 78*(1), 272-282.

Schindler, D. (1974). Europhication and Recovery in Experimental Lakes: Implications for Lake Management. *Science, New Series, 184*(4139), 897-899.

Schindler, D. (1977). Evolution of Phosphorous Limitation in Lakes. *Science, New Series, 195*(4275), 260-262.

Schindler, D. W. (2006). Recent Advances in the Understanding and Management of Eutriphication. *Limnology Oceanography, 51*(1, part2), 356-363.

Schindler, D., Hecky, R., Findlay, D., Stainton, M., Parker, B., Paterson, M., . . . Kasian, S. (2008). Eutrophicaton of Lakes Cannot be Controlled by Reducing Nitrogen Input: Results of a 37-year Whole-ecosystem Experiment. *Proceedings of the National Academy of Sciences (PNAS), 105*(32), 11254-11258.

Scott, K., & Kling, H. (2006). *South Lake, Riding Mountain National Park: Investigation into its Recent Limnological History using Radiodating and Microfossils in Sediments.* Wasagaming: Riding Mountain National Park.

Scott, K., & Sellers, P. (2005). *A Qualitative Investigation into the Planktonic Food Chains of Ominik Marsh and South Lake Riding Mountain National Park.*

Singleton, V. L., & Little, J. C. (2006). Designing Hypolimnetic Aeratio and Oxygenation Systems - A Review. *Environmental Science and Technology, 40*(24), 7512-7520. doi:10.1021/es060069

Snohomish County Surface Water Management. (2016). *Health Report: Lake Stevens.* Lake Stevens, WA: Snohomish County. Retrieved March 4, 2017, from https://snohomishcountywa.gov/DocumentCenter/Home/View/33534

Society for Ecological Restoration International Science and Policy Working Group. (2004). *The SER International Primer on Ecological Restoration.* Society for Ecological Restoration International Science and Policy Working Group.

Spears, B., Mackay, E., Yasseri, S., Gunn, I., Waters, K., Andrews, C., . . . Lurling, M. (2016). A meta-analysis of water quality and aquatic macrophyte responses in 18 lakes treated with lanthanum modified bentonite (Phoslock®). *Water Research, 97*, 111-121. Retrieved from http://dx.doi.org/10.1016/j.watres.2015.08.020

Stewart, G. A. (1880). Riding Mountain Area Land Survey. *Riding Mountain National Park Files*. Wasagaming, MB, Canada: Riding Mountain National Park. Retrieved 02 08, 2017

Tanaka, N., Asaeda, T., Hasegawa, A., & Tanimoto, K. (2004). Modelling of the tond-term competition between Typha angustifolia and Typha latifolia in shallow water - effects of eutrophication, latitude and initial advantage of belowground organs. *Aquatic Botany, 79*, 295-310. doi:10.1016/j.aquabot.2004.03.00

Thomas, R. (2011). *Tracking movements and habitat use of Norther Pike (Esox lucius) using VHF telemetry in Riding Mountain Nationa; Park.* Brandon University, Geography. Brandon: Brandon University.

Tilley, D. (2012). *Plant Guide for hardstem bulrush (Schoenoplectus acutus).* Idaho Plant Materials Centre. Aberdeen ID: USDA-Natural Resources Conservation Service.

Vanderploeg, H., Liebig, J., Carmichael, W., Agy, M., Johengen, T., Fahnensteil, G., & Nalepa, T. (2001). Zebra Mussel (Dreissena polymorpha) selective filtration promoted toxic Microcystis blooms in Saginaw Bay (Lake Huron) and Lake Erie. *Canadian Journal of Fisheries and Aquatic Sciences, 58*, 1208-1221. doi:10.1139/cjfas-58-6-1208

Vanderschuit, S. (1992). *Summary of Data ources Related to Aquatic Ecosystem Monitoring in Riding Mountain National Park.* Wasagaming, MB: Riding Mountain National Park.

Weber, C. A. (1907). Aufbau und Vegetation der Moore Norddeutschlands. *Bot Jahrb*, 40 (90):19-34.

Welch, E. B., & Cooke, G. D. (1999). Effectiveness and Longevity of Phosphorous Inactivation with Alum. *Journal of Lake and Reservoir Management, 15*(1), 5-27. doi:10.1080/0743814990935394

Welch, E. B., & Cooke, G. D. (2005). Intenal Phosphorous Loading in Shallow Lakes: Importance and Control. *Classic Lake and Reservoir Management, 21*(2), 209-217.

Wetzel, R. (2001). *Limnology: Lake and River Ecosystems* (3 ed.). San Diego, California, United States of America: Academic Press.

