Restoring a Culturally Eutrophic Shallow Lake: Case Study on Quamichan Lake in North Cowichan, British Columbia

by

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BES, University of Waterloo, 2017

Project Submitted in Partial Fulfillment of the
Requirements for the Degree of
Master of Science

in the

Ecological Restoration Program
Faculty of Environment (SFU)
and
School of Construction and the Environment (BCIT)

SIMON FRASER UNIVERSITY

BRITISH COLUMBIA INSTITUTE OF TECHNOLOGY
Spring 2019

Approval

Name:	**Kathleen E Moore**
Degree:	**Master of Science in Ecological Restoration**
Title:	**Restoring a Culturally Eutrophic Shallow Lake: Case Study on Quamichan Lake in North Cowichan, British Columbia**
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Abstract

Quamichan Lake is a culturally eutrophic shallow lake located in North Cowichan on Vancouver Island in British Columbia. My research project examined the current trophic status and water quality of Quamichan Lake and investigated a number of watershed and in-lake restoration methods to return the lake back to mesotrophic (nutrient rich) conditions. Based on the data collected, Quamichan Lake is currently in a hypertrophic state caused by excess phosphorus inputs that leads to Cyanophyte phytoplankton species (cyanobacteria) to dominate during the summer. Eutrophication is both an environmental and human health issue as cyanobacteria algal blooms can disrupt the lake ecology and are toxic to most mammals. The goal of my research was to provide the Municipality of North Cowichan and Vancouver Island Health Authority with a comprehensive restoration plan to contribute to the restoration of Quamichan Lake and other lakes in southern Vancouver Island that are experiencing cultural eutrophication.

Keywords: Restoration; Eutrophication; Limnology; Watershed Management; Cyanobacteria

Acknowledgements

This project would not have been possible without the support of both the Municipality of North Cowichan and the Vancouver Island Health Authority. I would like to thank the Municipality of North Cowichan for making it financially possible to perform rigorous chemical and biological water quality sampling throughout summer 2018. I would also like to specifically thank Shaun Chadburn at MNC for being a supportive point of contact throughout this project. I am grateful for Stacey Sowa at Island Health for providing support in applying to the VIHA Summer 2018 Student Research Grant Competition and for being my “Island Health collaborator” throughout the length of the project. I would like to express my gratitude to Island Health for financially supporting me by selecting my project for the Summer 2018 Student Research Grant Competition.

I would like to thank my supervisor, Dr. Ken Ashley, for all of his expertise and guidance throughout this experience. I also thank Dr. Ruth Joy for helping me make sense of some of my data and helping navigate the world of “R.” Many thanks to Dave Harper at the Rivers Institute for providing field equipment throughout the summer sampling season. I thank Dr. Dave Preikshot for collaborating on this project, sharing of data, in-field assistance, and knowledge of the Cowichan area. I am extremely grateful of Christopher Justice for taking me out on his boat all summer to sample, providing knowledge of the area, and for always asking critical questions about the lake.

Finally, I would like to thank my friends and family I am fortunate enough to have all across the country for providing encouragement and support from near and far; as well as the 2019 graduating class of MSc in Ecological Restoration for the constant positive support and countless memories.

Table of Contents

List of Tables

List of Figures

Chapter 1: Introduction

The purpose of this study is to create a comprehensive restoration plan for Quamichan Lake, a shallow culturally eutrophic lake near Duncan, BC in the municipality of North Cowichan. This study was completed in partnership with the Municipality of North Cowichan (MNC) and Vancouver Island Health to provide guidance for the restoration of Quamichan Lake, and potentially other lakes on southern Vancouver Island that are experiencing human accelerated eutrophication.

1.1. Site Location

Quamichan Lake is located within the municipality of North Cowichan, three kilometers east of the City of Duncan (Figure 1). The lake is the largest of three lakes in the District of North Cowichan on Vancouver Island and is important for recreation, fish and wildlife habitat, and freshwater source (QWWG, 2009; McPherson, 2006). Quamichan Lake has previously been characterized as a relatively shallow, mesotrophic-eutrophic (nutrient rich) lake with a mean depth of 4.7 meters and a maximum depth of 7.9 meters. The total surface area of the lake is 3.13 km^2 (Quamichan Lake Task Force, 2016).

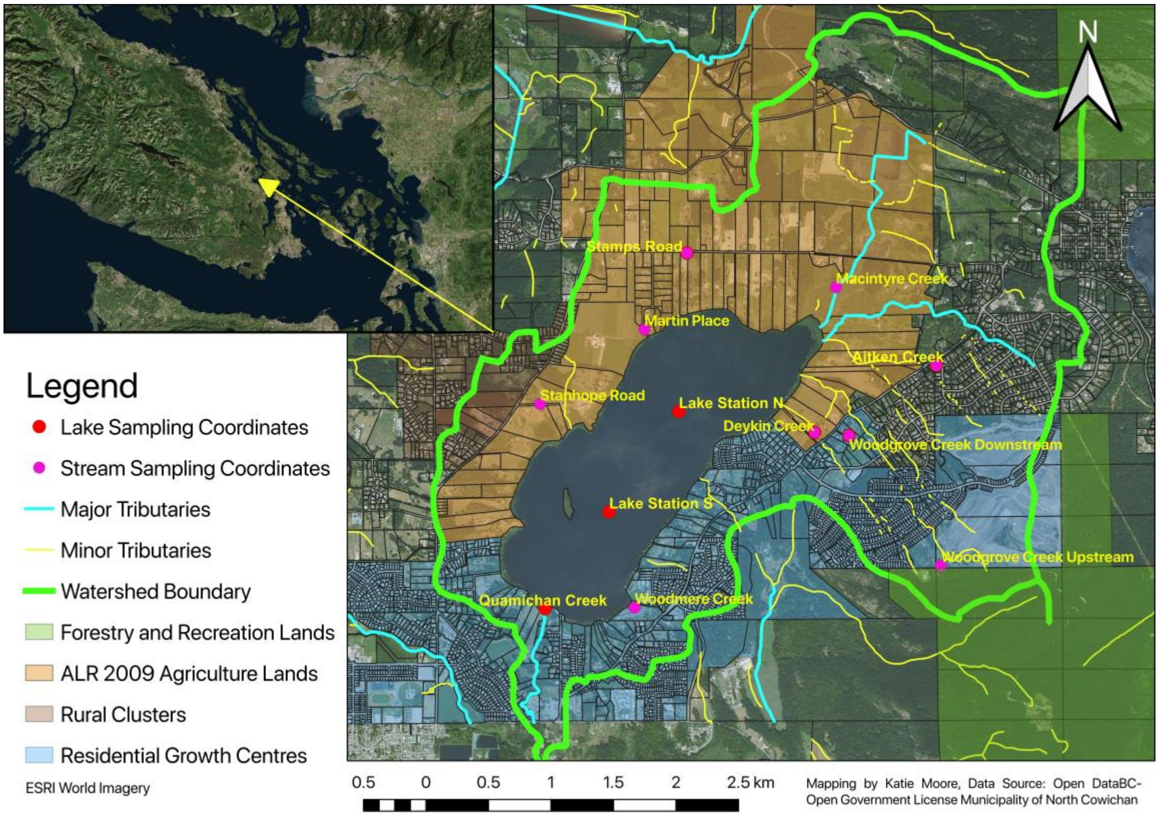

Figure 1. Map of Quamichan Lake watershed boundary and 2018 sampling locations with inset map showing location on Vancouver Island, BC.

The Cowichan region is in Canada's only Maritime Mediterranean climatic zone and has the warmest mean temperature of anywhere in Canada (CVRD, n.d.). Quamichan Lake and its surrounding ecosystem is located in the Coastal Douglas-Fir (CDF) biogeoclimatic zone (Nuszdorfer, Klinka, Demarchi, 1991). The CDF zone is characterized by dry hot summers and heavy precipitation in the rainy season which is typically between October and March (Nuszdorfer et al., 1991; Crawford, 2008). Based on Environment Canada's long-term climate data at Duncan Kelvin Creek Station (Climate ID: 1012573) 80% of the year's rainfall occurs between October and March with an average of 1,289.2 mm per year and increasing significantly from September to October (Environment Canada, 2018). The CDF zone is one of the most sensitive coastal ecosystems in BC and is dominated by *Pseudotsuga menziesii* (Douglas-fir), *Thuja plicata* (western red cedar), *Arbutus menzeisii* (arbutus), *Quercus garryana* (Garry oak) and *Alnus rubra* (red alder) forest stands. Quamichan Lake is located adjacent to some of the last remaining intact Garry oak woodlands in Canada (Pellatt, McCoy and Mathewes, 2015).

North Cowichan is located in the Nanaimo lowlands ecoregion in the Georgia Depression ecoprovince which is underlain by sedimentary rocks that have eroded from glacial retreat to form ridges and valleys in the region. The ridges are underlain by hard sandstone and valleys such as Somenos and Quamichan Lake basins are underlain by softer shale and siltstone (Jungen, Sanborn and Christie, 1985). Soil classes surrounding Quamichan Lake are dominated by Cowichan, Fairbridge, Mexicana, and Galiano soil types and major characteristics of each are summarized in the table below (BC SIFT, 2018; Jungen et al., 1985).

Table 1. Dominant soil types surrounding Quamichan Lake, adapted from BC MoE "Soils of Southeast Vancouver Island Duncan~Nanaimo Area" (Jungen et al., 1985).

Soil Name	Soil Texture	Soil Drainage	Dominant Classification
Cowichan (CO)	Silt loam	Poorly drained	Humic Luvic Gleysols
Fairbridge (FB)	Silt loam to silty clay loam	Imperfectly drained	Gleyed Eluviated Dystric Brunisols
Mexicana (ME)	Gravel sandy loam	Moderately well drained	Orthic Dystric Brunisols
Galiano (GA)	Gravelly loam	Well to moderately well	Orthic Dystric Brunisols

1.2. Site History

1.2.1. Pre-European Contact Era

Quamichan Lake is located within the traditional territory of the Cowichan Tribes, primary successor community to the Cowichan First Nation for at least the past 4,000 years (Pellat et al., 2015). The Cowichan people (the Hul'q'uni'num people) are part of the larger First Nations group referred to as the Coast Salish People. Historically, the Cowichan Nation territory once covered 376,308 ha encompassing the southern half of Vancouver Island, the Gulf Islands, and as south as Sumas and Nooksack in Washington state (Cowichan Tribes, n.d.). The Cowichan Tribes territory includes the regions of Cowichan Lake, the Cowichan and Koksilah River drainages, Cowichan Bay, Maple Bay, Shawnigan Lake, the southern Gulf Islands, and the south arm of the Fraser River (Cowichan Tribes, n.d.). Traditionally, Cowichan Tribes' home villages were along the Cowichan and Koksilah Rivers and had a population of up to 15,000 people spread over the whole territory (Cowichan Tribes, 2014). Quamichan (Kwa'mutsun) is a traditional village of Cowichan Tribes and Quamichan Lake was historically an important natural resource for the traditional people of the land. Cowichan Tribes is part of the Hul'qumi'num Treaty Group that has not been properly honoured by the government of Canada and British Columbia.

1.2.2. Post European Settlement

European settlement began on southern Vancouver Island in the mid 1800s, and the Duncan area was settled in the 1860s and developed as an agricultural center (Jungen, Sanborn and Christie, 1985). A paleoecology study was completed by Pellatt, McCoy, and Mathewes (2015) in which a sediment core was taken of Quamichan Lake and analyzed for pollen and charcoal deposits to understand vegetation change disturbance regimes in the past 500 years. Pollen analyses revealed a change in forest type from mixed oak forest to alder/conifer forest in the mid 1800's. Decreasing *Quercus* (oak) and increasing *Alnus* (alder), *Acer* (maple), *Poaceae* (grass), *Pteridium* (bracken) is likely the result of expansion of European settlement and clearing land due to logging in the watershed (Pellatt et al., 2015). Charcoal analysis indicated there were eight fire events occurring between 1745 to present, and fires frequently occurred on southern Vancouver Island simultaneously with land-clearing and slash burning agriculture that caused a rapid landscape change (Pellatt et al., 2015).

1.2.3. Quamichan Lake 1940's to Present

The charcoal analysis of the Quamichan Lake sediment core shows the lowest fire events from 1940s to present day and is interpreted as active fire exclusion programs which corresponds to the human development in this area (Pellatt et al., 2015). In the last 70 years, agricultural activities have resulted in extensive clearing and drainage around approximately half of Quamichan Lake that is designated as an agricultural land reserve (ALR) (McPherson, 2005). A large portion of the agriculture is pasture and forage crops and there is also a large number of dairy farms from Mill Bay to just north of Duncan (Jungen et al., 1985). At least 40% of the lower slopes surrounding Quamichan Lake have been modified for agricultural use (Burns, 2002). In addition, urban residential expansion has encroached into the catchment basin of the lake with growing communities in Duncan, North Cowichan, and Maple Bay.

Vancouver Island Health has reported Art Mann Park Beach at Quamichan Lake as being unfit for recreational bathing since 1986 as faecal coliform values increased between 1973 and 1995 (MoE, 1996). In addition, between 1985 and 2005 the average total phosphorus (TP) concentrations measured at 60 µg/L, ranging between 2-255 µg/L which exceed the CCME Criteria for Drinking Water and Recreation (10 µg/L) and Aquatic Life maximum (15 µg/L) (Crawford, 2008).

1.3. Current Problem and Project Significance

Quamichan Lake has a high recreational value hosting a wide array of activities including pleasure crafts, anglers, and swimmers. Most notably, the municipality of North Cowichan recently won the bid to become Rowing Canada Aviron's location for the new National Training Centre on Quamichan Lake, which is set to open in 2020. In addition, there are many waterfront properties surrounding the lake and there are three public access points that provide public access to water (Art Mann Park, Moose Road, and Sterling Ridge Park) (QWSS, 2011).

Despite the recreational importance of the lake, the water quality of Quamichan Lake has been deteriorating for the past 75 years (McPherson, 2006). The major land uses in the watershed are rural (42%), agricultural (18%), and residential (19%) (QWSS, 2011). Nutrients entering the lake, phosphorus and nitrogen, are thought to originate from runoff from outdated septic systems, agricultural and livestock operations, conversion of forested lands into grasslands or residential areas, lawn fertilizers etc. (Schindler and Vallentyne, 2008).

In August of 2016, a cyanobacterial *Microcytis* spp. algal bloom (commonly known as blue-green algae) began to develop in the lake and did not clear until early December 2016 (Quamichan Lake Task Force, 2016). The bloom formed a 0.5 cm thick floating mat that covered the whole lake. During the algae bloom of 2016, at least four dogs were killed after swimming in Quamichan Lake, likely by ingesting the blue-green algae (also known as cyanobacteria) which is toxic to many mammals (i.e. pets, livestock, and humans) (Paerl et al., 2001; Qin et al., 2015). Cyanotoxins are known to be responsible for continuous widespread poisoning of wild and domestic animals as well as human fatalities (Paerl et al., 2001).

As a result of the toxic algae bloom in 2016 and subsequent public outcry, the municipality of North Cowichan organized the Quamichan Lake Water Quality Task Force in an effort to understand the causes and discuss potential corrective measures for restoring the lake and improving water quality. The Task Force Report was released in December of 2016 summarizing the main sources of nutrient inputs as well as an overview of potential restoration options (Quamichan Lake Task Force, 2016). This applied research project report is meant to guide the long-term process of restoration of Quamichan Lake and is consistent with the recommendations from the Task Force Report.

Occurrences of cyanobacteria at high abundances are often referred to as harmful algae blooms (HABs) and can pose a serious health threat to both animals and humans (Waller, Bramburger and Cumming, 2016). The cultural eutrophication of Quamichan Lake is as much of an environmental concern as it is a human health concern, as many planktonic cyanobacterial species can produce neurotoxins, hepatotoxins, cytotoxins, and gastrointestinal toxins (Paerl et al., 2001; Qin et al., 2015). Cyanobacterial blooms can also disrupt food webs, drive hypoxia, reduce biodiversity, and produce secondary metabolites that are toxic to consumers (Qin et al., 2015). To reduce productivity, the source of the nutrients must be identified and reduced before in-lake treatments can be implemented. Both land-use management initiatives in the watershed and restoration treatments within the lake must be undertaken for Quamichan Lake to begin a long-term restorative process.

1.4. Quamichan Lake Ecology

1.4.1. Thermal Stratification and Shallow Lakes

Shallow lakes are typically defined as being less than 3 m deep and polymictic, implying that the water column mixes continuously, however they can be up to 5 m depth (Moss, 1998; Phillips, 2005). Within shallow lakes, the epilimnion, which is the upper layer where most plant growth takes place (Schindler and Vallentyne, 2008) is regularly in contact with sediment and can be easily influenced by fluctuations in the physical environment such as wind and temperature changes (Phillips, 2005). Rapid deposition of phytoplankton in shallow lakes provides a ready source of organic material and as phytoplankton decompose close to the sediment surface, they provide a source of soluble phosphorus and consume dissolved oxygen (Schindler and Vallentyne, 2008). For the purposes of this research, I classified Quamichan Lake as a shallow holomictic-dimictic lake, meaning the whole lake mixes twice per year in the spring and again in the fall based on field observations of thermal stratification data. In the summer and the winter the lake becomes thermally stratified, meaning the surface temperature (epilimnion) is much warmer than the bottom temperature (hypolimnion) and is separated by a thermocline.

Shallow lakes are more productive per unit area than deep lakes due to rapid recycling of nutrients that become available to primary producers (Beklioglu, Meerhoff, Sondergaard and Jeppeson, 2011). The role of phosphorus and nitrogen is essential to the growth of aquatic plants and are commonly limiting to plant growth, however, these two nutrients are also causal factors for eutrophication. When these limiting nutrients are in excess, plant growth increases significantly, especially phytoplankton growth in the epilimnion (Schindler and Vallentyne, 2008). In shallow freshwater, some of the major causes contributing to cyanobacterial occurrences are eutrophication, warm water temperatures, high light intensity, and stable weather conditions (Xie, Rediske, Hong, O'Keefe, Gillett, Dyble and Steinman, 2012).

1.4.2. The Role of Major Nutrients

Carbon, nitrogen, and phosphorus (C:N:P) are macronutrients needed for algal uptake and growth and are characterised by a stoichiometric ratio of 106: 16N: 1P (referred to as the Redfield Ratio) (Redfield, 1958); however, biological, geological, and physical long term processes determine the constant proportions of C:N:P ratios in lakes (Zhang, Song, Ji, Liu, Xiao, Cao and Zhou, 2018). The ratio of nitrogen (N) and phosphorus (P) play an important role in aquatic ecosystems and have strong influences on the biogeochemical cycle of the ecosystem (Zhang et al., 2018). These nutrients are also the main limiting nutrients to plant growth. The amount of nutrients in a lake can be determined by bedrock, vegetation cover, size, and human influences in the catchment area (Brönmark and Hansson, 1998). Nutrients, carbon and energy are transported upward in the food web, providing food to bacteria, algae and other primary producers that are then eaten by larger predatory organisms such as macroinvertebrates and fish (Brönmark and Hansson, 1998). Available nutrients can determine the predominant aquatic plant and organism communities present in an ecosystem.

