Synthesis of Water Quality Data and Modeling Non-Point Loading in Four Coastal B.C. Watersheds: Implications for Lake and Watershed Health and Management

by

Lisa Rodgers
B.Sc., Humboldt State University, 2009

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Supervisory Committee

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Abstract

I compared and contrasted nitrogen and phosphorus concentrations and land use differences in two oligotrophic lakes (Sooke and Shawnigan) and two meso-eutrophic lakes (St. Mary and Elk) in order to evaluate nutrient concentrations over time, and evaluate the relationship between in-lake nutrients and land use in the surrounding watershed. I used MapShed© nutrient transport modeling software to estimate the mass load of phosphorus and nitrogen to each lake, and evaluated the feasibility of land use modifications for reducing in-lake nutrients. In comparing nitrogen and phosphorus data in Sooke and Shawnigan Lakes, I determined that natural watershed characteristics (i.e., precipitation, topography, and soils) did not account for the elevated nutrient concentrations in Shawnigan verses Sooke Lake. Natural watershed characteristics indicated that external loads into Shawnigan Lake would be lesser-than or equal to those into Sooke Lake if both watersheds were completely forested. I evaluated trends of in-lake nutrient concentrations for Sooke and Shawnigan Lakes, as well as two eutrophic lakes, St. Mary and Elk. Ten to 30-year trends indicate that nitrogen and phosphorus levels in these lakes have not changed significantly over time. Time-segmented data showed that nutrient trends are mostly in decline or are maintaining a steady-state. Most nutrient concentration data are not precipitation-dependent, and this, coupled with significant correlations to water temperature and dissolved oxygen, indicate that in-lake processes are the primary influence on lake nutrient concentrations -- not external loading. External loading was estimated using, MapShed©, a GIS-based watershed loading software program. Model validation results indicate that MapShed© could be used to determine the effect of external loading on lake water quality if accurate outflow volumes are available. Based on various land-cover scenarios, some reduction in external loading may be achieved through land-based restoration (e.g., reforestation), but the feasibility of restoration activities are limited by private property. Given that most of the causal loads were determined to be due to in-lake processes, land-based restoration may not be the most effective solution for reducing in-lake nitrogen and phosphorus concentrations.

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Chapter 1: General Introduction

Problems with surface water quality include high turbidity, overabundance of nutrients, and fecal contamination. Poor quality sources for drinking water result in the need for extensive water treatment measures that may not be economically viable in rural communities. Small and rural communities are at greater risk for source water contamination because the construction of municipal storm water and sewer systems is also not economically feasible in rural communities due to a low tax base and scattered homesteads (Grey 2008). Water system operators must deliver safe drinking water to customers despite these challenges.

The multi-barrier approach to safe drinking water includes source water protection, drinking water treatment, and drinking water distribution. Source water protection is the first step in the multi-barrier approach. The Canadian Water Network determined seven categories of activities that are integral to protecting source waters across Canada (Simms et al. 2010). These include: surface and groundwater protection, drinking water and wastewater management, wetland and aquatic ecosystem protection, point source pollution management, land use planning, management of land use impacts, and land stewardship.

In southern Vancouver Island and the Gulf Islands (British Columbia, Canada) multiple government agencies are responsible for various aspects of source water protection. These include: Capital Regional District (CRD), Cowichan Valley Regional District (CVRD), the Islands Trust, Vancouver Island Health Authority (VIHA), and the British Columbia Ministry of Environment (BC MOE). Source water management is trending toward the formation of joint committees in which the various governance agencies are coordinating research and decision-making efforts that also include input from members of the local communities. The formation of the Salt Spring Island Watershed Protection Authority, which includes members of government, community scientists, and the public is one such example. In other cases, local government water managers cooperate closely with organizations such as the Shawnigan Lake Stewardship Society and the Victoria Golden Rod and Reels Society for Elk Lake, to coordinate water sampling and gain input on water management decisions.

The purpose of the research outlined herein is to address the protection and improvement of water quality in four freshwater lakes: Sooke (SOL), Shawnigan (SHL), St. Mary (SML), and Elk/Beaver (ELK). The focus of this research is on nitrogen and phosphorus inputs from non-point sources including: storm water runoff, residential areas, forestry, and agriculture. The research addresses the following questions:

- How much different are nitrogen and phosphorus concentrations in SHL compared to SOL, and can natural watershed characteristics account for these differences?
- Are the nitrogen and phosphorus concentrations in SOL, SHL, SML, or ELK increasing over time?
- Is MapShed modeling software a useful tool for evaluating the effect of external nutrient loads on in-lake water quality?
- Will watershed reforestation result in reductions in in-lake nitrogen and phosphorus concentrations?

In order to address questions around the protection and improvement of water quality in the four lakes (SOL, SHL, SML, and ELK), I quantified and compared trends of in-lake phosphorus and nitrogen (when available) concentrations in the four lakes, and used geographic information systems (GIS) data to conduct qualitative analyses and to model nitrogen and phosphorus transport into the lakes. Based on model outputs, I evaluated the feasibility of watershed reforestation for improving in-lake nutrient concentrations.

Watershed Land Use and Water Quality

The effect of watershed land use on water quality in lakes and streams has been well documented (Reckhow and Simpson 1980; Dillon and Kirchner 1975). Research has primarily focused on agricultural (Beaulac and Reckhow 1982; Carpenter et al. 1998; Parn et al. 2012), commercial forestry (Zhu 2005), and urban areas (Arnold and Gibbons 1996), with some localized study on the effects of forest age on non-point nutrient transport (Zhu 2008). Agricultural areas have been shown to be the source of excessive nitrogen transport, whereas urban areas tend to produce excessive phosphorus (Glandon et al. 1981; Wickham 2002). Impervious surfaces, such as paved roadways, sidewalks, and rooftops contribute significantly to the transport of nutrients and pollutants in watersheds because without the ability to percolate into the soil, substances flow directly over the hard surfaces, and pollutant concentrations are carried conservatively into receiving waters (Arnold and Gibbons 1996). The U.S. EPA estimates that surface runoff increases from an average 10% in natural areas to 55% in urban areas (Arnold and Gibbons 1996). This means that as the area of impervious surface increases, groundwater and aquifer recharge decreases, and more pollutants are deposited into surface waters.

In vegetated areas, plant roots stabilize the soil and retain nutrients so that nutrients and organic matter are not readily transported over the land surface or leached into ground water. Riparian vegetation buffers serve to reduce soil and nutrient inputs into lakes and streams because root systems slow the movement of water and nutrients within the soil (Phillips 1989). Delayed flow results in longer detention times for water and soluble nutrients which are then deposited, adsorbed, or assimilated prior to entering waterways. The risk of phosphorus export increases rapidly as forest cover declines below 90% (Wickham 2002). Vegetation clearing disrupts the mycorrhizal root associations which are important for nutrient absorption (Bunemann et al. 2011), and especially the retention of phosphorus (Bever et al. 2001).

Soil texture, the size distribution of soil particles, largely determines the erodibility, drainage, and permeability of soils (Mahmood-Ul-Hassan 2011). The erodibility factor indicates the relative sediment transport potential of each soil texture (Wall et al. 2002). Soil drainage indicates the ability of soil to hold water in the plant root zone for some period of time. Rapidly drained soils do not retain water except for immediately following precipitation events, whereas poorly drained soils retain moisture between precipitation events for most of the year. Well-drained soils retain some moisture, but not for a significant period. Soil drainage is expressed on a continuum of rapidly – well – mod. well – imperfectly – poorly – very poorly drained to describe moisture retention times from nearly none (rapidly) to surface saturation for most of the year (very poor) (Jungen et al. 1985). Soil permeability is based on measurements of the downward movement of water through a saturated soil. Permeability is a measured description of perviousness, which is the movement of water movement through cracks and pours in the soil matrix (Brady and Weil 2002). Like soil drainage, perviousness and permeability describe the water-retention capacity of a soil, and have mostly been related to the potential for plant (crop) growth in a given area. In the context of land-based contributions to in-lake nutrient loads, rapidly or well-drained soils with moderate to rapid permeability allow for rain water to enter the soil and percolate downward. This means there is less horizontal runoff to carry organic matter and surficial silts across the land and into lakes. In urban areas well-drained and permeable soils located between paved roadways and water surfaces reduce the quantity of pollutants entering natural waters. Alternatively, when considering the placement of septic systems within a watershed, a more moderate drainage and permeability regime allows nutrient particles to adhere to soil particles during percolation, thus reducing the discharge of nutrients into waters (van Vliet et al. 1987).

Water Quality Guidelines and Objectives

The BC MOE has established guidelines for water quality parameters (BC Ministry of Environment 1998). BC guidelines for temperature, pH, dissolved oxygen (DO), turbidity, total dissolved solids (TDS), total organic carbon (TOC), nitrogen (N), total phosphorus (P), coliform bacteria, and chlorophyll-a provide a basis for evaluating the condition of surface waters. BC MOE develops water quality objectives for waterbodies that may be affected by human activity now or in the future. Objectives are established on a site-specific basis to ensure the safety of the most sensitive water uses including: drinking water, public water supply, aquatic life and wildlife, agriculture, recreation, aesthetics, and industrial water supplies.

Nitrate and Total Nitrogen – Forms and WQ guidelines

Inorganic forms of nitrogen include nitrate (NO_3), nitrite (NO_2), ammonia (NH_3), and nitrogen gas (N_2). Blue-green algae, soil bacteria, and mycorrhizae (in plant roots) convert N_2 into NH_3 and NO_3 for use by aquatic organisms and terrestrial plants. Nitrate is water soluble, and is thus easily transported in streams and groundwater. Nitrate can be harmful to humans and fish in excessive amounts because once consumed, it is transformed to NO_2 which reacts with hemoglobin, and limits the ability of red blood cells to carry oxygen. This is known as "blue baby" syndrome in humans and "brown blood disease" in fish. Ammonia is unstable in water, and thus easily transformed to NO_3 in oxygenated waters, and NO_2 in oxygen deficient waters. Common sources of excessive nitrogen loading include fertilizers (lawns/agriculture), human sewage (septic systems), and animal waste (Murphy 2007). The BC Ministry of Environment has established guidelines for a number of water quality parameters, including nitrate. The recommended nitrate limits are 10 mg/L for drinking water and recreation, and an average of 40 mg/L for aquatic life (BC Ministry of Environment 1998).

Phosphate and Total Phosphorus – Forms and WQ guidelines

Phosphorus originates in rocks, and is found in soils and plants. Phosphorus is less soluble in water than nitrogen, and adheres readily with soil particles, thus natural concentrations of phosphorus in freshwater are relatively low. Due to the average pH of freshwater (pH 6-7), phosphorus is typically in phosphate (PO_4) form in waters. Phosphate is present in waters as organic phosphate, which includes microorganisms, human and animal waste, and organic pesticides and fertilizers, and as orthophosphate (inorganic), the form utilized by plants, but also found in detergents and some industrial discharge. Phosphates can cause digestive problems in very high amounts, but the primary concern over phosphate loading is the contribution to the nitrogen to phosphorus ratio as explained in the next section (Murphy 2007).

The BC Ministry of Environment recommends that TP not exceed 10 ug/L at spring overturn for drinking water and recreation, and that it remain between 5-15 ug/L for aquatic life (BC Ministry of Environment 1998).

Lake Eutrophication

Source waters are characterized as eutrophic, mesotrophic, and oligotrophic based upon the quantity of nutrients present. Highly productive lakes, i.e., those with high nutrient content, are eutrophic. Mesotrophic lakes have a moderate quantity of nutrients, and oligotrophic are relatively low in nutrients (Smith et al. 1999). In eutrophic waters, excess nutrient loading often results in algae blooms and an overall abundance of in-lake biomass. Nutrient enrichment further results in an imbalance among in-lake species (e.g., fish, phytoplankton, zooplankton), and long-term water quality management is compromised. Liebig's Law of the Minimum explains that the yield of plants (also algae and plankton) is limited by the vital nutrient of the lowest quantity available to the plant (von Liebig 1855). Thus, the productivity of organisms in source waters increases or decreases depending on the availability of nutrients (nitrogen (N), phosphorus (P)), sunlight, and dissolved oxygen utilized by organisms for growth.

Algae Blooms

Cyanobacteria (blue-green algae) are photosynthetic bacteria that form large "blooms" in lakes enriched with nutrients. Nitrogen enrichment is thought to initiate new blooms which compromise water quality by reducing water clarity. The decrease in clarity (secchi depth) interferes with the productivity of aquatic macrophytes, invertebrates, and fish. Once established, recurring algae blooms are linked to phosphorus enrichment, which is widely thought to be the nutrient that increases or limits algal biomass accumulation in freshwater ecosystems. Phosphorus enrichment especially favors nitrogen-fixing species that can supply their own nitrogen (Paerl and Otten 2013). Many cyanobacteria produce secondary metabolites that are harmful to humans, animals, fish and birds. Cyanobacteria produce neurotoxins and hepatotoxins which are deadly upon consumption over very short or recurring timeframes (Gray 2008) (Paerl and Otten 2013).

Frequent or persistent blooms of algae pose challenges to source water treatment, and harm the esthetic and recreational value of lakes. Algae blooms are a water quality nuisance that can be observed by the public, and are a typical driver for water quality research and remediation (Smith et al. 1999). It is widely understood that reducing inputs of N and P to freshwater effectively reduces algal biomass. The ratio of TN to TP determines whether increases in nitrogen or phosphorus are responsible for increased algae growth as the nutrient concentrations in the water approach the ratio of optimal growth for algae which generally ranges between 15:1 and 20:1 TN:TP (Knowles 1982). A low nitrogen to phosphorus ratio (~10:1) is the most significant factor driving the growth rate of nitrogen-fixing algae (Downing 1992) because in the absence of sufficient nitrogen, organisms that can “fix” atmospheric nitrogen into bioavailable nitrogen will dominate. Thus, P is often considered the limiting factor in the prevalence of extensive algal populations in eutrophic lakes. Algae growth is also enhanced by warm water temperatures, high organic matter concentrations, and high iron (Fe) content (Paerl and Otten 2013).

The presence of algae in source waters poses numerous source-to-tap challenges for the delivery of clean drinking water. Algae clog microstrainers, and rapid and slow sand filters. The production of carbon dioxide during respiration alters the pH of water, resulting in the potential for the passage of coagulant (such as aluminum sulfate) into drinking water. If allowed into the finished water, algae cause coloration, poor taste, and odor (Gray 2008).

Non-Point Loading Sources

Anthropogenic supplies (loads) of nutrients to source water include point sources which are specific areas or sources of loading, and non-point sources which are dispersed and difficult to monitor (Smith et al. 1999). Point sources include specific sites, such as industrial and municipal sewers, which have distinct outlets for loading. External loading of nutrients into source water occurs naturally by erosion and translocation of decomposed organic matter and soil. Anthropogenic activities such as agricultural and forestry operations contribute to increased erosion because they result in reduced plant root biomass. Thus, nutrients and organic matter are more readily translocated.

Storm Water Runoff

Storm water runoff is the driver of terrestrial erosion which transports material into source waters. In vegetated areas, plant roots stabilize the soil so that nutrients and organic matter are not readily transported over the land surface or leached into ground water. It is widely known that riparian vegetation buffers serve to reduce soil and nutrient inputs into lakes and streams because root systems slow the movement of water and nutrients within the soil. Delayed flow results in longer detention times for water and soluble nutrients which are then deposited, adsorbed, or assimilated prior to entering waterways. Mycorrhizal root associations increase the area from which plants can access soluble nutrients (phosphorus and nitrogen). Non-mycorrhizal roots typically absorb nutrients within 5 mm, whereas mycorrhizal roots can absorb nutrients several centimetres away (Bunemann et al. 2011). Arbuscular mycorrhizal fungi are the most common root symbionts, and are important because they facilitate the uptake of phosphorus (Bever et al. 2001).

The amount of storm water that enters surface waters is dependent on the soil type and frequency of precipitation. Naturally soils with large soil pore space, such as gravel and sand, will retain less water than silt and loam soils that have smaller pores. Gravelly and sandy soils also contain less organic matter and available ions, and do not attenuate nutrients as effectively. The texture and water regime of soils determines the type of vegetation that will grow in different soils. The ability of vegetation to obstruct, adsorb, or assimilate nutrients is primarily based the local slope gradient (Phillips 1989). Steeper slopes are associated with shallower soil depth (Sidle et al. 2000) and higher flow velocities which results in faster runoff of nutrients (Castillo 2009). Faster runoff velocities allow less time for nutrients to percolate into soil and be assimilated by plants.

MapShed© Modeling

MapShed© (PennState 2013) is a user-friendly open-source (free) software program used to predict non-point nutrient transport to streams and lakes from the surrounding land area. Many of the required input datasets are available as free downloads or are readily obtained from local governments. I used MapShed© to evaluate nitrogen and phosphorus inputs into the subject lakes because it runs on a GIS platform (either ArcMap or MapWindow) for which spatial data are easily acquired. The program is based on the established Generalized Watershed Loading Function (GWLF), the mathematical formulae are imbedded in the program and advanced mathematical modeling is not necessary. Although operational upgrade have been made to MapShed since it began as GWLF in 1992, all imbedded mathematical formulae can be found in the GWLF Manual (Haith et al. 1992). MapShed© is accessible to water managers with basic GIS skills and is available as on online download at no cost. Spatial data can be modified using MapWindow, which is an open-source GIS software program that operates very similar to the more expensive ArcMap products.

Summary of Research Objectives

The following research and research outcomes are guided by the three main Objectives:

Objective 1: Synthesize water quality and land use data for study watersheds
Long-term water quality data is available for each of the study watersheds. Data were synthesized and analyzed for each watershed and compared to the near-pristine Sooke Lake watershed. Data for each watershed were evaluated for statistical trends of in-lake water quality data, and correlated to land use.

Objective 2: Model transport of land-based non-point loading into source waters
The MapShed© model was used to integrate data on terrestrial biomass retention potential, soil retention and export potential, landscape slope, local precipitation, and land use in order to evaluate nutrient transport into source waters. Restoration scenarios that include the alteration of biomass, were subsequently modeled to evaluate the potential for reduced loading that may occur if the restoration is implemented.

Chapters 2, 3, and 4 address specific components of the research goals and objectives as follows:

Chapter 2 compares nitrogen and phosphorus in Sooke (SOL) and Shawnigan (SHL) Lakes and evaluates whether natural characteristics in the watersheds explain the elevated nutrient levels in SHL verses SOL.

Chapter 3 evaluates 30-year and time-segmented nitrogen and phosphorus in the four watersheds, SOL, SHL, St. Mary Lake (SML), and Elk Lake (ELK). The implications of the statistical analyses are discussed.

Chapter 4 provides evidence that the MapShed© model can be used to estimate the external loading trends, and how they would affect in-lake water quality. Watershed reforestation options are presented.

Chapter 2: A Comparison of In-Lake Nitrogen and Phosphorus in Sooke and Shawnigan Lakes, B.C. 2004-2013 and Potential Sources of Excess Nutrient Loading

Abstract

Ten years of water quality records for nitrate, total nitrogen (TN), phosphate, and total phosphorus (TP) were compiled and evaluated to determine how nutrient levels from water samples compared between Sooke and Shawnigan Lakes. The Sooke Lake watershed is protected, and land cover is 52.4% old forest, 31.7% recently logged forest, and 15.9% young forest. Shawnigan Lake is surrounded by low-density residential properties (13.7% of the watershed). The remainder of the watershed is 72.3% young forest, 10.1% recently logged forest, 2.1% old forest, 1.9% agriculture, and 0.5% barren land. The analyses showed that, with the exception of phosphate (2004-2009), nutrient levels were significantly higher in Shawnigan Lake. Annual mean nitrate was 2.5 times higher, TN was 1.9 times higher, and TP was 1.5 times higher in Shawnigan Lake compared to Sooke Lake. Nutrient levels in both lakes were well below BC Ministry of Environment guidelines. Precipitation trends, soils, slope, drainage, and land cover were compared for each watershed to determine if these factors account for the elevated nutrient levels in Shawnigan Lake compared to Sooke Lake. Annual precipitation (rain) was 10% higher, and there are more inflow drainages, and steeper slopes around Sooke Lake. Soils around Sooke Lake are also more likely to allow the passage of dissolved nitrogen, and exhibit less phosphorus sorption than Shawnigan Lake soils. Under natural conditions, these factors should result in greater nutrient transport into Sooke Lake. Based on a model of nitrogen export from different forest cover types in the Sooke watershed, estimated nitrogen export from the young forests in the Shawnigan watershed may contribute to elevated nitrogen in Shawnigan Lake. Anthropogenic factors, such as forest harvesting, residential clearing, the use of fertilizer on lawns, and faulty septic

systems are the likely causes of elevated nutrient levels in Shawnigan Lake compared to Sooke Lake.

Introduction

The effect of watershed land use on water quality in lakes and streams has been well documented (Reckhow and Simpson 1980; Dillon and Kirchner 1975). Research has primarily focused on agricultural (Beaulac and Reckhow 1982; Carpenter et al. 1998; Parn et al. 2012), commercial forestry (Zhu 2005), and urban areas (Arnold and Gibbons 1996), with some localized study on the effects of forest age on non-point nutrient transport (Zhu 2008). Interactions between vegetation type, soil type, and topography influence the extent of nutrient transport from watersheds into receiving waters (Sidle et al. 2000). Changes in land use further affect the extent of nutrient loading (Wickham and Wade 2006). Water quality monitoring conducted by the Victoria Capital Regional District (CRD) for Sooke Lake (SOL) and the Cowichan Valley Regional District (CVRD) for Shawnigan Lake (SHL) has resulted in more than 10 years of nearly continuous data for these lakes. The purpose of this data synthesis is to compare 10 years of water quality data (2004-2013) from two oligotrophic lakes, one in a nearly pristine watershed (SOL), and one within a neighboring and developed watershed (SHL). I compared annual and seasonal nutrient levels, and evaluated precipitation trends, soils, topography, and drainage to determine if these natural factors could account for differences in nitrogen and phosphorus concentrations in the two lakes. I show that differences in land cover are the primary factor influencing differences in in-lake nutrient concentrations.

Summary of Sooke and Shawnigan Watersheds

The two watersheds share an elevation divide where water flows south to SOL, and flows north to SHL. Both watersheds are located in the Coastal Grand Fir-Western Cedar Zone of southern Vancouver Island, British Columbia, within 44 km of Victoria.

SOL, located 30 km northwest of Victoria, supplies drinking water for the Greater Victoria Water Supply System, and is 98% owned and controlled (protected) by the Capital Regional District (CRD). The watershed area delineated for the following analyses is 71 km^2 including SOL (5.90 km^2), and includes approximately 65 km^2 of surrounding land area located upslope of SOL. Annual water temperatures in the lake range from 3^o C to 28^o C. The SOL watershed received an average of 1344 mm of rain annually during 2004-2008 (Government of Canada 2014). In addition to water withdrawals for public consumption, water is released from SOL to support fisheries in the Sooke River and Charter's River to the south. However, because the following analyses address water quality in SOL, only the upslope (contributing) watershed area is considered. The contributing land area is composed entirely of old growth, young, and selectively logged forest.

SHL is located in the Cowichan Valley Regional District (CVRD) 44 km northwest of Victoria. Average annual air temperatures range from 2.8^oC to 18.3^oC (Government of Canada 2014). The watershed area used in the following analyses is 71 km^2 including SHL (5.50 km^2), and includes approximately 65 km^2 of contributing land area. The SHL watershed received an average of 1110 mm of rain annually from 2004-2013 (Government of Canada 2014). Nearly 9,000 people reside in the watershed, with 600 lake-front properties, and 4,146 land parcels throughout the watershed

(WorleyParsons 2009). Land uses in the watershed include agriculture, residential areas, forest, and some barren surfaces (British Columbia, 2013). Agriculture uses include livestock/poultry, hay and grain, and food production (WorleyParsons 2009).