Whalen, S. C. (2005). Bioenergy of Methane Exchange between Natural Wetlands and the Atmosphere. *Environmental Engineering Science, 22*(1), 73-94. doi:10.1089/ees.2005.22.73

White, C. (2012). *Sedimentary Diatoms as Indicators of Water Quality and Ecosystem Change in Lakes of Riding Mountain National Park of Canada.* Okanagan: University of British Columbia.

Whitehouse, R. (2010). *Phosphorus Scavenging through Calcite Co-Precipitation Bringing Clarity to Clear Lake.*

Willis Cunliffe Tait DelCan. (1982). *Clear Lake Beach Study.*

Appendix

Appendix A. **DIN concentrations for in-lake samples at Clear Lake and South Lake. Bolded values indicate detection limit adjusted values (median of 0.0 and the detection limit (DL = 0.08 mg/L)).**

Clear Lake Station 2	**DIN mg/L (DL adjusted)**		
Date	**Epilimnion mg/L**	**Metalimnion mg/L**	**Hypolimnion mg/L**
2016-08-18	**0.040**	**0.040**	0.131
2016-09-06	0.058	**0.040**	0.046
2016-09-27	**0.040**	**0.040**	0.040
2016-10-20	**0.040**	**0.040**	0.047
2017-01-16	0.058	0.059	0.148
Clear Lake Station 1	DIN mg/L (DL adjusted)		
2016-08-18	**0.040**	**0.040**	0.113
2016-09-06	**0.040**	0.096	**0.040**
2016-09-27	**0.040**	**0.040**	**0.040**
2016-10-20	**0.040**	**0.040**	**0.040**
2017-01-16	0.052	0.050	0.063
Clear Lake Station 3	DIN mg/L (DL adjusted)		
2016-09-06	**0.040**	**0.040**	0.055
2016-09-27	**0.040**	**0.040**	**0.040**
2016-10-20	**0.040**	**0.040**	0.063
2017-01-15	0.054	0.053	0.050
2017-02-21	0.054	0.053	0.059
South Lake 1	DIN mg/L (DL adjusted)		
2016-08-24	0.095		
2016-09-07	0.108		
2016-10-27	0.376		
2017-01-15	0.102		
2017-02-21	0.104		
South Lake 2	DIN mg/L (DL adjusted)		
2016-08-24	0.057		
2016-09-07	0.046		
2016-10-27	0.069		
2017-01-15	0.565		
2017-02-21	0.685		

Appendix B. **SRP concentrations for in-lake samples at Clear Lake and South Lake. Bolded values indicate detection limit adjusted values (median of 0.0 and the detection limit (DL = 0.01 mg/L)).**

Clear Lake Station 2	**SRP mg/L (DL adjusted)**		
Date	**Epilimnion m/L**	**Metalimnion mg/L**	**Hypolimnion mg/L**
2016-08-18	**0.005**	**0.005**	**0.005**
2016-09-06	**0.005**	**0.005**	**0.005**
2016-09-27	**0.005**	**0.005**	**0.005**
2016-10-20	**0.005**	**0.005**	**0.005**
2017-01-16	**0.005**	**0.005**	**0.005**
Clear Lake Station 1	SRP mg/L (DL adjusted)		
2016-08-18	**0.005**	**0.005**	**0.005**
2016-09-06	**0.005**	**0.005**	**0.005**
2016-09-27	**0.005**	**0.005**	**0.005**
2016-10-20	**0.005**	**0.005**	**0.005**
2017-01-16	**0.005**	**0.005**	**0.005**
Clear Lake Station 3	SRP mg/L (DL adjusted)		
2016-09-06	**0.005**	**0.005**	**0.005**
2016-09-27	**0.005**	**0.005**	**0.005**
2016-10-20	**0.005**	**0.005**	**0.005**
2017-01-16	**0.005**	**0.005**	**0.005**
2017-02-17	**0.005**	**0.005**	**0.005**
South Lake 1	SRP mg/L (DL adjusted)		
2016-08-24	**0.005**		
2016-09-07	**0.005**		
2016-10-27	0.010		
2017-01-15	**0.005**		
2017-02-21	0.003		
South Lake 2	SRP mg/L (DL adjusted)		
2016-08-24	**0.005**		
2016-09-07	**0.005**		
2016-10-27	0.012		
2017-01-15	**0.005**		
2017-02-21	**0.005**		

Appendix C. **DIN concentrations measured at inflows and outflows of Clear Lake and South Lake. Bolded values indicate detection limit adjusted values (median of 0.0 and the detection limit (DL = 0.08 mg/L)).**