Phosphorus is essential for all organisms as it is used in storage and transfer of genetic information, cell metabolism, and cell energy (Brönmark and Hansson, 1998). Since phosphorus is generally limiting for the growth of plants, it is also the main determining factor for production. Total phosphorus is the sum of all particulate and dissolved forms. Sediments can have a high P content and factors such as pH affects the exchange of P between sediment and water; in lakes with pH values above 8, phosphate is exchanged with hydroxide ions (OH^-) which causes it to become soluble with water (Brönmark and Hansson, 1998). Due to human impact over the last 70 years, lakes close to urban development have much higher phosphorus concentrations than undisturbed lakes. The biologically available form of phosphorus is phosphate/ orthophosphate (PO_4^{3-}) which is the only form that can be directly taken up by organisms (Brönmark and Hansson, 1998).

Nitrogen enters the lake by precipitation, fixation of atmospheric nitrogen (N_2), or input from surface and groundwater drainage (Brönmark and Hansson, 1998). Nitrogen can be limiting in eutrophic lakes when P levels are high. Most nitrogen in a lake is bound in organisms (organic N) but also occurs as molecular N_2, nitrate (NO_3-N), nitrite (NO_2-N) and ammonium (NO_4-N). In eutrophic lakes where nitrogen is limiting, certain species (i.e. *Microcytis* spp.) of blue-green algae and bacteria are able to fix nitrogen from the atmosphere and can become the predominant algal species (Brönmark and Hansson, 1998).

1.4.3. Phytoplankton, Zooplankton, and Trophic Cascade

Aquatic plants are divided into macrophytes, phytoplankton (free-living), and periphytic (substrate dwelling) algae (Brönmark and Hansson, 1998). Emergent macrophytes are plants that photosynthesize above the water surface whereas submerged macrophytes grow under water attached to substrate. Phytoplankton are algae that range in shape and size from single-celled species to filamentous forms. Phytoplankton can have wide diversity and variability and many species have adaptations to prevent zooplankton grazing such as large size and shapes, spines, and mucous membranes (Brönmark and Hansson, 1998). Productivity (nutrient concentration) and pH are important factors that determine algal predominance in a community as certain groups are characterized among gradients of these factors; in highly productive lakes with a high pH, blue-green algae are likely to be predominate (i.e. *Microcystis* spp., *Aphanizomenon* spp., *Pseudanabaena* spp.) (Brönmark and Hansson, 1998).

There are two important groups of crustacean zooplankton in freshwater. Cladocerans are a group of zooplankton that are small and generally transparent. The dominant genus of Cladocerans are *Daphnia* and *Bosmina* which are pelagic filter-feeders mainly eating algae and bacteria to some extent (Brönmark and Hansson, 1998). Cladocerans are also an important food source for fish. In addition, Copepods are another group of zooplankton that are free-swimming and can be divided into *Calanoids*, *Cyclopoids*, and *Harpacticoids* (Brönmark and Hansson, 1998). Copepods have a varied diet but are known to graze on algae and other zooplankton (Brönmark and Hansson, 1998).

A lake food web can be simplified and broken down into four trophic levels: piscivores, planktivores, herbaceous zooplankton, and phytoplankton (Carpenter, Kitchell and Hodgson, 1985). The trophic cascade of a freshwater lake involves fish eating zooplankton, which eat phytoplankton to control lake species composition, biomass and productivity. Prey fish biomass declines as predator fish increase, and in turn, zooplankton biomass increases to reduce phytoplankton biomass (Carpenter et al., 1985). Herbivorous zooplankton alter phytoplankton species composition and size structure by selective grazing and nutrient recycling (Carpenter et al., 1985). A “top down” control of food web dynamics in theory states that zooplankton grazing can control phytoplankton abundance.

1.4.4. Alternative Stable States and Feedback Loops

Alternative states and feedback loops have been studied since the 1960s through the research of resiliency and adaptive capacity in which a community within an ecosystem can be found in one of several stable states (Holling, 1973; Beisner et al., 2003). Alternative stable states can occur in shallow lakes in the form of shifting abruptly from a turbid state with phytoplankton and suspended matter dominance to a clear state with dominance of submerged plants (Scheffer et al., 1990; Scheffer and van Nes, 2007; Beklioglu et al., 2011). The switch between these stable states is influenced by a number of mechanisms that can shift the system into the alternative state and is best described using the “ball and cup” diagram (Figure 2). In a low nutrient, clear water state, abundant submerged macrophytes promote habitat for macro-invertebrates that graze on phytoplankton, uptake nutrients otherwise available to phytoplankton, and oxidize the sediment that reduces the release of internal phosphorus (Williams, 2005; Scheffer and van Nes, 2007).

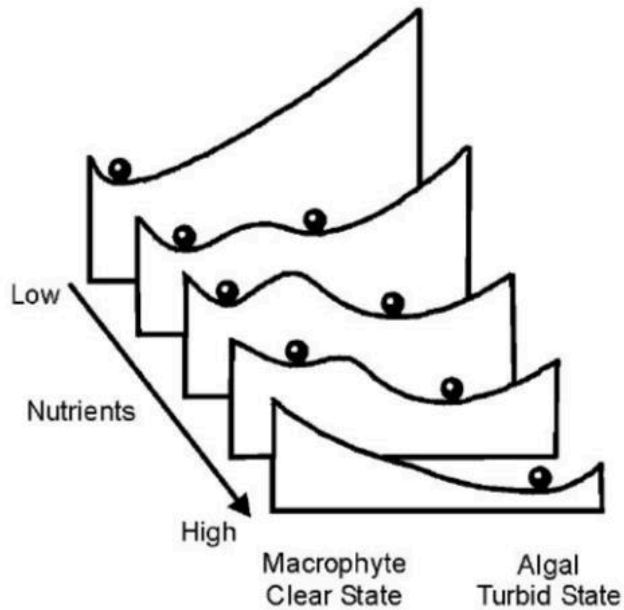

Figure 2. Two potential stables states in a shallow lake of macrophyte clear state and algal turbid state under a range of nutrient concentrations and influenced by multiple mechanisms. (Williams, 2005).

Alternatively, in a nutrient-rich and turbid water state, fish can promote phytoplankton by recycling nutrients and preying on zooplankton that would otherwise clear the water of phytoplankton. Phytoplankton can inhibit the development of macrophytes, increasing the turbidity of the water, and reducing habitat for grazing invertebrates (Scheffer and van Nes, 2007). Phytoplankton predominated water states are enhanced when the phytoplankton species present consist of large filamentous or colonial blue-green algal species that are inedible to zooplankton (Williams, 2005). Each stable state in the lake has a number of mechanisms that maintain the state and through trophic cascade of fish, plankton, invertebrates, macrophytes and nutrients in a reciprocal feedback mechanism that determines which state is present and but also when a shift between states happens (Williams, 2005). Understanding the mechanisms that determine each alternative state in a lake can aid in restoration efforts as it is possible to force a shift of states but the complex feedback loops within a lake can also inhibit the success of lake restoration.

1.5. Ecological Stressor: Eutrophication

In the past 50 years, eutrophication has constituted the most serious environmental threat to lakes worldwide (Beklioglu et al., 2011). Eutrophication has become a common issue in many urban lakes on Vancouver Island including Langford Lake, Elk/Beaver Lake, and Quamichan Lake with deteriorating water quality that is a concern for the ecosystem as well as human health.

Eutrophication is often described as the natural ageing process of lakes in which external sources of nutrients, organic matter, and sediment from the surrounding land accumulate in the lake, gradually filling it and increasing its biological production (Phillips, 2005). Under natural circumstances this process takes place in geological time, however under human influences this process can accelerate to mere decades, which is known as cultural eutrophication (Schindler and Vallentyne, 2008). Nutrient enrichment in a lake from nitrogen and phosphorus causes an accelerated growth of algae and other plant life that produces an imbalance of predominant organisms and water quality (Dokulil and Teubner, 2011). Major symptoms of cultural eutrophication include excessive growth of algae, degraded water quality, and increased sedimentary fluxes of decomposing algae that can induce oxygen deficits (Phillips, 2005). In eutrophic lakes, as productivity increases, deep water oxygen depletes and becomes anoxic and cannot support aquatic organisms, especially cold-water fish that require high dissolved oxygen levels such as salmonids (Schindler and Vallentyne, 2008).

Rapid deposition of phytoplankton in shallow lakes provides a source of organic material and its decomposition close to the sediment surface provides a source of soluble phosphorus (Phillips, 2005). The ability of phosphorus to enter the water column from sediment depends on the oxygen concentration gradient across the sediment-water interface and amount of physical and biological disturbance, which can significantly increase the rates of release and dispersion (Boström et al., 1982). In a shallow lake, there is likely to be a persistent return of phosphorus to the water column, which can continue to support the growth of phytoplankton for many years following a reduction in external nutrient loading (Phillips, 2005). This process known as ‘internal loading’ makes restoration of shallow eutrophic lakes very challenging.

Nutrient loading into lakes is strongly affected by the hydrology of the watershed as well as catchment characteristics such as soil types and land use. Shallow lakes in lowland areas are vulnerable to nutrient enrichment through the conversion of terrestrial ecosystems to agricultural and residential lands that flow into the lake (Kosten et al., 2009). Phosphorus loading from non-point sources is the biggest threat to Quamichan Lake. The land use zones surrounding Quamichan Lake such as agriculture, hobby farms and residential areas contribute to phosphorus run-off into the lake as well as point source pollution from wastewater effluent, waste disposal sites, construction sites and more (Quamichan Lake Task Force, 2016). Within a shallow lake such as Quamichan, phosphorus builds up within the sediments from continuous deposition, sinking of dead surface biomass, and prolonged internal loading which mixes twice per year (fall and spring) bringing phosphorus to the surface again resulting in a positive feedback loop (Quamichan Lake Task Force, 2016).

Excess primary production causes growth of algal species, particularly cyanobacteria, resulting in algal blooms and deterioration of water quality (Zębek and Napiórkowska-Krzebietke, 2016). These processes of algal growth are effectively limited by reducing nutrient inflow especially phosphorus, using lake-basin restoration methods, and including external protective methods in the lake’s catchment area (Zębek and Napiórkowska-Krzebietke, 2016). However, it is difficult to establish an effective restoration plan for a specific lake, and selection of restoration methods should consider the lakes morphological and hydrological conditions, management of catchment area, and local conditions (Zębek and Napiórkowska-Krzebietke, 2016). Understanding when cyanobacteria is present and environmental factors associated with HABs is critical to the minimization of health risks and effective lake management (Xie et al., 2012). For Quamichan Lake to be restored back to a mesotrophic state, the nutrient inputs from external point and non-point sources and internal sources must be identified and controlled. Only after the input of phosphorus from the surrounding watershed is reduced, in-lake restoration techniques can be initiated.

1.6. Desired Future Conditions

Currently the ecological trajectory of Quamichan Lake is in a hypereutrophic state and more frequent and longer lasting cyanobacterial algae blooms can be expected if the current phosphorus levels remain without restoration intervention. The desired future state of Quamichan Lake is to move from a turbid nutrient-rich stable state dominated by phytoplankton towards a clearwater state dominated by submerged macrophytes. The goal for restoration of Quamichan Lake is to create a suitable ecosystem for aquatic life as well as a valued recreational area for human use that has not been possible for about 30 years.

Since all lakes can vary in their ecology, a reference site is not appropriate to use but rather it is recommended to use reference conditions as a goal for restoration. Canadian guidelines outline 'trigger ranges' for desired phosphorus concentrations and if the upper limit of the range is exceeded it indicated a potential problem that "triggers" further investigations (Table 2) (CCME, 2004). The current TP levels in Quamichan Lake are triple that of what CCME classifies as hyper-eutrophic. The aim is to reduce the TP in the lake towards a meso-eutrophic status (20-35 µg/L) and eventually back to a mesotrophic state (10-20 µg/L) (CCME, 2004).

Table 2. Total phosphorus trigger ranges for Canadian lakes and rivers (CCME, 2004).

Trophic Status	Total Phosphorous (µg/L)
Ultra-oligotrophic	< 4
Oligotrophic	4 – 10
Mesotrophic	10 – 20
Meso-eutrophic	20 – 35
Eutrophic	35 – 100
Hyper-eutrophic	> 100

1.7. Project Goal and Objectives

The goal of this research is to provide the municipality of North Cowichan with a comprehensive restoration plan to restore Quamichan Lake to historic conditions of a mesotrophic, moderately nutrient-rich lake. In addition, I will provide valuable recommendations to MNC and VIHA to alleviate environmental and health concerns surrounding eutrophic lakes.

The key objectives for achieving this goal are as follows:

1. Perform baseline data collection of Quamichan Lake including physical water quality characteristics, nutrient water quality analysis, phytoplankton identification and algae biomass using chlorophyll-a analysis; as well as surrounding landscapes land use inventory;

2. Identify tributary streams and sites of high nutrient input through fall “first flush” sampling;

3. Conduct research on reducing non-point nutrient sources through catchment basin management options and in-lake restoration techniques to mitigate internal loading;

4. Create a restoration plan to begin to guide the long-term recovery of Quamichan Lake back to historical/ mesotrophic conditions;

5. Create a long-term monitoring plan to measure the level of success of ongoing restoration treatments and maintenance;

Chapter 2. Site Assessment

2.1. Methods

To create a restoration plan for Quamichan Lake, current conditions were assessed by various field sampling techniques to gather baseline data. In the field, a series of water quality parameters were sampled for both physical and chemical metrics as well as biological samples of phytoplankton. In addition to water quality samples, phytoplankton samples and chlorophyll-a samples were sent to a third-party laboratory for analysis. Pertinent literature was reviewed was used to gather background information on limnological processes and potential lake restoration techniques.

2.1.1. Field Methods

Water quality parameters were measured from April 2018 to November 2018 at two locations on Quamichan Lake (Lake Station S, and Lake Station N) as well as a third sample at the mouth of the lake's output, Quamichan Creek (Figure 1). Sample locations were chosen based on previous monitoring data collected by the province of British Columbia and ensuring good coverage over the lake. During the summer, when the lake is stratified, two water samples were taken at each sampling location, a surface sample (~0.5 m) and deep sample (5 m). These stations were sampled twice a month from May-September when the lake is most susceptible to algae blooms. A spring lake sample was taken in April and fall sample taken in November from one lake surface location since the lake was isothermal. A fourth sample at Stamps Road (a known input to Quamichan Lake) was also to be sampled throughout the summer, however, it was found to be dry from May- December 2018. In addition, three streams Deykin Creek, Woodgrove Creek S, and Aitken Creek were found to be flowing in June and were sampled from July-November 2018. Water clarity was measured using a 20 cm Secchi disk and measuring turbidity (FNU) at each lake station throughout the sampling season. Secchi depth is used to provide a visual measure of water clarity and optical depth (CCME, 2011). Turbidity is a measure of total suspended solids in the water column. Both parameters can be an indication of lake trophic status and which stable state the lake is currently in.

At each sampling station, water quality was measured for temperature, dissolved oxygen, conductivity and pH. In addition, water samples were obtained and sent by an overnight courier in coolers on ice to ALS Labs in Burnaby for analysis of dissolved orthophosphate and total dissolved phosphate, total phosphorus, dissolved inorganic nitrogen (nitrate, nitrite, ammonia), total nitrogen, and alkalinity. At the lake stations a turbidity meter (LaMotte 2020) was used to measure total suspended solids, and a 20 cm diameter Secchi disk was used to measure the clarity of the water.

In addition to water quality samples, biological samples of phytoplankton and chorophyll-a were obtained from Lake Station N locations. Both samples were obtained using an integrated tube sampler (1.25 in. diameter) lowered to 3 m to get a heterogenous sample of the epilimnion for analysis. The tube sampler was lowered to 3 m twice and emptied into a bucket. Part of the water sample was transferred to a 1 L amber bottle and preserved with Lugol's solution (EPA, 2010). This sample was sent to Biologica Environmental Services Ltd. in Victoria for taxonomic identification and biomass estimates for all taxa present. The remaining water sample was filtered through a glass microfiber filter (0.7 µm pore size, 25 mm diameter) using a 60 mL syringe and frozen for later chlorophyll-a analysis in the lab (Appendix C3).

2.1.2. Laboratory Methods

All chemical water quality samples were sent to ALS Laboratory in Burnaby and measured according to corresponding method and detection limit (Table 3).

Table 3. Parameters analyzed at ALS Labs and methods and detection limits for each. Modified from quote issued on March 23, 2018.

Parameter	Method Reference	Reporting Detection Limit	Units
Alkalinity, Total (as CaCO3)	APHA 2320 Alkalinity	1	mg/L
Ammonia, Total Dissolved (as N)	J. ENVIRON. MONIT., 2005,7,37-42, RSC	0.0050	mg/L
Nitrate (as N)	EPA 300.1 (mod)	0.0050	mg/L
Nitrite (as N)	EPA 300.1 (mod)	0.0010	mg/L
Orthophosphate-Dissolved (as P)	APHA 4500P Phosphorus	0.0010	mg/L
Phosphorus (P)- Total	APHA 4500P Phosphorus	0.0020	mg/L
Phosphorus (P)- Total Dissolved	APHA 4500P Phosphorus	0.0020	mg/L

Total Nitrogen	APHA4500-P(J)/NEMI9171/USGS03-4174	0.030	mg/L

Preserved phytoplankton samples were sent to Biologica Environmental Services Ltd. in Victoria for taxonomic analysis of predominant taxa as well as biovolume measurements for all taxa present. Algal taxa were identified by counting algal cells (single cells, colonies, coenobia, or filaments) from a randomly located field of view (FOV) using a Zeiss Axio Vert A.1 inverted phase microscope at 400x magnification until a minimum of 300 units were enumerated (Biologica, 2018). The mean number of cells per unit (=1 for single cells, >1 for all other algal forms) were estimated for all taxa. Phytoplankton were identified to genus, where possible. Species level identifications were only identified for taxa where there are reliable taxonomic references (Biologica, 2018). Only viable cells (those that appeared to be alive at the time of collected) were included in the density count.

Biovolume measurements depend on a number of assumptions including mean algal dimensions, conversion of linear to three-dimension measurements, and variation in conversion of biovolume to biomass and cell carbon (Bellinger and Sigee, 2010). Biovolume of the phytoplankton samples was measured by Biologica Environmental Services Ltd. by measuring at least 10 specimens of each taxon for each batch of samples and applying standard geometric formulas best fitted for the shape of the cell (e.g. cylinder, prolate spheroid, rectangular box etc.) (Biologica, 2018). For multicellular taxa (filaments or colonies) biovolumes were obtained for individual cells and multiplied by the mean number of cells in the colony or filament. All biovolume estimates were expressed as microliters per liter (µL/L) and cell measurements were taken using a Zeiss Axio Vert A.1 inverted phase contrast microscope at 400x magnification and Zen 2.3 (blue edition) software (Biologica, 2018).

Chlorophyll-a samples were sent to ALS for analysis using procedures modified from EPA method 445.0. Filters were processed by a routine acetone extraction followed with analysis by fluorometry using the non-acidification procedure. This method is not subject to interferences from chlorophyll-b. The reporting detection limit of 0.010 µg/L was used. It is to be noted that all samples analyzed exceeded ALS Labs’ recommended hold time of 28 days and therefore, hence the results are subject to some error.