Water flows from the Sooke-Shawnigan watershed divide north into SHL, and south into SOL. SHL is the through-flow for Shawnigan Creek, which originates in the Elkington forest, and flows north out of SHL, then east to Mill Bay. SHL receives water from two unnamed drainages to the east. In addition, two unnamed drainages, McGee Creek, Round House Creek, as well as, an inflow at the “west arm”, flow in from the west. Numerous drainages flow into SOL. Among these are Judge Creek, Begbie Creek, Magee Creek, Whiskey Creek, and Jones Creek for the northern basin. Rithet Creek, Horton Creek, and Highball Creek flow into the middle basin from the west, and Daisy Creek flows into the east side of the middle basin. The southern basin receives water from Trestle Creek. The SHL contributing area includes agriculture, urban areas, forest of various age, and some barren surfaces. A comparison of the area and percent cover of land cover types in the Sooke and Shawnigan watersheds is provided in Table 2.1.

Methods

Data Acquisition and Analyses

I obtained temperature and precipitation (rain) data collected at Shawnigan Lake Station #1017230 and Sooke Lake North Station #1017563 from the Government of Canada’s web-based climate database (Government of Canada 2014). Mean monthly temperatures calculated from daily temperatures (^{o}C) recorded at SHL Station #1017230, and ranged from 2.8^{o}C in December to 18.3^{o}C in July. .I obtained geographic data from DataBC (British Columbia 2013) for land cover, and Agriculture and Agri-Food Canada for soils (Agriculture and Agri-Food Canada 2012). I used ArcMap 10.0 to compare land cover and soil data between the two lakes. I obtained nitrogen and phosphorus data from the Water and Aquatic Sciences Research Lab at the University of Victoria for SOL sample point #SOL-04 and SHL sample point #SHL-01.

Using the results of water samples taken from the epilimnion (0-6 m depth) and hypolimnion (~45 m depth), I synthesized nitrogen and phosphorus records for years 2004-2013, and conducted analyses based on the annual and seasonal means. Nitrate and phosphate data were available for 2004-2009, and total nitrogen (TN) and total phosphorus (TP) data were available for 2006-2013. Results were not recorded for either lake in December 2006/2007 and January 2007, nor for SHL in January 2009. Additional data were available for south-lake sample points in each lake (SOL-01 and SHL-02), as well as, SOL-00 which was added in January 2009. However, statistical analysis determined that the trends at these points did not differ from SHL-01 and SOL-04, and there were many gaps in sampling at these points, thus they were omitted from further analysis.

Watershed Delineation

The land area (watershed) used to evaluate SHL is based on the major watershed polygon downloaded from DataBC (British Columbia 2013). I used the watershed delineation toolset in ArcMap 10.0 to verify the watershed boundaries, and returned a very similar watershed (contributing area) extent. I edited the major watershed polygon to include only contributing drainage areas, and omitted downstream drainages and topography directing water downstream using the digital elevation model (DEM) for the area. The resulting land area used to evaluate SOL is a combination of the area returned by the ArcMap 10.0 watershed delineation tool, and the sub-watershed polygons downloaded from DataBC. The final contributing land area used is of similar size to the watershed used in the evaluation conducted by Zhu and Mazumder (2008), and includes all upstream tributaries as obtained from the National Hydro Network (GeoBase, 2011).

Results

Nitrogen and Phosphorus Annual Means

I calculated the annual means from monthly and bi-monthly measurements (as recorded) of nitrate (2004-2009) and total nitrogen (2006-2008; 2011-2013) in SOL and SHL. As shown in Figure 2.1, both nitrate and TN annual means were notably higher in both the epilimnion and hypolimnion of SHL compared to SOL each year. Differences in epilimnion annual means ranged from 19.3 ppb (ug/L equivalent) nitrate in 2005 to 133.4 ug/L TN in 2012. Epilimnion nitrate was an average 2.5 times higher, and TN was an average 2.1 times higher, in SHL compared to SOL. Differences in hypolimnion annual means ranged from 21.5 ppb (ug/L equivalent) nitrate in 2006 to 128.8 ug/L TN in 2007. Hypolimnion nitrate was also an average 2.5 times higher, and TN was an average 1.9 times higher in SHL compared to SOL. Paired two sample test for means (t-test) showed significant ($p<0.001$) differences in both nitrate and TN in both the epilimnion and hypolimnion between the two lakes.

The recommended nitrate limits are 10 mg/L for drinking water and recreation, and an average of 40 mg/L for aquatic life (BC Ministry of Environment 1998). Nitrate levels recorded in SHL and SOL from 2004 to 2009 were consistently at or below 0.1 mg/L. Total nitrogen never exceeded 0.27 mg/L in SHL and was at or below 0.1 mg/L in SOL from 2006-2013.

The BC Ministry of Environment recommends that TP not exceed 10 ug/L at spring overturn for drinking water and recreation, and that it remain between 5-15 ug/L for aquatic life (BC Ministry of Environment, 1998). Phosphate levels recorded in SHL and SOL from 2004 to 2009 were consistently at or below 1.0 ug/L. Total phosphorus recorded in SHL was consistently below 10 ug/L with the exception of Summer 2007, late spring 2011, and one anomalous recording on 13 March 2013, which may have been a sample error. Total phosphorus in SOL was generally below 5 ug/L with a few exceptions, and did not exceed 10 ug/L in any sample record. When the entire data set is plotted, as presented in Figure 2.2, the resulting graph shows that nitrate levels were nearly the same, and close to zero, for both lakes during the summer months, whereas rainy season spikes were more severe at SHL.

As shown in Figure 2.3, the recorded TN was only rarely below 50 ug/L at SOL-04, but was rarely below 100 ug/L and typically above 150 ug/L at SHL-01. Interestingly, the overall relative trend was very similar for both lakes.

Phosphate and Total Phosphorus

The annual means were calculated from monthly measurements of phosphate (2004-2009) and TP (2006-2008; 2011-2013) in SOL and SHL. Annual mean phosphate was higher in the epilimnion of SOL by an average 0.21 ppb in 2004-2009. TP was an average 1.4 times higher in the epilimnion of SHL). Records were available for TP in the hypolimnions only from 2006-2008. Annual mean TP was 1.6 times higher in the hypolimnion of SHL during that time (Figure 2.4). Statistical analysis showed a significant difference ($p<0.001$) between TP in both the epilimnion and hypolimnion between the two lakes.

Figure 2.5 shows the recorded TP was only rarely below 2 ug/L in SOL, but spanned primarily from 4-8 ug/L in SHL. The overall relative trend was very similar for both lakes. TP in SHL decreased in late 2011 to 2013 compared to previous years.

Nitrogen and Phosphorus Seasonal Means

Given that annual mean TN and TP were notably higher in SHL, and statistical analyses comparing all data entries for both datasets showed significant differences, we expect seasonal means to also be higher. Differences in seasonal mean TN in SHL were on average twice that of SOL (Figure 2.6). With the exception of Spring 2006, seasonal mean TP was an average 1.5 times higher in SHL (Figure 2.7).

Precipitation

Mean monthly rainfall (mm) was calculated based on daily rainfall data (Government of Canada, 2014). The comparison of SOL and SHL precipitation records showed that the SOL watershed received approximately 10% more rainfall than SHL from 2004 to 2009 (Table 2.2), and the maximum recorded monthly rainfall was 500 mm in SOL verses 432 mm in SHL. The mean differences in monthly precipitation during the rainy season (October-April) range from nearly 12% in October to 27% in January (Table 2.3).

Soils

Soils in the SOL watershed are predominantly gravelly sandy loam. As presented in Table 2.4, these soils have a relatively low erodibility factor (0.009), indicating less erosion (sediment transport) than soils like the silty gravel soils (0.025) that comprise 47.1% of the SHL watershed. Gravelly sandy loam soils have secondary dominance in the SHL watershed, covering 41.9% of the watershed area. Soils in the SOL watershed are predominantly rapidly drained with moderate permeability, whereas the SHL watershed consists primarily of well-drained soils (Table 2.5).

Agriculture and Agri-food Canada maintains soil survey reports for British Columbia. The reports for Vancouver Island (Agriculture and Agri-Food Canada, 2012) were prepared by the BC Ministry of Environment during the early 1980's and provide the most comprehensive information on soils. Soil survey reports classify soils into named units, and describe soil traits such as the erodibility, drainage, and permeability previously discussed. Soils around the perimeter of SHL are primarily Dashwood soils, with Shawnigan soils to the northeast, Chemainus soils on the southern shore, and Qualicum soils at the tip of the "west arm". Soils around the perimeter of SOL are primarily Shawnigan to the north, Quinsam soils on the west and south shores, and Shepherd and Squally soils on the east shore.

Most BC soil survey reports characterize only the type, texture, and likely locations of each named soil unit. The *Soils of Southeast Vancouver Island, Duncan-Nanaimo Area* (Jungen, et al., 1985) contains additional soil chemistry information for some soils as presented in Table 2.7. Soil chemistry information was not available for Shepherd and Squally soils (Sooke).

Discussion

Precipitation and Watershed Drainage

Precipitation contributes to erosion and increased runoff over land into lakes and into the drainages that transport nutrients into lakes. Rainfall intensity and the corresponding dilution effect results in lower nitrogen and phosphorus concentrations in runoff (Kleinman et al. 2006). The SOL watershed receives 17-27% more monthly precipitation than SHL between December and April (Table 2.3). Monthly mean rainfall intensity, expressed as millimeters per square kilometer (mm/km^2) between 2004 and 2012, was greater in all months of the year, but neither the maximum difference in January of 0.6 mm/km^2 nor lesser differences would likely contribute to substantial dilution in SOL runoff compared to SHL.

Drainages and creeks carry nutrients, especially nitrogen, from upland areas into the receiving water (lakes). Nutrient transport in linear drainages is more conservative than that of overland flows because nutrients are not intercepted by vegetation, and due to flow depth and velocity, do not adhere as readily to soil particles. It would follow that the watershed with a larger number of direct drainage channels should receive a higher quantity of nutrients. The SOL and SHL watersheds are very similar in size at 65.14 and 65.08 km^2 respectively. SOL has 27 channelized drainage inflow points compared to eight in SHL. The total linear length of streams in the SOL watershed is 93 km verses 65 km in the SHL watershed. SOL receives water from many more direct drainage channels than SHL, yet SHL nitrogen and phosphorus levels were consistently higher than SOL over the past 10 years.

Local Topography

Local topography is a major factor in nutrient transport within watersheds because steeper slopes result in greater flow velocities (Sidle et al. 2000), and carry more nutrients (Castillo 2009) whereas more minor slope gradients slow flow velocity and allow water to percolate into the soil. Soil particles carried by drainage channels will more readily settle when flow velocities are lower (i.e., in areas with lesser slope). The topography within the SOL watershed ranges from 500 m above mean sea level (amsl) at the top of the watershed to 190-200 m amsl at the shoreline. Topography within the SHL watershed ranges from 560-590 m amsl to the south and west, 180 m amsl to the east, and 120 m amsl at the shoreline. Topography around SOL is steeper than that of SHL in the vicinity of the lake shores. Within 500 m of the lake edge, the percent slope around SHL ranges from 3-13%, and from 10-64% around SOL. SHL has a much wider area of gently sloping topography adjacent to the shoreline. This should result in slower flow velocities and greater retention of nutrients by soils and vegetation.

Soils

The different soil types around each lake do not likely account for the differences in TN and TP. Although the silty gravel soils immediately surrounding SHL have a higher erodibility factor and somewhat slower percolation (although still primarily well-drained) than the gravelly sandy loam soils present around two-thirds of SHL, these soils are also more likely to attenuate nutrients and reduce loading. Mahmood-Ul-Hassan (2011) found that soils with larger macropores adsorb less TP and allow for the rapid movement of water soluble nitrogen through the soil. The gravelly and rapidly-drained soils surrounding SOL have larger macropores than the silty gravel soils around SHL. Thus, SHL soils will most likely adsorb more TP and retain more TN than SOL soils due to soil textures. The 3-13% slopes around SHL compared to 10-64% slopes around SOL indicate that vegetation uptake and particle settling should be higher in near SHL.

As shown in Table 2.6, the SHL soils, Qualicum, Dashwood, Shawnigan, and Chemainus, have the highest mean phosphorus content at 90.4, 30.4, 19.8, and 15.4 ppm respectively. In a study of phosphate mobility and persistence in septic effluent it was determined that because metal salts such as alum and iron chlorides aid in the precipitation of phosphate in soil, they also control the quantity of phosphate that is transported out of septic drain fields (Robertson, et al., 1998). The soil chemistry information (Table 2.7) supports the idea that the soils around SHL retain phosphorus due to the relatively high mean phosphorus (indicating natural attenuation) and the presence of iron and aluminum. Because soil chemistry information is not available for the entire SOL perimeter, a direct comparison is not possible. However, Quinsam and Shawnigan soils were lower in phosphate than the Dashwood soils that surround most of SHL. A general evaluation of phosphorus and mineral content in soils indicates that naturally-occurring soil chemistry does not explain the higher phosphorus levels in SHL compared to SOL.

Land Cover

The comparison of precipitation, topography, and soils does not explain the elevated levels of in-lake nutrients in SHL compared to SOL given the natural characteristics of the two watersheds. Land cover and land use in the SHL watershed is likely the most influential factor contributing to higher nutrient loading into SHL compared to SOL.

The SOL contributing area consists entirely of old growth (52%), young forest (16%), and recently (<10 years) logged forest lands (32%). The SHL contributing area includes agriculture (2%), urban areas (14%), young forest (72%), old growth forest (2%), recently logged forest lands (10%), and some barren surfaces (0.5%) (Table 2.1) (British Columbia 2013)

Zhu (2005) found that forest harvesting may contribute to increased nutrient concentrations in streams because the reduction in cover (shade) results in increased soil temperature which facilitates organic matter decomposition and increases water soluble nitrogen. Decomposing harvest waste (slash) left on site increases the organic matter, and thus increases nutrients available for transport via soil solution (Zhu, 2005). Zhu and Mazumder (2008) modeled nitrogen export in the SOL watershed based on forest cover and soil types. The study found that forests less than 20 years old and with less than 40% canopy cover result in more nitrogen export than mature forests. Nitrogen export rates from young forest (<20 years) were an average 4.00 kg ha^{-1} yr^{-1} (400 kg/km^2) verses nitrogen export rates for old growth forests (>120 years), which averaged between 1.28 and 1.74 kg ha^{-1} yr^{-1} (128 kg/km^2 and 174 kg/km^2). Nitrogen export rates from forests with less than 60% canopy cover averaged 2.78 kg ha^{-1} yr^{-1} (278 kg/km^2), and from bare land averaged 1.26 kg ha^{-1} yr^{-1} (126 kg/km^2) (Zhu, 2008). The dominance of young forest in the Shawnigan watershed may explain some of the elevated nitrogen levels. Because the absence of forest cover (shade) and associated increased soil temperatures result in higher water soluble nitrogen, forest harvesting and residential clearing around SHL are likely contributors to elevated TN.

In a study of phosphorus export in 31 southern Ontario watersheds, which included the compilation of similar studies on 43 additional watersheds in North America and Europe, Dillon and Kirchner (1975) found significant differences between phosphorus export from forested areas verses areas with pasture and forest. The study found that phosphorus export from forested areas averaged 11.7 mg m^{-2} yr^{-1}, and phosphorus export in areas with forest and pasture averaged 23.3 mg m^{-2} yr^{-1}. Based on studies of phosphorus export in 74 watersheds world-wide, twice as much phosphorus was transported from lands containing forest and pasture than completely forested land

(Dillon and Kirchner 1975). Agricultural areas, which are typical sources of fertilizer and animal waste, comprise only 2% of the SHL watershed. Low-density residential areas comprise 14% of the watershed, and are directly adjacent to the lake shore along >90% of the perimeter. The residential areas contribute to direct run-off due to impervious surfaces such as roofs and driveways. The urban area at the north end of the lake would also contribute excess phosphorus to the lake (Glandon et al. 1981; Wickham 2002).

Common sources of excessive nitrogen and phosphorus loading include fertilizers (lawns/agriculture), human sewage (septic systems), and animal waste (Murphy, 2007). The placement of septic systems around the lake perimeter increase the risk of excessive nitrogen loading in the event of outdated systems, over-use, and seasonally-elevated groundwater levels. Bacteria source tracking analyses conducted on SHL (Water and Aquatic Sciences Research Program, 2012) found that 14.3% of *E.coli* isolates in raw water samples taken in 2011 were human, indicating that septic leachate is reaching the lake. Caffeine and prescription drugs have also been detected in the lake (Mazumder pers. comm., 2014), and are another indicator of septic intrusion. The bacterial source tracking analyses also found that 18.6% of *E.coli* isolates in raw water samples taken in 2011 were from horses, and horse manure may also contribute additional nitrogen into the lake.

Conclusion

The comparison of nitrate, TN, phosphate, and TP over 10 years showed that annual mean nitrate, TN, and TP were higher in SHL than SOL. Between 2004 and 2009, annual mean nitrate was 2.5 times higher in SHL. Between 2006 and 2013, annual mean TN was 1.9 times higher, and annual mean TP was 1.5 times higher in SHL. SOL and

SHL datasets were significantly different ($p<0.001$) for TN and TP. Phosphate was slightly (0.21 ppb) higher in SOL from 2004 to 2009. Seasonal mean TN was on average double and seasonal mean TP was 1.5 times higher in SHL between 2006 and 2013.

The nitrate limits recommended by the BC Ministry of Environment are 10 mg/L for drinking water and recreation, and an average of 40 mg/L for aquatic life (BC Ministry of Environment, 1998). Nitrate levels in both lakes were consistently at or below 0.1 mg/L. Total nitrogen never exceeded 0.27 mg/L in SHL and was at or below 0.1 mg/L in SOL from 2006-2013.

The BC Ministry of Environment recommends that TP not exceed 10 ug/L at spring overturn for drinking water and recreation, and that it remain between 5-15 ug/L for aquatic life (BC Ministry of Environment, 1998). Phosphate levels in both lakes were consistently at or below 1.0 ug/L. Total phosphorus in SHL was consistently below 10 ug/L. Total phosphorus in SOL was generally below 5 ug/L with a few exceptions, and did not exceed 10 ug/L in any sample record.

I evaluated differences in precipitation, watershed drainage, topography, soils, and land cover between the SOL and SHL watersheds to determine if these factors account for the elevated nutrient levels in SHL compared to SOL. Although rainfall intensity, and therefore dilution, is higher in the SOL watershed than the SHL watershed, the differences are minimal, and precipitation alone does not account for the additional nutrient loading into SHL compared to SOL. There are more drainages (creeks and unnamed features) flowing into SOL. Drainage channels carry dissolved nutrients from upland areas into downstream waters. Given that SOL has 30% more linear meters of streams than SHL, the number of incoming drainages does not explain the excess nutrients in SHL. Soil texture in the vicinity of the lake edge differs between the two watersheds, and the silty gravel adjacent to SHL is more erodible than the gravelly sandy loam that surrounds two-thirds of SOL, but is also more likely to adsorb and retain TP and reduce the movement of TN through the soil. The ability of vegetation to obstruct, adsorb, or assimilate nutrients is primarily based the local slope gradient (Phillips 1989). Topography around SOL is much steeper than around SHL, and 3-13% slopes around SHL would reduce erosion and increase the potential for nutrient attenuation in the soils around the lake edge. Precipitation, slope, drainage, and soils in the two watersheds do not account for the differences in nutrient levels between the lakes.

The SOL contributing area consists entirely of old growth, young forest, and selectively logged forest lands. The SHL contributing area includes agriculture, urban areas, forest of various age, and some barren surfaces. The mix of young forest, residential areas, and the urban area to the north of SHL are the only factors account for the additional nutrient loading into the lake.

The elevated levels of TN and TP in SHL verses SOL are most likely due to anthropogenic factors, such as forest harvesting, residential clearing, the use of fertilizer on lawns, and faulty septic systems. Given that 18.6% of *E.coli* came from horses, the presences of horses (manure) may also contribute to elevated nitrogen.

Tables

Table 2.1. Comparison of Sooke and Shawnigan land cover. Land cover types are expressed by area (km^2) and percent of total watershed.

Land Cover Type	SOL		SHL	
	(km^2)	%	(km^2)	%
Agriculture	None	0	1.26	1.9%
Barren Surfaces	None	0	0.30	0.5%
Young Forest	10.36	15.9%	47.27	72.3%
Old Forest	34.28	52.4%	1.36	2.1%
Recently Logged	20.71	31.7%	6.59	10.1%
Selectively Logged	0.008	<0.01%	---	0.0%
Urban	None	0%	8.57	13.7%
Total Land Use[1]	65.36 km^2		65.35 km^2	
Fresh Water	5.74	8.1%	5.48	8.4%
Total with Lake	71.1 km^2		70.83 km^2	

Table 2.2 Annual difference in precipitation. The Sooke watershed received more precipitation between 2004 and 2009 than the Shawnigan watershed each year as expressed in mm/year and percent difference.

	2004	**2005**	**2006**	**2007**	**2008**	**2009**	**Mean Annual Difference**
mm/yr	197.2	184.0	279.0	383.2	136.9	207.6	233.85
%	9.1%	13.5%	8.4%	-1.1%	13.9%	10.1%	10.1%

Table 2.3 Mean difference in monthly precipitation. The Sooke watershed received more precipitation in each month between 2004 and 2009 than the Shawnigan watershed as expressed in mm/month and percent difference.

2004-2009	**mm/month**	**%**
January	62.88	26.6%
February	21.35	16.7%
March	36.53	21.8%
April	18.72	27.0%
May	3.97	0.4%
June	0.87	-8.4%
July	2.63	4.1%
August	-0.68	-21.2%
September	6.90	12.3%
October	15.88	11.9%
November	33.18	12.8%
December	31.62	17.2%

[1] Note that minor discrepancies in total land area for land cover and soils (Shawnigan 0.20 km^2 and Sooke 0.16 km^2) are due to minor inconsistencies in the "Clip" tool used in ArcMap 10.0 to designate only land cover and soils within the watersheds.

Table 2.4. Proportion of soil textures by watershed, erodibility, drainage, and permeability. Soil textures for the Shawnigan (SHL) and Sooke (SOL) watersheds are expressed by area (km^2) and percent of total watershed, with each associated erodibility factor (higher=more erodible), and drainage (indicates how water will move through the soil).

Soil Texture	**SHL**		**SOL**		**Erodibility Factor**	**Drainage**
	(km^2)	**%**	**(km^2)**	**%**		
Gravel	0.94	1.4%	---	0.0%	0.001	rapid
Gravelly Sandy Loam	27.45	41.9%	43.21	66.3%	0.009	rapid
Gravelly Loamy Sand	5.04	7.7%	---	0.0%	0.005	well
Sandy Loam	0.60	0.9%	3.28	5.0%	0.017	well
Silty Gravel	30.85	47.1%	18.71	28.7%	0.025	mod. well
Loam	0.46	0.7%	---	0.0%	0.040	mod.well
Peat	0.21	0.3%	---	0.0%	0	very poor
Total Soil Area	65.55		65.20			

Table 2.5. Proportion of drainage and permeability classes in the Shawnigan (SHL) and Sooke (SOL) watersheds. Soil drainage (indicates how water will move through the soil) and permeability (indicates how easily water can enter the surface of the soil) are expressed by area (km^2) and percent of total watershed.