Date	West Beach	Aspen 1	Aspen 2	Aspen 3	Spruces	Glen Baeg	North Shore 1	North Shore 3	EE Culvert	Wishing Well	Deep Bay	Octopus	South Lake Channel	EE Launch	Wasamin
2016-08-24													0.081		
2016-08-30					0.046	0.313	**0.040**		0.072	0.112	0.136	0.047			0.045
2016-09-01	0.200	0.075	0.050	0.054		0.314		0.085							
2016-09-07													0.156		
2016-09-21	0.179	0.050	0.045	0.070	0.040	0.135	**0.040**	0.107	0.068	0.123	0.130	0.105			**0.040**
2016-10-13	0.429	0.057	0.051	0.056	0.057	0.312	0.125	0.249	0.094	0.063	0.062	0.057		0.050	**0.040**
2016-10-27													0.079		
2016-11-15		0.050	0.045	0.049	0.055	0.448	0.154	0.212		0.145	0.099	0.048		0.103	**0.040**
2017-01-17										0.202					0.045
2017-02-21															0.048

Appendix D. **SRP concentrations for inflows and outflows of Clear Lake and South Lake. Bolded values indicate detection limit adjusted values (median of 0.0 and the detection limit (DL = 0.01 mg/L)).** e

SRP	West Beach	Aspen 1	Aspen 2	Aspen 3	Spruces	Glen Baeg	North Shore 1	North Shore 3	EE Culvert	Wishing Well	Deep Bay	Octopus	South Lake Channel	EE Launch	Wasamin
2016-08-24													0.108		
2016-08-30					0.022	0.014	0.015		**0.005**	0.016	0.022	**0.005**			**0.005**
2016-09-01	0.015	0.073	0.041	**0.005**		0.005		0.013							
2016-09-07													0.102		
2016-09-21	0.027	0.075	0.035	0.005	0.020	0.014	0.012	0.012	0.005	0.014	0.018	**0.005**			**0.005**
2016-10-13	0.054	0.036	0.028	0.027	0.017	0.015	0.016	0.005	0.005	0.015	0.025	**0.005**		0.029	**0.005**
2016-10-27													0.022		
2016-11-15		0.029	0.024	0.020	0.017	**0.005**	0.016	0.018		0.016	0.027	**0.005**		0.011	**0.005**
2017-01-17										0.005					**0.005**
2017-02-21															**0.005**

Appendix E. In-lake profile data for Station 1 in Clear Lake.

Date	Depth (m)	Temp	pH	DO mg/L	DO %	Connectivity	Comments
Sept 6 2016	0.1	18.4	8.7	8.6	92.1	380	
	1.2	18.4	8.7	8.5	90.4	380	
	2.2	18.4	8.7	8.5	90.2	380	
	3.1	18.4	8.7	8.5	90.1	380	
	4.1	18.4	8.7	8.4	90.0	380	
	5.1	18.4	8.7	8.4	89.7	380	
	6.0	18.4	8.7	8.4	89.3	380	
	7.2	18.4	8.7	8.4	89.4	380	
	8.1	18.4	8.7	8.4	89.2	380	
	9.1	18.4	8.7	8.4	89.2	380	
	10.1	18.4	8.7	8.3	88.9	380	
	11.2	18.4	8.7	8.3	88.7	380	
	12.1	18.4	8.7	8.3	88.4	380	
	13.2	18.3	8.7	8.2	87.5	380	
	14.1	17.6	8.5	6.9	72.3	384	
	15.2	16.2	8.2	3.6	36.3	392	
	16.1	15.4	8.0	2.4	24.1	394	
	17.1	14.1	7.9	1.5	14.5	397	
	18.1	12.9	7.8	0.7	6.6	398	
Sept 27 2016	0.2	14.0	8.5	8.6	83.8	390	Very windy
	1.0	14.0	8.5	8.4	81.1	390	
	2.1	13.9	8.4	8.3	80.7	390	
	3.1	13.9	8.4	8.3	80.5	389	
	4.0	13.9	8.4	8.3	80.1	390	
	5.1	13.8	8.3	8.2	79.4	390	
	6.1	13.8	8.3	8.2	79.1	390	
	7.1	13.8	8.3	8.2	79.0	390	
	8.1	13.8	8.3	8.2	79.0	390	
	9.0	13.8	8.3	8.2	79.2	390	
	10.1	13.8	8.3	8.2	79.3	390	
	11.0	13.8	8.3	8.2	79.4	390	
	12.1	13.8	8.3	8.2	79.4	390	
	13.1	13.8	8.3	8.2	79.0	390	
	14.0	13.8	8.3	8.1	78.6	390	
	15.1	13.8	8.3	8.1	78.3	390	
	16.1	13.8	8.3	8.1	78.1	390	
	17.1	13.7	8.3	8.1	78.1	390	
	18.1	13.7	8.3	8.1	77.8	391	
	19.0	13.7	8.3	8.1	77.9	390	