2.2. Physical and Chemical Limnology

2.2.1. Temperature and Dissolved Oxygen

Isothermal (mixing) conditions in the lake were observed in May 2018 (although a vertical temperature profile was not taken prior to June 2018) with surface temperature readings of 17.9 °C (Lake Station N) and 16.8°C (Lake Station S) and deep temperature readings of 17.3 °C (Lake Station N) and 13.3°C (Lake Station S). The lake was thermally stratified from June to August 2018 (Figure 3). The highest surface temperature readings were observed in July at 25.7 °C (Lake Station N) and 24.9 °C (Lake Station S). The upper lethal temperature limit for various salmonid species (cold-water species) in British Columbia has been recorded as ~25 °C and ideal temperature ranges from ~12-14 °C (Environment Canada, 2014). The upper lethal temperature represents the highest temperatures a fish can be acclimated above the lethal limit, survival depends on duration of exposure and life history stages in species. Increased temperature has been shown to lead to increased productivity of phytoplankton, with temperatures 15 to 20 °C ideal for growth, and temperatures above 35 °C allowing for Cyanophyta become the predominant phylum (Environment Canada, 2014). Isothermal conditions were observed again in September and November as temperature of the whole lake was 18.5 °C and 11.4 °C, respectively (Figure 3). In addition, increasing water temperature decreases the solubility of dissolved oxygen, freshwater is saturated with 14.6 mg/L of DO at 0 °C and declines to 8.3 mg/L DO at 25 °C (Ministry of Environment, 1997).

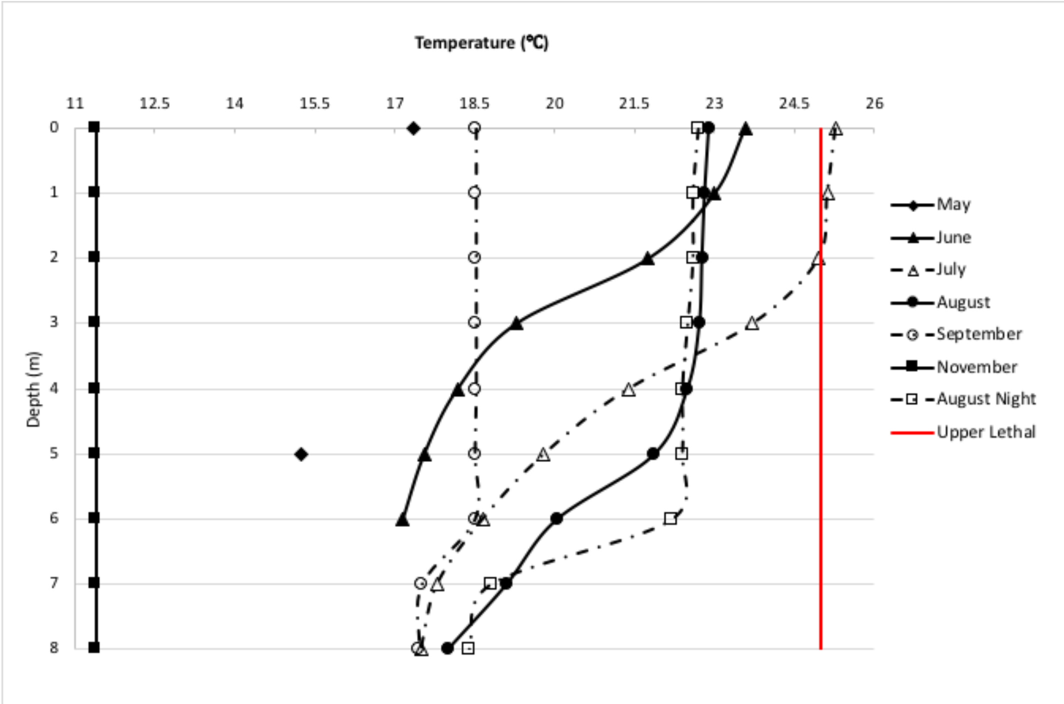

Figure 3. A vertical lake depth profile of temperature as varying depths showing thermal stratification June-August and isothermal mixing in September and November.

Dissolved oxygen (DO) concentrations correlated with thermal stratification as the DO was constant when the lake was isothermal (April-May and September-November) and a gradient of declining DO was observed when lake was thermally stratified. Anoxic conditions are defined as DO <0.01 mg/L at which sugar cannot be respired to carbon dioxide and water, and products of anaerobic metabolism (i.e. ethanol and organic acids) accumulate instead (Pokorny and Kvet, 2004). In addition, the provincial water quality guidelines states that aquatic life need a long-term average DO concentration of > 8 mg/L to survive without deleterious effects (MOE, 2018). Anoxic conditions were observed in June and July with lake bottom DO measured at less than 0.01 mg/L (Figure 4; Appendix B). In addition, DO measurements were rarely above 8 mg/L below 3 m depth for most sampling events (Figure 4; Appendix B).

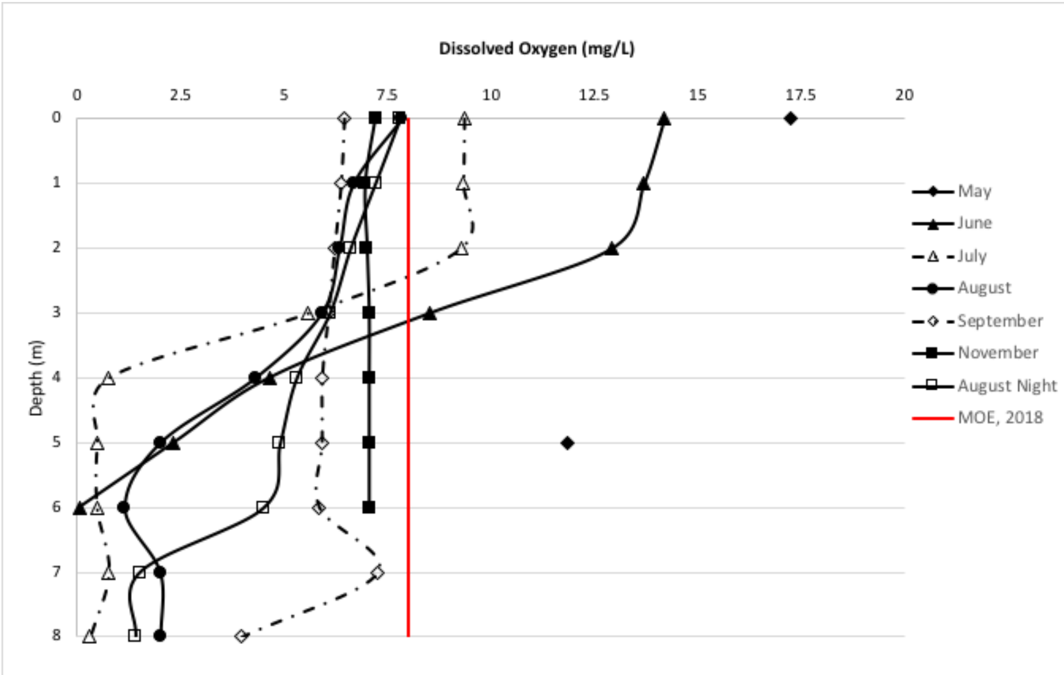

Figure 4. A vertical lake depth profile of dissolved oxygen at varying depths showing anoxic conditions near sediment surface.

A fish kill was observed on August 13 that consisted mostly of *Lepomis gibbosus* (pumpkinseed) fish (Appendix C1). High water temperature at the surface of the lake and DO concentrations below 8 mg/L for whole water column and close to anoxic at 5 m is a probable causal factor for the fish kill. In addition, the pH on August 13 was average 9.2 at surface and 7.7 at the bottom (Appendix B) which could have contributed as well. The development of cyanobacteria in spring which leads to a high pH in the whole water column that in turn, increases the fraction of unionized ammonia (Levit, 2010). In freshwater, dissolved ammonia exists in two forms, NH_3 (unionized) and NH_4^+ (ionized), ionized ammonia does not easily cross fish gills and is less bioavailable than the unionized form (Levit, 2010). However, the unionized form of ammonia can cross from water into fish, and once inside can convert to ionized form and cause cellular damage in fish. The fraction of NH_3 converting to NH_4^+ increases by 10 times for each one unit rise in pH (Environmental Protection Agency, 2013; Levit, 2010) (Figure 5).

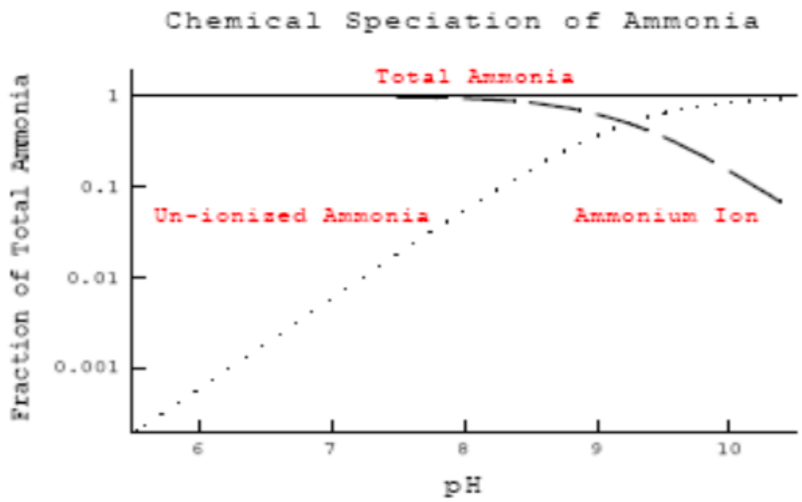

Figure 5. Fraction of chemical speciation of ammonia present with increase in pH at 25°C. The increase in unionized ammonia with increased pH is one reason why toxicity of total ammonia increases as pH increases (EPA, 2013).

2.2.2. Water Clarity

Natural variation of water quality within a season can occur in general as temperature rises, productivity increases, the lake becomes more turbid, and as a result reduced water clarity is observed. The deepest Secchi depth was observed on May 23 at 3.7 m and most shallow on August 13 at 0.3 m (Figure 6). Secchi depth fluctuated between sampling events as the lake system experienced clear water and turbid states. Turbidity was measured using Formazine Nephlometric Units (FNU) which is the equivalent to Nephlometric Turbidity Units (NTU). The mean turbidity over the sampling period was 8.7 FNU, and some outliers were observed. Quamichan Creek measured 58.7 FNU on May 9 and 24.9 FNU on June 7 (Figure 7). Lake Station S (Shallow) measured 30 FNU on July 18. The BC Ambient Water Quality Guidelines for turbidity adopted standards from the Ministry of Health state that for recreational and aesthetic use the water should have a Secchi reading of at least 1.2 m and a maximum limit of 50 NTU (Ministry of Environment, 2001).

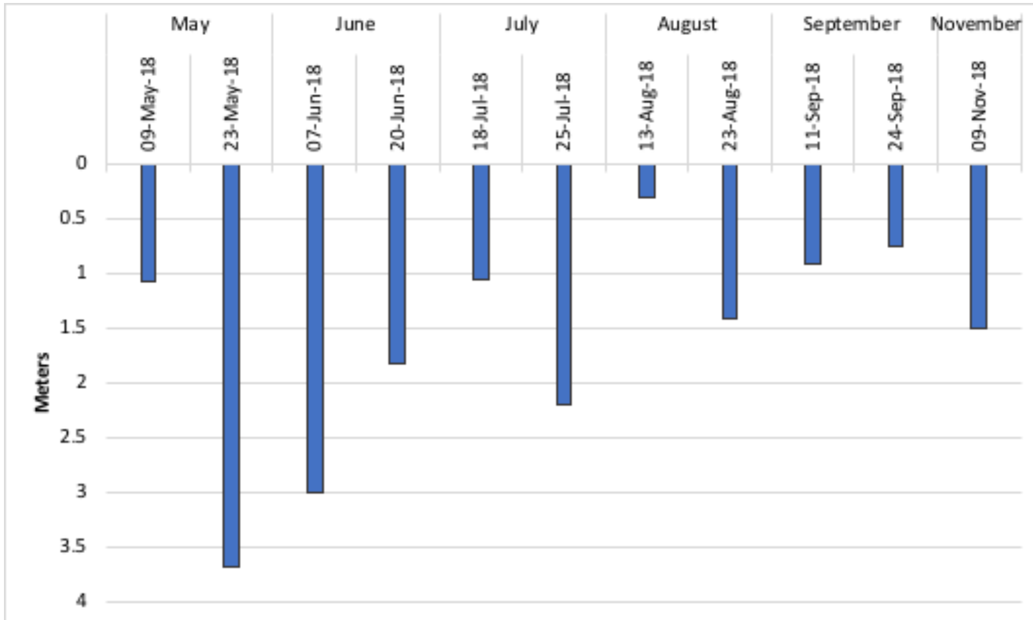

Figure 6. Secchi depths in meters (averaged between Lake Station N and S) from May to November 2018.

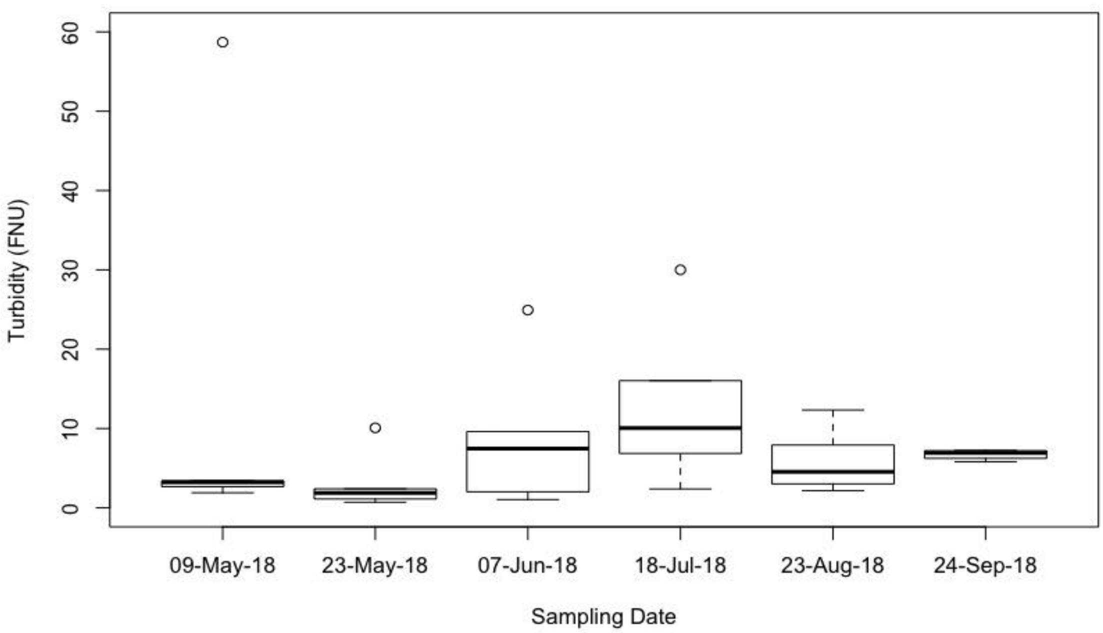

Figure 7. Turbidity measurements in FNU from May to September 2018 for all three sampling locations in the lake, from 5 measurements per sampling date. Outliers are presented as circular dots that are outside the inter-quartile range of 1.5.

2.2.3. Phosphorus

All forms of phosphorus were analyzed in a lab for the lake sampling stations. Total phosphorus is the measure of all particulate and dissolved forms of phosphorus. Although much of it is not bioavailable to algae, it is a useful parameter to assess overall trophic status of a lake. The recommended British Columbia total phosphorus thresholds for aquatic life is 5 to 15 µg/L (MOE, 2018) and for recreational activity is 10 µg/L (MOE, 2017). Based on the results, Quamichan Lake TP levels have far exceeded the recommended guidelines (Figure 8). Lake Station N (n=22) soluble reactive phosphorus (SRP) averaged 217.6 ± 62 µg/L, TDP averaged 253.4 ± 65 µg/L; Lake Station S (n=20) SRP averaged 233.6 ± 69 µg/L, TDP averaged 271.8 ± 73 µg/L; Quamichan Creek (n= 12) SRP averaged 180.7 ± 44 µg/L, TDP averaged 216.9 ± 48 µg/L (Appendix D). The high pH measured in the lake throughout the summer (Appendix B) corresponds with the high phosphorus concentrations because pH above 8 causes phosphate to become soluble with water, contributing to internal loading (Brönmark and Hansson, 1998).

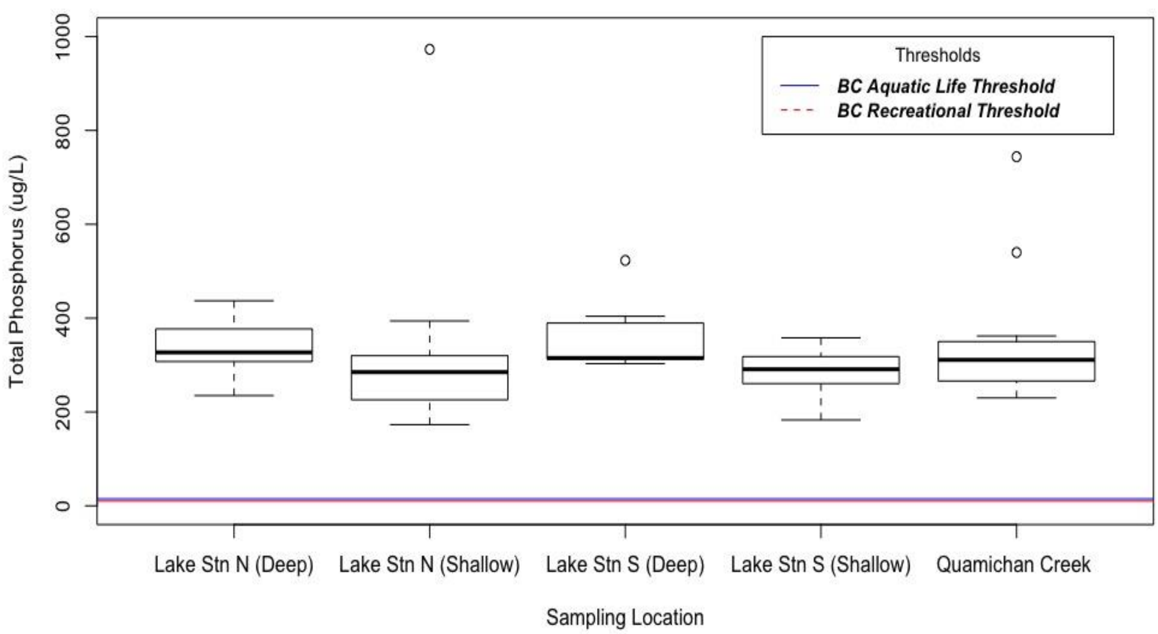

Figure 8. A boxplot with whiskers that show the total phosphorus (µg/L) measurements taken at each sampling station from April- November 2018. Outliers are presented as circular dots that are outside the inter-quartile range of 1.5. The recommended British Columbia threshold for recreational use based on aesthetic water clarity and protection of aquatic life are displayed as a red and blue line.

The Government of BC has been monitoring the surface water quality of Quamichan Lake at multiple stations as early as 1988 and as recently as 2018. There are two provincial sampling stations in the lake, Quamichan Lake Center (EMS ID: E207466), Quamichan Lake South (EMS ID: E207465), that are comparable to the locations sampled in the summer 2018 (Lake Station N and Lake Station S). These sites have been tested for forms of phosphorus, nitrogen and some chlorophyll-a analyses (DataBC). The TP concentration of the epilimnion has increased significantly since August 2015 (468 µg/L) and the two highest concentrations were recorded in July 2018 (744 µg/L) and August 2018 (973 µg/L). The TP concentration of the hypolimnion was recorded as unusually high in 1988 (>1,500 µg/L), but lower values were recorded in 1992 (<50 µg/L). High TP concentrations were recorded in August 2015 (1,220 µg/L) and August 2016 (1,060 µg/L) and the highest recorded concentration in 2018 was in August as well (731 µg/L) (Figure 9).