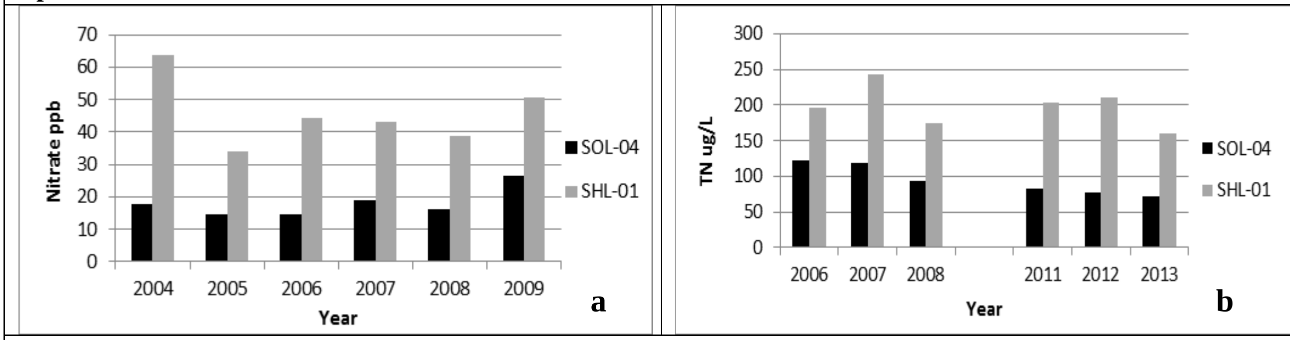
Drainage	**Shawnigan**		**Sooke**		**Permeability**	**Shawnigan**		**Sooke**	
	km^2	**%**	**km^2**	**%**		**km^2**	**%**	**km^2**	**%**
very poor	0.21	0.3%	---	0.0%	moderate	64.6	98.6%	65.2	100.0%
mod. well	31	47.8%	19	28.7%	rapid	0.94	1.4%	---	0.0%
well	5.6	8.6%	3.3	5.0%					
rapid	28	43.3%	43	66.3%					

Table 2.6. Soil Composition from Soil Survey Reports. Named soils for which soil chemistry was available are expressed as the percent (%) of the total watershed. Soil chemistry information was not available for all soils (≠100%). Organic carbon, nitrogen, iron and aluminum are expressed in % of soil mass. Phosphorus is expressed in parts per million (ppm).

Soil Association	SHL %	SOL %	pH 1:1 H_2O	Organic Carbon Mean %	Nitrogen Mean %	Phosphorus Mean ppm	Iron %	Aluminum %
Arrowsmith	0.3%	0.0%	5.4	43.8 (35.0-50.9)	1.94 (1.81-2.06)	ND	ND	ND
Chemainus	0.7%	0.0%	5.8	1.12 (0.21-2.26)	0.08 (0.01-0.10)	15.4 (7.9-25.6)	0.36 (0.06-0.18)	0.37 (0.24-0.49)
Dashwood	21.5%	0.0%	5.8	1.6 (0.3-3.3)	0.07 (0.01-0.19)	34.0 (8.7-80.6)	0.18 (0.02-0.43)	0.14 (0.09-0.22)
Qualicum	1.4%	0.0%	5.2	1.9 (0.3-1.9)	0.03 (0.01-0.07)	90.4 (29.4-137.8)	0.17 (0.01-0.35)	0.31 (0.17-0.76)
Quinsam	0.0%	2.7%	5.6	1.7 (0.7-2.36)	0.05 (0.03-0.10)	11.3 (7.7-18.6)	0.29 (0.13-0.55)	0.45 (0.28-0.69)
Shawnigan	25.5%	28.7%	5.8	1.1 (0.3-1.6)	0.04 (0.01-0.09)	19.8 (9.6-44.3)	0.16 (0.04-0.30)	0.28 (0.15-0.41)

Figures

Epilimnion



Hypolimnion

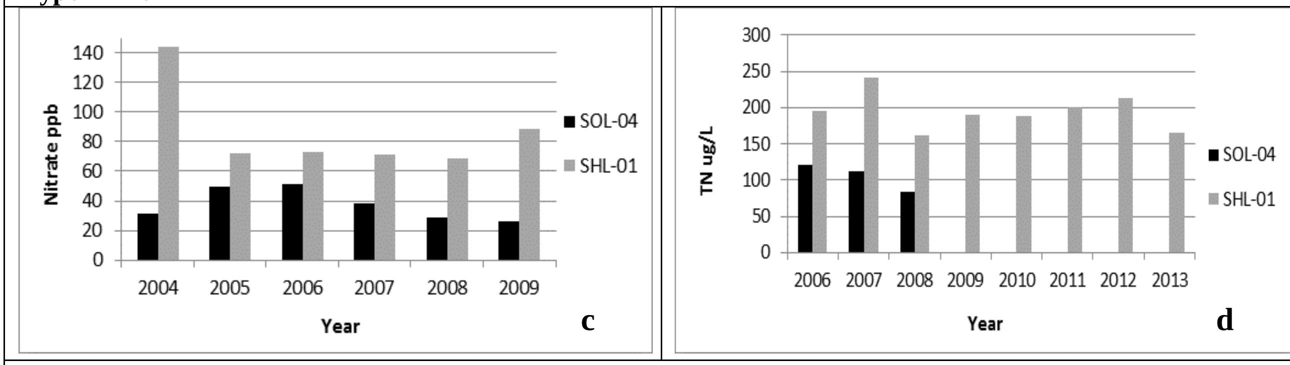


Figure 2.1. Annual mean nitrate and total nitrogen (TN). Comparison of nitrate annual mean concentrations (ppb) for years 2004-9009 and TN annual mean observed concentration (ug/L) for years 2006-2008 and 2011-2013 at sample points SOL-04 and SHL-01. Note: Due to gaps in monthly sampling at SHL-01 in 2009 and 2010, these years were excluded from the epilimnion TN annual mean comparison.

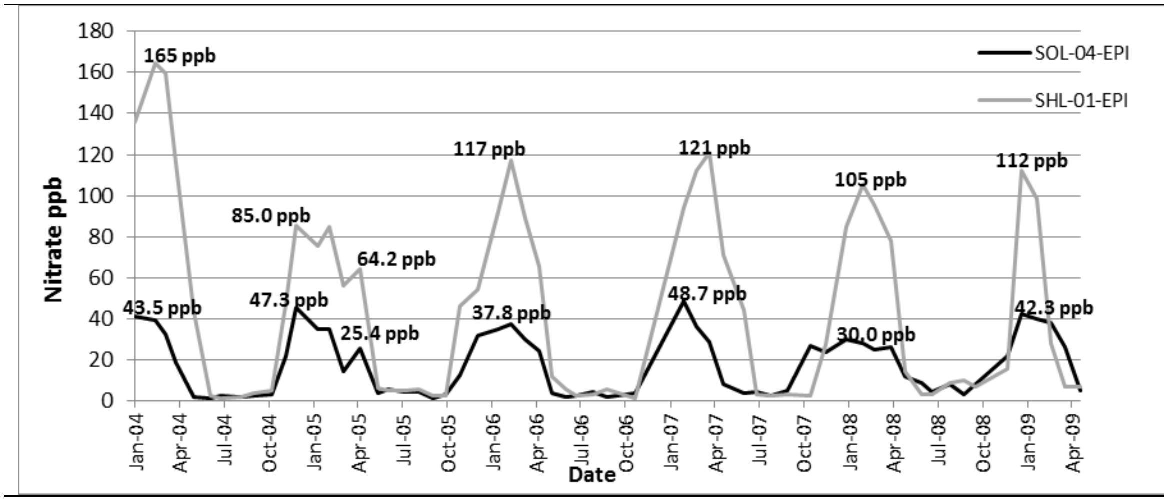


Figure 2.2. Nitrate (ppb) trends from all data records in Sooke (SOL) and Shawnigan (SHL) lakes at sample points SOL-04 and SHL-01 from 2004-2009.

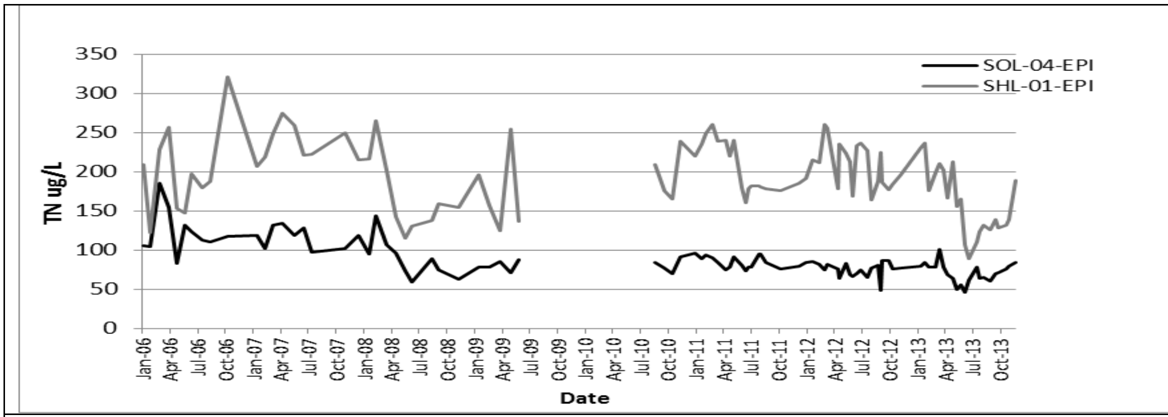
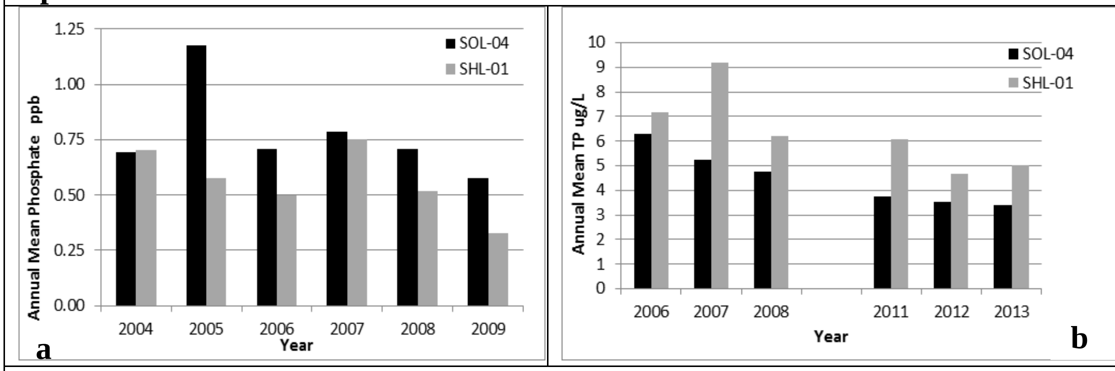


Figure 2.3. Total nitrogen (TN) trends from all epilimnion data records for Sooke (SOL) and Shawnigan (SHL) lakes taken at sample points SOL-04 and SHL-01 from 2006-2009 and 2011-2013.

Epilimnion



Hypolimnion

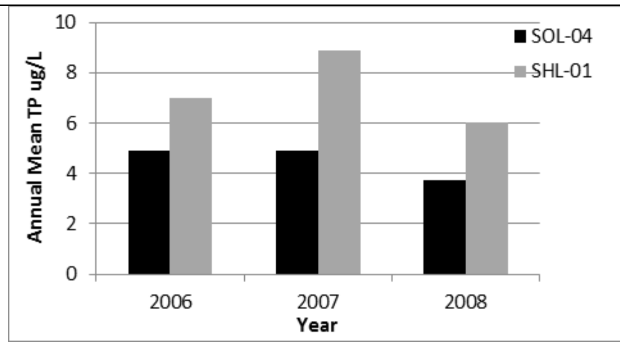


Figure 2.4. Total phosphorus (TP) annual mean concentration in ug/L for years 2006-2008 and 2011-2013 in Sooke (SOL) and Shawnigan (SHL) lakes recorded at sample points SOL-04 and SHL-01.

Note: Due to gaps in monthly sampling at SHL-01 in 2009 and 2010, these years were excluded from the epilimnion TP annual mean comparison.

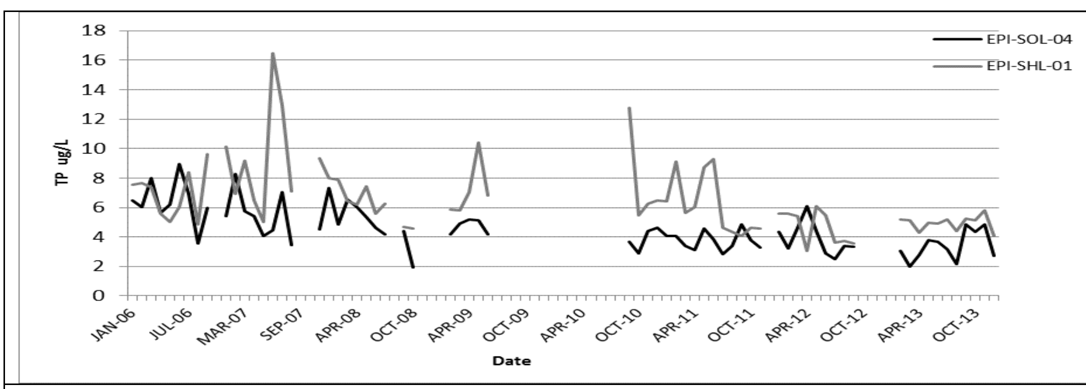


Figure 2.5. Total phosphorus (TP) trends from all epilimnion available data records for Sooke (SOL) and Shawnigan (SHL) lakes recorded at sample points SOL-04 and SHL-01 .

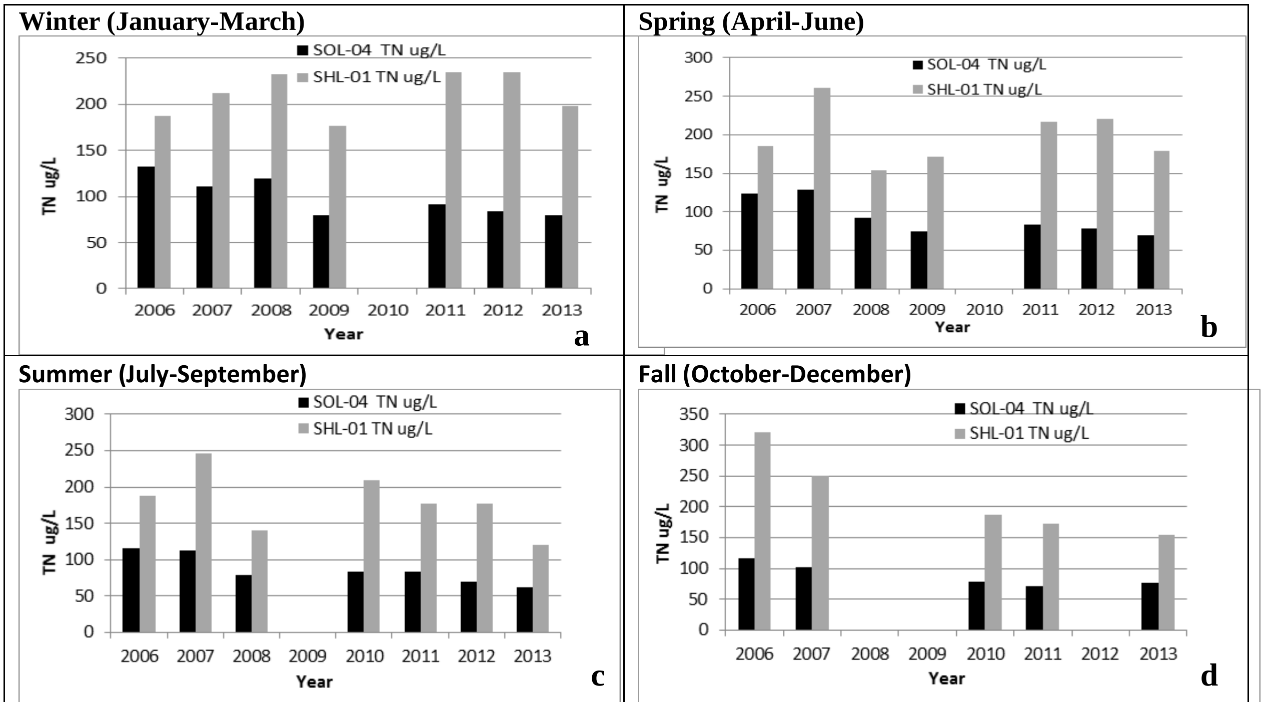


Figure 2.6. Differences in seasonal mean total nitrogen (TN) between Shawnigan (SHL-01) and Sooke (SOL-04). (a) Winter (January-March) mean TN (ug/L). (b) Spring (April-June) mean TN (ug/L). (c) Summer (July-September) mean TN (ug/L). (d) Fall (October-December) mean TN (ug/L).

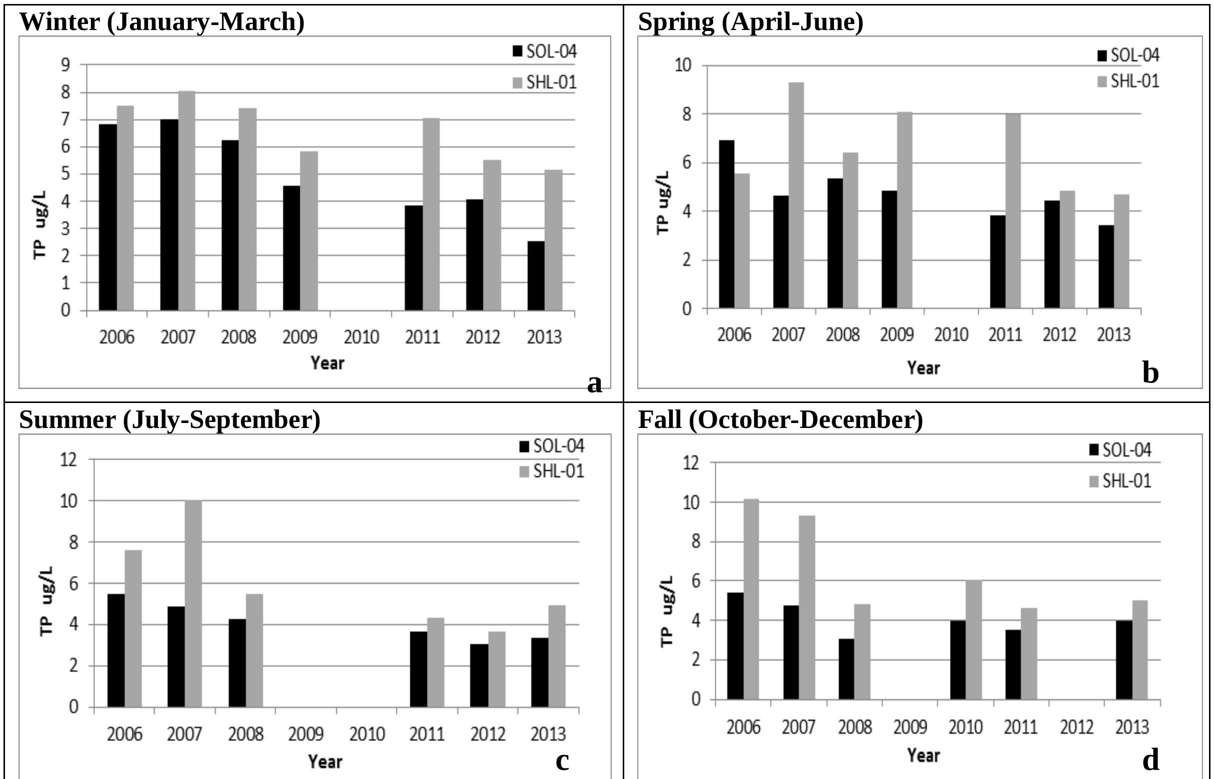


Figure 2.7. Differences in seasonal mean total phosphorus (TP) between Shawnigan (SHL-01) and Sooke (SOL-04). (a) Winter (January-March) mean TP (ug/L). (b) Spring (April-June) mean TP (ug/L). (c) Summer (July-September) mean TP (ug/L). (d) Fall (October-December) mean TP (ug/L).

Chapter 3: Phosphorus and Nitrogen trends in four coastal freshwater lakes and implications for causal loading factors

Abstract

I performed linear regression analyses to determine whether phosphorus and nitrogen concentrations have changed over time and to evaluate nutrient-precipitation dependence in four coastal B.C. lakes. Monthly sample data were data were acquired for two oligotrophic lakes, Sooke and Shawnigan, and two eutrophic lakes, St. Mary, and Elk. Long-term trends were evaluated beginning in the 1970's for Shawnigan and St. Mary Lake, the 1980's for Elk Lake, and from 2006-2013 for Sooke Lake. With the exception of Elk Lake, long-term data show no significant changes in in-lake nutrient concentrations. Linear regression analyses were also performed on time-segmented monthly concentration data for winter and summer for both shallow (epilimnion) and deep water (hypolimnion) samples. The results show that phosphorus and nitrogen concentrations declined or exhibited no change in the oligotrophic lakes. The average nitrogen to phosphorus ratio in these lakes has increased from 17:1 to 25:1 since 2006, indicating a trend toward lower in-lake productivity. Nutrient concentrations in the oligotrophic lakes were generally not precipitation dependent, and this indicates that the declining trends are due to declining in-lake productivity. St. Mary Lake data showed that from 1979-1990 phosphorus concentrations were significantly correlated to precipitation, and were in decline, indicating a reduction in external loading during that time period. From 2006-2014 phosphorus concentrations no longer correlated to precipitation, and monthly samples exhibited greater variation, particularly in the winter. Winter phosphorus showed no change. Summer phosphorus showed an increasing trend in the epilimnion, and a decreasing trend in the hypolimnion. Aeration operation in several years is shown to correspond to especially low summer deep water phosphorus.

Phosphorus concentrations were lower in the spring than in the late fall in the majority of years, and this, coupled with no precipitation-dependence indicate that the cause of phosphorus variation is internal nutrient cycling rather than external loading. Correlations with algal abundance indicate that a persistent algae population is likely sequestering phosphorus in the water column for most of the year. Elk Lake phosphorus concentration data show no change in the epilimnion, but a significant increase in the hypolimnion since 1986. The results are influence by deep water phosphorus samples obtained in 2014 in which phosphorus levels were 3 times previous records. Additional sampling is recommended to determine whether the very high phosphorus concentrations represent the lake condition. Statistical relationships between phosphorus, water temperature, and dissolved oxygen were evaluated, and the results support the determination that phosphorus concentrations are mostly influenced by in-lake productivity.

Introduction

Declining water clarity and frequent algae blooms in both the winter and summer have led to water quality monitoring in four small community watersheds: Sooke (SOL), Shawnigan (SHL), St. Mary (SML), and Elk/Beaver (ELK) Lakes. Lake productivity is primarily a function of the relative amounts of nitrogen and phosphorus (N:P ratio) in lake water, and secondarily linked to lake water residence times and warmer year-round temperatures (Romo, 2013). With predictions of increasing winter temperatures and rainstorm intensity in the Pacific Northwest due to climate change (IPCC, 2013), watershed managers need to better understand the factors contributing to lake productivity in order to prevent oligotrophic (low productivity) lakes from becoming eutrophic (highly productive), and to improve the water quality in eutrophic lakes that already experience frequent, and often toxic, algae blooms. Understanding the factors that are contributing to the quality of source lake water is essential to informing cost-effective and beneficial management decisions so that appropriate actions are implemented and resources are not wasted.

Water residence times are very positively correlated with the retention of both nitrogen and phosphorus by lakes (i.e., long residence times = high nutrient retention) (Brett and Benjamin 2008; Saunders and Kalff 2001). Drought-years and the significant reduction in precipitation during the summer months result in longer residence times, especially for small hydrologically isolated lakes like SML and ELK, which are entirely dependent on wet season precipitation for water inflow[2]. Oligotrophic lakes typically

[2] In contrast, higher elevation interior lakes receive water (inflow) from snowmelt, often throughout the summer dry season.

have shorter residence times than productive lakes (Romo, 2013). The shorter (~2-year) residence times of SOL and SHL result in less nutrient accumulation than the longer (~7-year) residence times of SML and ELK, and as a result, lakes with longer residence times are more productive.