Oct 20 2016	0.3	7.9	8.3	12.1	101.7	387	
	1.2	7.9	8.3	11.8	99.9	387	
	2.2	8.0	8.3	11.8	99.4	387	
	3.2	8.0	8.3	11.7	99.0	387	
	4.1	8.0	8.3	11.7	98.8	387	
	5.1	8.0	8.3	11.7	98.8	387	
	6.1	7.9	8.3	11.7	98.8	387	
	7.0	7.9	8.3	11.7	98.9	387	
	8.0	7.9	8.3	11.7	98.9	387	
	9.0	7.9	8.3	11.7	98.8	388	
	10.1	7.9	8.3	11.7	98.8	387	
	11.1	7.9	8.3	11.7	98.7	387	
	12.0	7.9	8.3	11.7	98.7	387	
	13.0	7.8	8.3	11.7	98.5	388	
	14.0	7.7	8.3	11.7	98.2	388	
	15.1	7.7	8.3	11.7	97.9	388	
	16.1	7.3	8.3	11.7	97.6	388	
	17.2	7.2	8.3	11.8	97.6	388	
	18.1	7.2	8.3	11.9	98.5	388	
	18.4	7.1	8.3	11.9	98.8	388	
Jan 16 2017	0.7	0.2	8.7	13.5	92.5	419	
	1.1	0.3	8.7	13.3	91.5	415	
	2.1	0.3	8.6	13.2	90.9	414	
	3.0	0.4	8.6	13.1	90.8	413	
	4.1	0.4	8.5	13.1	90.9	413	
	5.1	0.4	8.5	13.1	90.7	412	
	6.1	0.4	8.5	13.1	90.5	412	
	7.0	0.5	8.5	13.0	90.5	410	
	8.0	0.6	8.4	12.9	90.1	408	
	9.0	0.7	8.4	12.8	89.5	406	
	10.1	0.8	8.4	12.7	88.9	404	
	11.1	0.9	8.4	12.6	88.2	403	
	12.1	1.0	8.4	12.4	87.4	405	
	13.1	1.1	8.4	12.3	86.5	409	
	14.1	1.1	8.4	12.1	85.6	411	
	15.1	1.2	8.4	12.0	84.8	411	
	15.1	1.2	8.4	11.9	84.1	411	
	16.1	1.2	8.4	11.7	82.9	412	
	17.0	1.3	8.4	11.5	81.6	414	
	18.1	1.3	8.4	11.4	80.6	414	
	18.6	1.5	8.4	10.9	78.0	414	

Appendix F. In-lake profile data for Station 2 in Clear Lake.

Date	Depth (m)	Temp	pH	DO mg/L	DO %	Connectivity	Comments
Sept 6 2016	0.1	18.7	8.7	8.8	94.5	381	
	1.1	18.7	8.7	8.7	93.1	381	
	2.2	18.7	8.7	8.7	92.8	381	
	3.2	18.7	8.7	8.6	92.7	381	
	4.2	18.7	8.7	8.6	92.4	381	
	5.1	18.7	8.7	8.6	92.2	381	
	6.1	18.6	8.7	8.6	91.7	381	
	7.1	18.6	8.7	8.5	91.4	381	
	8.1	18.6	8.7	8.5	90.8	381	
	9.1	18.5	8.7	8.3	88.8	382	
	10.3	18.5	8.7	8.3	88.7	381	
	11.0	18.5	8.7	8.3	88.9	381	
	12.2	18.4	8.7	8.4	89.7	381	
	13.2	18.2	8.7	8.1	86.5	381	
	14.2	18.2	8.6	8.0	84.8	381	
	15.2	17.6	8.4	5.1	53.4	388	
	16.2	17.2	8.3	4.4	45.4	390	
	17.3	16.4	8.1	3.4	35.2	392	
	18.3	14.7	8.0	2.1	21	396	
	19.3	13.2	7.9	1.4	13	398	
	20.1	12.2	7.8	1.2	11.3	398	
	21.1	11.6	7.8	0.9	8.3	399	
	22.1	10.7	7.7	0.3	3	400	
	23.1	10.6	7.7	0.2	1.5	400	
	24.1	10.3	7.7	0.1	1.3	401	
	25.2	10.2	7.7	0.1	1.2	402	
	26.3	10.2	7.7	0.1	1.1	402	
	27.2	10.1	7.7	0.1	1	402	
	28.3	10.0	7.7	0.1	0.9	404	
	29.2	10.0	7.7	0.1	0.9	404	
	30.2	10.0	7.7	0.1	0.8	404	
	31.2	10.0	7.7	0.1	0.8	404	
	32.1	9.9	7.7	0.1	0.8	413	
Sept 27 2016	0.1	14.2	9.0	8.9	86.6	390	Very windy; profile not complete
	1.0	14.2	8.4	8.2	80.3	390	
	2.1	14.2	8.4	8.2	80.2	390	
	3.1	14.2	8.4	8.2	80.1	390	