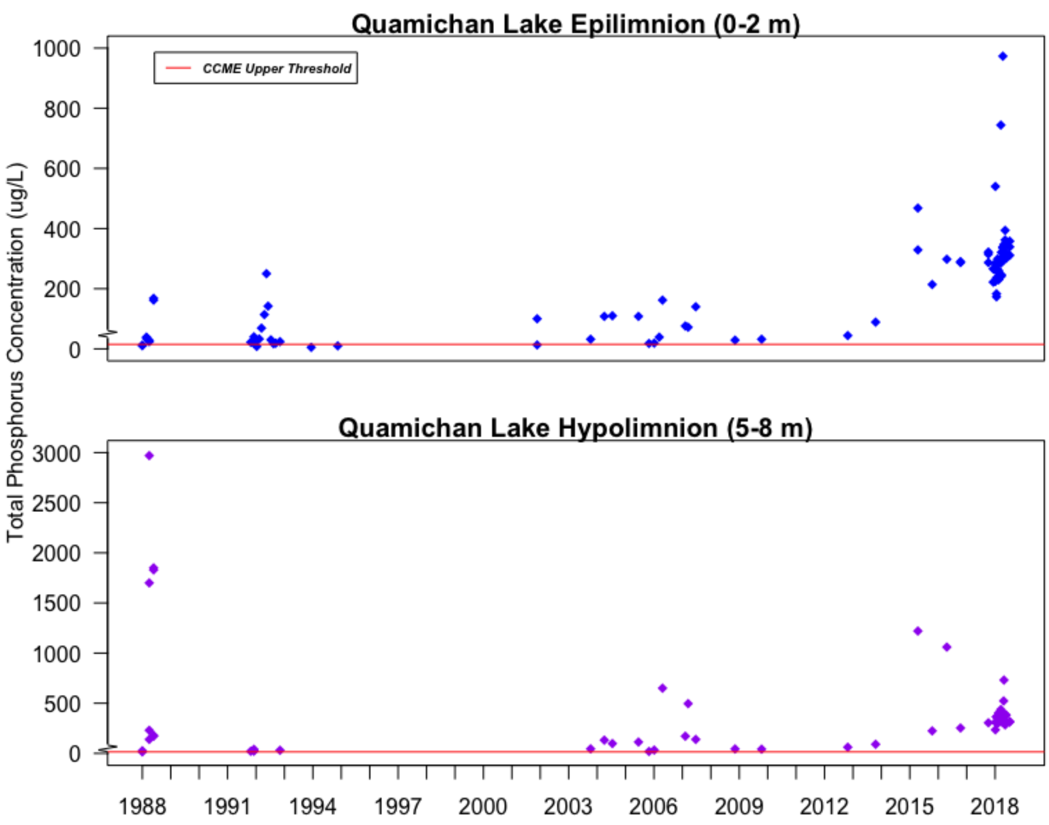

Figure 9. Historical total phosphorus (µg/L) data for Quamichan Lake epilimnion (0-2 m depth) and hypolimnion (5-8 m depth) sampled from 1988 to 2018 at two provincial sampling stations (EMS ID: E207465, E207466) and 2018 sampling stations (Lake Station N and Lake Station S).

2.2.4. Nitrogen

All forms of nitrogen were analyzed in a lab for the sampling locations. Total nitrogen (TN) is a measure of all particulate and dissolved forms of nitrogen. Although there is no threshold for the amount of total nitrogen in a system, the ratio between N and P should be approximately 16N:1P for a balanced system (Redfield, 1958). Based on the results of total nitrogen measured, Quamichan Lake is a nitrogen limited system (Figure 10).

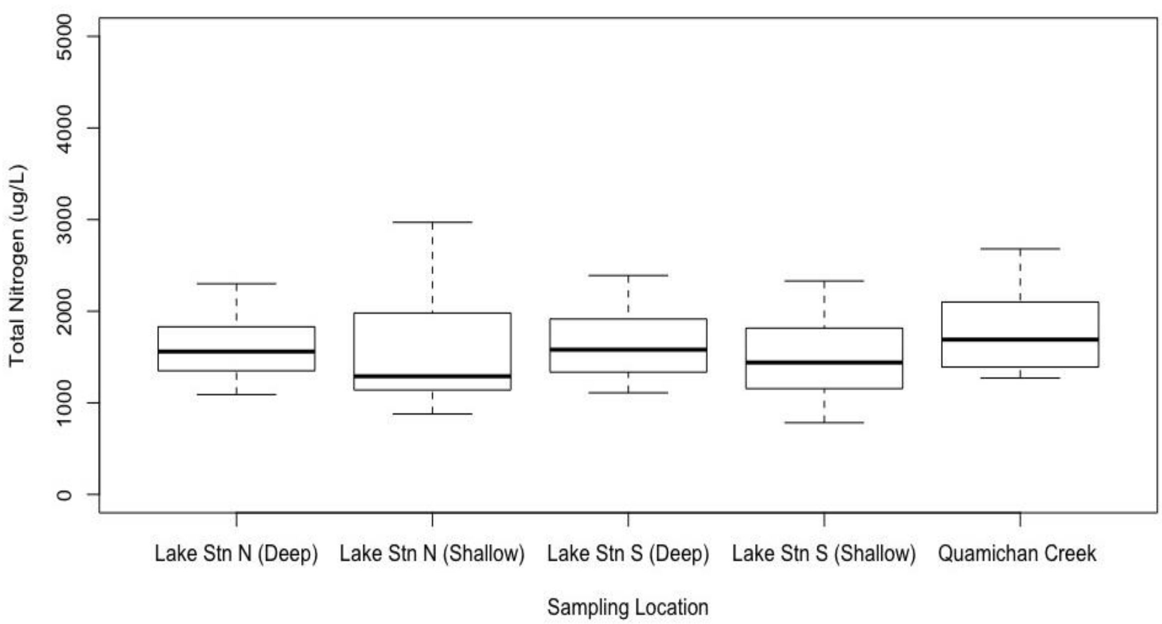

Figure 10. A boxplot with whiskers that show the total nitrogen (μg/L) measurements taken at each sampling station from April- November 2018. Plot was created using an inter-quartile range of 1.5.

2.2.5. N:P Ratio and Bioavailability

The ratio of ambient nitrogen to phosphorus in an ecosystem is important to determine whether the system is limited by one macronutrient over the other or if it co-limited (Ashley and Stockner, 2003). The ratio of N:P is best approximated by the bioavailable forms of nitrogen and phosphorus which are most readily taken up by phytoplankton in the system. These forms are dissolved inorganic nitrogen (DIN) (ammonia + nitrite + nitrite) and SRP (orthophosphate). A DIN:SRP ratio of 10:1 (atomic weight) or 4.5:1 (weight to weight) is used as a measure of balanced nutrients for algal plant growth (Ashley and Stockner, 2003). Minimum target DIN:SRP ratios should aim for 7.5: 1 (i.e. 30 µg/L DIN: 4 µg/L SRP) to ensure the system does not become co-limited by N (Ashley and Stockner, 2003). Based on the DIN:SRP weight to weight ratio of Quamichan Lake, it is clear that the lake is nitrogen limited as all samples were < 4.5:1 (Figure 11). Therefore, the lake is susceptible to cyanobacteria blue-green algae blooms that are able to fix nitrogen from the atmosphere when their source is limited.

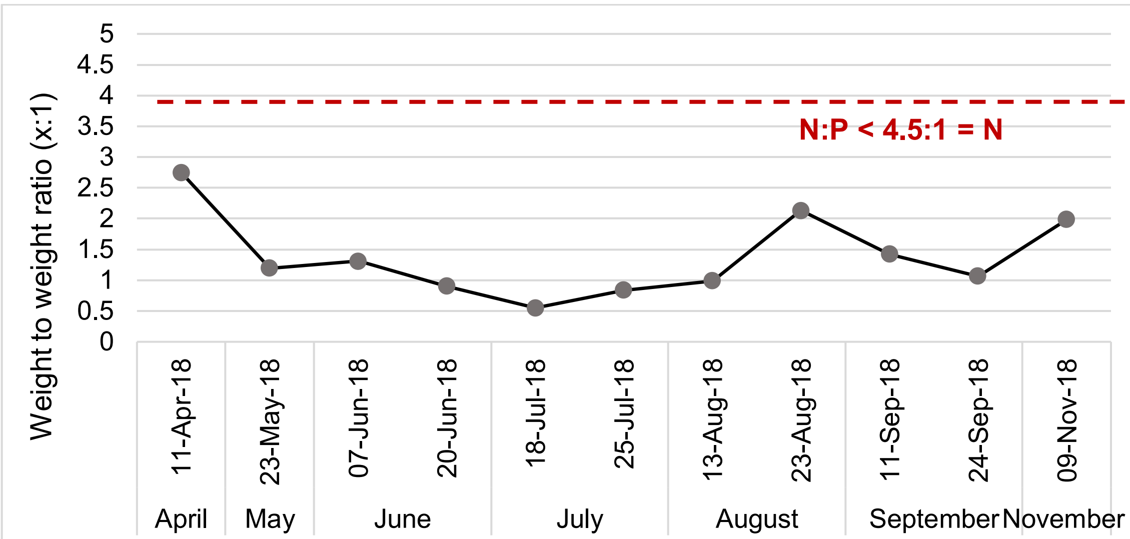

Figure 11. The average DIN: SRP ratio at each sampling date for all three lake stations in 2018. The red dashed line indicates the point below which a system is N-limited.

2.3. Biological Samples and Trophic Status

Biological samples were collected from the lake from May- September 2018 during the growing season, when primary productivity was highest. Phytoplankton taxonomy and chlorophyll-a analysis was used to determine algal biomass and trophic status of the ecosystem.

2.3.1. Phytoplankton Taxonomy

Predominant phytoplankton taxa and their cell biovolume was determined by Biologica Environmental Services Ltd. Analysis was completed for a total of 9 samples from May-September. Analysis of phytoplankton community from the epilimnion of a lake can provide information about aquatic conditions in general and is the basis of broad categorization of the lake's trophic status in terms of their overall productivity and species composition (Bellinger and Sigee, 2010).

Phytoplankton groups can be categorized according to their size and taxonomic features and can be grouped into four classes: picoplankton, nanoplankton, microplankton, and macroplankton (Bellinger and Sigee, 2010) (Table 4). Zooplankton prefer to prey on species smaller than 30 μm but blue-green algae (Cyanophyta) are not edible because of their toxicity. Phytoplankton phyla present in freshwater include Cyanophyta (blue-green algae), Chlorophyta/ Charophyta (green algae), Euglenophyta (euglenoids), Dinophyta (dinoflagellates), Cryptophyta (cryptomonads), Chrysophyta (chrysophytes), and Bacillariophyta (diatoms) (Bellinger and Sigee, 2010).

Table 4. Four categories of phytoplankton and their respective sizes (Bellinger et al., 2010)

Size Category	Size (μm)	Unicellular Organisms	Colonial Organisms
Picoplankton	0.2-2	Blue-green algae (*Synechococcus, Synechocystis*)	
Nanoplankton	2-20	Blue-green algae Cryptophytes (*Cryptomonas, Rhodomonas*)	
Microplankton	20-200	Dinoflagellates (*Ceratium, Peridinium*)	Diatoms (*Asterionella*)
Macroplankton	>200		Blue-green algae (*Anabaena, Microcystis*)

Phytoplankton species composition can be related to trophic status in three main ways: seasonal succession, biodiversity, and determination of bioindices (Bellinger and Sigee, 2010). Throughout the growing season, there is a natural successional cycle of algae. Phytoplankton growth starts with early spring diatom bloom, which is followed by zooplankton growth increase by grazing on the diatoms and results in a "spring clearwater phase" (Bellinger and Sigee, 2010). During early summer, green-algae becomes predominant followed by late summer cyanobacteria bloom and in the fall dinoflagellate and diatoms become predominant again. The natural successional cycle of algae can be influenced by increased nutrients, temperature and dissolved oxygen level. In eutrophic lakes, spring diatom bloom is limited, followed by a spring clearwater phase and resulting in a mid-summer bloom in which large unicellular (*Ceratium*), colonial filamentous (*Anabaena*) and globular (*Microcystis*) blue-green algae dominate the water column (Bellinger and Sigee, 2010).

In eutrophic and hypereutrophic lakes, during the summer growth phase, species diversity rises progressively but falls mid-summer where a small number of species outcompetes other algae. The total phytoplankton density (cells/L) of Quamichan Lake in May show some diversity in species (48% green algae, 49% blue-green algae, 2% cryptomonads) and by end of June and July samples were dominated almost entirely by Cyanophytes (99.8% and 99.8%, respectively) (Figure 12). The largest cyanobacteria bloom was observed on August 13, 2018 (99.6%) and dominated mostly by *Spirulina spp*. (Figure 12) (Appendix E).

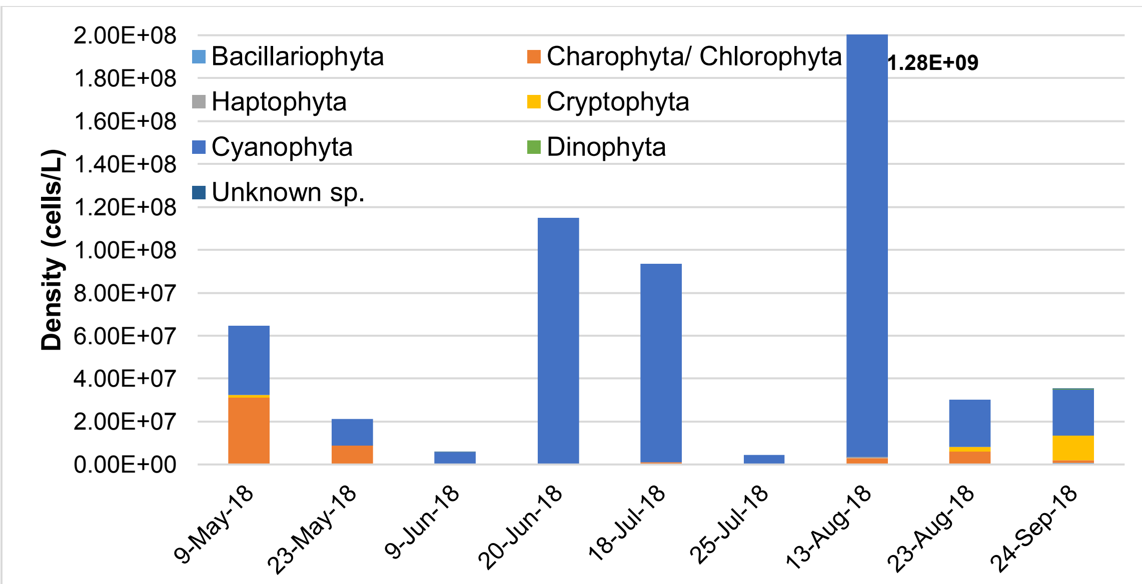

Figure 12. Total phytoplankton density (cells/L) for samples taken from May-September 2018 at Lake Station N.

Phytoplankton community structures were analyzed using bioindices of species diversity calculated from species counts (density). Species richness and dominance were calculated as standard univariate statistical analysis. Species richness (d) was calculated using the Margalef (1958) index and a combination of richness and dominance was calculated using the Shannon-Weiner diversity index (H).

The statistical bioindices were calculated as follows:

Species richness (d) = $(S\text{-}1)/\log_e N$
S= number of taxa

N= number of individuals

$$\text{Shannon-Weiner diversity (H)} = \sum_{i=1}^{S} - (P_i * \ln P_i)$$

P_i = fraction of the entire population made up of species i
S = numbers of species encountered
$\sum$ = sum from species 1 to species S

Results show that phytoplankton species diversity (H) was lowest from June 9-August 13. During these months the unique taxa counts were low as the water column was dominated by blue-green algae (*Microcystis* spp., *Woronichinia* spp., *Pseudanabaena* spp.) (Appendix E) (Table 5). On August 13 the total number of cells/L counted and species richness was high (8.95), but the Shannon-Weiner index was very low (0.21) (Table 5) due to the very high abundance of *Spirulina* spp. (96%) relative to other phytoplankton observed in the sample. September 24 had the highest unique taxa count and a high species richness (10.94) this was because diatoms (*Aulacoseira granulate, Stephanodiscus* spp., *Asterionella formosa*), green algae (*Oocystis* spp., *Pediastrum boryanum*), and dinoflagellates (*Ceratium hirundinella*, *Gymnodinium* spp.) started to emerge in the water column again (Appendix E). This high measure of phytoplankton is also correlated with high biovolume and chlorophyll-a measurements in Figure 15. The most diverse (richness and dominance) phytoplankton communities were observed on August 23 (1.74) and September 24 (1.34) (Table 5).

Table 5. Phytoplankton species richness and diversity throughout the growing season (May-September) 2018 at Lake Station North, Quamichan Lake.

Sampling date	Total unique taxa (S)	Total individuals (N)	Species richness (d)	Shannon-Weiner Diversity (H)
9-May-18	7	64,572,095	6.94	1.3
23-May-18	9	21,250,595	8.94	1.40
9-Jun-18	6	5,857,652	5.94	0.98
20-Jun-18	6	114,961,780	5.95	1.13
18-Jul-18	5	93,553,970	4.95	1.02
25-Jul-18	7	4,380,737	6.93	0.88
13-Aug-18	9	1,287,595,125	8.95	0.21
23-Aug-18	9	30,263,258	8.94	1.74
24-Sep-18	11	35,211,124	10.94	1.34

Density of species is useful for determining the populations of individual algal species and how the species composition changes over a growing season however, species counts do not give comparative information on the relative contribution of different species to overall phytoplankton biomass. Cell biovolume is a measure of the volume of a single alga within a population (Bellinger and Sigee, 2010). It is measured by the volume occupied by a single species populations per unit volume of lake water and was measured in microliters per liter (µL/L) in this study (Biologica, 2018). Biovolume can be used as an approximate index of algal biomass. Green algae dominated 78.8% of the phytoplankton biovolume in May 2018, but transitioning into June and July, blue-green algae had the largest biovolume (78.9% and 75.7%, respectively). The mean biovolume of blue-green algae between June and July was 6.22 x 10^0 µL/L. On September 24, the algal biomass was dominated by cryptomonads (72%), dinoflagellates (13.7%), and diatoms (7%) (Figure 13). This is a result of the system becoming isothermal and the natural algal succession where diatoms and dinoflagellate species become predominant.