Internal Loading Verses In-Lake Productivity

The term "internal loading" refers to the release of phosphorus from the sediments in the lake bottom. Carey and Rydin (2011) found that phosphorus sequestration in lake sediments varies predictably between eutrophic, mesotrophic, and oligotrophic lakes. Total phosphorus (TP; includes organic and inorganic soluble and non-soluble forms of phosphorus) measured in the sediment of 94 lakes (to 30 cm depth) showed that eutrophic lakes exhibit high TP in surface sediment, with a rapid decrease in TP along the sediment depth gradient. Oligotrophic lakes exhibit low TP in surface sediment, with a moderate rate of increase in TP with depth. Mesotrophic lakes exhibit relatively constant TP with depth. The study found that the TP trend within the sediment corresponded with the water column TP concentrations that characterize lake trophic status. Iron (Fe-III) and aluminum are fundamental to phosphorus sequestration by soils and lake sediment. Low hypolimnic oxygen levels, which are typical in productive lakes during summer stratification, hinder the ability of iron and aluminum to sequester phosphorus due to resulting molecular state changes (e.g., Fe-II to Fe-III). This occurs under low oxygen conditions due to the chemical precipitation wherein Fe (II) unbinds from the sediments and releases Fe (III) ions along with phosphorus (as PO_4-) into the overlying water (Carey & Rydin, 2011). This results in a "load" of phosphorus in the deep water (hypolimnion). Low hypolimnic oxygen levels in productive lakes are caused by summer stratification which isolates the hypolimnion from atmospheric oxygen exchange, coupled with high biological productivity which depletes the dissolved oxygen (DO) in the lake. DO concentrations have been strongly correlated to the release of phosphorus from lake sediments (Carey and Rydin, 2011), and internal loading is expected at DO concentrations < 0.5 mg/L (Nurnberg 1984). Sedimentary bacteria that thrive in anoxic conditions also contribute to the release of phosphorus from lake sediments (Gachter 1986), and benthic bacteria and cyanobacteria contribute to sediment to water column phosphorus cycles through productivity in which they sequester and release phosphorus (Brunberg 1995). Due to higher DO levels (typically >5 mg/L), internal loading is minimal in oligotrophic lakes. Unlike the mesotrophic and eutrophic lakes where TP can be 6 or more times higher in the hypolimnion than the epilimnion in summer months, TP concentrations in the oligotrophic lakes are similar (within 1 ug/L by average) in the epilimnion and hypolimnion in the winter and summer. The oligotrophic lakes do stratify in the summer, and can exhibit similar maximum surface-to-deep-water temperature differences (~13^{o}C). However, the oligotrophic lakes maintain much higher DO levels in the hypolimnion (SHL minimum DO = 4.2 mg/L verses SML minimum DO = 0.01 mg/L). This is because biological productivity is less, and therefore oxygen consumption is also less, in oligotrophic lakes than mesotrophic or eutrophic lakes.

Evaluating the change in nutrient concentrations over time and determining whether a statistical correlation with precipitation exists give insight into the factors which are most affecting water quality and lake productivity. External loading can only occur by the transport of nutrients in runoff from watersheds into lakes. If nutrient concentrations are changing over time, and external loading is the primary driver, then in-lake nutrient concentrations would correlate with monthly precipitation. If nutrient concentrations are changing over time and nutrient concentrations do not correlate with precipitation, then in productive lakes (mesotrophic and eutrophic) previous research indicates that internal loading from sediments is the primary driver. However, in SML, TP in the hypolimnion (Hypo-TP) in the summer is in a declining trend, and this indicates that internal loading does not explain the increased variation in TP in the epilimnion (Epi-TP) in recent years. Similarly, in oligotrophic lakes where TP cycling from the sediment is minimal, nutrient cycling from sediments cannot explain the decrease in TP. Thus, cycles of biological productivity of organisms in the lakes (referred to herein as "in-lake" causal load) must be driving nutrient concentrations such that cycles of high biological productivity result in increased nutrient concentrations, and cycles of low productivity lead to reduced nutrient concentrations because the nitrogen will flow out of the lake and phosphorus will adsorb to lake sediments. For additional evidence of this, hypolimnic DO, and in some cases water temperature, were found to correlate statistically with TP concentration when internal loading, ambient mass, and in-lake processes were identified as the causal loads. Warm water temperatures promote biological growth, and low DO indicates the use of oxygen by organisms in the lake. Finally, when there is no change in nutrient concentration over time, then either the ambient load is persisting if precipitation is not a factor, or there is a net balance in inflow mass and outflow mass if nutrient concentrations are precipitation dependent. Figure 3.1 demonstrates these links between loading and in-lake nutrient trends, and guides interpretation of the results presented in Figures 3.2a-3.15b.

The TP trends for SOL, SHL, SML, and ELK are presented in the figures and tables at the end of Chapter 3. TN trends are also presented for the oligotrophic lakes (SOL and SHL) because changes in the N:P ratio over time indicate whether these lakes are moving toward or away from increased productivity (Knowles, 1982; Downing, 1992). The meso-eutrophic lakes (SML and ELK) are already highly productive and TP concentration is of greater importance than TN, which has only rarely been documented during water quality sampling.

Influential Weather Patterns

Weather patterns that influence the climate, both temperature and precipitation, in the subject watersheds include Pacific Decadal Oscillation (PDO), El Nino Southern Oscillations (ENSO), and La Nina. PDO is a 20-30 year temperature oscillation in the northern Pacific Ocean (California Institute of Technology 2015). A “warm” phase was thought by oceanographic scientists to have lasted from 1977-1999, after which the “cool” phase began. During the warm phase, the west Pacific Ocean was cooler, and during the current “cool” phase, a pattern of warm ocean water is evident in the north, west, and southern Pacific Ocean. ENSO occur every two-seven years and last from 6 to 18 months (Environment and Climate Change Canada 2015). ENSO is centered in the tropics. During ENSO, the Pacific Ocean on the west coast of BC is relatively warm, and this results in milder (warmer) weather in the winter and spring. Ocean temperatures on the west coast of BC are cooler during La Nina, resulting in below-normal air temperatures. A four-year ENSO occurred in the early 1990’s (Environment and Climate Change Canada 2015). ENSO also occurred in 1977-78, 1979-80, 1987-88, spring 2005, 2006-2007 and 2009-2010 (National Weather Service 2015). Warmer temperatures and high precipitation would have occurred during those years. La Nina oscillations were prevalent from 1974 to early 1976, 1988-89, 2007 to mid-2008, and from July 2010 to April 2012 (National Weather Service 2015). Cooler drier winters occurred during La Nina years.

Summary of Subject Watersheds

SOL and SHL are oligotrophic lakes located within adjacent watersheds approximately 35 km north of Victoria on Vancouver Island, BC. SML is a mesotrophic lake located on Salt Spring Island, 32 km northwest of the SOL/SHL divide. ELK is a eutrophic lake located on the Saanich Peninsula on Vancouver Island, BC approximately 11 km north of Victoria. SOL lies within an undeveloped and protected watershed, and provides 90% of the drinking water for the city of Victoria. SHL lies within a partially developed watershed where logging operations are ongoing, and is the drinking water source for the SHL community. SML lies within a watershed subdivided into rural residential parcels, and provides drinking water for much of north Salt Spring Island. ELK lies within a watershed mostly subdivided into rural residential parcels, and containing annual cropland and pasture lands. ELK is not current managed for public consumption. All four watersheds share very similar climate and geology, and are within the Coastal Douglas-fir biogeoclimatic zone.

Table 3.1 summarizes the general characteristics of the subject lakes. The surrounding watersheds of SOL and SHL are of similar size, approximately 65 km^2. The drainage area to lake surface area ratios are 1:11 and 1:12. SOL has a greater lake volume, 160.3 x 10^6 m^3 compared to that of SHL with a volume of 71.9 x 10^6 m^3, a greater mean depth, 19.5 m compared to 13 m, and a faster flushing rate of 0.48

compared to 0.30 due both to winter overflow volumes and water withdrawals for consumption. The ELK watershed is slightly larger than SML, 7.82 km^2 verses 5.44 km^2. The drainage are to lake surface area ratios are 1:2.9 and 1:3.5 respectively. ELK has a mean volume of 18.8 x 10^6 m^3 and SML has a mean volume of 15.9 x 10^6 m^3. SML has a slightly faster flushing rate of 0.17 compared to 0.14 at ELK. According to the metrics established by Nurnberg (1996), SOL and SHL are oligotrophic, SML is mesotrophic, and ELK is eutrophic.

Methods

Water Quality and Climate Data

Water quality data were acquired from the BC Environmental Management System (BCEMS) database, the University of Victoria Water and Aquatic Sciences Research Lab (UVic), and the North Salt Spring Waterworks District (NSSWD). Sample dates and continuity of sample data vary for each lake. Data were analyzed for SOL from 2006-2013, for SHL from 1976-2014, for SML from 1979-2014, and for ELK from 1986-2015. Parameters include TP, TN, DO, Chl-a, algae, and water temperature. Table 3.2 summarizes the data sources used for the water quality analyses. Precipitation data were acquired from the Environment Canada Canadian Climate database (Government of Canada 2015), as summarized in Table 3.3.

Statistical Analyses

Microsoft Excel was used to perform linear regression on climate data and water quality data for the epilimnion (0-6 m) and hypolimnion (> 12 m) in all lakes. When sufficient data were available, analyses were performed separately by time segments (oldest-late 1970s) and more recent past (<10 years) to evaluate differences in trends. Normal probability plot outputs showed all data sets to be normally distributed. Multiple separate analyses were conducted to determine whether statistically significant relationships exist for the following when enough samples were available:

- Changes in TP and TN over time;
- Changes in precipitation over time;
- Relationships between TP and TN, precipitation, water temperature, and dissolved oxygen (DO);
- Relationships between TP, algae, and Chlorophyll-a;

Flushing Rates and Residence Times

Flushing rates were calculated using measured outflow in SOL (CRD 2015) and SML (NSSWD 2015). Dam overflow months were recorded by the CRD for SOL. The precipitation fraction in these months was used to weight each month's contribution to total overflow (Q). SOL withdrawal for consumption (W) were recorded monthly by the CRD. SML outflow volumes were recorded by the NSSWD monthly for both Q and W no estimations were necessary. The outflow volume from SHL was estimated using a lake level line graph produced by the CVRD for 2006-2009 estimates. The 2015 lake level line graph (Cowichan Valley Regional District, 2015) shows the outflow rate (L/s) that corresponds to the lake level, and this was used to estimate lake-outflow volume based upon lake levels in the 2006-07 and 2007-08 water years. Outflow volumes for 2010-11 through 2012-13 water-years were estimated by calculating the percent precipitation in those years verses 2006-2007 and applying that to the outflow. SHL outflow volume was weighted for winter months based upon known lake overflow months as documented for SOL. Outflows from ELK were estimated based on hydraulic modeling of daily outflows produced by MOE from data collected from May 2014 to

April 2015. Outflows for ELK in other years were estimated by determining the proportion of precipitation for the estimated year versus the measured/modeled year, and then applying the percentage (garnered from precipitation) to the measured outflow volume in order to estimate the outflow, based on precipitation, for each year where flows were estimated. This method was evaluated for accuracy using the measured SOL data (CRD 2015) and the estimated data for SOL presented by Nowlin et al. (2004). A paired t-test found no significant difference ($p = 0.096$) in individual estimates, and a comparison of means found no difference in the averages of 9 years of residence times.

Annual flushing rates for each lake were calculated by dividing the outflow volume by the entire lake volume. The result describes the percentage of water flowing out of the lake each year. The resulting residence time was calculated as 1/flushing rate, and provides an estimate of the number of years mass substances remain in the lakes.

Results

Results are limited by the frequency of sampling in ELK, and limited historical data for SOL were acquired. Results for the best available datasets are presented below.

Table 3.4 summarizes the results of the linear regression analyses as they relate to Figure 3.1, and as presented in Figures 3.2a-3.15b. Linear regression analyses conducted on in-lake nutrient data from the oligotrophic lakes (SOL and SHL) show either stable or declining TP or TN. Although cyclical fluctuations are evident, no long-term significant changes have occurred since 1976 in SHL (Figure 3.3a), and in-lake nutrient concentrations did not correlate significantly to precipitation. Results of linear regression analyses conducted for the meso-eutrophic lakes (SML and ELK) were more variable. Results showed no linear increase or decrease for SML TP in the winter between 2006 and 2014 (Figure 3.8a). Cyclical patterns are evident, which do not correspond to precipitation. SML summer TP concentrations declined in the hypolimnion but increased in the epilimnion (Figure 3.8b). Aerator operation in SML from 1987-1990 and 2009-2011 likely contributed to the low summer mean TP. Data for the ELK epilimnion show no change in Epi-TP since 1986. ELK Hypo-TP data show increasing TP; however, as shown in Figure 3.9b, the most recent sampling effort (2014-2015) resulted in Hypo-TP concentrations three times higher than previous samples, and this affected the linear regression outcomes.

In-Lake Phosphorus Trends Over Time

Trends over time are presented in one month increments in order to conserve the largest number of sample dates possible for each given data set. TP over time analyses that yielded a p-value of <0.05 indicate that TP concentrations have changed in the lake over time. Trend-lines with positive slope indicate increasing TP and trend-lines with negative slope indicate declining TP concentrations.

The linear regression results for winter (Nov.-Apr.) and summer (May-Oct.) Epi-TP and Hypo-TP over time are presented in Figures 3.2a-3.9b. In general, trends for all lakes showed a decrease in TP between the 1970's and early 1990's. TP trends for the most recent 10-15 years vary for each lake.

Phosphorus Trends in the Oligotrophic Lakes (SOL and SHL)

SOL data were evaluated for 2006-2013 Epi-TP. SOL Hypo-TP data were limited to 2006-2008. Linear regression analyses showed a decrease in both SOL Epi-TP and Hypo-TP in the winter months (Nov.-Apr.) (Figure 3.2a). Summer (May-Oct.) Epi-TP also decreased over time. Summer Hypo-TP data showed no change in TP concentrations in 2006-2008 (Figure 3.2b). The range of SOL Winter Epi-TP values decreased from 5.4-8.3 ug/L in 2006-07 to 2.1-4.8 ug/L in 2012-13. Low to high variability ranged from 1.2 ug/L in 2009-10 to 2.9 ug/L in 2007-08.

Long-term (1976-2014) winter and summer TP data for SHL showed no overall change in TP (Figures 3.3a, 3.3b). Time-segmented data illustrate a possible cycle of declining Hypo-TP from around 8 ug/L to 4-5 ug/L in 1976-1990, however limited sampling in the 1980's restricts the certainty of a declining trend. Winter Hypo-TP values for 2006-2014 indicate no significant changes, and the range of monthly values each winter all ranged between 5-9 ug/L. Both summer and winter Epi-TP values for 2006-2014 showed a significant decline over time (R^2=0.45; 0.27). Summer Hypo-TP also showed time-segmented trends of decline both for 1976-2003 and 2006-2013.

Nitrogen Trends in the Oligotrophic Lakes (SOL and SHL)

Epilimnic nitrogen trends from 2006-2013 were evaluated for the oligotrophic lakes. Figure 3.4 shows a decline in SOL Epi-TN in both winter and summer months. Figure 3.5 shows no significant change, but slight declining trends in SHL Epi-TN in both winter and summer months. SOL Epi-TN declined from an average of 125 ug/L to 60 ug/L in both winter and summer 2006-2013 (Figure 3.4). The variation in monthly sample results also decreased after 2009. SHL Epi-TN has remained stable with monthly samples generally ranging between 125 ug/L and 250 ug/L.

Trends in the Nitrogen to Phosphorus Ratios of the Oligotrophic Lakes (SOL and SHL)

Trends in the nitrogen to phosphorus ratio (N:P) of the oligotrophic lakes indicate whether or not the lakes are moving toward greater productivity, which would increase as the N:P approaches 15:1 or less (Downing, 1992). Figure 3.6 shows a significant increasing trend in SOL winter N:P, and no change in summer N:P. Figure 3.7 shows no significant change, but a slight increasing trend, in SHL winter N:P, and a significant increasing trend in summer N:P. Because the y-intercept values are greater than 15, and trend-lines indicate increasing N:P, a decline in in-lake productivity is currently occurring in both SOL and SHL. However, given the evident cyclical nature of the TP cycles the lakes could enter a cycle of increased productivity in future years.

Phosphorus Trends in the Meso-Eutrophic Lakes (SML and ELK)

No linear increases or decreases in TP concentrations are occurring in during the winter in SML (Figure 3.8a). Cyclical variation in TP concentrations are evident, but do not correlate with precipitation. A paired two-sample t-test for SML showed no significant difference between winter Epi-TP and winter Hypo-TP. Although linear regression shows a significant increase in Summer Epi-TP in SML (Figure 3.8b), the R^2 value is weak (R^2=0.08) and the statistical outputs are influenced by above average TP concentrations in 2012. The decline in SML summer Hypo-TP corresponds to the operation of the aeration system in the summers of 2009-2011. Despite the decline in summer Hypo-TP, the monthly variability in summer Epi-TP was greatest from 2011-2013, and this contributes to a statistically positive correlation, but does not indicate an overall increase in TP concentration over time. .

ELK data showed no statistically significant change in winter or summer Epi-TP between 1986 and 2015 (Figures 3.9a and 3.9b). ELK winter and summer Hypo-TP data for years 1986-2015 showed a significant increase; however, average winter and summer TP concentrations during the 2014-2015 sampling period are 6x greater in summer and 2x times greater in winter than the average recorded between 1986 and 2007. The high concentrations recorded for summer 2014 Hypo-TP are within the range of sediment pore water concentrations recorded in SHL, which has much lower TP in the water column. The implications of this are discussed in the Discussion section for ELK. A paired two-sample t-test for ELK confirmed that winter Epi-TP and winter Hypo-TP data were significantly different ($p = 0.006$) for years 2006-2015.

Nutrient to Precipitation Relationships in the Oligotrophic Lakes (SOL and SHL)

SOL TP data show no relationship to precipitation from 2006-2013. SHL data show a significant relationship between winter Epi-TP and precipitation only from 2006-2014. No relationship to precipitation is evident for winter and summer Epi-TN in SOL and SHL.

Total Phosphorus to Precipitation Relationships in the Meso-Eutrophic Lakes (SML and ELK)

Epi-TP to Precipitation Relationships

Winter 2005-2014 Epi-TP and Hypo-TP showed no statistical correlation to precipitation. Long-term data did show an overall significant relationship between winter Epi-TP and precipitation for 1974-2014. SML summer Epi-TP concentrations showed a significant negative relationship to monthly precipitation for years 1979-1991, and a positive relationship in the long-term (1980-2014); however, there was no statistically significant relationship for summer Epi-TP between 2005-2014.

ELK data showed no significant relationship between TP and precipitation in any season or date range.

Annual Precipitation Over Time

SOL and SHL climate data showed no significant change in annual precipitation over the past 10 years. Similarly, SML and ELK climate data showed no significant change in annual precipitation when data were analyzed for the past 20 years. SML data for 2004-2014 showed the only significant change in precipitation. Precipitation falling at SML have declined by 43.5 mm per year.

As demonstrated in Figure 3.10, annual precipitation exhibits variability over time, and it is not possible to know whether the current decline in precipitation at SML is a new long term trend or is merely a snapshot of a natural cycle in which precipitation will again increase in the coming years.

External Loading

If land-based loads are a main source of phosphorus to the lakes, then we would expect to see an increase in Epi-TP when the majority of precipitation falls between November and April. Figure 3.11 shows that, in fact, in 9 out of 10 years when water quality samples taken in November are compared to those taken the following April, the spring samples show significantly less P than is present in the fall. T-test statistics showed a definite difference ($p = 0.003$) between corresponding November and April Epi-TP concentrations in SML and ELK. It should be noted that Epi-TP concentrations typically do not increase incrementally over the winter months, and May/June concentrations can be much lower than those recorded in April. Use of only November and April TP values are not sufficient to determine winter loading.

It has been informally implied by others conducting research on SML that the November Epi-TP concentrations may reflect TP that will re-settle to the lake bottom, and thus does not provide an accurate comparison to April Epi-TP concentrations in regard to making conclusions about external loading. To further evaluate this, January Epi-TP concentrations in SML are also compared to April Epi-TP concentrations in

Figure 3.11. The results show that when January and April Epi-TP concentrations are compared, April Epi-TP is lower in 8 out of 14 years. However, elevated April Epi-TP concentrations may be due to winter algae abundance in at least some of these years as presented in Figure 3.14.

Internal Loading

The extent of internal loading has been evaluated based on the difference between early summer (May) and late summer (Oct.) TP concentrations (Nurnberg, 1984). Because internal loading typically increases toward the end of the summer, and since summer flows out of the lakes are minimal, comparing May to Oct. TP-concentrations are a good indicator of internal loading. SOL and SHL data indicate that limited internal loading does occur in some years, but unlike SML in which phosphorus is released from sediments, the differences between May and Oct. TP concentrations are likely the result of biological productivity (Figure 3.12). SML data indicate various proportions of TP in the hypolimnion between May and October, but clearly experiences internal loading in all years. ELK sampling was not frequent enough to evaluate internal loading in this manner.

Phosphorus Algae Relationships

Figure 3.13 shows the Epi-TP to algae relationship for SML, and the Epi-TP to Chlorophyll-a (Chl-a) relationship for SHL and ELK. Data were limited for SHL, but no significant relationship was determined for 2012-2014. There is a definite positive relationship between algae and Epi-TP in SML all year. There is also a definite positive relationship between Chl-a and Epi-TP during the summer in ELK[3]. Interestingly, Chl-a in SHL was typically higher during winter months with an average of 2.14 ug/L, verses

[3] ELK winter sampling was not frequent enough to include in this analysis.

1.76 ug/L in the summer. As demonstrated by Figure 3.14, in some years, algal abundance in SML is higher in December/January than April/May. Elevated winter algal biomass was also found during a study by Davies (2004) in ELK and other proximate BC lakes.

Total Phosphorus Related to Temperature and Dissolved Oxygen

Table 3.5 summarizes the statistical analyses conducted for TP, temperature, and DO. Data for SOL and ELK were not sufficient to perform these analyses. When the result returned 95% certainty ($p < 0.05$) a definite relationship is assumed and the percentage certainty is not noted in the table. A (+) indicates a positive correlation, and a (-) indicates a negative correlation. The SHL winter Hypo-TP/DO relationship is shown as significant at 85% certainty ($p = 0.15$) because the R^2 value is equal to the summer Hypo-TP R^2 value (0.22).

SHL summer Epi-TP exhibits some correlation to water temperature ($p=0.10$; $R^2=0.14$). SHL Hypo-TP exhibits a minor positive correlation to DO in the winter ($p=0.15$; $R^2=0.22$) indicating that oxygen available for in-lake productivity. A stronger but negative correlation in the summer ($p<0.05$; $R^2=0.22$) indicates that in-lake productivity contributes to a decline in DO in the summer.

SML data show significant negative correlations to DO, especially Hypo-DO in winter and summer 2006-2013. TP data show positive correlations to temperature in the winter, but not the summer, in both shallow and deep water samples.

Flushing Rates (all lakes) and Water Quality

Specific details on the derivation of flushing rates for each lake are provided in the Methods section. Flushing rates for SOL and SHL have been previously determined by calculating the difference between inflow volumes and the change in lake level and estimated evaporation (Nowlin, et al., 2004). This yielded much faster flushing rates, and shorter residence times than the estimates presented in Tables 3.6-3.9. However, because the focus of this research is the conservation of a substance (TP) in the lake water, it is logical that we would evaluate loss based only on ways in which water carries TP out of the lake. Evaporation would not remove TP from the lakes, rather, extensive evaporation during summer months could explain a small portion of the increase in TP evident in the TP concentration data for all lakes.