	4.1	14.1	8.4	8.2	79.9	390	
	5.1	14.1	8.4	8.2	79.7	390	
	6.0	14.1	8.4	8.2	79.5	390	
	7.0	14.1	8.4	8.2	79.5	390	
	8.2	14.1	8.4	8.2	79.5	390	
	9.2	14.0	8.4	8.2	79.5	390	
	10.2	14.0	8.4	8.2	79.6	390	
	11.2	14.0	8.2	8.8	85	389	
	12.0	14.0	8.3	8.7	84.1	389	
	13.0	14.0	8.3	8.7	84.3	389	
	14.0	14.0	8.4	8.7	84.4	389	
	14.0	13.9	8.4	8.7	84.5	389	
	15.1	13.9	8.4	8.6	83.1	389	
	16.2	13.9	8.4	8.5	82.5	389	
	17.0	13.9	8.4	8.5	82.3	389	
	18.1	13.9	8.2	8.8	84.8	390	
	19.1	13.8	8.3	8.5	82.1	390	
	20.0	13.8	8.3	8.5	81.6	390	
	21.0	13.8	8.3	8.4	81.1	390	
	22.1	13.7	8.3	8.4	80.8	390	
	23.0	13.7	8.3	8.4	80.7	390	
	24.0	13.7	8.3	8.4	80.6	391	
Oct 20 2016	0.2	8.4	8.3	12.1	103.7	388	DO probe was acting up and YSI sent in after this day.
	0.2	8.4	8.3	12.0	102.6	388	
	1.2	8.4	8.3	11.9	101.4	388	
	2.1	8.5	8.4	11.7	100.2	388	
	3.1	8.5	8.4	11.6	99.4	388	
	4.1	8.5	8.4	11.6	99.1	388	
	5.1	8.5	8.4	11.6	98.9	388	
	6.2	8.5	8.4	11.5	98.6	388	
	7.1	8.4	8.4	11.5	98.5	388	
	8.1	8.3	8.4	11.5	98.3	388	
	8.0	8.3	8.4	11.5	98.2	388	
	8.9	8.3	8.4	11.5	98.1	388	
	9.9	8.3	8.4	11.5	98.1	388	
	10.6	8.3	8.3	11.5	98	388	
	11.5	8.3	8.3	11.5	98.1	388	
	12.9	8.2	8.3	11.6	98.3	388	
	13.9	8.2	8.3	11.6	98.2	387	
	14.8	8.2	8.3	11.6	98.2	388	
	15.5	8.2	8.3	11.6	98.2	387	

	16.8	8.2	8.3	11.6	98.1	388	
	18.0	8.2	8.3	11.6	98.3	388	
	20.1	8.1	8.3	11.6	98.3	388	
	20.8	8.1	8.3	11.6	98.4	388	
	21.9	8.1	8.3	11.6	98.5	388	
	23.0	8.1	8.3	11.6	98.6	388	
	24.1	8.1	8.3	11.6	98.5	388	
	25.1	8.1	8.3	11.6	98.5	388	
	26.2	8.1	8.3	11.6	98.4	388	
	27.1	8.1	8.3	11.6	98.3	388	
	27.9	8.1	8.3	11.6	98.1	388	
	28.0	8.1	8.2	11.7	98.8	388	
	29.1	8.0	8.3	11.6	98.2	388	
	30.1	8.0	8.3	11.6	97.7	388	
	31.1	8.0	8.3	11.5	97.1	388	
	32.0	8.0	8.3	11.5	96.9	388	
16-Jan-17	0.4	-0.1	8.5	13.0	88.9	375	
	1.0	0.2	8.5	13.8	95.0	365	
	2.0	0.5	8.4	13.9	96.6	391	
	3.1	0.6	8.4	13.9	97.0	378	
	4.0	0.6	8.4	13.9	97.0	376	
	5.1	0.7	8.4	13.9	96.9	397	
	6.1	0.8	8.4	13.8	96.6	397	
	7.1	0.8	8.4	13.7	95.8	401	
	8.1	0.9	8.4	13.6	95.3	401	
	9.1	0.9	8.4	13.4	94.3	402	
	10.1	1.0	8.4	13.3	93.4	405	
	11.1	1.0	8.4	13.2	93.0	407	
	12.1	1.0	8.4	13.2	93.0	408	
	13.1	1.0	8.4	13.3	93.6	408	
	14.1	1.0	8.4	13.3	93.4	409	
	15.1	1.1	8.3	13.2	93.2	410	
	16.0	1.1	8.3	13.1	92.5	411	
	17.0	1.1	8.3	13.0	92.1	412	
	18.1	1.1	8.3	13.0	91.6	414	
	19.0	1.2	8.3	12.9	91.2	415	
	20.0	1.2	8.3	12.8	90.7	415	
	21.0	1.3	8.3	12.7	90.2	414	
	22.0	1.3	8.3	12.6	89.2	415	
	23.0	1.3	8.3	12.4	88.0	415	
	24.1	1.4	8.3	12.3	87.3	415	
	25.0	1.4	8.3	12.0	85.8	415	