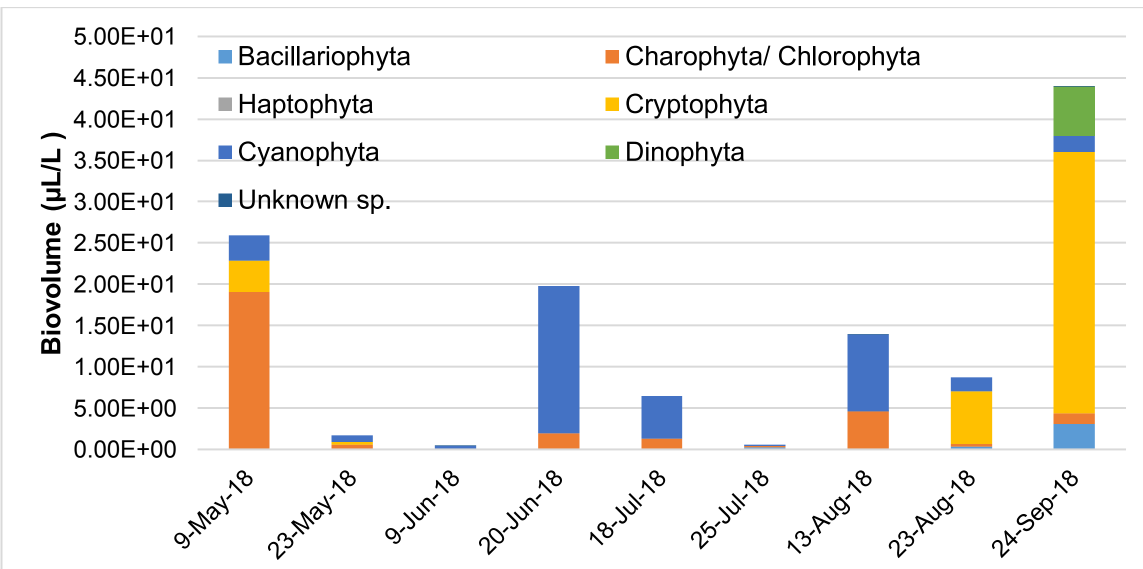

Figure 13. Total phytoplankton biovolume (µL/L) for samples taken from May-September 2018 at Lake Station N.

Cyanophyta (blue-green algae) are widespread and a key indicator of eutrophic and hypereutrophic lake systems. Blue-green algae are a group of algae lacking nucleus and other cell structures and therefore, technically prokaryotic (Brönmark and Hansson, 1998). Many filamentous blue-green algae (i.e. *Anabaena* spp., *Aphanizomenon* spp.) are able to form heterocytes that allow for nitrogen fixation from the atmosphere and when they die and decompose, they release their toxins becoming a human and animal health issue (EPA, n.d.; Brönmark and Hansson, 1998). Therefore, in nitrogen limiting lake systems, such as Quamichan Lake, nitrogen fixing algal species are able to dominate.

Total predominant species taxa within the Cyanophytes measured over the growing season include *Microcystis* spp. which made up 23% of the Cyanophyte biomass (Figure 14). This species can form large globular mucilaginous colonies and are well known for producing microcystins and lipopolysaccharides which affect the liver (hepatotoxins) and potentially carcinogenic to humans (Bellinger and Sigee, 2010; Paerl, Fulton, Moisander, and Dyble, 2001). *Microcystis* blooms can also induce phosphorus release from the sediment and enhance internal loading leading to a positive feedback loop (Xie et al., 2012). *Dolichospermum* spp. *and Woronichinia* spp. each made up 61.3% and 4.3% of Cyanophyte biomass respectively and are both anatoxins which affect the central nervous system (neurotoxins) (EPA, n.d.) (Figure 14). *Aphanizomenon* spp. made up 8.55% of the biomass and has been associated with saxitoxins (Paralytic Shellfish Poisoning toxins), anatoxins, cylindrospermopsin, and lipopolysaccharides (EPA, n.d.; Paerl et al., 2001) (Figure 14). Understanding when cyanotoxins are present and the environmental factors associated with a cyanobacteria bloom formation are critical to effective lake restoration and management and minimizing the human health risks (Xie et al., 2012).

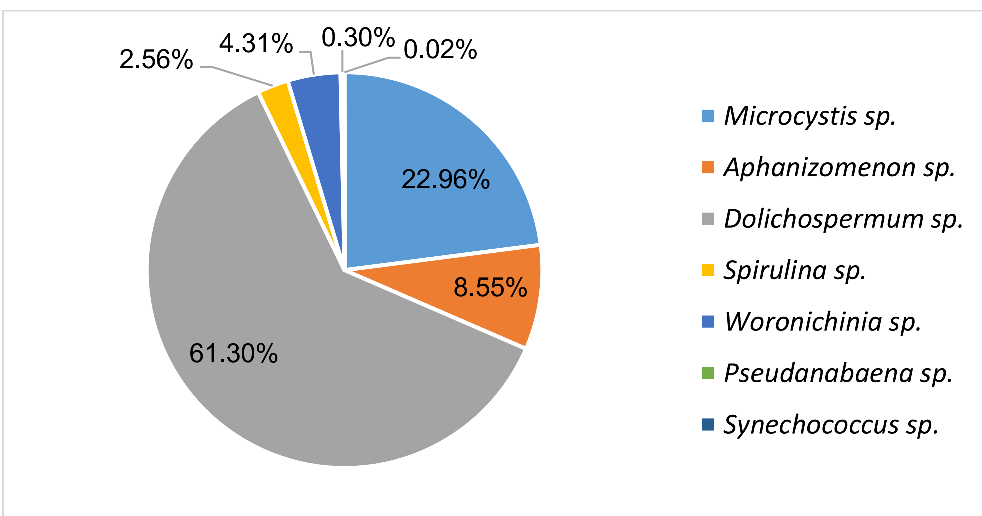

Figure 14. Species predominant taxa percentage of total cyanophyte biovolume (μL/L) over the 2018 growing season at Lake Station N.

2.3.2. *Chlorophyll-a*

Chlorophyll- a is common to all photosynthetic organisms and is relatively easy and rapid to quantify and its concentration is used extensively for estimating phytoplankton biomass (Felip and Catalan, 2000). Chlorophyll- a samples were taken at Lake Station N from May-September 2018. Measurements of chlorophyll-a gradually increased throughout the summer season with maximum levels recorded on August 23 (52.5 µg/L) and September 24 (200 µg/L) (Figure 15). Two measures of biomass (chlorophyll-a and phytoplankton biovolume) were plotted together to show that they generally correlate with each other. Algal biomass measurements indicate that the lake experienced a significant algal bloom at the end of August and throughout September (Figure 15).

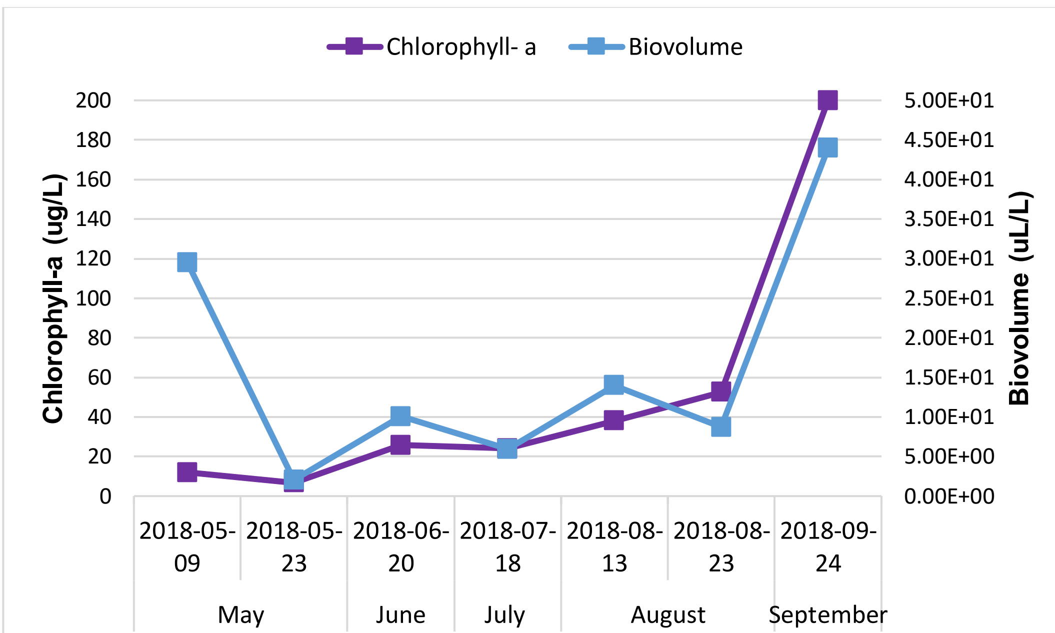

Figure 15. Total phytoplankton biomass in form of chlorophyll-a (µg/L) and phytoplankton biovolume (µL/L) measured from May-September 2018 at Lake Station N.

Chlorophyll-a has been intermittently measured by the province at two stations (EMS ID: E207465, E207466) since 2004. Although sample sizes were variable, and data was limited, results shows that chlorophyll-a concentrations have remained relatively constant over the past 14 years, with the outlier of 200 µg/L (September 2018) (Figure 16). However, due to small sample sizes in previous years, it is likely that algal blooms were missed as a result of the sampling regime (in some years, only one sample was taken). The concentration in September 2018 (200 µg/L) is the highest level on record.

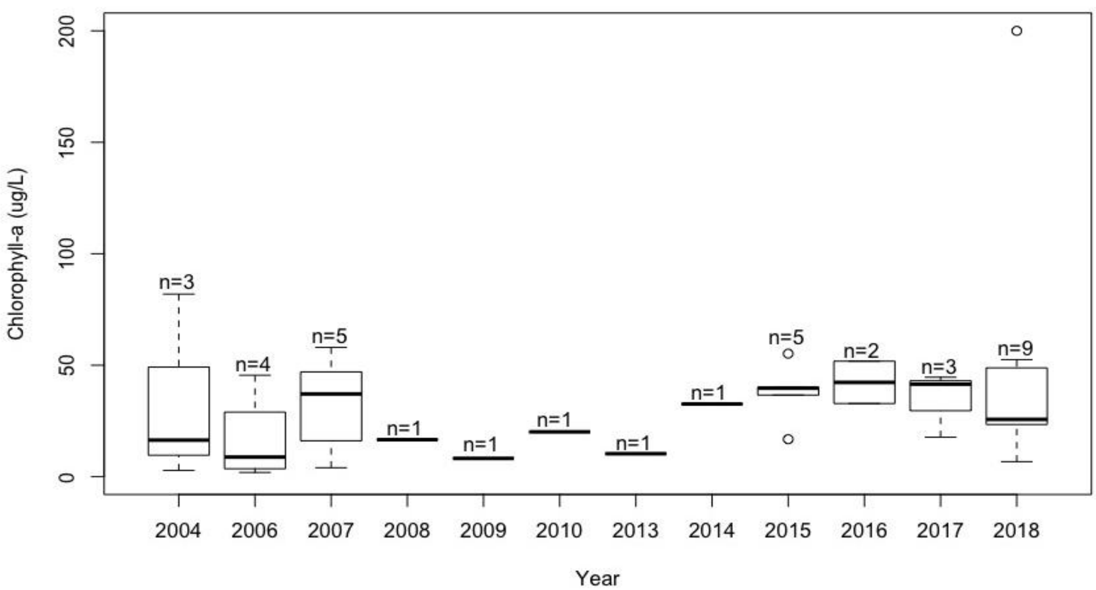

Figure 16. Boxplot of chlorophyll-a (µg/L) measured over time from 2004 to 2018 at two provincial sampling stations (EMS ID: E207465, E207466) and 2018 sampling stations (Lake Station North and Lake Station South). Sample sizes for each year were variable and listed above each box and whisker point. Outliers are presented as circular dots that are outside the inter-quartile range of 1.5.

Based on the mean levels of chlorophyll-a sampled in 2018 (51.3 µg/L) and median (25.7µg/L) Quamichan Lake is considered hypereutrophic in relation to the boundary value criteria of trophic categories by the OECD (1982) (Table 6).

Table 6. Trophic status according to annual mean chlorophyll (Chl_m; µg/L) and annual maximum chlorophyll (Chl_{max}; µg/L) adapted from OECD 1982.

	Ultra-Oligotrophic	Oligotrophic	Mesotrophic	Eutrophic	Hypereutrophic
Chl_m	<1	1-2.5	2.5-8	8-25	>25
Chl_{max}	<2.5	2.5-8	8-25	25-75	>75

2.4. Tributary Nutrient Loading and Critical Areas

In June 2018, three streams were found to have very low flow, Deykin Creek, Woodgrove Creek S (Woodgrove Creek Downstream), and Aitken Creek, and were sampled from July- November 2018 (Figure 1). A one-way ANOVA test was done for the data collected during low-flow season to determine if there were any significant differences in soluble reactive phosphorus (SRP) (bioavailable phosphorus) between the three creeks. The test results show that the mean SRP concentration was greater in Woodgrove Creek S than Deykin Creek and Aitken Creek for the low-flow season (July-November) (F= 8.6, P-*value*= 0.003; Figure 17).

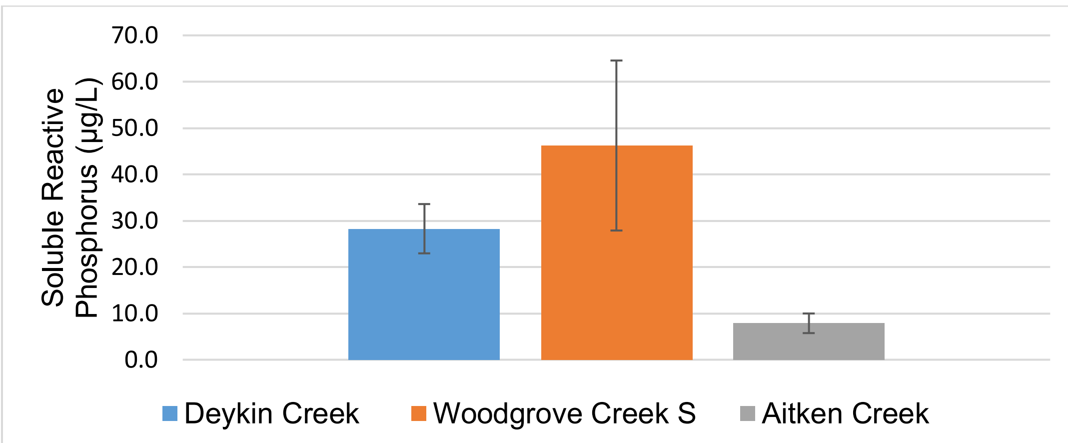

Figure 17. Mean soluble reactive phosphorus (µg/L) concentrations (with 95% CI) for low flow analysis of three creeks from July-November 2018. Sample sizes for Deykin Creek= 7, Woodgrove Creek S= 7 and Aitken Creek=5.

Chemical water quality data of tributaries entering the lake was collected on December 21, 2018, when flow was significant. A total of nine tributaries were identified to be flowing at the time of sampling, locations of sampling is shown in Figure 1. Flow data was not obtained during the stream sampling event, and annual flow data is not available to determine annual flow-weighted mean concentrations. A phosphorus loading study done in 2008 (McPherson, 2008) analyzed surface phosphorus data from 15 tributaries sampled around Quamichan Lake. This study found that creeks draining agricultural and rural areas (McIntyre Creek and Stamps Road) had highest phosphorus concentrations and some high concentrations in residential areas (Woodgrove Creek South). In this study, it is clear that creeks in agricultural zoned areas are contributing significantly more phosphorus than residential areas, most notably Stamps Road and McIntyre Creek, which is consistent with the findings of the study done in 2008 (McPherson, 2008) (Figure 18). In the McIntyre Creek sample total phosphorus was measured at 890 µg/L but SRP measured at 279 µg/L (Figure 18). This could indicate there is a lot of organic matter and sediment contributing to particulate phosphorus, however the SRP concentration is still unusually high. Waters that have high suspended-sediment load will have higher total phosphorus concentrations because of the high proportion of P being in the particulate phase and not the bioavailable SRP form (Jarvie, Withers, and Neal, 2010). It is noted that Aitken Creek, Woodgrove Creek, and Deykin Creek have low flow throughout the summer while all other creeks remain dry and the results of "first flush" sampling for these tributaries could be perceived lower than the actual input nutrient concentration as they are "flushing" nutrients year round.

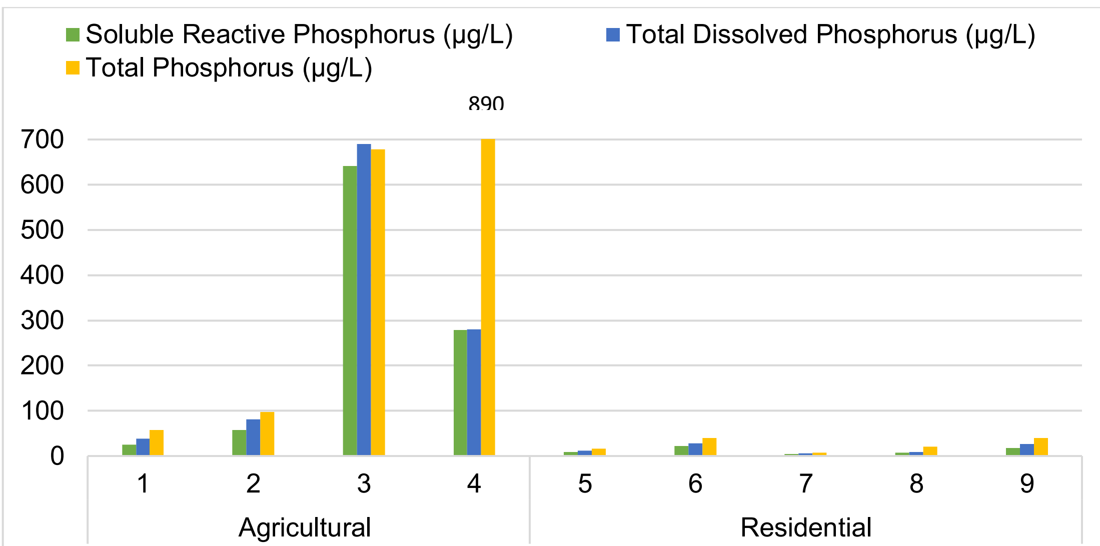

Figure 18. Concentration of all forms of phosphorus (µg/L) for tributaries sampled December 21, 2018. 1= Stanhope Rd, 2= Martin Place, 3= Stamps Rd, 4= McIntyre Creek, 5= Aitken Creek, 6= Upper Woodgrove Creek, 7= Lower Woodgrove Creek, 8= Deykin Creek, 9= Woodmere Creek.

Although there are limitations with the stream data collected, it is clear from the data collected and previous studies that critical loading areas occur in agricultural zoned lands especially McIntyre Creek and Stamps Road and need to be addressed in restoration prescriptions. Based on the low flow stream analysis, Woodgrove Creek is another potential critical loading area within the residential area and should also be addressed.

Chapter 3. Restoration Plan

Shallow lakes are mostly confined to lowland areas and are vulnerable to nutrient enrichment through the conversion of land to agriculture or urban uses with major effects of nutrient flows (Beklioglu et al., 2011). Once a lake's trophic status has been changed, it is very difficult to fully restore a lake back to original state and in many cases restoration attempts have been too limited (Beklioglu et al., 2011). Extensive restoration of whole floodplains rather than isolated lake basins is needed and, in some cases, where full restoration is not possible, rehabilitation is necessary to move the lake to a more acceptable trophic status than the current one (Beklioglu et al., 2011). Based on the physical, chemical, and biological water quality results of Quamichan Lake and its hypereutrophic status, it will require many years of intensive restoration using a full suite of both watershed management and in-lake measures to return it to a eutrophic-mesotrophic state. A diagnosis-feasibility-implementation approach to lake restoration is considered to be the most successful way to restore a lake (Olem and Flock, 1990). Consideration of the whole watershed as an interconnected ecosystem will reduce wasting money on ineffective temporary management measures in the future.