The flushing rates and resulting residence times calculated for SHL are much higher than the approximately 1 year residence time estimated previously. The results in Table 3.7 are, however, derived from the best available data as described in the Methods section. One problem with the outflow estimates is that no winter outflow volume data were available. Another is that statistical analyses conducted on lake level verses precipitation in SML concluded that there is no statistically significant relationship between these factors. This is likely because of the volume of water required to refill the lake after summer drawdown and the minimum downstream flow requirements which require water to be released from the lake even in the summer when precipitation is minimal. Based on previous estimates of residence times in both SOL and SHL which were both nearly 1 year, and based on the revised average SOL residence time presented in Table 3.6, it is likely that flushing rates in SHL are somewhat higher than those shown in Table 3.7, and the resulting average residence times is likely less than 3 years, or approximately 2.5 years.

Discussion

General overview and conclusions

Although the data trends indicate decadal cycles of both increase and decline, analyses of the data from the 1970's to 1990's compared to data for the past 10 years show that TP concentrations are neither higher nor lower in recent years compared to 35 years ago. The most recent 7-year dataset shows SOL TP concentrations and both SOL and SHL TN concentrations are declining over time, while the in-lake N:P ratios in both lakes are increasing. The TP concentrations in the SML hypolimnion may be artificially low and declining due to periodic operation of an aeration system since the ELK Hypo-TP concentrations are increasing with no aeration system in place. As explained in greater detail below TP and TN in SOL, and to some extent nutrients in SHL, are declining due to a cycle of naturally low in-lake productivity, and the loss of TN and TP through lake water outflow, which contributes to further reductions in in-lake productivity. Limited TP samples taken in ELK between 1986 and 2015 show that while Hypo-TP concentrations are increasing, no significant change in Epi-TP is evident.

The decreasing TP trends in the oligotrophic lakes indicate a decline in in-lake productivity rather than internal loading. As discussed above and explained more specifically below for SOL and SHL, the cycle of low productivity coupled with the loss of nitrogen and phosphorus via lake outflow contributed to a decline in lake productivity in the oligotrophic lakes. The internal dynamics of SML and ELK are more complex because they are naturally more productive due to long residence times (Brett and Benjamin 2008; Saunders and Kalff 2001), and experience significant internal loading as evidenced by the variation in May verses October Hypo-TP. As a result, the causal loads in these lakes vary by season between internal loading in the summer and the "ambient" load perpetuated by nearly year-round biological productivity (e.g., algae blooms).

Oligotrophic Lakes: Sooke (SOL) and Shawnigan (SHL)

TP trends in the oligotrophic lakes demonstrate an overall decline, with some seasonal steady-state ("ambient") conditions in TP concentrations. Relative TP and TN concentrations are discussed for SOL and SHL because the N:P ratio is the underlying factor contributing to growth rates of all aquatic organisms. The ratio of TN to TP determines whether increases in nitrogen or phosphorus are responsible for increased biological productivity as the nutrient concentrations in the water approach the ratio of optimal growth for algae which generally ranges between 15:1 and 20:1 TN:TP (Knowles, 1982). A low nitrogen to phosphorus ratio (~10:1) is the most significant factor driving the growth rate of nitrogen-fixing algae (Downing, 1992) which transform molecular nitrogen (N_2) into biologically available nitrogen (typically nitrate NO^3- or ammonium NH_4) when there is a shortage of biologically available nitrogen from other sources.

SOL Phosphorus and Nitrogen Trends: Implications for Biological Productivity

Figure 3.15 summarizes the results of the TP analyses conducted for SOL based on Figure 3.1 and the results presented in Table 3.4 and Figures 3.2a and 3.2b,. Results for SOL indicate that winter and summer Epi-TP and winter Hypo-TP declined significantly between 2006 and 2013 (Figures 3.2a, 3.2b). Summer Hypo-TP did not change between 2006 and 2008. Epi-TN also decreased in winter and summer between 2006 and 2013 (Figure 3.4). TP and TN did not correlate to precipitation in any strata or season. The decline in lake nutrient concentrations, with no correlation to precipitation, and minimal internal loading, indicates that a reduction in biological productivity is occurring. This is supported by the increasing N:P ratio over the same time period (Figure 3.6), which is trending from near 15:1 to near 25:1, or away from the optimal ratio for biological productivity. The y-intercept values for winter Epi-TP ($y=-0.03x+5.8$) and winter Epi-TN ($y=-0.57x+121$) indicate a mean ratio of 21:1 where both TP and TN are each declining at a relative rate of 0.5% per month. The decline is less significant for TN due to the much higher concentrations of TN relative to TP, and the result is a higher N:P ratio initiating a decline in biological productivity.

Long-term water quality records were not evaluated for SOL; however, it is likely that the snapshot of time between 2006 and 2013 is evidence that the watershed is still recovering from forestry operations (land clearing) that began in 1915, and occurred intensively between 1954 and 1997 when over 26 km^2 were continuously logged and replanted (Smiley, et al., 2014). The past commercial forestry operations (land clearing) in the SOL watershed likely resulted in external loading of nitrogen and phosphorus into the lake (Zhu, 2005). Since forestry operations were discontinued around 1997, the growth of young recovering forest has reduced the overland transport of nutrients by stabilizing the soil, and by utilizing soil nutrients for rapid growth (Zhu, 2008). The declining TN and TP concentrations from 2006 to 2013 are not correlated to a reduction in external loading (via a precipitation relationship) because after 10 years (more in some areas) of forest growth, land-based nutrient transport has been reduced such that the decline in TN and TP concentrations in the lake is due to annual nutrient losses via TP sequestration by lake sediments, dam overflow, and water consumption. The loss of nitrogen and phosphorus via lake outflow is significantly greater than the gain from overland flows and as a result nutrient concentrations are declining, the N:P ratio is increasing, and SOL is approaching a less productive stable-state.

Since summer outflows are much lower than in the winter, the decline in summer Epi-TP can be more strongly attributed to the decline in N:P ratio which facilitates a decline in lake productivity, as opposed to the mass outflow that occurs in the winter. Data for summer Hypo-TP were limited to 2006-2008 and no change in TP is apparent. Mass loss through outflows and lake productivity will be more evident in the epilimnion where the majority of water movement occurs and where light availability and summer temperature is higher. The hypolimnion thus maintains relatively stable nutrient concentrations.

SHL Phosphorus and Nitrogen Trends: Implications for Biological Productivity

Figure 3.16 summarizes the results of the analyses conducted for SHL based on Figure 3.1 and the results presented in Table 3.4 and Figures 3.3a and 3.3b.

Past Trends 1976-2003

SHL results show no change in Epi-TP between 1976 and 2002 (Figures 3.3a and 3.3b). Winter Epi-TP did not correlate to precipitation, but oddly summer Epi-TP did. The reason for this is unclear, but the overall indication is that the lake maintained stable-state TP concentrations due to a balance of in-lake productivity, as well as relatively equal external loading and outflow mass. Hypo-TP decreased during the same time period (Figures 3.3a and 3.3b), and was not precipitation dependent. This indicates a decline in biological productivity, possibly due to an increase in the N:P ratio at that time.

Current Trends 2006-2014

SHL results for 2006-2014 data showed a decline in Epi-TP (Figures 3.3a and 3.3b). Winter Epi-TP was precipitation dependent, and the decreasing TP concentrations are likely due to improved forestry management practices, a reduction in the frequency and/or area of land clearing, and possibly the result of riparian restoration conducted in the watershed. Summer Epi-TP was not precipitation dependent, and likely relates to the increased N:P ratio (Figure 3.7) which, as described for SOL, leads to lower in-lake productivity. The same was true for summer Hypo-TP which also correlates with DO (Table 3.5). Winter Hypo-TP (Figure 3.3a), and Epi-TN (Figure 3.5) showed no change and are not precipitation dependent. Although winter Hypo-TP also correlated with DO. These factors are additional evidence to support recurring stable-state biological productivity. Like SOL, the decline of biological productivity in the summer is also supported by evidence that the N:P ratio is increasing (Figure 3.7). Summer N:P in SHL is trending from 25:1 to over 35:1, and is moving further from the optimal ratio for biological productivity (15:1).

Long-Term Trends and Land Use

Extensive land-clearing both for commercial forestry and for residential development has occurred in the SHL watershed more intensively in the past. Although forestry practice is ongoing in a portion of the watershed, 44% of the watershed currently has at least 75% forest cover, and open lands (37%) are also maintained with vegetative cover. TP trends for the past 40 years show no change in TP concentrations. Land use activities have not contributed to increased nutrient concentrations in SHL for at least the past 40 years. Improved land management (and possibly septic system) regulations and practices may be contributing to the recent declines in Epi-TP, and as long as the lake maintains a relatively short average residence time, the increasing N:P will continue to limit biological productivity in the future.

Mesotrophic and Eutrophic Lakes: St. Mary and Elk

Figure 3.17 summarizes the results of the analyses conducted for SML based on Figure 3.1 and the results presented in Table 3.4 and Figures 3.8a and 3.8b. Additional discussion on SML follows Figure 3.17. Figure 3.18 summarizes the results of the analyses conducted for ELK based on Figure 3.1 and the results presented in Table 3.4 and Figures 3.9a and 3.9b.

SML Phosphorus Trends and Precipitation Relationships

No significant change in winter Epi-TP occurred from 2005-2014 (Figure 3.8a), and winter Epi-TP was not precipitation dependent ($p=0.134$, $R^2=0.04$).

Figure 3.8a (panel iii) shows mean winter Epi-TP and total winter precipitation (rain). A “warm” PDO phase was in effect from 1977 until 1999, after which the “cool” phase began. During the “warm” phase temperature and precipitation in BC would have been lower than normal. A four-year ENSO occurred in the early 1990’s (Environment and Climate Change Canada 2015) which would have had a warming influence. With the exception of 1990, precipitation and mean winter Epi-TP may have been influenced by the PDO as both were lower than in recent years. Warmer temperatures and higher rainfall are expected during the current PDO phase which began after 1999. Mean winter Epi-TP and precipitation were similar to 1988-1991 levels between 2007 and 2010, so no PDO influence is evident. ENSO occurred in spring 2005, 2006-2007 and 2009-2010 (National Weather Service 2015), and higher precipitation would have been expected during those years, but does not appear to be substantially different. La Nina oscillations were prevalent from 2007 to mid-2008, and from July 2010 to April 2012 (National Weather Service 2015), and would have resulted in cooler drier winters, but neither total winter precipitation nor mean winter Epi-TP concentrations reflect this weather pattern. Overall, there does not seem to be evidence that the PDO, ENSO, or La Nina weather patterns account for variation in winter precipitation or Epi-TP concentrations.

The reduction in summer Hypo-TP that has occurred since 2009 can be attributed to the upgrade and operation of the aerators (Figure 3.8b). Although the aerators were only operational during the summers of 2009-2011, DO in the hypolimnion was increased, and as a result, much less phosphorus was released from the lake sediments. This is also evident in TP concentrations from 1987-1990 when the previous system was operational (Figure 3.8b). The effects of aeration appear to persist for several years after operation as shown by a minimal increase in summer Hypo-TP from 2012-2014. However, it is likely that Hypo-TP concentrations would return to 2005-2006 levels or higher if the use of the system is permanently discontinued.

SML Persistent "Ambient" Load (2005-2014 Winter Epi- and Hypo-TP/Summer Epi-TP)

If external loading were the main factor currently influencing TP concentrations in SML, we would expect to find a significant relationship to precipitation, and would expect that Epi-TP concentrations measured in April would be greater than those measured the previous November. Figure 3.8a shows no TP to precipitation relationship for 2006-2014 winter Epi-TP or Hypo-TP. Figure 3.11 shows that Epi-TP concentrations in were higher in November than in April in 9 out of 10 years. In the event that the Epi-

TP concentrations measured in November still contain residual summer TP that will re-settle to the lake-bottom, comparisons of January and April Epi-TP are also presented in Figure 3.11, and January Epi-TP concentrations are higher than those measured in April in 8 out of 14 years. Although this could indicate that the lake receives some excess phosphorus from external sources in some years, Figure 3.14 shows that in each year when April Epi-TP was higher than November Epi-TP, and corresponding data on algae abundance are available (i.e., 2006, 2009, 2011-2013), algae abundance was also higher in April than in November. This directly supports the hypothesis that the persistent "ambient" or steady-state load evident in 2005-2014 winter TP data is due to the sequestration of phosphorus by a persistent algae population suspended in the water column, and not external loading. This is supported by Brunberg (1995) who found that benthic bacteria and cyanobacteria contribute to sediment to water column phosphorus cycles through productivity in which they sequester and release phosphorus.

As shown on Figure 3.13, there is a statistically significant ($p<<0.001$, $R^2=0.26$) algae to phosphorus relationship in SML. This was evaluated for all winter and summer months (combined for analyses) for which algae and Epi-TP data were both available. Variability in lake flushing rates between 3 and 9 years (based on lake outflows; Table 3.7) result in fluctuations in nutrient retention from year to year (Brett & Benjamin, 2008) (Saunders & Kalff, 2001). In addition, the growth of *Anabaena* and *Aphanizomenon* have been very positively correlated with both water residence time and increases in lake water temperature (Davis, et al., 2009) (Elliott, 2010) (Romo, 2013). In years when outflow volume is greater (i.e., high flushing rate/low residence time) it is expected that algae abundance is lower, and this is demonstrated by the average annual algae concentration (cells/mL) presented in Table 3.8. Winter Epi-TP and Hypo-TP are both positively correlated with temperature and negatively correlated with DO (Table 3.5). Warm water temperatures (>4^{o}C above ambient) promote biological productivity (Davis, et al., 2009), and low DO indicates use of oxygen by organisms. So the statistical correlations summarized in Table 3.5 further support the idea that algae are sequestering phosphorus in the water column in the winter.

SML Internal Loading (2005-2014 Summer Hypo-TP)

As described previously, internal loading of phosphorus is the result of the chemical precipitation of Fe-II to Fe-III in lake sediments which releases phosphorus (PO^{4}-) into the water column, and this process occurs more readily under low DO conditions (Carey & Rydin, 2011). The analyses provided in Figure 3.8b demonstrate the effect of hypolimnic aeration on summer deep water TP concentrations. The SML aeration system was operational in the summers of 1987-1990 and 2009-2011, when an equipment malfunction in 2012 caused increase turbidity and high sulfur content in the water, and operations were subsequently discontinued (Squires, 2015). It is clear, however, that increasing DO levels via aeration resulted in much lower average TP concentrations in the summer hypolimnion.

Internal loading in the summer between 2006 and 2014 is also evident in the May to October comparison presented in Figure 3.12. TP data were not available for the summers of 2009-2010, but unlike any other year, 2011 and 2012 data show negative internal loading occurred between May and October. This fact provides additional support for the effectiveness of hypolimnic aeration when functioning properly. Even with the inclusion of summers that the aeration system was operated, summer Epi-TP and

Hypo-TP show a significant negative correlation ($p<0.05$; $R^2=0.48, 0.30$) with DO at 12m depth (Table 3.5).

ELK Phosphorus Trends and Precipitation Relationships

Results for ELK indicate that Epi-TP concentrations have not changed significantly since 1986 (Figures 3.9a and 3.9b) and are not precipitation dependent (Figures 3.15a and 3.15b). This indicates external loading is not contributing significantly to TP concentrations in the lake. Instead, the ELK epilimnion is maintaining recurring steady-state or "ambient" TP concentrations. Hypo-TP data show an increase in both winter and summer between 1986 and 2015 (Figures 3.9a and 3.9b), with no precipitation dependence (Figures 3.15a and 3.15b), and this indicates that the primary causal load is internal loading from lake sediments. It is important to note, however, that Hypo-TP data showed no significant change in TP concentrations when based only on 1987-2007 data records. Winter Hypo-TP concentrations recorded in 2014-2015 were twice the average of the 1987-2007 data set. Sample results recorded from August to October 2014 were 2 to 3 times higher than the maximum recorded during the summers of 1987-2007, and twice the maximum recorded at SML. This sharp increase is evident in the line graph at the bottom of Figure 3.9b. There is the possibility that increased turbidity at the lake bottom may be occurring due to the presence of benthic fish, resulting in excess lake sediment particles being included in deep water samples. Lake sediment sample data from SHL show TP concentrations as high as 1300-3500 ug/g (BC Ministry of Environment, 2015), and given that the August to October 2014 Hypo-TP samples from ELK ranged from 1190 to 1490 ug/L, these samples may have in fact been acquired from a turbid zone above the sediment-water interface. Future data collection is needed to confirm the high Hypo-TP concentrations, and total dissolved phosphorus should be measured rather than TP.

Similar to SML, the presence of persistent ambient TP concentrations in the epilimnion is supported by the significant (p=0.03, R^2=0.32) Epi-TP to Chlorophyll-a relationship (Figure 3.13). Algae blooms have been observed in ELK in January and February since 2011 (Nordin, 2015). As described in more detail for SML, it is likely that that algae are sequestering phosphorus in the ELK water column in the winter.

Conclusion

All of the lakes are generally maintaining their natural steady-state productivity. No shift in productivity (e.g. from oligotrophic to eutrophic) has occurred in the recent past.

Lake and watershed management in SOL and SHL is currently succeeding in reducing TP and TN concentrations, and as a result increasing in-lake N:P ratio. With current residence times, increased biological productivity is not likely to occur in these lakes. Seasonal increases in both TP concentration and sample variation identified in SML (and ELK) cannot be explained by Pacific Ocean weather patterns or TP-precipitation relationships. The increased can be attributed to the persistence of a year-round algae population which may have been dormant during the winter in the past. There is some evidence of extensive internal loading in ELK, but given the magnitude of the increase, this should be verified through the acquisition of several more years of dissolved phosphorus (not total phosphorus) data. There is also evidence that the aeration system in SML is contributing to lower summer Hypo-TP concentrations than would be occurring if the aerators had not been operating properly up to 2011. Unlike that of oligotrophic lakes, the sediment in eutrophic lakes has very limited phosphorus sequestration capacity, and it is this factor, rather than increased external loading, that determines the availability of phosphorus to cycle back to the water column (internal cycling) (Carey & Rydin, 2011), and controls the TP concentrations in the lake water.

Continuous long-term water quality sampling is essential to the evaluation of lake health. Linear and cyclical trends cannot be determined if there are long gaps between sample occurrences. Monthly samples are sufficient for evaluating long-term trends (>10 years), but more frequent sampling is needed to detect inter-annual and seasonal variation. Frequent sampling should focus on the most severe annual conditions such as periods of summer lake draw-down, fall turnover, and the first extreme rain events of the fall/winter, in addition to year-round monthly sampling. This will give a better understanding of lake nutrient regimes and regime-shifts that last more than a year or two can be evaluated to determine the best lake or watershed management strategy.

Tables

Table 3.1. Summary of Sooke (SOL), Shawnigan (SHL), St. Mary (SML) lake and watershed characteristics. The drainage area (km2) is the contributing area (watershed). The Drainage to Surface Ratio is the proportion of Lake Surface Area (km^2) to the contributing area. Maximum and mean lake depth is shown in meters (m), and lake volume in million cubic meters (x106 m3). Flushing rate is the percentage of outflow/lake volume per year, and Residence time = 1/flushing rate. Annual average total phosphorus (TP) concentration is shown in ug/L. Lake trophic status is oligotrophic (O), mesotrophic (M), or eutrophic (E).

Lake	Drainage Area (km^2)	Lake Surface Area (km^2)	Drainage to Surface Ratio	Max Depth (m)	Mean Depth (m)	Lake Volume ($x10^6 m^3$)	Flushing Rate ($year^{-1}$)	Residence Time (years)	Annual Avg. TP ug/L	Trophic Status[4]
SOL	65.14	5.90	1:11	61	19.5	160.3	0.48	2.2	4.5	O
SHL	65.08	5.50	1:12	52	13.0	71.9	0.30	2.5-3.4*	6.4	O
SML	5.44	1.82	1:2.9	17	8.0	15.9	0.17	6.8	36	M
ELK	7.82	2.24	1:3.5	19	7.7	18.8	0.14	7.3	73	E

*The residence time for SHL has previously been calculated to be approximately 1 year; however, based on changes in lake level (Cowichan Valley Regional District, 2015), and outflows measured in 2000-01 (Nowlin, et al., 2004), residence time is greater than previously estimated. The 3-year residence time, derived from calculations and shown in Table 7, is likely due to an underestimation of winter outflows from available data, and the residence time is probably slightly higher than SOL (~2.5 years).

[4] Studies on lake productivity (i.e., trophic status) categorize oligotrophic (O) or low nutrient lakes as those with average TP concentrations of <10 ug/L, mesotrophic (M) lakes as those with average TP concentration of 10-30 ug/L, and eutrophic (E) lakes as those with average TP concentrations between 24-75 ug/L (Nurnberg 1996).

Table 3.2. Summary of sources of data. Data were acquired from the University of Victoria Aquatic Sciences Research Lab (UVic), the BC Ministry of Environment Environmental Monitoring System (BCEMS), and the North Salt Spring Water District (NSSWD) for the sample points at Sooke (SOL), Shawnigan (SHL), St. Mary (SML), and Elk (ELK) lakes. The latitude (Lat.) and longitude (Long.) in decimal degrees, and years of available data are shown for total phosphorus (TP), total nitrogen (TN), temperature (Temp.), dissolved oxygen (DO), algae, and Chlorophyll-a (Chl-a).

Lake	Source	Sample Point	Lat.	Long.	Years	Parameters
SOL	UVic	SOL-04	48.57	-123.69	2006-2013	TP, TN
SHL	BCEMS	#1199901	48.64	-123.63	1976-82, 1984, 1986-2003	TP, Temp., DO, Chl-a
	UVic	SHL-01[5]	48.64	-123.63	2006-2014	TP, TN
SML	NSSWD/ BCEMS	#1100104	48.89	-123.54	1979-81, 1986-91, 2005-14	TP, Temp., DO, algae
ELK	BCEMS	#1100844	48.53	-123.40	1986, 1988-95, 1997, 1999-2003, 2005-06, 2008, 2013-15	TP, Temp., DO, Chl-a

Table 3.3. Summary of climate stations used for precipitation data. Government of Canada climate station numbers from which precipitation (rain) data were acquired are shown for Sooke (SOL), Shawnigan (SHL), St. Mary (SML), and Elk (ELK) lakes. The latitude (Lat.) and longitude (Long.) in decimal degrees and elevation in meters of the climate stations are shown.

Lake	Station #	Station Name	Latitude	Longitude	Elev. (m)
SOL	1017563	SOL NORTH	48.58	-123.64	231
SHL	1017230	SHL	48.65	-123.63	159
SML	1016995	SALTSPRING ST MARY'S L	48.89	-123.55	45.7
ELK	1018620	VICTORIA INT'L A	48.65	-123.43	19.5

[5] BCEMS point and UVic point are in the same location.

Table 3.4. Summary of linear regression results for Sooke (SOL), Shawnigan (SHL), St. Mary (SML), and Elk (ELK) total phosphorus (TP) and total nitrogen (TN) trends. The change in TP is shown as decreasing or no change along with the p-value, R^2 value, and linear equation produced by the statistical analysis. For precipitation, linear regression analyses resulted in $p>0.05$ when TP or TN were not correlated to precipitation (N), and $p<0.05$ when TP or TN were correlated to precipitation (Y). The "Causal Load" is as described in the text.