	26.1	1.5	8.3	11.9	84.8	418	
	27.1	1.7	8.3	11.3	81.4	422	
	28.1	1.8	8.2	10.7	76.8	424	
	29.1	2.2	8.2	8.1	58.8	422	
	30.1	2.3	8.1	6.8	49.4	430	
	30.7	2.5	8.1	5.4	39.3	432	

Appendix G. In-lake profile data for Station 3 in Clear Lake.

Date	Depth (m)	Temp	pH	DO mg/L	DO %	Connectivity	Comments
Sept 6 2016	Surface	18.32	8.69	8.74	93.2		Turn over has happened
	1.0	18.3	8.7	8.7	92.3		Sample taken 0.5m from surface
	2.0	18.3	8.7	8.6	91.9		
	3.0	18.3	8.7	8.6	91.8		
	4.0	18.3	8.7	8.6	91.6		
	5.0	18.3	8.7	8.6	91.4		
	6.0	18.3	8.7	8.6	91.4		
	7.0	18.3	8.7	8.6	91.2		
	8.0	18.3	8.7	8.6	91.0		
	9.0	18.3	8.7	8.6	91.1		
	10.0	18.3	8.7	8.6	91.2		
	11.0	18.2	8.7	8.6	91.2		Sample taken at 11.5m from bottom
	12.0	18.2	8.7	8.6	91.2		
Sept 27 2016	0.3	14.2	8.4	9.0	87.4	390.0	Data collected from YSI
	1.0	14.2	8.4	8.8	85.6	390.0	Very windy; could not complete profile
	2.0	14.1	8.3	8.7	85.0	390.0	
	3.0	13.9	8.2	8.7	84.6	391.0	
	4.0	13.7	8.2	8.8	84.6	391.0	
	5.1	13.6	8.3	8.9	85.7	391.0	
	6.0	13.6	8.3	9.0	86.6	390.0	
Oct 20 2016	Surface	8.2	8.4	11.4	96.9	386.0	Overcast skies, wind SW <5km, Temp 4
	1.0	8.3	8.3	11.4	96.7	386.0	
	2.0	8.3	8.2	11.4	96.9	386.0	
	3.0	8.3	8.2	11.4	97.0	387.0	
	4.0	8.3	8.2	11.4	97.2	387.0	
	5.0	8.3	8.2	11.4	97.1	386.0	
	6.0	8.2	8.2	11.4	97.2	387.0	
	7.0	8.3	8.2	11.4	97.4	386.0	
	8.0	8.3	8.2	11.5	97.4	386.0	
	9.0	8.3	8.2	11.5	97.5	386.0	

	10.0	8.3	8.2	11.5	97.5	387.0	
	11.0	8.3	8.2	11.5	97.4	387.0	
	12.0	8.3	8.2	11.5	97.5	387.0	
	13.0	8.2	8.2	11.5	97.5	387.0	
	14.0	8.2	8.2	11.5	97.5	387.0	
	15.0	8.2	8.2	11.5	97.5	386.0	
	16.0	8.2	8.2	11.5	97.3	387.0	
	17.0	8.2	8.2	11.5	97.4	387.0	
	18.0	8.2	8.2	11.5	97.3	387.0	
Jan 17 2017	0.4	0.1	9.1	13.2	90.4	413.0	
	1.1	0.4	8.9	12.8	88.7	410.0	
	2.1	0.5	8.8	12.7	87.8	408.0	
	3.1	0.5	8.8	12.6	87.2	408.0	
	4.1	0.5	8.7	12.5	86.7	407.0	
	5.0	0.6	8.7	12.4	86.4	406.0	
	6.0	0.7	8.6	12.3	86.1	405.0	
	7.1	0.8	8.6	12.2	85.8	404.0	
	7.4	0.9	8.6	12.1	85.1	415.0	

Appendix H. **In-lake profile data for South Lake. S. Lake Stn 1 = South Lake Station 1, S. Lake Stn 2 = South Lake Station 2.**