3.1. Restoration Goals and Objectives

Goal 1: Reduce the amount of nutrients entering the lake through catchment basin management techniques

Objective 1.1: Creation of wetlands in critical loading areas in catchment of McIntyre Creek and Stamps Road and natural floodplain areas

Objective 1.2: Implementation of bio-retention basins and rain gardens in any new residential developments and catchment minor tributaries and storm water drainage sites

Objective 1.3: Adjustment of land use management of agricultural land use and riparian areas

Goal 2: Prevent continual internal phosphorus loading through in-lake restoration techniques

Objective 2.1: Dredge upper 2-3 m of surficial sediment using a hydraulic suction dredge

Objective 2.2: Inactivate the sediment surface phosphorus release with chemical dosing of aluminum salts

Objective 2.3: Destratify the water column in the summer using a fine bubble linear aeration system

Objective 2.4: Conduct a feasibility study of the food web to determine biomanipulation method

Goal 3: Ensure restoration success by creating a public outreach and stewardship plan

Objective 3.1: Identify main stakeholders and participatory actions

3.2. Catchment Basin Management Techniques

Quamichan Lake does not have significant point-source pollution since the nutrients entering the lake originate from diffuse runoff from agriculture and livestock operations, residential areas, failing septic systems, and sediment erosion. As a result, management of the surrounding watershed will be a critical step to restoring the lake. To be as cost effective as possible, watershed management practices will be directed towards priority areas.

3.2.1. Constructed Wetlands in Critical Loading Areas

Wetlands around a lake have occasionally been used for wastewater treatment because they can function as a biological filter to remove silt, organic matter, and nutrients from an inflowing stream to the lake to improve water quality (Olem and Flock, 1990). There have been a number of studies that indicate constructed and natural wetlands can increase the retention of nutrients and pollutants using a number of mechanisms including transport and settling of suspended solids, nutrient uptake by marsh vegetation, and denitrification (Janse, Ligtvoet, Van Tol, and Bresser, 2001). Wetland zones can also serve as spawning and nursery areas for predatory fish.

A constructed wetland generally consists of an inlet zone (a constructed sedimentation pond located upstream of the wetland), macrophyte zone (extensive emergent vegetation), and a high flow bypass channel (Mangangka, Liu, Goonetilleke and Egodawatta, 2016). The inlet zone sedimentation pond can stabilize times of high flow, specifically the first flush phenomenon, where the first storm event of the season carries the highest pollutant loads (Manganka et al, 2016). A simulation model of the relationship between pelagic and wetland zones by Janse et al (2001) found that a substantial decrease of summer peaks in chlorophyll-a and increase in submerged macrophyte vegetation occurred at wetland areas of 0.5 times the lake area. In addition, a wetland size of >10% of the lake is suitable as spawning and overwintering habitat for predatory fish (Janse et al., 2001). Constructed wetlands can be efficient in pollutant removal by settling, vegetation uptake, adsorption, filtration and biological decomposition; however, the nutrient uptake can be variable as it depends on rainfall characteristics, pollutant loading rate and hydraulic retention time (Manganka et al, 2016). For example, a study by Li, Liang, Gao, and Li (2016) found that a constructed wetland designed as 350 mm subsurface flow depth had a high variability of dissolved phosphorus reduction (9.66-37.37%). Therefore, the switch from phytoplankton dominated systems to lakes with submerged vegetation is more successful when wetlands are present only if the wetland size is substantial, nutrient loading is moderate, and sufficient mixing is present (Janse et al., 2001).

Furthermore, wetland plants can take up nutrients through their well-developed root system and accumulate significant amount of nutrients in their biomass (Manganka et al, 2016). *Juncus* spp. (common rush) and *Typha latifolia* (common cattail) both have large biomass above and below the surface of the substrate and can accumulate a large amount of nutrients from inflowing water. The uptake capacity of nutrients in emergent macrophytes can be up to 50-150 kg/ha of P and 1,000-25,000 kg/ha of N (Herath and Vithanage, 2015; Brix, 1994). Thus, constructed wetlands and the species of macrophytes planted can be a form of phytoremediation by removing nutrients from inflowing water before reaching the lake. A restoration project of Delavan Lake, Wisconson (Robertson, Goddard, Helsel and MacKinnon, 2000) found that enhancing a small 6 ha wetland to a 38 ha prairie marsh with three sedimentation ponds retained 2,800 kg of P input through the winter months for 2 years following restoration.

Based on the results of the tributary sampling event, Stamps Road and McIntyre Creek are areas where critical nutrient loading is taking place. Therefore, these areas are good candidates for wetland creation in order act as a catchment for excess nutrient and sediment loading, as well as areas that are known to be natural floodplains (Figure 19). All of the waterfront properties on the lake are privately owned and therefore, stewardship agencies should consider land acquisition or alternatively partnering with landowners in restoration efforts through available incentive programs. It is noted that several wetlands were constructed in 2012 in partnership with Nature Conservancy Canada and Quamichan Lake Stewards (estimated area is shown in Figure 19) however it is unknown the status of the effectiveness and if any monitoring has been done on these wetlands.

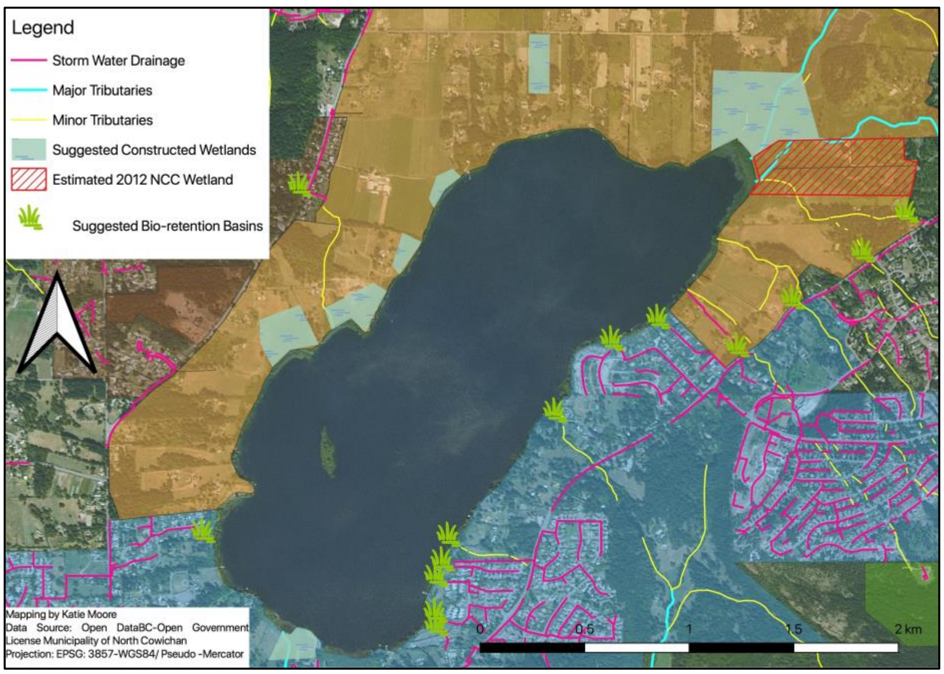

Figure 19. A map of suggested areas for constructed wetlands and bio-retention basins as catchment basin management techniques.

3.2.2. Bio-Retention Basins

Bio-retention basins, bio-swales and rain gardens are increasingly being installed in urban and residential areas to reduce overland flow of storm water into waterways that can carry pollutants including excess nutrients and heavy metals (Mangangka et al., 2016). Bio-retention basins work by intercepting runoff from impervious surfaces through a vegetated layer that removes course to medium sediments, and filtration media that removes finer particulates and pollutants through filtration, infiltration, adsorption, and biological uptake (Mangangka et al., 2016). There are generally three layers within the basin: the filter media (coarse sand or fine gravel), transition layer, and drainage layer (Mangangka et al., 2016) (Figure 20). Bioretention basins are efficient in reducing peak runoff from storm events as well as removing nutrients; studies found that these systems can remove 56% of total nitrogen concentrations (Chen, Peltier, Sturum and Young, 2013) and 50% of total phosphorus concentrations (Mangangka, Liu, Egodawatta and Goonetilleke, 2015).

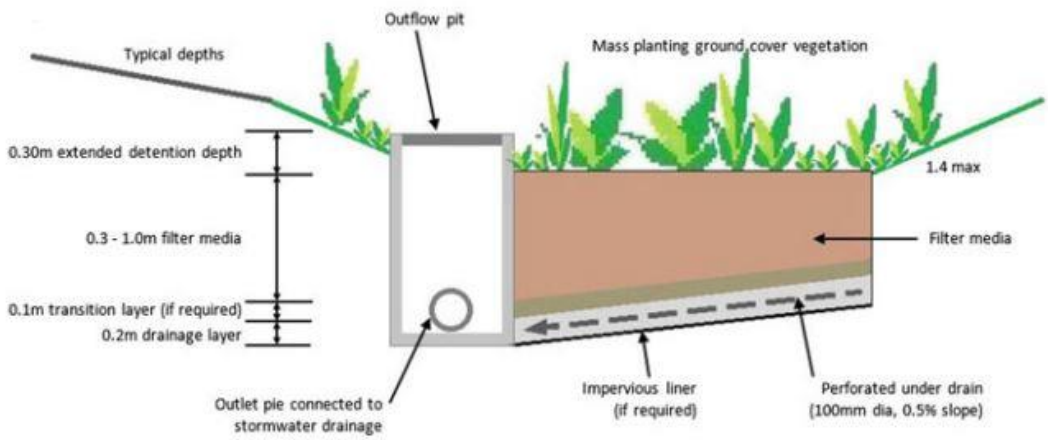

Figure 20. A schematic of a conceptual bio-retention basin (Mangangka et al., 2016).

In 2005 the top of Mt Tzouhalem was cleared for a golf-course development and has remained untouched for over 10 years with little vegetation regrowth and dominated by invasive *Cytisus scoparius* (Scotch broom). This is likely to be a source of sediment and nutrient input into the lake from Woodgrove Creek that flows from the top of Mt Tzouhalem and into Quamichan Lake, and which measured high SRP concentrations throughout summer 2018. In 2017, Municipality of North Cowichan Council approved a new residential development plan entitled “Kingsview at Maple Bay” to be constructed in the cleared area. It is encouraged that the new housing development implement rain-gardens and bio-retention basins throughout the complex. In addition, bio-retention basins should be created further downstream of Woodgrove Creek and also alongside any storm water drainage ditch as bio-swales that flows directly into the Lake (Figure

19). Raingardens can also be implemented by homeowners on their own property, and workshops should be created to inform and educate the public about creating them. Storm water diversion can reduce the phosphorus income to the lake by 50-60% and is a necessary step for management but insufficient for restoration (Olem and Flock, 1990).

3.2.3. Land Use Policy and Management

Within the Quamichan Lake watershed, agricultural and residential land use have had significant impacts on how the land has been altered. It was observed by sampling the tributaries entering the lake, many had disturbed and reduced riparian vegetation. The Riparian Areas Regulation (RAR) (2005) was designed to protect riparian fish habitat of any watercourse that contains water on perennial or seasonal basis, supports fish, or drain into a water body that supports fish (including ponds, lakes, creeks, brooks) (MFLNRO, 2016). Local governments have responsibility for land use decisions which relate to the protection, conservation and enhancement of the environment within their jurisdictions under the RAR. It is recommended for MNC to ensure and enforce a 30 m buffer from top of bank (or 10 m buffer from top of ravine bank) as per the RAR surrounding streams and seasonal storm water drainage sites that drain into the lake. In addition, the shoreline of Quamichan Lake has been encroached by residential settlement and farming operations often extending to the shoreline. Riparian area management of the shoreline and tributaries is recommended to increase connectivity and filtering benefits of riparian vegetation. Re-establishing riparian ecosystems around the lake would involve incorporating riparian vegetation, structures (large woody debris), and fencing to exclude livestock from accessing the riparian buffer. It is recommended that MNC conduct a riparian area assessment of the shoreline and major tributaries to Quamichan Lake including detailed mapping, to further inform future riparian planting and restoration.

The RAR does not apply to farm practices as defined by the *Farm Practices Protection Act*, however RAR can still be enforced in Agricultural Land Reserve (ALR) parcels for lands that are not being used for farm practices (MFLNRO, 2016). In addition, on February 28, 2019 the Code of Practice for Agricultural Environmental Management replaced the Agricultural Waste Control Regulation (AWCR) in efforts of a more science-based approach to nutrient management for all agricultural operations in British Columbia (*Environmental Management Act*, 2004). Within the new regulations, there are nutrient application restrictions in which operators must not apply nutrients to land covered with standing water, on snow or frozen ground; and self-prepared nutrient plans are required for operations with high soil nitrate and phosphorus levels as to not exceed the nutrient needs of crops (*Environmental Management Act*, 2004). MNC is encouraged to promote and educate farmers on these new regulations. For working farmlands, best management practices on agricultural lands should be implemented to control the interactive processes of erosion, nutrient runoff, and pesticide/toxin runoff. Some BMPs include animal waste management, conservation tillage, crop rotation, flood storage (runoff detention), and streamside management zones (buffer strips) (Olem and Flock, 1990).

Data was not obtained for the amount of properties on Quamichan Lake that are currently on septic tank systems versus sewer systems for wastewater treatment. However, previous studies stated that residential areas on the north-east side of the lake are still on septic systems as well as the properties within the agricultural zoned north-west and west areas of the lake (Crawford, 2008). Lakeside lots are inappropriate for septic systems and lake nutrient influxes has conclusively been associated with septic system failures (Olem and Flock, 1990). Unsuitable soils (poorly drained clays/ lacustrine soils), high water tables and steep slopes that occur around lakes contribute to factors that make lakeside lots unsuitable for septic systems as they are ineffective in removing organic matter, bacteria, and nutrients (Olem and Flock, 1990). Since the municipality has a current serviced sewer system to the south and south-east areas around the lake, it is recommended to tie the residential areas north-east of the lake to the public sewer system. For sites that are not able to be tied to the existing sewer system, likely the land parcels in the ALR, alternative on-site wastewater treatment techniques should be considered. Some on-site wastewater treatment methods include aerobic treatment units (ATUs) which use an aeration chamber to allow bacteria to break down contaminants and filtration units. Another option is septic tank-sand filters in which effluent is trickled down materials such as peat moss, sand, or synthetic medium (MAFRA, 2011). These are some examples of ways to retrofit the conventional septic tank without the high costs of upgrading the land parcels to a full sewer system.

3.3. In-lake Restoration Techniques

The community resilience of phytoplankton causes a lag between algal biomass response and decreased phosphorus loads. When phosphorus loading is reduced phytoplankton biomass will not respond as long as the lake is internally loading. In shallow lakes, the lag time is prolonged because phosphorus bound to sediment previously accumulated remains bioavailable. Significant nutrient load reduction and exhaustion of internal phosphorus eventually leads to a linear decrease in algal biomass capacity. An analysis of 22 case studies found a time lag of <10-15 years to achieve TP loading reduction and <5-10 years to achieve N loading reduction (Jeppesen, Søndergaard, Meerhoff, Lauridsen and Jensen, 2007). Internal loading of bioavailable phosphorus and the lag time of biomass after nutrient reduction is why further in-lake restoration techniques must be implemented after external nutrient sources are reduced.

3.3.1. Sediment Dredging

In shallow lakes, there is likely to be a persistent return of phosphorous to the water column for many years after external nutrient sources are reduced, and therefore the physical removal of nutrient rich sediments can produce substantial benefits (Phillips, 2005). Sediment dredging is the partial or complete removal of sediment layers rich in nutrient and organic matter. In shallow lakes, the highly enriched sediment have a negative impact on the oxygen budget, and interstitial soluble phosphorus concentrations are often highest in the uppermost 10 cm of the sediment (Hupfer and Hilt, 2008; Phillips, 2005). Internal phosphorus release is reflective of historical loadings and high susceptibility of the lake to hypolimnetic oxygen depletion and wind mixing. High TP concentrations observed in the Quamichan Lake hypolimnion compared to TP concentrations measured from inflowing streams is indicative that the lake is internally loading phosphorus as a result of 75 years of nutrient input.

The surficial sediments in shallow lakes consists of a semifluid layer of sediment particles, algal cells including benthic species and diatoms but can also include recently sedimented phytoplankton such as cyanobacteria of the *Oscillatoria* spp. group (Phillips, 2005). The ratio of iron: phosphorus in the sediment is a good predictor of the internal phosphorus loading extent (Phillips, 2005). When the sediment surface is aerobic, iron complexes in the sediment counter the release of phosphorus but in a shallow lake, wind stress may disturb the sediment surface sufficiently to transport P into the water column (Reynolds, 1992; Phillips, 2005).

There are several types of dredging equipment that can be used. A lake can be dredged by dry or wet excavation, hydraulic and pneumatic dredging, the latter two being more common (Olem and Flock, 1990). Hydraulic dredges are equipped with a cutter head auger to loosen sediments that is mixed with 80-90% water through a pipeline to a remote disposal area (Hupfer and Hilt, 2008; Olem and Flock, 1990). Pneumatic dredging is a relatively new technique that uses air pressure to pump sediment out of the lake (Hupfer and Hilt, 2008).

A successful sediment dredging project was completed on Burnaby Lake in Burnaby, British Columbia. Burnaby Lake was experiencing similar eutrophic conditions to Quamichan Lake and significant macrophyte growth over the whole lake. Burnaby Lake is 3.11 km^2 and was dredged using a hydraulic suction dredge (up to 2-3 m) in 2010. The lake was split into 16 dredge zones with turbidity barriers separating each zone to allow for fish and organism use and recreational activities to continue (Burnaby Lake Rejuvenation Project, 2012). Overall, 215,000 m^3 of sediment was removed from Burnaby Lake at approximately $70 to $80 per cubic meter, the total cost was approximately $18 to $20 million (Ashley, 2008).

Dredging is a very expensive method to remove phosphorus from a lake however it can be a viable option for bodies of water that have suffered encroachment of sediment accumulation and internal loading (Klapper, 2005). Re-use of excavated materials as natural fertilizer or soil-conditioner for construction sites may help offset the cost of lake restoration. However, the use of the dredged material would need to be used outside the Quamichan Lake watershed to ensure the nutrients are not recycled back into the system. The dredged material would also have to be analyzed for heavy metals (particularly copper and arsenic which are known to be used in herbicides), chlorinated hydrocarbons, and/or iron sulfides to use or dispose of accordingly (Olem and Flock, 1990). It is important to note that dredging a lake can have negative impacts as it can increase the turbidity and lead to the release of nutrients and heavy metals that can effect downstream areas (Hupfer and Hilt, 2008). It is difficult to understand the exact rate of release of P from the sediments unless a sediment core is taken, and therefore, it is recommended that sample sediment cores are taken from the lake to measure the depth of bioavailable P and dredging test areas before applying the treatment to the whole lake.

3.3.2. Phosphorus Inactivation

A subsequent treatment to dredging is increasing the P retention capacity of the sediment by chemical dosing. This can be done using iron, aluminum, or calcium compounds and high doses can remove P from the water but also increase the P-binding capacity in the sediment so internal release of P is decreased (Hupfer and Hilt, 2008). Phoshorus inactivation is one of the most widely used lake restoration techniques and has been used throughout western Europe and the United States (Ashley, 2008).