Lake	Sample Years	Metric	Season	Change in TP over Time				Precipitation		Causal Load
				TP Status	P-Value	R^2	Linear Equation	Dependent?	R^2	
SOL	2006-13	Epi-TP	Winter	Decrease	p << 0.001	0.42	y=-0.03x+6	N	0.00	in-lake
			Summer	Decrease	p << 0.001	0.38	y=-0.03x+5	N	0.00	in-lake
	2006-08	Hypo-TP	Winter	Decrease	p = 0.041	0.31	y=-0.08x+6	N	0.00	in-lake
			Summer	No change	p = 0.445	0.04	y=-0.02x+4	N	0.03	ambient
	2006-13	Epi-TN	Winter	Decrease	p <<0.001	0.40	y=-0.6x+121	N	0.00	in-lake
			Summer	Decrease	p <<0.001	0.52	y=-0.6x+113	N	0.00	in-lake
SHL	1976-90	Epi-TP	Winter	No change	p = 0.159	0.14	y=-0.01x+6	N	0.03	ambient
	1976-02		Summer	No change	p = 0.306	0.00	y=-0.002x+6	Y	0.25	= in/out
	1976-90	Hypo-TP	Winter	Decrease	p = 0.008	0.39	y=-0.02x+8	N	0.03	in-lake
	1976-03		Summer	Decrease	p = 0.001	0.42	y=-0.007x+7	N	0.00	in-lake
	2006-14	Epi-TP	Winter	Decrease	p <<0.001	0.45	y=-0.04x+8	Y	0.15	external
			Summer	Decrease	p = 0.003	0.27	y=-0.08x+9	N	0.03	in-lake
		Hypo-TP	Winter	No change	p = 0.191	0.05	y=-0.01x+7	N	0.00	ambient
			Summer	Decrease	p = 0.004	0.23	y=-0.04x+7	N	0.01	in-lake
	2006-13	Epi-TN	Winter	No change	p = 0.153	0.06	y=-0.4x+226	N	0.08	ambient
			Summer	No change	p = 0.145	0.06	y=-0.3x+193	N	0.01	ambient
SML	2005-14	Epi-TP	Winter	No change	p = 0.711	0.00	y=0.03x+33	N	0.04	ambient
			Summer	Increase	p = 0.035	0.08	y=0.14x+14	N	0.01	internal
	2006-14	Hypo-TP	Winter	No change	p = 0.130	0.05	y=0.13x+26	N	0.00	ambient
			Summer	Decrease	p = 0.001	0.17	y=-1.7x+222	N	0.00	internal
ELK	1986-15	Epi-TP	Winter	No change	p = 0.117	0.09	y=0.03x+18	N	0.07	ambient
	1987-14		Summer	No change	p = 0.311	0.06	y=-0.01x+16	N	0.00	ambient
	1986-15	Hypo-TP	Winter	Increase	p < 0.001	0.51	y=0.08x+14	N	0.14	internal
	1987-14		Summer	Increase	p = 0.004	0.42	y=2x+47	N	0.12	internal

Table 3.5. Summary of linear correlation results for total phosphorus verses temperature (Temp.) and dissolved oxygen (DO) in Shawnigan (SHL), St. Mary (SML), and Elk (ELK) lakes. The first four columns are taken from Table 3.4. Linear correlation analyses resulted in $p>0.05$ when TP was not correlated to temperature or DO (N), and $p<0.05$ when TP was correlated to temperature or DO (Y).

Lake	Sample Years	Metric	Season	Precip. Dep.?	Causal Load	Temp. Dep?	DO Dep.?
SHL	1976-13	Epi-TP	Winter	N	ambient/ external	N	N
			Summer	N	in-lake	Y (90%)(-) R^2=0.14	N
		Hypo-TP	Winter	N	in-lake/ ambient	N	Y (85%)(+) R^2=0.22
			Summer	N	in-lake	N	Y (-) R^2=0.22
SML	2006-14	Epi-TP	Winter	N	ambient	Y (+) R^2=0.25	Y (-) R^2=0.48
			Summer	N	internal	N	Y (-) (w/12m) R^2=0.19
		Hypo-TP	Winter	N	ambient	Y (+) R^2=0.22	Y (-) R^2=0.30
			Summer	N	internal	N	Y (-) R^2=0.21
ELK	1986-15	Epi-TP	Winter	N	ambient	N (-)	N (+)
	1987-14		Summer	N	ambient	UNK	UNK
	1986-15	Hypo-TP	Winter	N	internal	UNK	UNK
	1987-14		Summer	N	internal	UNK	UNK

Table 3.6. Sooke Lake water balance. Flushing rate is the percentage of outflow/lake volume per year, and Residence time = 1/flushing rate. Outflow includes both spill-over and water withdrawals for consumption. Water volume is expressed in cubic decameters (dam^3).

Hydro Year	Precip. (mm)	Spill over (dam^3)	Withdraw (dam^3)	Lake Volume (dam^3)	Flushing Rate ($year^{-1}$)	Res. Time (years)
2004-05	1397	0	55,848	160,320	0.35	2.87
2005-06	1433	17,118	57,362		0.46	2.15
2006-07	1842	61,445	55,242		0.73	1.37
2007-08	1133	12,950	53,718		0.42	2.40
2008-09	867	0	54,294		0.34	2.95
2009-10	1520	29,382	51,564		0.50	1.98
2010-11	1306	48,300	48,532		0.60	1.66
2011-12	1133	36,923	47,912		0.53	1.89
2012-13	1110	33,041	48,694		0.51	1.96
2013-14	993	15,882	46,315		0.39	2.58
				Average	**0.48**	**2.18**

Table 3.7. Shawnigan Lake water balance. Flushing rate is the percentage of outflow/lake volume per year, and Residence time = 1/flushing rate. Average withdrawal was calculated from changes in the lake level and represents outflow for the flushing rate calculation. Water volume is expressed in cubic decameters (dam^3).

Hydro Year	Precip. (mm)	Average Withdrawal (dam^3)	Lake Volume (dam^3)	Flushing Rate ($year^{-1}$)	Res. Time (years)
2006-07	1423	24,322	71,900	0.34	2.96
2007-08	993	20,269		0.28	3.55
2008-09	669	19,038		0.26	3.78
2010-11	1280	23,471		0.33	3.06
2011-12	1100	22,483		0.31	3.20
2012-13	1110	19,701		0.27	3.65
			Average	**0.30**	**3.37**

Table 3.8. St. Mary Lake water balance. Flushing rate is the percentage of outflow/lake volume per year, and Residence time = 1/flushing rate. Outflow includes both water withdrawals for consumption and the volume of water that leaves the lake via Duck Creek. Water volume is expressed in cubic decameters (dam^3). Average algae cells (cells/mL) are shown as algal abundance has been linked to lake residence time.

Hydro Year	Precip. (mm)	Duck Creek Out (dam^3)	Withdraw (dam^3)	Lake Volume (dam^3)	Flush Rate ($year^{-1}$)	Res. Time (years)	Avg. Algae (cells/mL)
2007-08	798	1,348	570	15,900	0.12	8.29	ND
2008-09	681	1,132	606		0.11	9.15	2,032
2009-10	1,138	4,368	608		0.31	3.20	1,983
2010-11	990	2,779	595		0.21	4.71	1,949
2011-12	794	1,042	588		0.10	9.76	3,581
2012-13	1,004	2,493	583		0.19	5.17	ND
2013-14	806	1,564	579		0.13	7.42	ND
				Average	**0.17**	**6.81**	

Table 3.9. Elk Lake water balance. Flushing rate is the percentage of outflow/lake volume per year, and Residence time = 1/flushing rate. Water flowing out of the lake via the Colquitz River is the outflow used to calculate the flushing rate. Water volume is expressed in cubic decameters (dam^3).

Hydro Year	Precip. (mm)	Colquitz River Out (dam^3)	Lake Volume (dam^3)	Flush Rate ($year^{-1}$)	Res. Time (years)
2005-06	805	2,494	18,900	0.13	7.58
2006-07	1,100	3,408		0.18	5.55
2007-08	793	2,457		0.13	7.69
2008-09	653	2,024		0.11	9.34
2009-10	991	3,072		0.16	6.15
2010-11	1,015	3,147		0.17	6.01
2011-12	736	2,282		0.12	8.28
2012-13	913	2,830		0.15	6.68
2013-14	758	2,348		0.12	8.05
2014-15	817	2,533		0.13	7.46
			Average	**0.14**	**7.28**

Figures

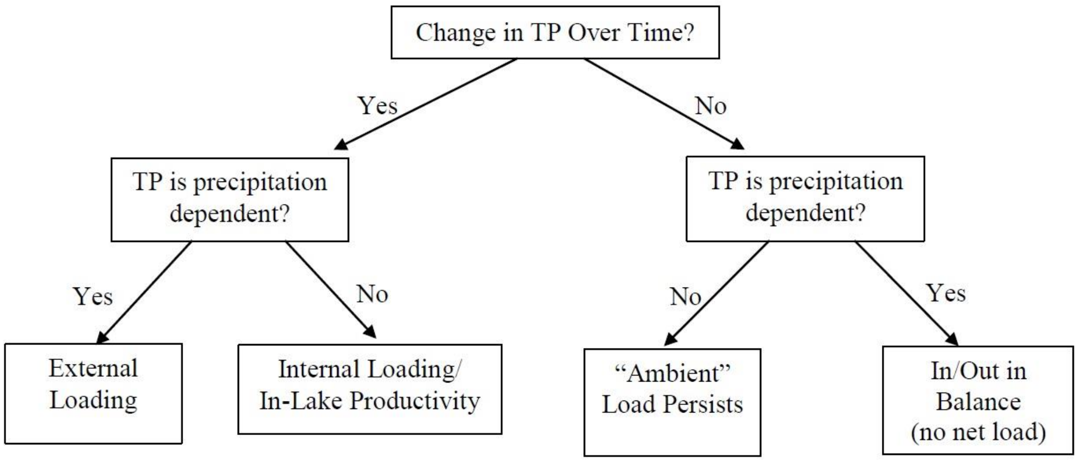


Figure 3.1. Causal load flow chart. Primary cause of total phosphorus (TP) loads in SHL are shown based on the flow chart.

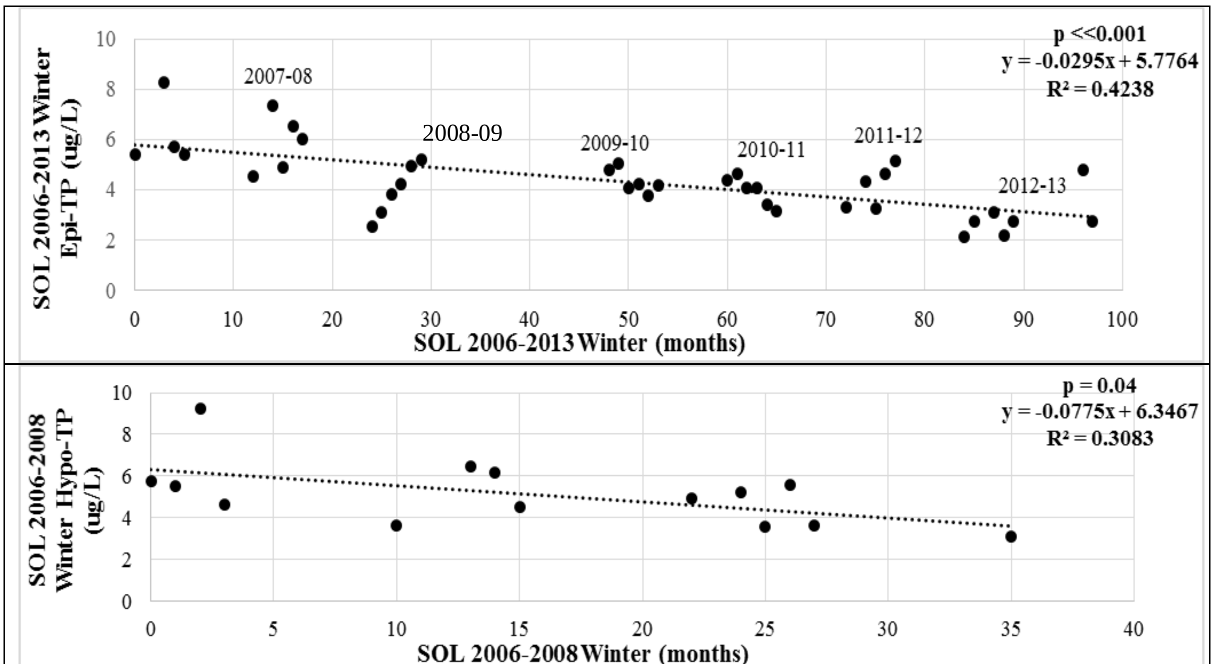


Figure 3.2a. Sooke Lake (SOL) winter (Nov.-Apr.) total phosphorus (TP) trends. Winter TP trends in the epilimnion (Epi-TP) for years 2006-2013 (upper) and the hypolimnion (Hypo-TP) for years 2006-2008 (lower).

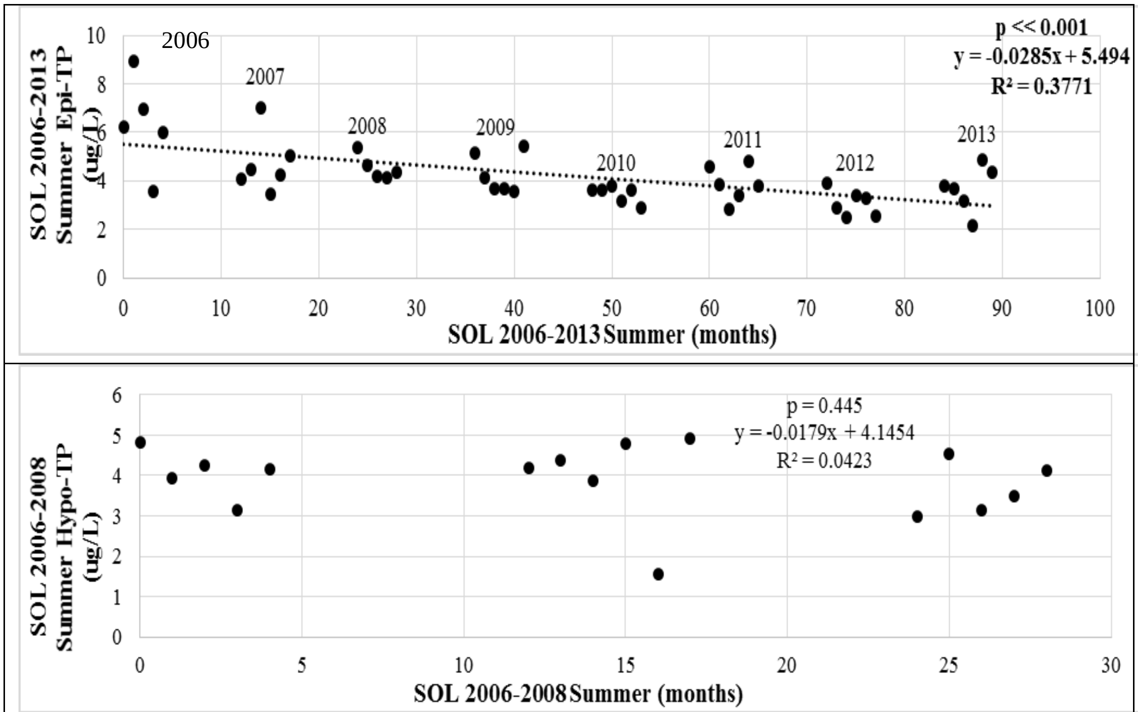


Figure 3.2b. Sooke Lake (SOL) summer (May-Oct.) total phosphorus (TP) trends. Summer TP trends in the epilimnion (Epi-TP) for years 2006-2013 (upper) and the hypolimnion (Hypo-TP) for years 2006-2008 (lower).

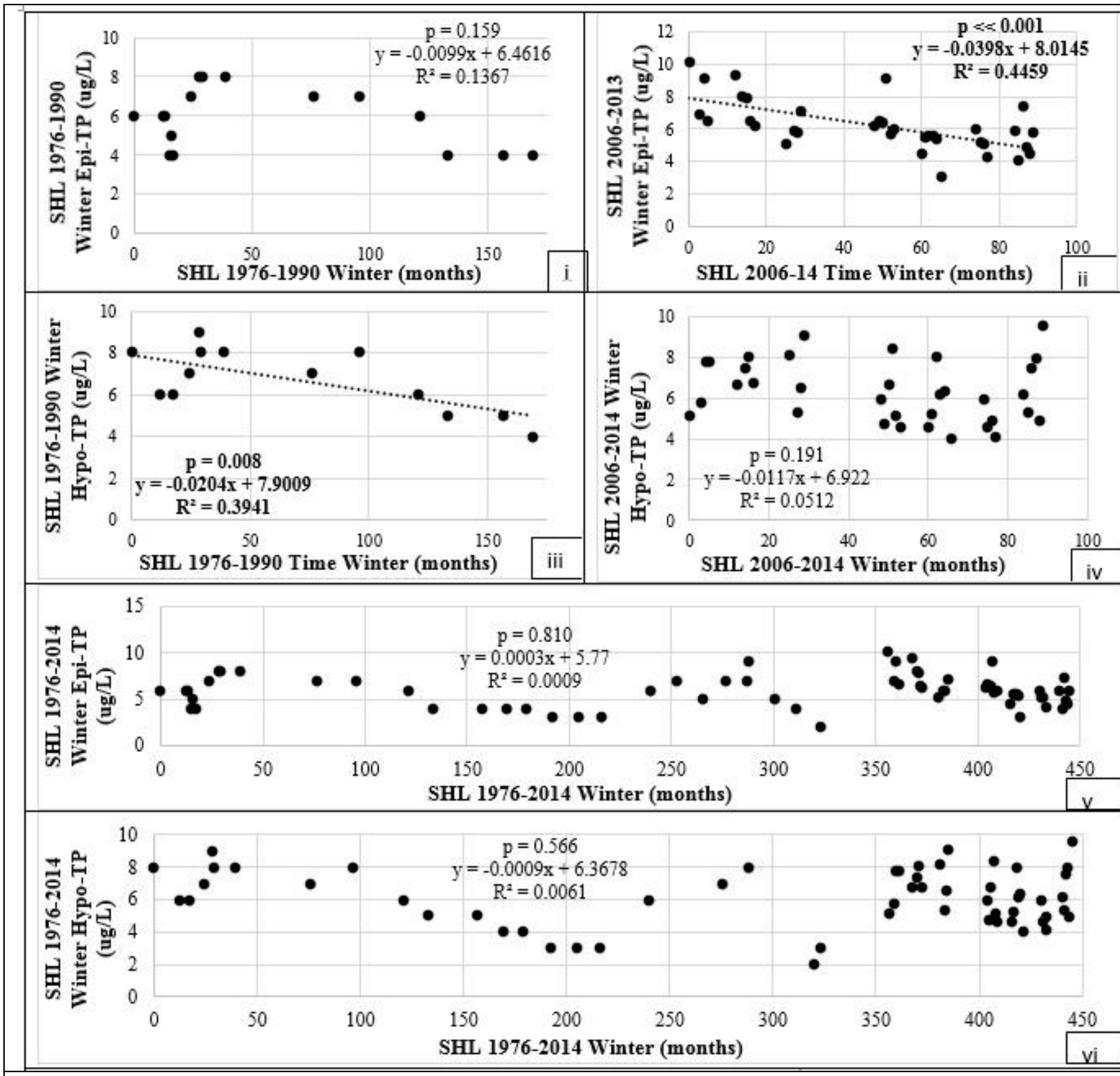


Figure 3.3a. Shawnigan Lake (SHL) winter (Nov.-Apr.) total phosphorus (TP) trends. Winter TP trends in the epilimnion (Epi-TP) and hypolimnion (Hypo-TP) for years 1976-1990 (i) and (iii) and 2006-2014 (ii) and (iv). The past 38 years of winter TP sampling showed no change linear trend for Epi-TP (v) or Hypo-TP (vi).

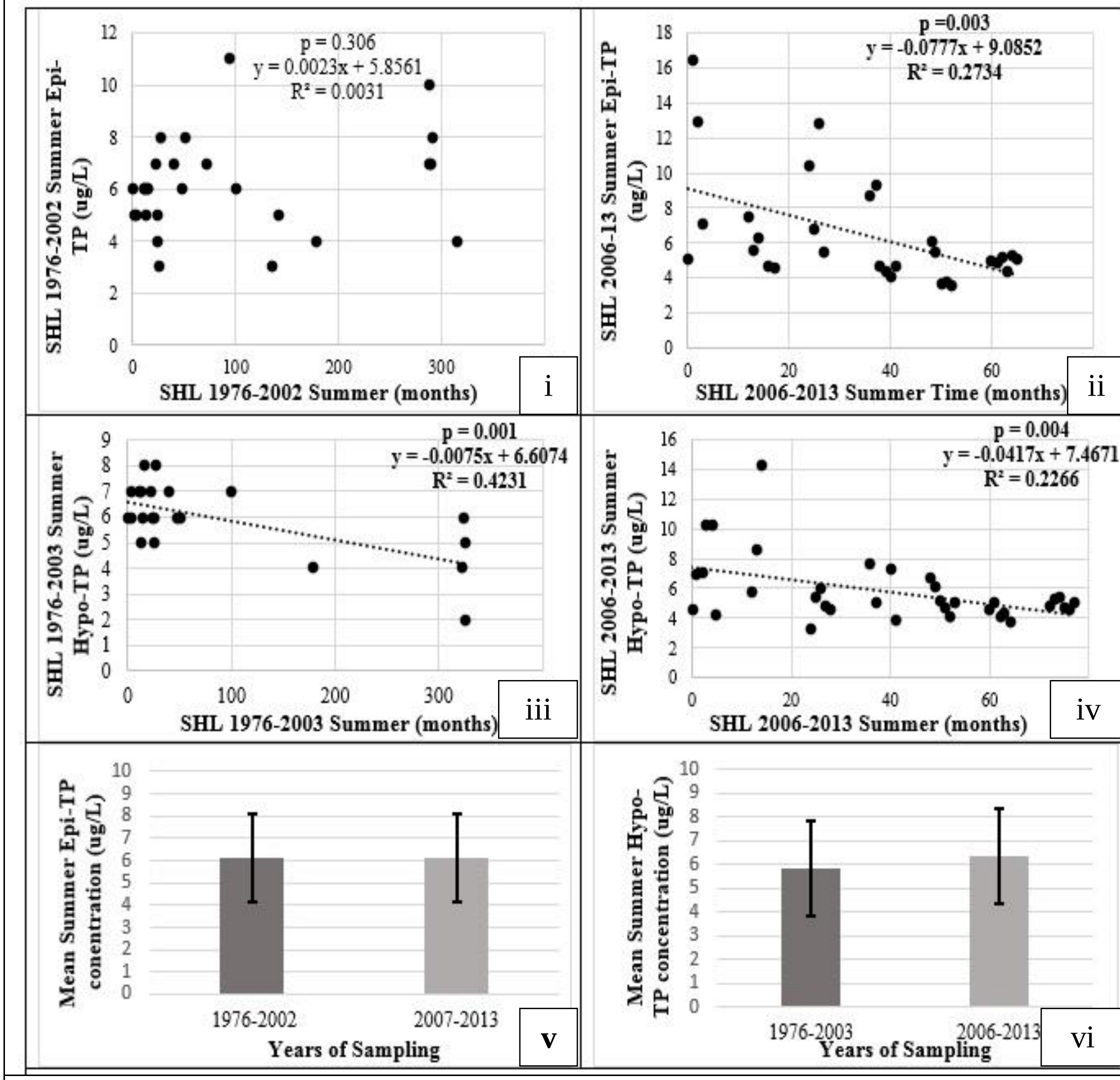


Figure 3.3b. Shawnigan Lake (SHL) summer (May-Oct.) total phosphorus (TP) trends. Summer TP trends in the epilimnion (Epi-TP) and hypolimnion (Hypo-TP) for years 1976-2002 (i) and (iii) and 2006-2013 (ii) and (iv). Mean summer TP concentration (ug/L) was no different in 1976-2002 (v) compared to 2006-2013 (vi).