Site	Date	Depth (m)	Temp	pH	DO mg/L	DO %	Conductivity	Comments
S. Lake Stn 1	Aug 24th	0.5	18.6	7.9	5.8	61.9		Outside temp 17 degrees, cloudy with chance of rain
S. Lake Stn 2	Aug 24th	0.5	18.7	8.9	9.4	100.4		
		1.2	18.6	8.9	9.1	98.0		
S. Lake Stn 1	Sept 7th	0.5	16.4	8.3	10.3	104.3	293	
S. lake Stn 2	Sept 7th	0.5	16.1	9.0	10.8	109.9	229	
		1.0	16.1	9.1	10.8	110.0	229	
S. Lake Stn 1	Oct 27th	0.5	4.3	8.3	13.2	102.9	339	Sunny; low wind
S. Lake Stn 2	Oct 27th	0.5	4.3	8.5	13.4	103.2	295	
		1.0	4.3	8.4	13.5	103.7	295	
S. Lake Stn 1	17-Jan-17	0.3	0.4	7.8	1.6	11.5	590	site #80 on YSI 2 depths
S. Lake Stn 2	17-Jan-17	0.3	0.5	8.4	9.1	63.0	521	site #77 on YSI 1 depth
		1.1	2.6	8.2	3.6	26.2	499	Windy day light cloud 0-2 degrees

Appendix I. Sub-basin runoff data. See 6.3 Results for a discussion on calculation methods. South Lake's total percentage runoff was added to Octopus Creek due to the connectivity through the South Lake Channel

Basin Name	Area (m^2)	CN	Rainfall Mean annual P (in)	Potential Retention S (in)	Initial abstraction I (in)	Runoff (in)	Runoff ratio	Total Runoff	Percentage of total runoff	Average SRP (mg/L)	Average DIN (mg/L)	SRP %	DIN %
Octopus Creek	34361430	75	20.43	3.33	0.67	16.91	0.83	1.48E+10	41.35%	0.04	0.085	0.017	0.035
West Bay	4092927	65	20.43	5.38	1.08	15.14	0.74	1.57E+09	3.48%	N/A	N/A	N/A	N/A
South Lake	9122202	75	20.43	3.33	0.67	16.91	0.83	3.92E+09	**8.67%**				
Aspens	4409724	65	20.43	5.38	1.08	15.14	0.74	1.70E+09	3.75%	0.03	0.054	0.001	0.002
West Shore	2755511	65	20.43	5.38	1.08	15.14	0.74	1.06E+09	2.35%	N/A	N/A	N/A	N/A
Ministic Lake	13273590	65	20.43	5.38	1.08	15.14	0.74	5.10E+09	11.30%	0.02	0.158	0.002	0.018
South Shore	10306149	61	20.43	6.39	1.28	14.36	0.70	3.76E+09	8.32%	0.02	0.133	0.002	0.011
North Shore	8183338	65	20.43	5.38	1.08	15.14	0.74	3.15E+09	6.97%	0.01	0.127	0.001	0.009
Glen Beag	2674949	65	20.43	5.38	1.08	15.14	0.74	1.03E+09	2.28%	0.01	0.295	0.000	0.007
Spruces	12266075	65	20.43	5.38	1.08	15.14	0.74	4.72E+09	10.44%	0.02	0.050	0.002	0.005
North 61A	1978944	65	20.43	5.38	1.08	15.14	0.74	7.61E+08	1.68%	N/A	N/A	N/A	N/A
Moose Bog Spring	1549240	65	20.43	5.38	1.08	15.14	0.74	5.96E+08	1.32%	N/A	N/A	N/A	N/A
Aspens Wetland	22853	65	20.43	5.38	1.08	15.14	0.74	8.79E+06	0.02%	N/A	N/A	N/A	N/A
Spruces Corner	99846	65	20.43	5.38	1.08	15.14	0.74	3.84E+07	0.09%	N/A	N/A	N/A	N/A
Pumphouse	69908	65	20.43	5.38	1.08	15.14	0.74	2.69E+07	0.06%	N/A	N/A	N/A	N/A
First Bay	2402402	65	20.43	5.38	1.08	15.14	0.74	9.24E+08	2.05%	N/A	N/A	N/A	N/A
Eagle Point	2294899	65	20.43	5.38	1.08	15.14	0.74	8.82E+08	1.95%	N/A	N/A	N/A	N/A
Frith Beach	2327641	70	20.43	4.29	0.86	16.06	0.79	9.49E+08	2.10%	N/A	N/A	N/A	N/A
Cancun	574429	65	20.43	5.38	1.08	15.14	0.74	2.21E+08	0.49%	N/A	N/A	N/A	N/A

Appendix J. Average dissolved oxygen concentrations over summer DO profile sampling in 2015. Values marked in bold had incomplete profiles, values used in calculations were simply the value for the next deepest profile. Red text indicates the period of declining hypolimnetic dissolved oxygen.