Aluminum salts are attractive because they are not sensitive to redox changes and P (dissolved inorganic) binds tightly to its salts over a wide range of ecological conditions such as anoxia (Phillips, 2005; Olem and Flock, 1990). Aluminum sulfate ($Al_2(SO_4)_3$), sodium aluminate ($Na_2Al_2O_4$) or aluminum chloride ($AlCl_3$) at dosage between 3 and 30 g/m^3 is added to the water column and aggregates of aluminum hydroxide ($Al(OH)_3$) are formed that precipitate and settle to the sediment, once it is settled on the sediment the layer will reduce the internal release of phosphorus (Olem and Flock, 1990; Hupfer and Hilt, 2008). There have been several whole-lake experiments that have had success with alum application in reducing P concentrations for up to 14 years (Galvez-Cloutier et al, 2012; Cooke et al., 1982; Holz & Hoagland, 1999); however Al can reduce the pH of water that causes $Al(OH)_2$ and dissolved elemental aluminum (Al^{+3}) both of which are toxic to many organisms (Olem and Flock, 1990). The potential for toxicity problems is related to the alkalinity and pH of the lake because both these parameters will decrease during alum application. Therefore, well-buffered, hard-water lakes are good candidates for this treatment because large dose can be given without risk of creating toxic forms of aluminum (Olem and Flock, 1990). The pH of Quamichan Lake in the summer growing season averaged 8.7($\pm$0.9) on the surface and 7.9($\pm$0.6) at 5 m and alkalinity (as $CaCO_3$ mg/L) averaged 52.85($\pm$4.47) on the surface and 55.20($\pm$3.72) at 5 m.

In water with a pH of 6-8 aluminum hydroxide readily forms, however in water with pH above 8 the aluminate ion can become prevalent which has a higher solubility and can lead to the release of P (Olem and Flock; Ashley, 2008). Quamichan Lake has a high alkalinity (>20 mg/L) which makes it less sensitive to declines in pH when alum is added without additional buffers. Lakes that are good candidates for P inactivation using chemical dosing are those that have had external nutrient reduction and proven to have a high internal P load (Olem and Flock, 1990). Iron addition is not a suitable restoration method as it is redox sensitive and requires an aerobic hypolimnion.

A study on applying aluminum sulfate to Eau Galle Lake, Wisconsin (Kennedy, Barko, James, Taylor and Godshalk, 1987) determined the quantity of alum needed as 5 times the average summer internal phosphorus load; 23.5 ha of the lake was successfully treated with 11.3 g/m^2 aluminum sulfate at $553/ha. This study found that after alum was added, internal P rate and abundance of blue-green algae were significantly reduced. However, in contrast a study on Delavan Lake, Wisconsin found a dramatic decrease of P following the addition of alum (from > 600 μg/L to <150 μg/L) but was short lived as P concentration increased to 670 μg/L 2 years after restoration (Robertson et al., 2000). Failure of this restoration prescription was attributed to underestimating the internal P loading rate and subsequently the alum dosage. Dosage of aluminum salts is based on the absolute internal P loading rates from the sediments that can be determined from a sediment core sample (Robertson et al., 2000). A drawback of this treatment is that the initial costs of P inactivation using aluminum salts can be high and often has to be repeated every few years.

3.3.3. Aeration

Aeration can be considered a feasible restoration option only after external nutrient loading has been significantly decreased. Aeration is often applied to stabilize or recover a disturbed oxygen regime and destratification or artificial circulation is used as an aeration method by using compressed air or mechanical mixers to destratify the water column. Typically, destratification is accomplished by the injection of compressed air through perforated pipes along a part of the bottom of the lake (Gibbs and Howard-Williams, 2018) (Figure 20). Rising bubbles act to mix the water column, and oxygen gas transfer occurs at contact with the atmosphere (Gibbs and Howard-Williams, 2018). At low oxygen concentrations, the anaerobic decomposition of organic matter leads to the production of toxic substances (eg. Hydrogen sulfide (H_2S), ammonium (NO_4-N), and nitrite (NO_2-N)) (Ashley and Nordin, 1999). An improved oxygen supply in the hypolimnion accelerates the decomposition of organic matter and provides a higher DO concentration for organisms and conditions for cold-water fish and increases the phosphorus sorption capacity of oxidized iron (Hupfer and Hilt, 2008). Artificial circulation can suppress cyanobacteria in favour of diatoms and green algae by increasing the waters CO_2 and decreasing pH to benefit the recreational uses of the lake (Klapper, 2005; Olem and Flock 1990). Destratification has also been shown to increase macrophyte species abundance, expand fish habitat, and reduce overall internal P loading (Ashley and Nordin, 1999).

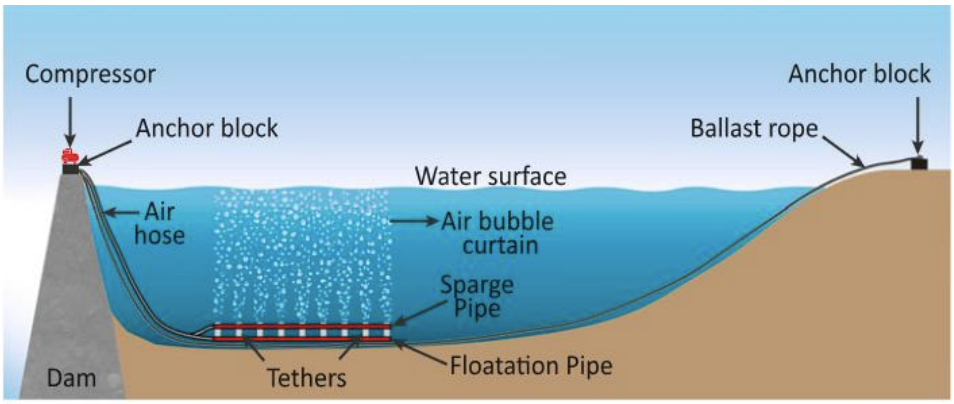

Figure 21. Schematic diagram of a fine bubble linear aeration system using compressed air and diffusers to destratifiy the water column in a reservoir (Gibbs and Howard-Williams, 2018).

An obvious drawback of artificial circulation is the loss of cold-water refuge for fish species such as salmonids that require ~12-14 °C (Environment Canada, 2014). However, it was observed at Quamichan Lake that during months of stratification temperatures below 18 °C were observed only at depths below 5 m which were anoxic conditions and fish would not be able to survive (Figure 3). The rate of oxygen depletion was estimated based on the volume of the hypolimnion calculated by Crawford (2008) as 220,500 m^3 (as no updated bathymetric data was accessed). Based on the dissolved oxygen measurements on May 9 (16.5 mg/L) during spring overturn and June 20 (1.7 mg/L) during summer thermal stratification, the mass of oxygen below 5 m on May 9 was 3638.25 kg and on June 20 was 374.85 kg. The oxygen depletion was approximately 3,263.4 kg over 43 days (76 kg/day). A rule of thumb in limnology is to double the oxygen demand of a lake in order to account for the induced oxygen demand of decomposing bacteria and other mechanisms (Ashley, personal communication, 2018); therefore, at least 152 kg/day of oxygen is required for the lake to prevent anoxic conditions in the hypolimnion.

Fine bubble linear aeration to destratify the water column in the summer would be suitable for Quamichan Lake because the lake becomes thermally stratified with anoxic conditions and internal P loading in the hypolimnion. When the DO in the hypolimnion is below 7 mg/L aeration should be on continuously (Gibbs and Howard-Williams, 2018); in the case of Quamichan Lake this would mean continuous aeration from beginning on May to end of September each year (Appendix B). Quamichan Lake is located in an urban environment and three-phase electricity is accessible within 2 km of the lake which make aeration a feasible restoration option. Costs for destratification are relatively low and are primarily for the compressor and installation of pipes and diffuser (Olem and Flock, 1990). Specific sizing and design of the aeration system and exact oxygen demand rates should be further investigated.

3.3.4. Biomanipulation

Shallow lakes appear to be more sensitive to trophic interactions such as the top down control of phytoplankton by grazing zooplankton (Phillips, 2005). In many phytoplankton rich lakes, planktivorous fish are likely to limit the grazing potential of zooplankton; although, the edibility of the phytoplankton is also a factor in zooplankton grazing as they cannot graze on most Cyanophytes due to their toxicity (Meijer, Boois, Scheffer, Portielje and Hosper, 1999). Biomanipulation is defined as a technique of using biological interactions within lakes to control algal abundance and species composition (Shapiro, Lamarra and Lynch, 1975; Hosper, Meijer, Gulati and Van Donk, 2005). Stocking piscivorus fish or removing planktivorous fish as a form of biomanipulation can have relatively long-term control pf phytoplankton biomass in shallow lakes (Phillips, 2005). This is because organic debris, benthic and epiphytic algae can provide an alternative food supply preventing the collapse of zooplankton population when phytoplankton are exhausting and therefore, maintaining the grazing pressure (Phillips, 2005).

It is possible for biomanipulation to trigger a shift from a stable turbid water state to an alternative stable clear water state, however the reduction of external and internal nutrient load is an important prerequisite for biomanipulation (Hosper et al., 2005). An analysis of 18 case studies on the effectiveness of biomanipulation on shallow lakes was completed in the Netherlands by Meijer et al (1999) and found that improved water clarity was observed after >75% of planktivorous fish reduction. However, the study also found that several lakes experienced decline in water quality after ~4 years and concluded that management would have to continue every 3-4 years to maintain water clarity (Meijer et al., 1999).

Quamichan Lake has been stocked with *Oncorhynchus mykiss* (rainbow trout) and *Oncorhynchus clarkii* (cutthroat trout) with records of stocking dated back to 1990 (Fosker and Philip, 2004). A detailed stocking assessment was done on Quamichan Lake in 2004 by the province of British Columbia funded by the Freshwater Fisheries Society of BC. The results from the stocking assessment fish survey using an overnight gill net found pumpkinseed sunfish dominated 80% of the catch (Fosker and Filip, 2004) (Table 9). Pumpkinseed sunfish were also observed in the fishkill of August 2018 indicating that this species is tolerating the high nutrient levels of the lake, and further contributing to high predation pressure on zooplankton. It is recommended that another stocking assessment be completed on Quamichan Lake to determine the feasibility of reducing the planktivorous pumpkinseed fish species in efforts of biomanipulation purposes to release zooplankton from predation.

Table 7. Summary of catch from Quamichan Lake Stocking Assessment in 2004, table adapted from Fosker and Filip (2004).

Species	Common Name	Sample Size	Percentage of Catch
Oncorhynchus clarkii	Cutthroat Trout	11	1.3%
Ameiurus nebulosus	Brown Catfish	137	15.7%
Cottus spp.	Sculpin	23	2.6%
Lepomis gibbosus	Pumpkinseed sunfish	700	80.4%

This restoration method can be relatively inexpensive and an attractive method for managing eutrophic lakes; however, more research is needed on the current fish community in Quamichan Lake and the reduction of inedible cyanobacteria populations to ensure *Daphnia* zooplankton grazers can promote a clear water phase. Assessment of the current fish community will also guide decisions of the aeration restoration treatment. Biomanipulation can be a feasible management option but must be closely monitored and cannot be used as a standalone restoration option.

3.4 Public Outreach and Stewardship

The restoration of Quamichan Lake and the surrounding watershed is a multi-scale issue and will take many years to accomplish through a number of phases. For restoration to be successful, a full suite of stakeholders' participation is necessary. There are many government agencies, stewardship societies, and groups of people that have a responsibility towards the restoration of the lake and who are affected by the current status of the water quality. Stakeholder engagement can empower participants, promote social learning, and increase the likelihood of accounting for a diversity of values and needs and recognizing the complexity of human-environmental interactions (Reed, 2008). It is argued that participation enables interventions to be better adapted to local socio-cultural and environmental conditions (Reed, 2008).

Stakeholder engagement and public outreach can occur through many forms such as workshops, consultation, education, interpretive signage and more. It is important for lake restoration programs to become collaborative projects that recognize and empower co-governors, co-leaders, co-researchers working towards a holistic goal. Based on the social-ecological system of the Quamichan Lake watershed, a number of stakeholders and key participants have been identified and associated contributions towards restoration are suggested in Figure 22.

Province of British Columbia	• RAR engagement and enforcement (MFLNRO) • Enforce the Code of Practice for Agricultural Environmental Management (MECC) • Conduct updated fish stocking assessment survey
Municipality of North Cowichan	• Conduct riparian assessment and enforce 30 m riparian buffer as per RAR • Commit to long-term water monitoring program • Engage and consult public and Cowichan Tribes prior to restoration
Vancouver Island Health Authority	• Commit to biological water sampling • Public education on water, human health, and recreation • Invest in interpretive signage
Cowichan Tribes	• Inclusion on any planting to ensure culturally significant plants incorporated • Engagement and consultation in all aspects of management and restoration activities
Quamichan Stewards	• Conduct raingarden workshops • Investment in constructed wetlands and bio-retention basins through funding applications
Somenos Marsh Wildlife Society	• Implementing restoration and community outreach through Somenos and Quamichan Lakes Clean Water Action Project
Homeowners	• Citizen science water monitoring (ie Secchi depth) • Investment in home rain gardens • Retrofit septic tanks
Farmers/ Landowners	• Self-prepared nutrient plans • Engagement in incentives for farmlands BMPs • Retrofit septic tanks

Figure 22. Suggested key stakeholders and associated contributions towards stewardship and restoration of the social-ecological Quamichan Lake watershed.

3.5 Monitoring and Management

A long-term water quality monitoring plan will provide valuable temporal data to assess any changes in the lake's chemistry. Monitoring pre and post restoration is an integral part of the restoration process and is one of the most cost-effective lake management activities, but it is rarely done in the long-term (Olem and Flock, 1990). Without long term monitoring data as baseline data, it is practically impossible to assess restoration success of the lake. Success for Quamichan Lake will be defined by phases of successful reduction of external P loading, elimination of internal P loading, and participatory engagement of stakeholders within the system.

3.5.1 *Lake Water Quality Monitoring*

It is recommended that the Municipality of North Cowichan and Island Health invest in long-term water quality monitoring program for Quamichan Lake for all stages before and after restoration to ensure the success of water quality improvement. The province has been monitoring Quamichan Lake sporadically since 1988, however there is currently no regular sampling plan in place for the lake. The monitoring should take place over at least a 5-year period before and after restoration from early spring to late fall each year to capture the algal succession and changes in thermal stratification of the lake. Sampling locations should stay consistent over this time using the coordinates of locations sampled in the 2018 season at Quamichan Creek (48.7888°, -123.6731°) Lake Station South (48.79575°, -123.6661°), and Lake Station North (48.80295°, -123.6585°).

Physical and chemical water quality should be measured from April-November at Lake Station N, Lake Station S, and Quamichan Creek. In the summer when the lake is thermally stratified, a shallow (~ 0.5 m) and a deep (~ 5 m) sample should be taken at Lake Station N and S. Sampling should occur once a month (Table 8) and should include physical parameters (temperature, pH, DO, conductivity) and all forms of phosphorus and nitrogen (Table 9). Secchi depth is a valuable yet easy parameter to measure and should be measured every 2 weeks during the summer and can be measured by lake residents as a way to utilize citizen science.

Table 8. Sampling schedule for chemical and physical water quality samples for a long- term monitoring plan.

	Lake Station N (Surface)	Lake Station N (Deep)	Lake Station S (Surface)	Lake Station S (Deep)	Quamichan Creek
April			X		X
May	X	X	X	X	X
June	X	X	X	X	X
July	X	X	X	X	X
August	X	X	X	X	X
September			X		X
October			X		X
November			X		X
Total Cost[1]	**$624.00**	**$624.00**	**$1248.00**	**$624**	**$1248.00**
Annual Total[1]	**$4368.00**				

[1]Excluding taxes.

Table 9. Costs per chemical parameter as provided by ALS Labs.

Chemical Parameter	Cost per Sample
SRP (orthophoshate)	$10.00
TD (total dissolved phosphorus)	$13.00
TP (total P)	$12.00
DIN (nitrate)	$15.00
DIN (nitrite)	$10.00
DIN (ammonia)	$15.00
TN (total nitrogen)	$30.00
Total Metals (incl Ca, Mg)	$35.00
Alkalinity ($CaCO_3$)	$11.00
Handling/Disposal fee	$5.00
TOTAL[1]	**$156.00**

[1]Excluding taxes.

Biological parameters that should be included in the monitoring plan are chlorophyll-a and fecal coliform concentrations. It is recommended that Island Health start to regularly monitor for these biological parameters during the summer season when recreation is high and bacteria and algal blooms are at the greatest risk. Chlorophyll-a should be sampled once a month over the growing season from May-September using an integrated tube sampler lowered to 3 m for a composite sample of the epilimnion (Table 10). Chlorophyll-a samples can be analyzed by ALS Labs at $55.00 per sample (excluding taxes and handling fee). Currently, there is no regular fecal coliform (*E. coli*) sampling being done on Quamichan Lake. Island Health should be monitoring for this as it is a human health risk especially when Rowing Canada starts to utilise the lake and reactional activity on the lake increases significantly. Fecal coliform monitoring should occur once a month from May- September near Art-Mann Park where there is public access (Table 10). Indicator bacteria *E.coli* samples can be analyzed by ALS Labs at approximately $25.00 per sample (excluding taxes and handling fee).

Table 10. Costs per biological parameter as provided by ALS Labs.

	Chlorophyll-a	Faecal Coliform Bacteria
May	$55.00	$25.00
June	$55.00	$25.00
July	$55.00	$25.00
August	$55.00	$25.00
September	$55.00	$25.00
Parameter Total[1]	**$275.00**	**$125.00**

[1]Excluding taxes and ALS Labs $5.00 handling fee.

3.5.2 Major Tributary and Storm Water Drainage Monitoring

Major tributary and storm water drainage sampling should be monitored over all seasons for discharge measurements and chemical parameters to obtain accurate flow-weighted mean concentrations of loadings from these sources into Quamichan lake. First flush stream sampling occurred in December 2018 but did not include flow measurements and was a one-time sampling event resulting in limitations for data analysis. Most lake responses to watershed loadings is a function of both water quality and quantity; and therefore, it is good practice that whenever water samples are collected in a stream, concurrent flow rate must also be known (Wedepohl, Knauer, Wolbert, Olem, Garrison and Kepford, 1990). A long-term flow and concentration study is important to accurately understand the difference in nutrient loading from each source. From this study, it is clear that McIntyre Creek and Stamps Road are critical concentration areas but the amount of nutrient loading each stream contributes is still unknown. This study also did not analyze any storm water drainage data which could be an important nutrient source if these sites have significant flow.

In December 2018, 8 streams were determined to have significant flow. These streams should be measured several times a year for all forms of P and N as well as heavy metals. Each time the streams are sampled, instantaneous flow must be measured in order to calculate the stream discharge. The velocity-area measurement of discharge is calculated by determining the mean velocity of water passing through the cross-sectional area of the channel (Wedepohl et al., 1990). The streams should be sampled several times when the streams have flow in order to develop a rating curve (minimum of 5 direct stream gagings needed) that can be plotted to determine a stage-discharge relationship (Wedepohl et al., 1990). Once a stage-discharge rating curve is developed, flows can be determined based on the water levels in the stream channel

(Wedepohl et al., 1990). It is noted that based on the climatic zone of Quamichan Lake, many of the streams are seasonal and do not have flow during the summer. In summer of 2018, Deykin Creek, Woodgrove Creek and Aitken Creek were found to have low-flow, of possible groundwater source. These streams should be sampled during the summer into the winter, while all other creeks should be sampled during the rainy season to get a minimum of 5 measurements for each stream over a year (Wedepohl et al., 1990) (Table 11).

Table 11. A stream sampling monitoring plan with sampling frequency and associated costs for tributaries of Quamichan Lake.