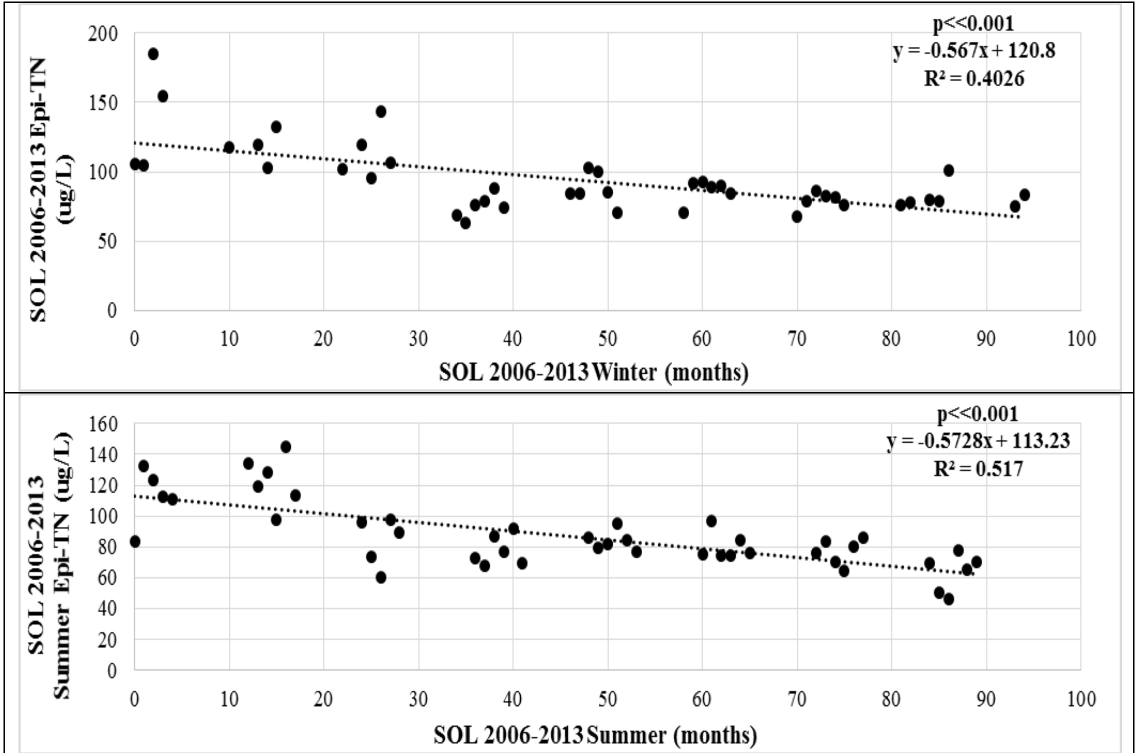


Figure 3.4. Sooke Lake (SOL) winter (Nov.-Apr.) and summer (May-Oct.) total nitrogen (TN) trends. Winter TN trends (upper) and Summer TN trends (lower) in the epilimnion are shown for years 2006-2013.

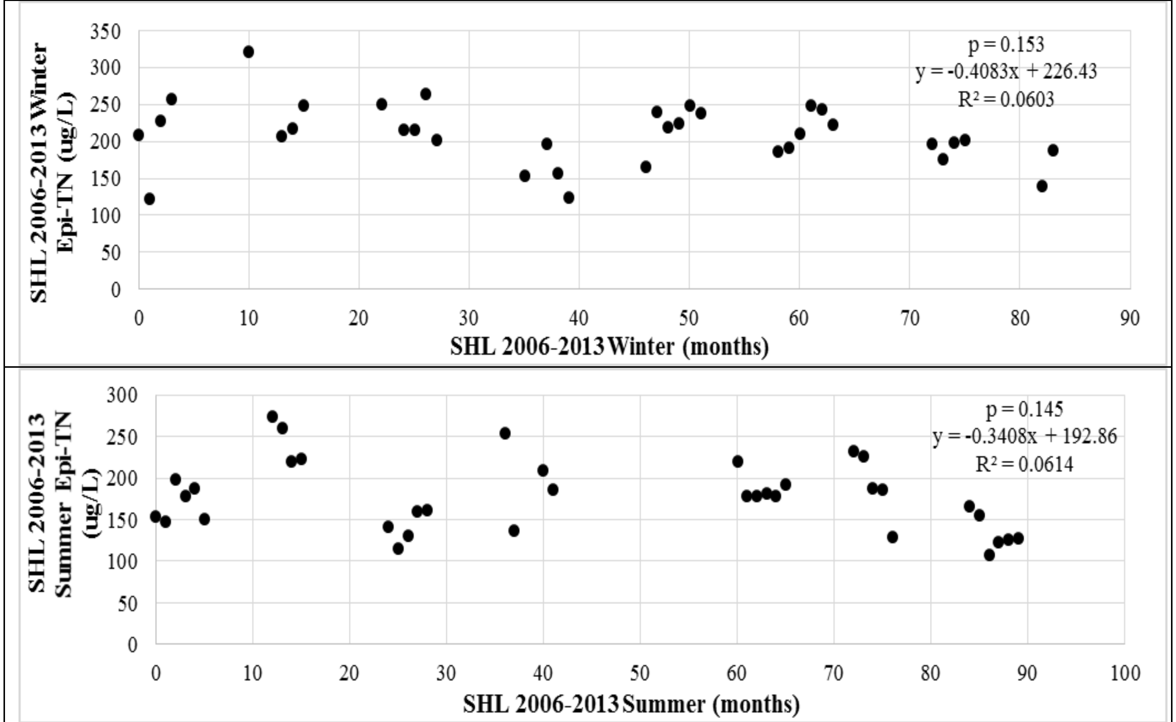


Figure 3.5. Shawnigan Lake (SHL) winter (Nov.-Apr.) and summer (May-Oct.) total nitrogen (TN) trends. Winter TN trends (upper) and Summer TN trends (lower) in the epilimnion are shown for years 2006 and 2013.

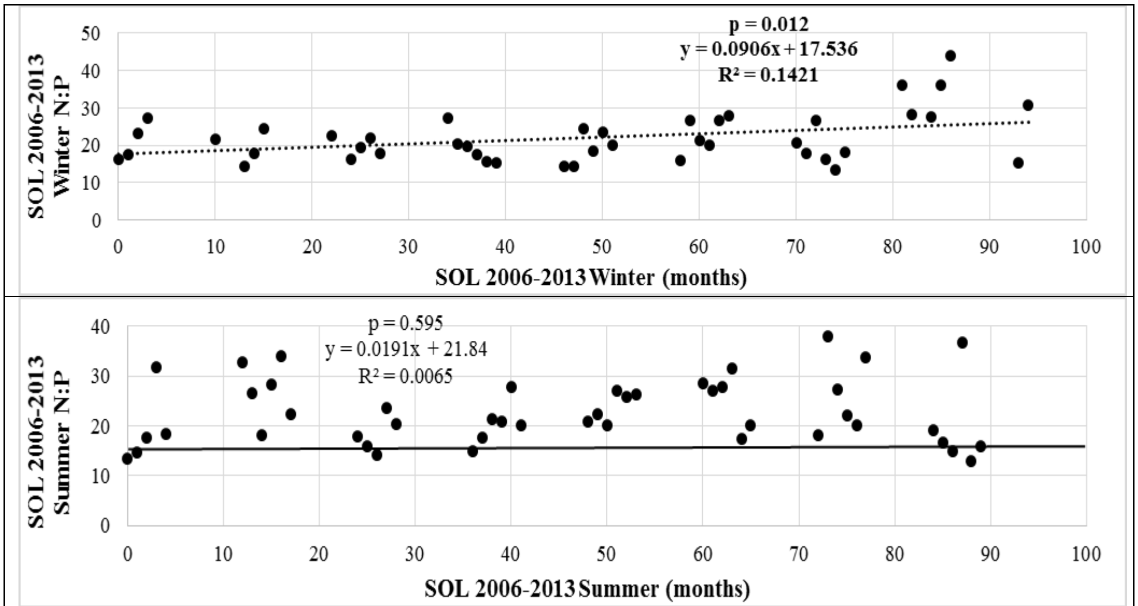


Figure 3.6. Sooke Lake (SOL) winter (Nov.-Apr.) and summer (May-Oct.) trends in nitrogen to phosphorus ratio (N:P). Winter N:P trends (upper) and Summer N:P trends (lower) in the epilimnion are shown for years 2006-2013.

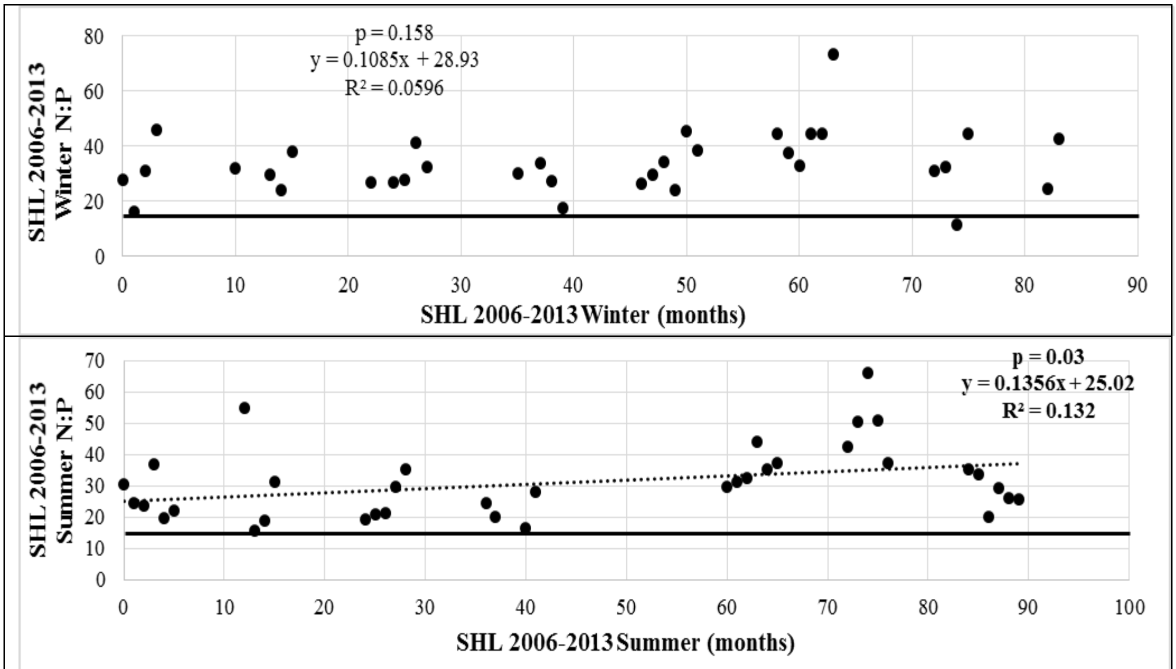


Figure 3.7. Shawnigan Lake (SHL) winter (Nov.-Apr.) and summer (May-Oct.) trends in nitrogen to phosphorus ratio (N:P). Winter N:P trends (upper) and Summer N:P trends (lower) in the epilimnion are shown for years 2006-2013.

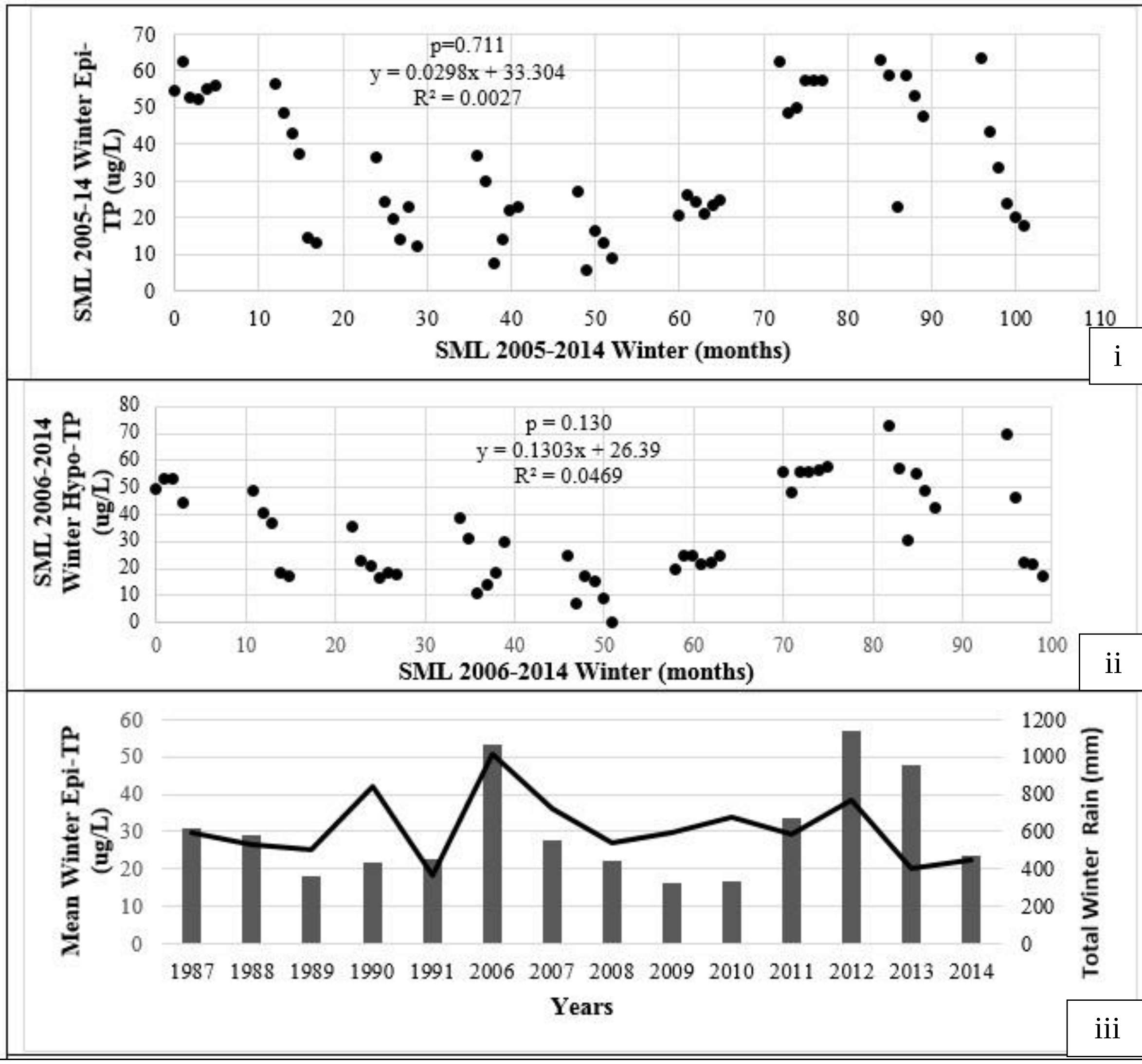


Figure 3.8a. St. Mary Lake (SML) winter (Nov.-Apr.) total phosphorus (TP) trends. Winter TP trends in the epilimnion (Epi-TP) (i) and hypolimnion (Hypo-TP) (ii) for years 2006-2014. Mean winter Epi-TP shows a cyclical trend that does not correspond to winter precipitation (iii).

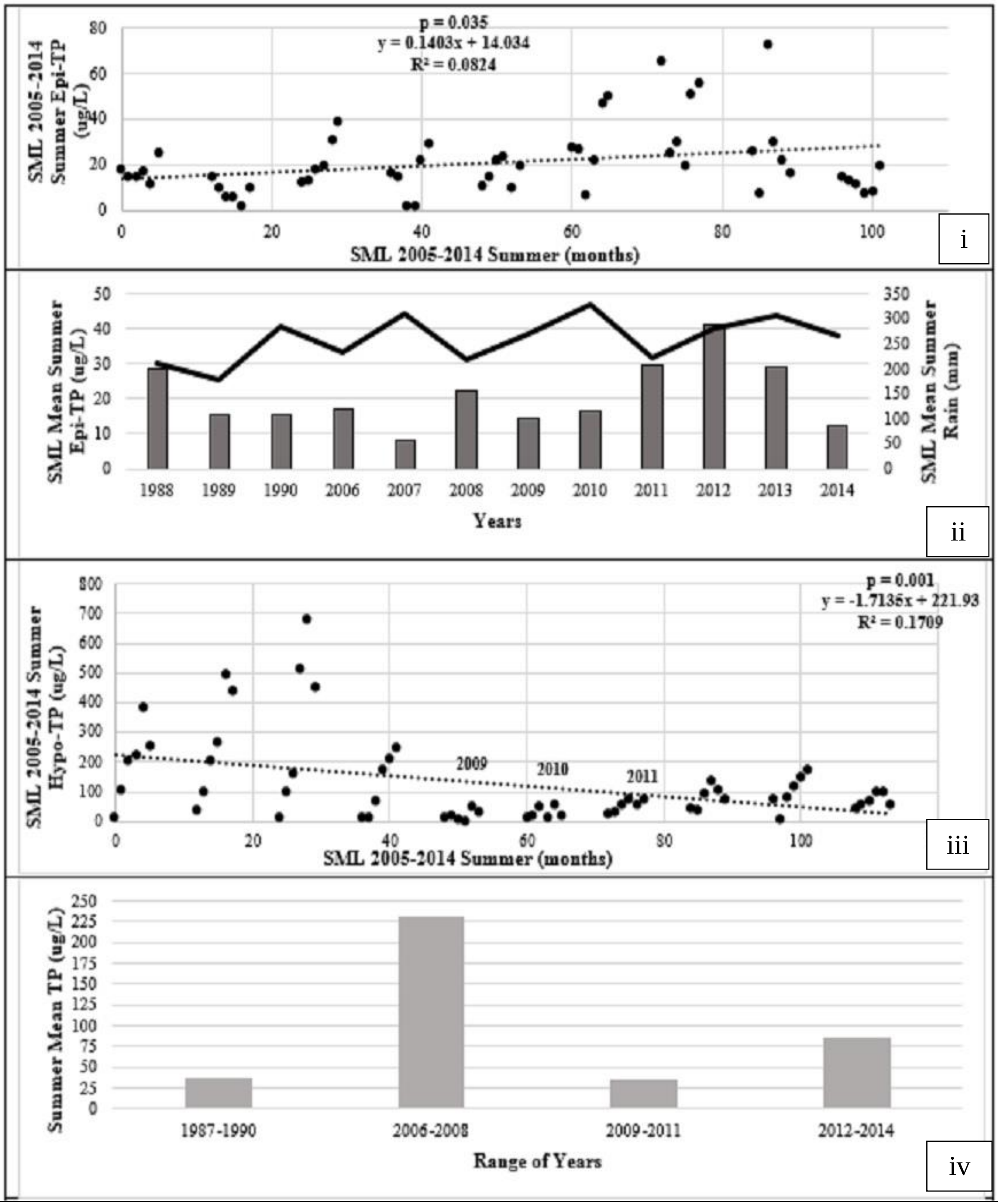


Figure 3.8b. St. Mary Lake (SML) summer (May-Oct.) total phosphorus (TP) trends. Summer TP trends in the epilimnion (Epi-TP) (i) and hypolimnion (Hypo-TP) (iii) for years 2006-2014. Mean summer Epi-TP shows a cyclical trend that does not correspond to winter precipitation (ii). Summer mean TP (ug/L) was lowest in 1987-1990 and 2009-2011 when the aeration system was operational (iv).

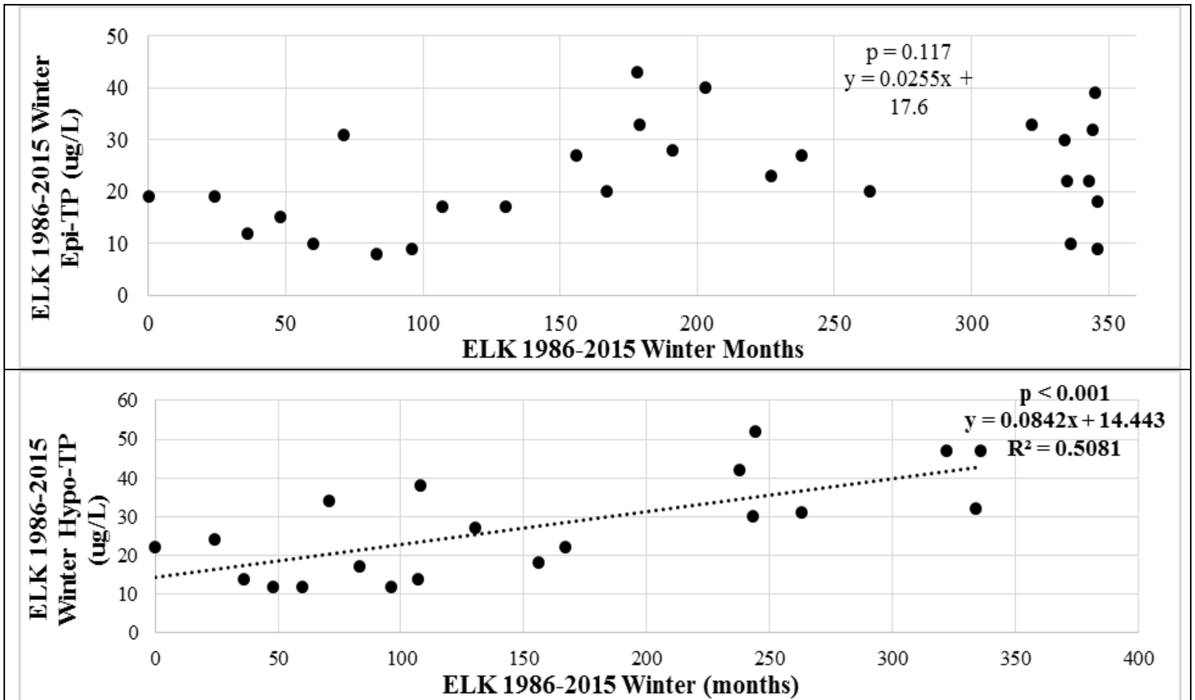


Figure 3.9a. Elk Lake (ELK) winter (Nov.-Apr.) total phosphorus (TP) trends. Winter TP trends in the epilimnion (Epi-TP) (upper) and the hypolimnion (Hypo-TP) (lower) for years 1986-2015.

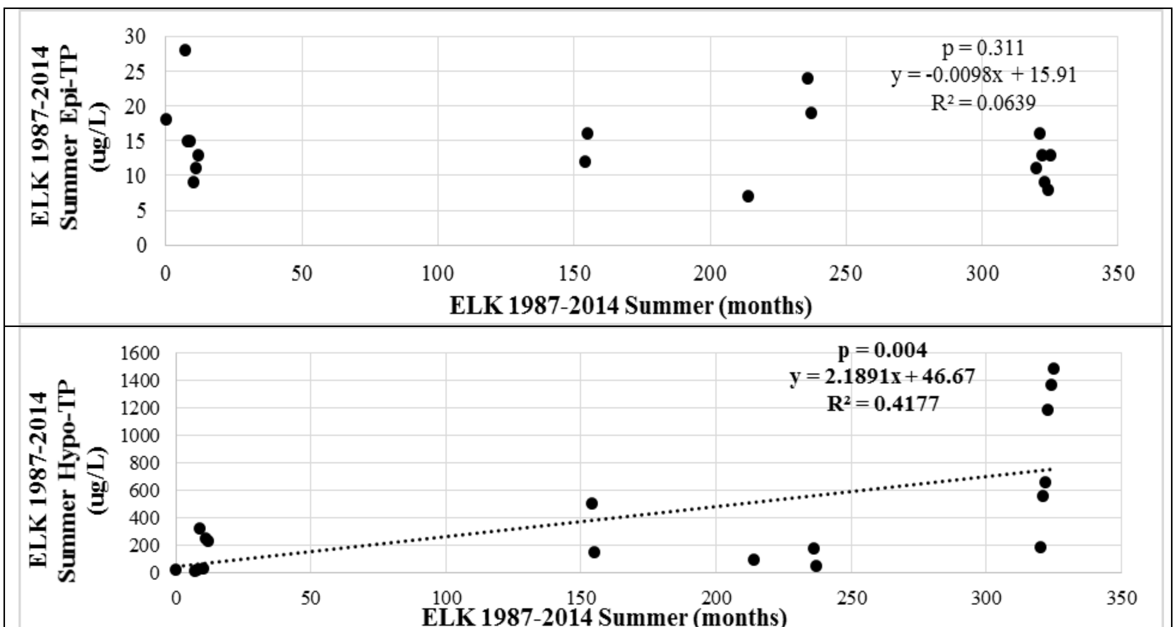


Figure 3.9b. Elk Lake (ELK) summer (May-Oct.) total phosphorus (TP) trends. Summer TP trends in the epilimnion (Epi-TP) (upper) and the hypolimnion (Hypo-TP) (lower) for years 1987-2014.