Strata (m)	Volume (m3)	Mean O2 (mg/L)	Total O2 in strata (metric tonne)
15-19	46728883.67	8.81	411.77
20-24	69178645.26	8.03	555.74
25-29	34311857.40	7.35	252.02
30-34	20369788.83	5.74	116.92
06-Mar-15		Total	1336.45
Strata (m)	Volume (m3)	Mean O2 (mg/L)	Total O2 in strata (metric tonne)
15-20	46728883.67	13.25	619.34
20-25	69178645.26	12.67	876.63
25-30	34311857.40	12.00	411.67
30-34	20369788.83	**12.00**	244.44
28-May-15		Total	2152.09
Strata (m)	Volume (m3)	Mean O2 (mg/L)	Total O2 in strata (metric tonne)
15-20	46728883.67	17.14	800.78
20-25	69178645.26	16.81	1162.78
25-30	34311857.40	15.83	543.16
30-34	20369788.83	10.79	219.79
11-Jun-15		Total	2726.50
Strata (m)	Volume (m3)	Mean O2 (mg/L)	Total O2 in strata (metric tonne)
15-20	46728883.67	10.38	485.23
20-25	69178645.26	9.63	666.33
25-30	34311857.40	9.20	315.74
30-34	20369788.83	9.05	184.35
18-Jun-15		Total	1651.65
Strata (m)	Volume (m3)	Mean O2 (mg/L)	Total O2 in strata (metric tonne)
15-20	46728883.67	9.40	439.44
20-25	69178645.26	8.81	609.35
25-30	34311857.40	8.01	274.92
30-34	20369788.83	**8.01**	163.16
23-Jun-15		Total	1486.87
Strata (m)	Volume (m3)	Mean O2 (mg/L)	Total O2 in strata (metric tonne)
15-20	46728883.67	8.04	375.89
20-25	69178645.26	7.29	504.31
25-30	34311857.40	5.92	203.06
30-34	20369788.83	5.35	108.98
21-Jul-15		Total	1192.24

Strata (m)	Volume (m3)	Mean O2 (mg/L)	Total O2 in strata (metric tonne)
15-20	46728883.67	5.99	280.00
20-25	69178645.26	5.01	346.70
25-30	34311857.40	4.32	148.06
30-34	20369788.83	**4.32**	88.00
04-Aug-15		Total	862.75
Strata (m)	Volume (m3)	Mean O2 (mg/L)	Total O2 in strata (metric tonne)
15-20	46728883.67	5.87	274.45
20-25	69178645.26	4.44	306.81
25-30	34311857.40	3.52	120.89
30-34	20369788.83	3.13	63.66
17-Aug-15		Total	765.81
Strata (m)	Volume (m3)	Mean O2 (mg/L)	Total O2 in strata (metric tonne)
15-20	46728883.67	7.08	330.93
20-25	69178645.26	3.16	218.74
25-30	34311857.40	0.91	31.22
30-34	20369788.83	0.21	4.28
23-Sep-15		Total	585.18
Strata (m)	Volume (m3)	Mean O2 (mg/L)	Total O2 in strata (metric tonne)
15-20	46728883.67	8.28	386.92
20-25	69178645.26	5.73	396.05
25-30	34311857.40	0.77	26.56
30-34	20369788.83	0.16	3.26
30-Sep-15		Total	812.78
Strata (m)	Volume (m3)	Mean O2 (mg/L)	Total O2 in strata (metric tonne)
15-20	46728883.67	8.85	413.32
20-25	69178645.26	8.35	577.36
25-30	34311857.40	2.49	85.37
30-34	20369788.83	0.22	4.48
07-Oct-15		Total	1080.53

Appendix K. **Full lift hypolimnetic aerator sizing based on calculated DO consumption and the recommended doubling of DO demand.**

	Calculated DO consumption	2X Calculated DO consumption
Consumption per day mg/day	1455000.00	2910000.00
water flow m3/day	727500.00	1455000.00
water flow m3/second	8.42	16.84
Tube radius	1.49	2.11
tube diameter	2.99	4.23
Entrance loss	0.04	0.04
Exit loss	0.07	0.07
Friction loss	0.01	0.01
Total head loss	0.12	0.12
Density of air:water mix (kg/m3)	995.12	995.27
Air flow rate Eq10 constant	1.86	1.86
Air flow rate m3/s	0.02	0.04
Air flow rate m3/min	1.33	2.58
Pressure check	1.68	3.37
Air flow check		
Pressure required	3.23	3.23
	4.58	4.58
Box dimensions (m)		
Length	10.8	15.2
Width	4.7	6.7
Height	3.5	5.0
flotation requirements	3112.4	6041.8