	Deykin Creek	Woodgrove Creek	Aitken Creek	Stamps Road	Martin Place	Stanhope Rd	Woodmere Creek	McIntyre Creek
January	X	X	X	X	X	X	X	X
February				X	X	X	X	X
March				X	X	X	X	X
April								
May	X	X	X					
June								
July								
August	X	X	X					
September								
October	X	X	X					
November				X	X	X	X	X
December	X	X	X	X	X	X	X	X
Total Cost[1]	**$780**	**$780**	**$780**	**$780**	**$780**	**$780**	**$780**	**$780**
Annual Total[1]	**$6240**							

[1]Excluding taxes.

3.6 Conclusions and Future Considerations

It is predicted that due to the effects of climate change, eutrophic lakes will reach a more turbid state with high temperatures also leading to high TP concentrations and faster rate of deoxygenation (Jeppesen et al., 2007) . In addition, increasing temperature that promotes thermal stratification will benefit cyanobacteria algal blooms (specifically *Anabaena* spp. and *Microcystis* spp.) and higher drought periods of reduced precipitation will decrease the residence time of the lake to flush out nutrients (O'Neil, Davis, Burford, and Gobler, 2012). Eutrophication is a result of cumulative effects that leads to the impairment and deterioration of the services that lakes provide and will be magnified by climate change (Dokulil and Teubner, 2003). Climate change will make restoration objectives difficult to achieve without undertaking additional efforts to reduce nutrient loading and rigorous in-lake restoration treatments. Watershed management practices will be extremely important to reduce fertilizer use and manure application in agriculture and residential zones, moving toward less intensive farming, and better sewage treatment. Fostering adaptive management from long-term monitoring data will be important when undertaking restoration of a culturally eutrophic lake such as Quamichan Lake. The long-term monitoring plan will play a key role as it will detect any changes in the physio-chemical and biological conditions of the lake.

The results of my research indicate that Quamichan Lake is in a hypereutrophic state with excessive nutrients being both internally and externally loaded and is nitrogen limited causing Cyanophyte phytoplankton to become predominant during the growing season. The current trophic state of the lake is of serious concern for both human health and lake ecology processes. The high recreational value of the lake and the anticipated effects of climate change will amplify the urgency to restore Quamichan Lake back to a mesotrophic state. The water quality of Quamichan Lake has been deteriorating for over 75 years and long-term financial investment, monitoring plan, and comprehensive restoration is required to begin the restoration process.

As part of the monitoring and management process of restoration, a number of further research and studies should be completed on Quamichan Lake. A sediment core study was completed in 2015 by Pellatt et al. (2015), however the study focused on the fire-history of the region's Garry Oak ecosystem by analyzing pollen and charcoal presence. If this core is still available, it should be analyzed for lake biota and pollutants to understand the long term and recent human impacts on the lake, climatic changes, and overall succession of the lake. Sediment cores can also determine the amount of bioavailable phosphorus in the upper sediment layer to examine the feasibility of dredging as well as aluminum salt application. In addition, a fish survey and benthic invertebrate survey should be completed on the entire lake to better understand the current food web dynamics and determine the feasibility of biomanipulation as a restoration treatment. A groundwater survey should also be completed in addition to rigorous stream and storm water drainage sampling to contribute to an accurate phosphorus loading study. Mapping the current vegetation around the lake would give an indication to the extent of the current riparian area buffers to the lake.

Successful restoration of Quamichan Lake and the surrounding watershed will require many decades of restoration, management, and participatory engagement. This case study on Quamichan Lake as a hypereutrophic shallow lake is meant to be used as a guide to further the long-term studies and efforts to restore the lake and others that are experiencing similar problems in southern Vancouver Island. This study confirms the importance of treating the sources of the eutrophication problems and not just the symptoms to successfully restore an ecosystem.

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Appendix A. Bathymetric Map

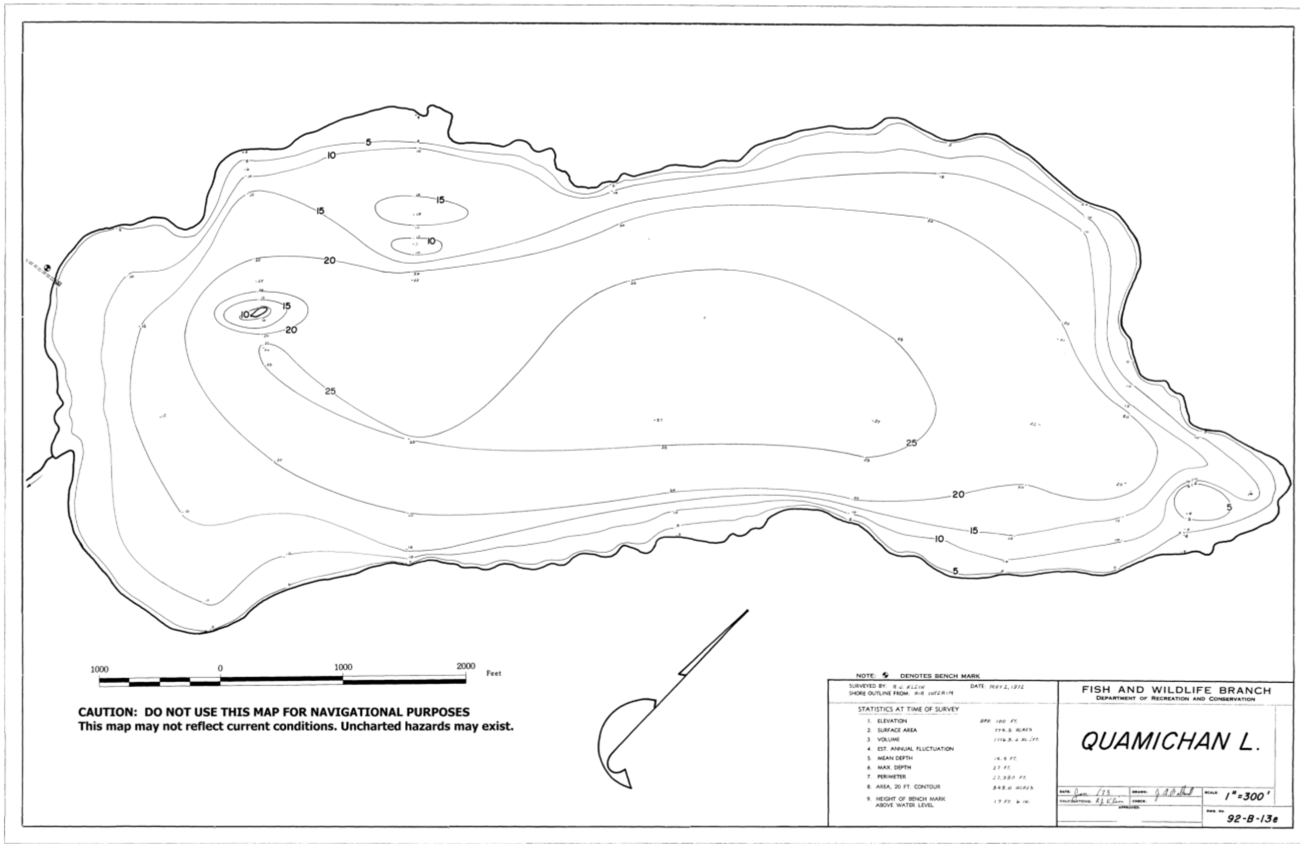

Figure A1. Quamichan Lake Bathymetric Map (Fish and Wildlife Branch, Victoria, BC).

Appendix B: Raw Data for Vertical Lake Profile

		Lake Station South				Lake Station North			
Date	Depth	Temp (°C)	DO (mg/L)	Cond. (µs/cm)	pH	Temp (°C)	DO (mg/L)	Cond. (µs/cm)	pH
09-May-18	0 m	17.9	17.4	187.6	9.92	16.8	17.1	177.4	9.4
	5 m	17.3	16.5	178.2	9.74	13.2	7.2	182.1	8.7
23-May-18	0 m	22.5	8.7	173.9	9.09	23.3	8.6	166.1	9.25
	5 m	14.6	0.04	190.5	8.4	15.3	0.2	179.9	8.23
07-Jun-18	0 m	18.9	7.5	185.9	7.2	18.4	6.9	185.8	7.2
	5 m	18.1	6.7	185.9	7.3	17.9	6.1	186.1	7.2
20-Jun-18	0 m	23.3	14.5	185.3	9.64	23.9	13.9	186.2	9.5
	1 m	22.7	14.3	184.6	9.5	23.3	13.2	185	9.44
	2 m	21	11.5	183.4	9.18	22.5	14.3	180	9.29
	3 m	19.1	8.1	187.1	8.53	19.5	8.9	185.8	8.51
	4 m	17.75	2.8	190.8	7.99	18.6	6.5	187.6	8.07
	5 m	17.4	1.7	191.2	7.8	17.7	3.0	190	8
	6 m	17.1	0.1	193	7.7	17.2	0	194.7	7.8
18-Jul-18	0 m	24.9	10.1	217.5	9.45	25.7	8.9	218.7	9.49
	1 m	24.7	9.9	217.3	9.56	25.4	9.3	218.6	9.57
	2 m	24.6	9.9	217	9.59	25	9.3	217.9	9.59
	3 m	23.7	7.6	215.4	9.22	24.4	7.3	215.7	9.36
	4 m	21.6	2.3	217.6	8.3	20.5	0.5	218.2	8.54
	5 m	20.3	1.3	220.2	7.73	19.3	0.4	223.5	7.95
	6 m	18.9	1.3	228.3	7.31	18.5	0.3	232.3	7.47
	7 m	17.8	1.2	240.3	6.71	17.8	0.3	240.2	7.12
	8 m	-	-	-	-	17.5	0.3	274.3	6.68
25-Jul-18	0 m	25.3	9.2	189.3	9	25.2	9.3	189.1	9.1
	1 m	25.3	9	189	9.1	25.1	9.1	189	9.1
	2 m	25.2	9	189.3	9.1	25	8.9	189.1	9.1
	3 m	22.9	2.1	190.5	8.6	23.8	5.4	189	8.9
	4 m	21.8	0.1	192.6	8.2	21.6	0.1	194.3	8.4
	5 m	19.9	0.1	198	7.8	19.7	0.1	197.4	8
	6 m	18.6	0.1	204.4	7.6	18.6	0.1	204.8	7.7
13-Aug-18	0 m	22.9	7.4	190.4	9.2	23.4	11.2	192	9.3
	1 m	22.8	6.4	190.7	9.1	23	7.8	190.5	9.2
	2 m	22.8	6	190.8	9.1	22.9	7	190.4	9.2
	3 m	22.8	5.9	190.8	9	22.9	6.8	190.5	9.2

	4 m	22.7	5.5	191.4	9	22.4	2	196	8.6
	5 m	21.8	3	200	8.5	21.9	0.3	199.2	8.3
	6 m	19.3	0.2	217.5	7.6	19.9	0.3	213.3	7.8
23-Aug-18 DAY	0 m	22.5	6.9	189	7.9	22.8	5.9	187.9	8.4
	1 m	22.6	7.0	189.4	8.1	22.9	5.5	188.1	8.46
	2 m	22.5	6.8	189.6	8.1	22.9	5.6	188.4	8.46
	3 m	22.5	6.2	190	7.9	22.7	4.7	189.6	8.25
	4 m	22.4	5.9	190.4	7.8	22.5	3.7	190	8
	5 m	21.4	3.3	201.5	7.1	22.4	1.4	192	7.43
	6 m	20.4	3.2	209.3	6.8	20.6	0.9	208.9	6.78
	7 m	18.9	3.2	225.3	6.6	19.3	0.8	220.4	6.59
	8 m	17.8	3.2	379.1	6.4	18.2	0.8	321.7	6.39
23-Aug-18 NIGHT	0 m					22.7	7.8	186.1	8.02
	1 m					22.6	7.2	188.2	8.03
	2 m					22.6	6.6	189.1	8.01
	3 m					22.5	6.1	187.4	7.85
	4 m					22.4	5.3	189.3	7.5
	5 m					22.4	4.9	190.6	7.37
	6 m					22.2	4.5	191.9	7.07
	7 m					18.8	1.5	228.1	6.15
	8 m					18.4	1.4	279.9	6.04
11-Sep-18	0 m	19.4	4.9	196.8	7.7	19.5	4.9	196.9	7.5
	1 m	19.4	4.9	196.9	7.7	19.5	5	196.8	7.5
	2 m	19.4	4.8	196.8	7.6	19.5	4.6	196.9	7.5
	3 m	19.4	4.7	196.8	7.6	19.5	4.5	196.9	7.5
	4 m	19.4	4.5	196.9	7.6	19.5	4.5	196.9	7.4
	5 m	19.4	4.4	197.1	7.5	19.5	4.5	196.9	7.4
	6 m	19.4	4.3	197.1	7.5	19.5	4.4	196.9	7.4
24-Sep-18	0 m	17.5	8.5	211.4	7.35	17.7	7.5	210	7.67
	1 m	17.5	8.4	211.4	7.35	17.7	7.2	210.4	7.6
	2 m	17.5	8.4	211.4	7.37	17.7	7.1	210.5	7.54
	3 m	17.5	8.4	211.4	7.37	17.7	6.8	210.7	7.5
	4 m	17.5	8.2	211.5	7.37	17.7	6.5	210.8	7.46
	5 m	17.4	8.2	211.5	7.37	17.7	6.6	210.9	7.46
	6 m	17.4	8.1	211.4	7.36	17.7	6.7	210.9	7.45
	7 m	17.4	7.9	211.6	7.35	17.6	6.6	211.1	7.44
	8 m	17.4	6.2	237.5	7.13	17.5	1.7	366.1	6.64
09-Nov-18	0 m	11.4	7.3	192.7	7.2	11.4	7.1	193.2	7.3
	1 m	11.4	7.3	192.7	7.2	11.4	6.6	193.3	7.2

2 m	11.4	7.2	192.8	7.2	11.4	6.8	193.3	7.2
3 m	11.4	7.2	192.8	7.2	11.4	6.9	193.3	7.2
4 m	11.4	7.2	192.8	7.2	11.4	6.9	193.3	7.1
5 m	11.4	7.2	192.8	7.2	11.4	6.9	193.3	7.1
6 m	11.4	7.2	192.8	7.2	11.4	6.9	193.6	7.1

Appendix C: Field Photos

Figure C1. Quamichan Lake fish kill observed August 13, 2018.

Figure C2. Quamichan Lake algae bloom observed July 18, 2018.

Figure C3. Chlorophyll-a filtered sample obtained August 23, 2018.

Appendix D: Summary Statistics for Chemical Parameters

Table D1. Summary statistics for chemical parameters at Lake Station North, n= 22.

	Alkalinity (mg/L)	Total Nitrogen (mg/L)	Orthophosphate (mg/L)	Total Dissolved Phosphorus (mg/L)	Total Phosphorus (mg/L)
Minimum	37.8	0.878	0.108	0.15	0.173
Median	54.05	1.35	0.2075	0.2585	0.303
Mean	53.54	2.078	0.2176	0.2534	0.3318
Maximum	58.3	14	0.334	0.39	0.973
Standard Deviation	4.62	2.71	0.062	0.065	0.16

Table D2. Summary statistics for chemical parameters at Lake Station South, n= 20.

	Alkalinity (mg/L)	Total Nitrogen (mg/L)	Orthophosphate (mg/L)	Total Dissolved Phosphorus (mg/L)	Total Phosphorus (mg/L)
Minimum	48.2	0.783	0.13	0.156	0.183
Median	54.35	1.465	0.2235	0.261	0.3115
Mean	54.47	1.517	0.2336	0.2718	0.3214
Maximum	63.8	2.39	0.417	0.472	0.3214
Standard Deviation	4.62	0.43	0.069	0.073	0.071

Table D3. Summary statistics for chemical parameters at Quamichan Creek, n= 12.

	Alkalinity (mg/L)	Total Nitrogen (mg/L)	Orthophosphate (mg/L)	Total Dissolved Phosphorus (mg/L)	Total Phosphorus (mg/L)
Minimum	49.3	1.27	0.0702	0.117	0.23
Median	54.6	1.675	0.1915	0.219	0.314
Mean	54.37	2.786	0.1807	0.2169	0.3535
Maximum	58.6	14.7	0.224	0.292	0.744
Standard Deviation	2.99	3.77	0.044	0.048	0.15

Appendix E: Total Phytoplankton Density (cells/L)

Common Name	Species	9-May-18	23-May-18	9-Jun-18	20-Jun-18	18-Jul-18	25-Jul-18	13-Aug-18	23-Aug-18	24-Sep-18
Diatoms	*Aulacoseira granulata*									334,919
	Aulacoseira sp.								334,919	267,935
	Stephanodiscus sp.						13,397			133,967
	Asterionella formosa									66,984
	Total Diatoms	**0**	**0**	**0**	**0**	**0**	**13,397**	**0**	**334,919**	**803,805**
Green Algae	*Closterium sp.*				143,537					
	Staurastrum sp.				143,537		13,397			
	Elakatothrix gelatinosa			160,761						
	Sphaerocystis sp.	12,479,073								
	Oocystis sp.	803,805	150,713					3,014,269	66,984	803,805
	Ankyra sp.	50,238	8,339,477						5,425,684	
	Pediastrum duplex	3,818,074								
	Pediastrum boryanum									401,902
	Planktosphaeria sp.	13,915,874	150,713			1,116,396			133,967	
	Total Green Algae	**31,067,063**	**8,640,904**	**160,761**	**287,073**	**1,116,396**	**13,397**	**3,014,269**	**5,626,635**	**1,205,707**
Golden Algae	*Chrysochromulina sp.*							502,378		
	Total Golden Algae	**0**	**0**	**0**	**0**	**0**	**0**	**502,378**	**0**	**0**
Cryptomonads	*Cryptomonas sp.*	1,356,421	100,476						2,277,447	11,387,237
	Plagioselmis nanoplanctica		100,476							
	Total Cryptomonads	**1,356,421**	**200,951**	**0**	**0**	**0**	**0**	**0**	**2,277,447**	**11,387,237**
Blue-green Algae	*Microcystis sp.*		6,983,056	3,780,898	38,467,810	35,724,666	3,000,872	11,052,319	10,382,481	17,348,791
	Aphanizomenon sp.	32,148,611	3,617,122							
	Dolichospermum sp.			331,570	56,266,350	3,572,467		28,133,175	3,483,155	2,344,431
	Spirulina sp.						107,174	1,237,859,691	3,000,872	
	Woronichinia sp.		301,427	20,095	4,449,635	49,121,416	107,174	502,378	0	0
	Pseudanabaena sp.		1,507,134	1,483,948	15,490,912	4,019,025	53,587	3,014,269	5,157,749	1,920,201
	Synechococcus sp.						1,085,137	2,511,891		
	Total Blue-green Algae	**32,148,611**	**12,408,740**	**5,616,510**	**114,674,707**	**92,437,574**	**4,353,944**	**1,283,073,722**	**22,024,257**	**21,613,423**
Dinoflagellates	*Ceratium hirundinella*									66,984
	Gymnodinium sp.									66,984
	Total Dinoflagellates	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**133,967**
Other Algae	*Flagellated algal cell (< 10µm in length)*			60,285				1,004,756		66,984
	Non-flagellated algal cell (< 10µm in length)			20,095						
	Total Other Algae	**0**	**0**	**80,380**	**0**	**0**	**0**	**1,004,756**	**0**	**66,984**
	Total Cells	**64,572,095**	**21,250,595**	**5,857,652**	**114,961,780**	**93,553,970**	**4,380,737**	**1,287,595,125**	**30,263,258**	**35,211,124**