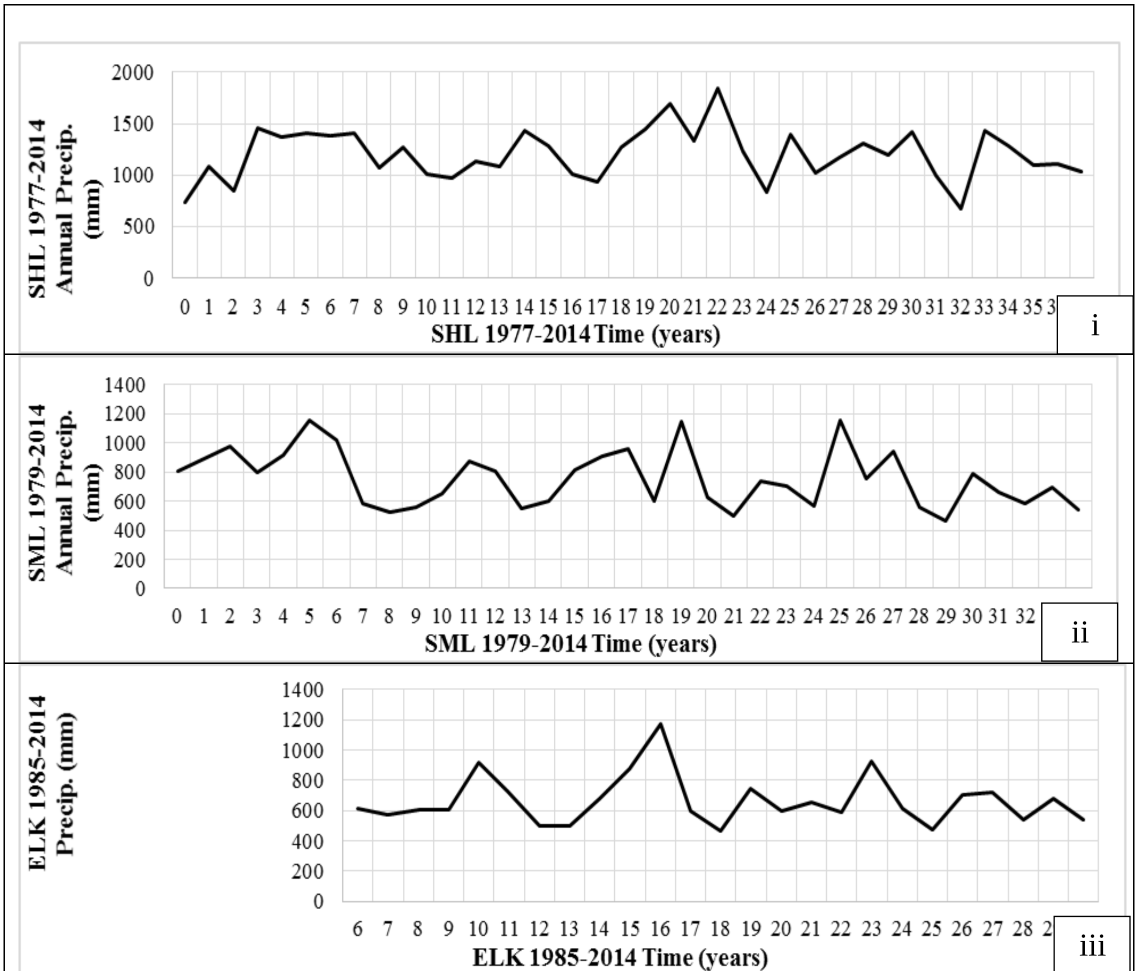


Figure 3.10. Precipitation trends in Shawnigan (SHL), St. Mary (SML), and Elk (ELK) watersheds. Precipitation (mm) over time (years) are shown for SHL (i), SML (ii), and ELK (iii) for years that correspond to total phosphorus analyses.

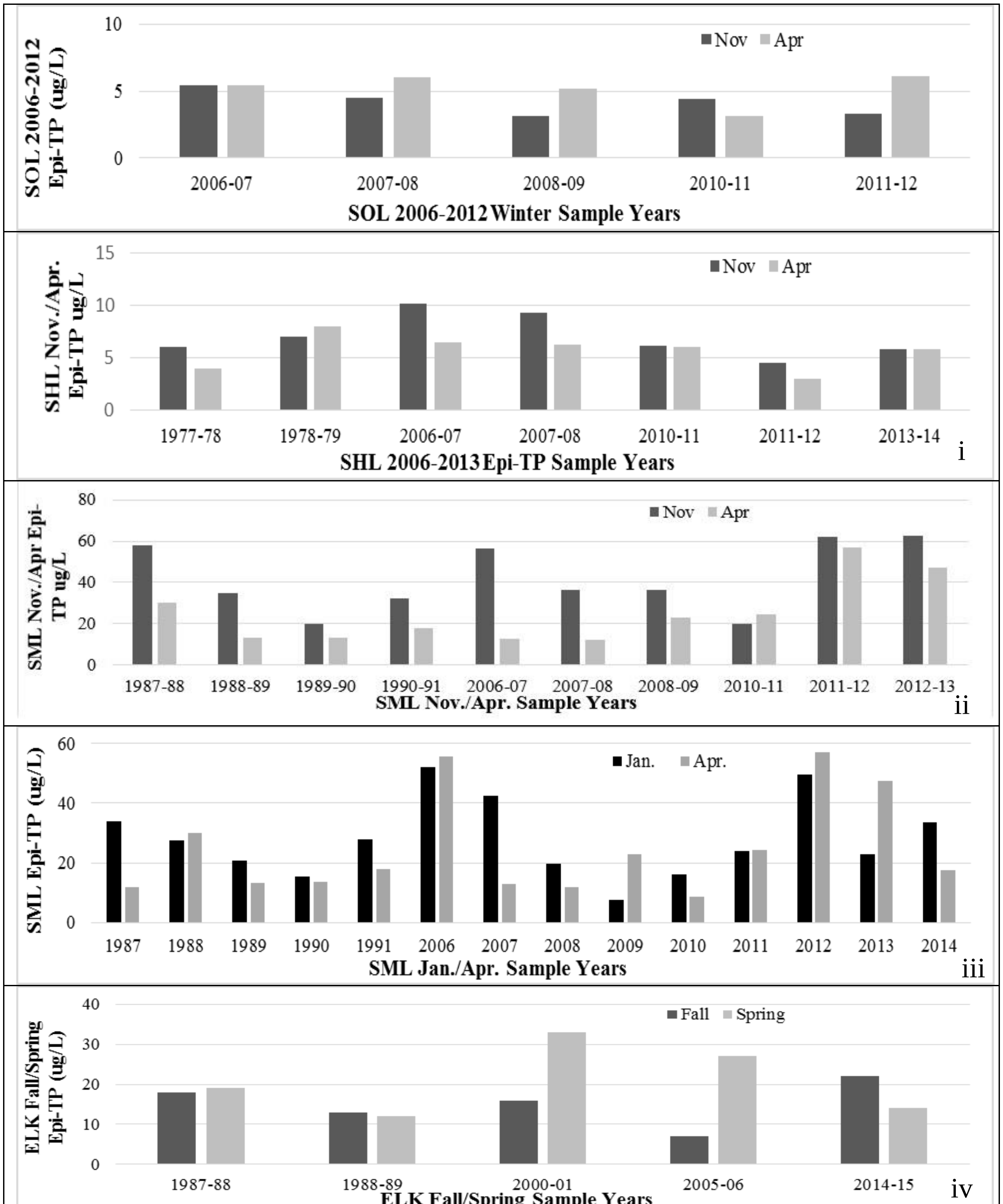


Figure 3.11. Monthly/seasonal total phosphorus (TP) concentrations in the epilimnion (Epi-TP) of Sooke (SOL), Shawnigan (SHL), St. Mary (SML), and Elk (ELK) lakes. November verses April Epi-TP for SOL (i), SHL (ii), and SML (iii). January verses April Epi-TP for SML to account for resettling of TP following fall turnover (iv). Monthly TP data for ELK for Fall was either October or November and for Spring was March or April depending on each year's sampling frequency (v).

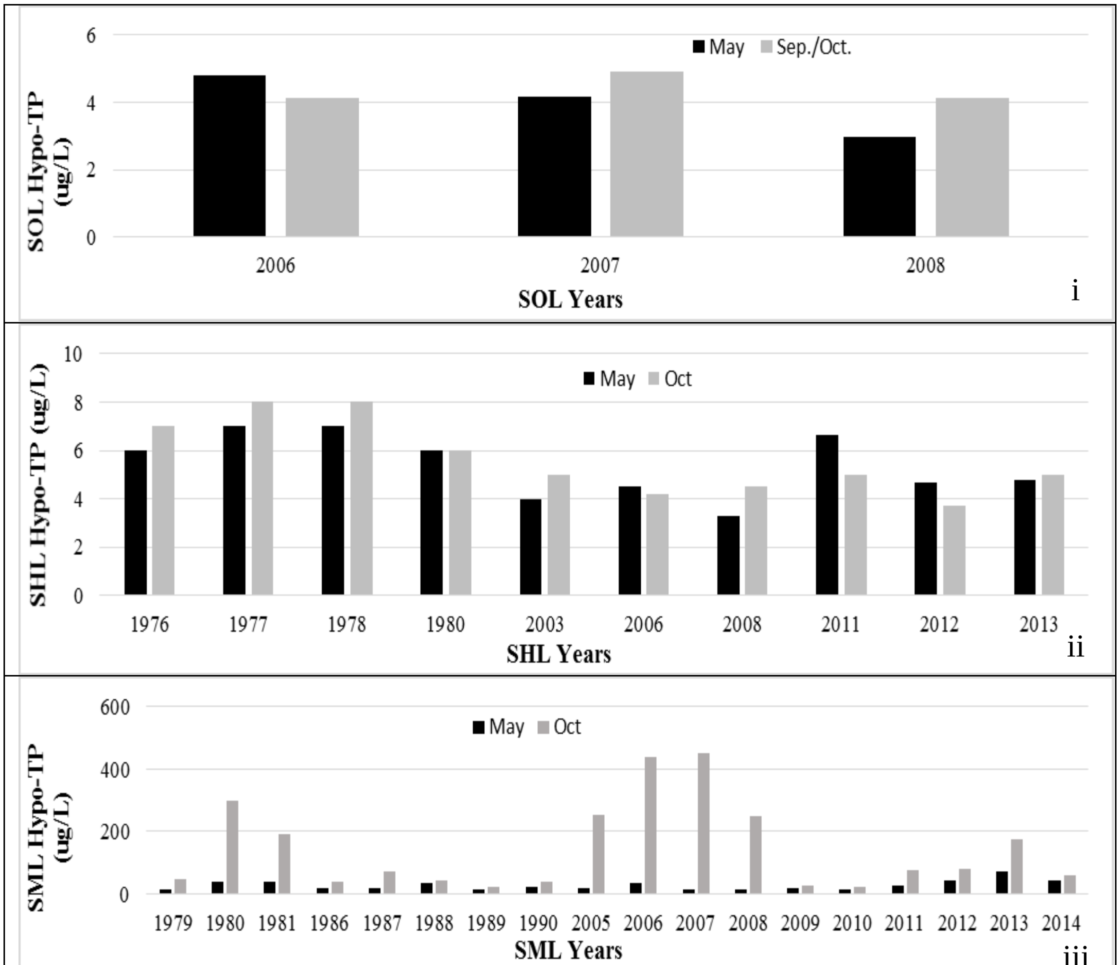


Figure 3.12. May verses October total phosphorus (TP) concentrations in the hypolimnion (Hypo-TP) of Sooke (SOL), Shawnigan (SHL), and St. Mary (SML) lakes. May verses October Hypo-TP for SOL 2006-2008 (i), SHL 1979-81, 1986-90, and 2005-2014 (ii), and SML 1979-81, 1986-90, and 2005-2014 (iii).

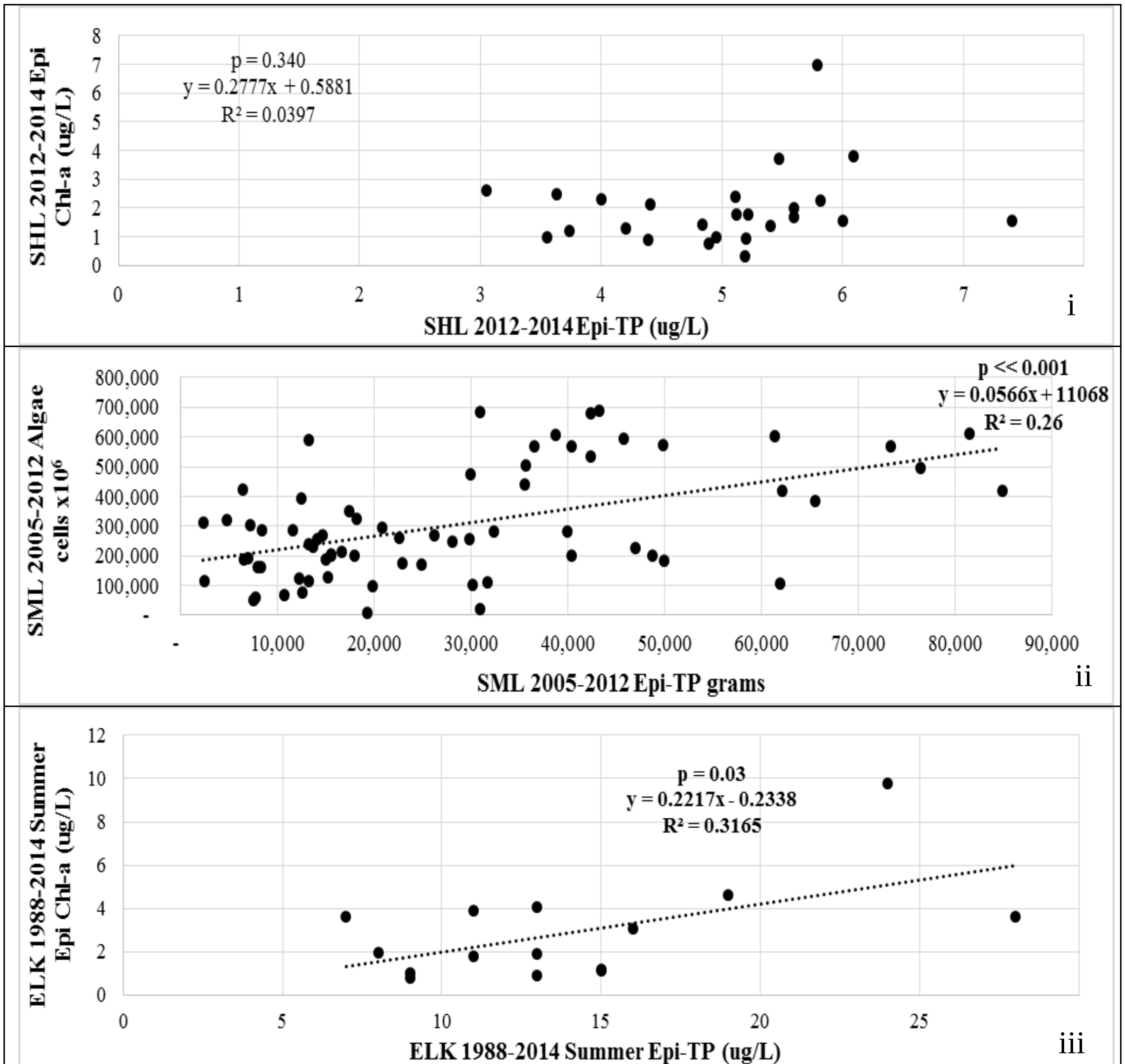


Figure 3.13. Total phosphorus (TP) and algae/Chlorophyll-a (Chl-a) relationships in Shawnigan (SHL), St. Mary (SML), and Elk (ELK) lakes. (i) No relationship between TP (ug/L) and Chl-a (ug/L) in SHL. (ii) Positive relationship between TP (grams) and algae cells (cells x10^6) in SML. (iii) Positive relationship between TP (ug/L) and Chl-a (ug/L) in ELK.

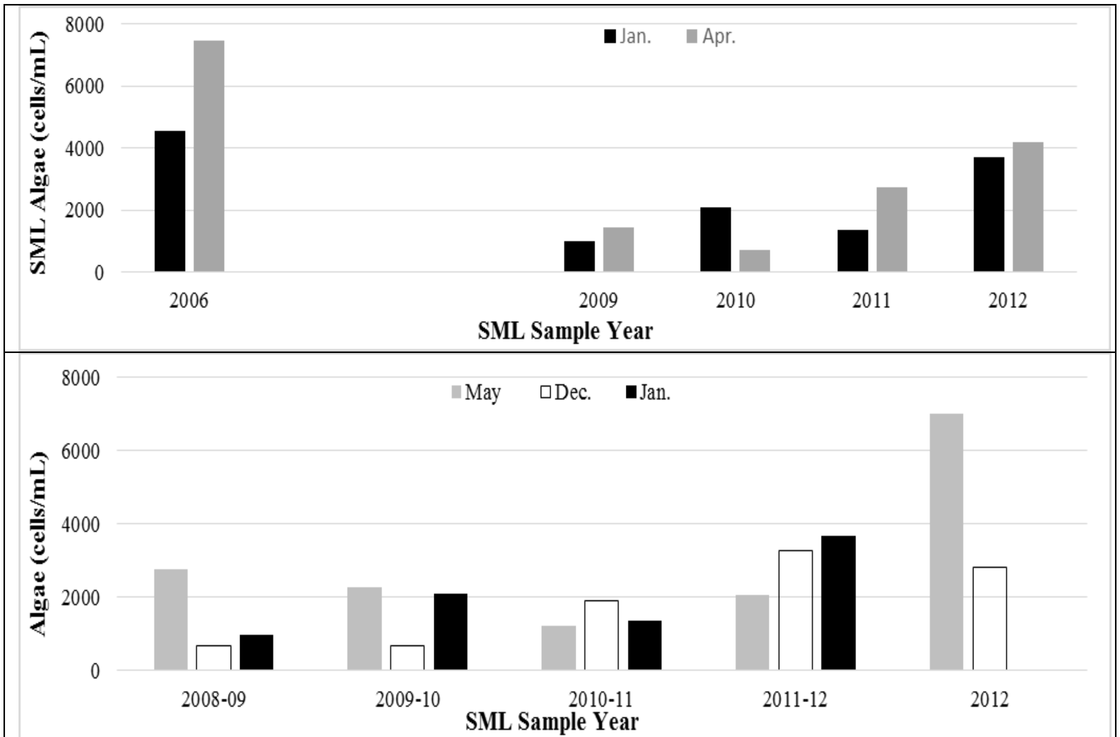


Figure 3.14. Seasonal algae abundance in St. Mary Lake (SML). (upper) Winter (Januar verses April) algae abundance (cells/mL) for 2006, 2009-2012. (lower) Late spring (May) verses early winter (December) and early spring (April) algae abundance (cells/mL) for 2008-2012.

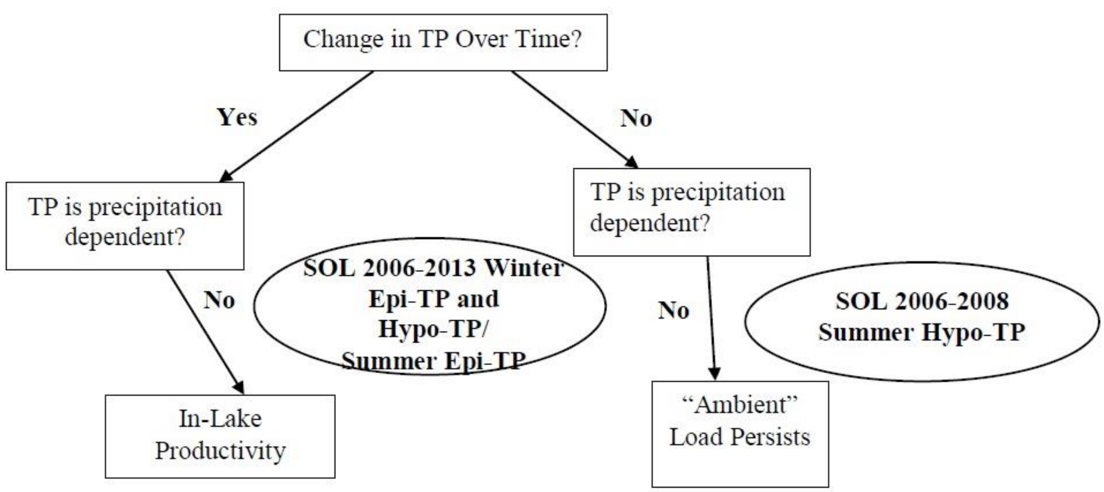


Figure 3.15 Sooke Lake (SOL) causal load flow chart. Primary cause of total phosphorus (TP) loads in SOL are shown based on the flow chart.

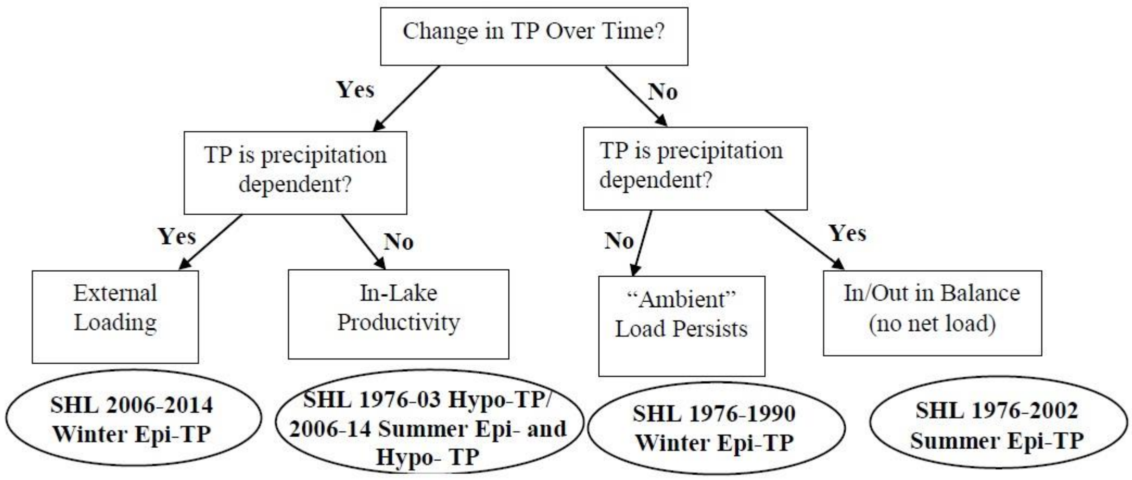


Figure 3.16 Shawnigan Lake (SHL) causal load flow chart. Primary cause of total phosphorus (TP) loads in SHL are shown based on the flow chart.

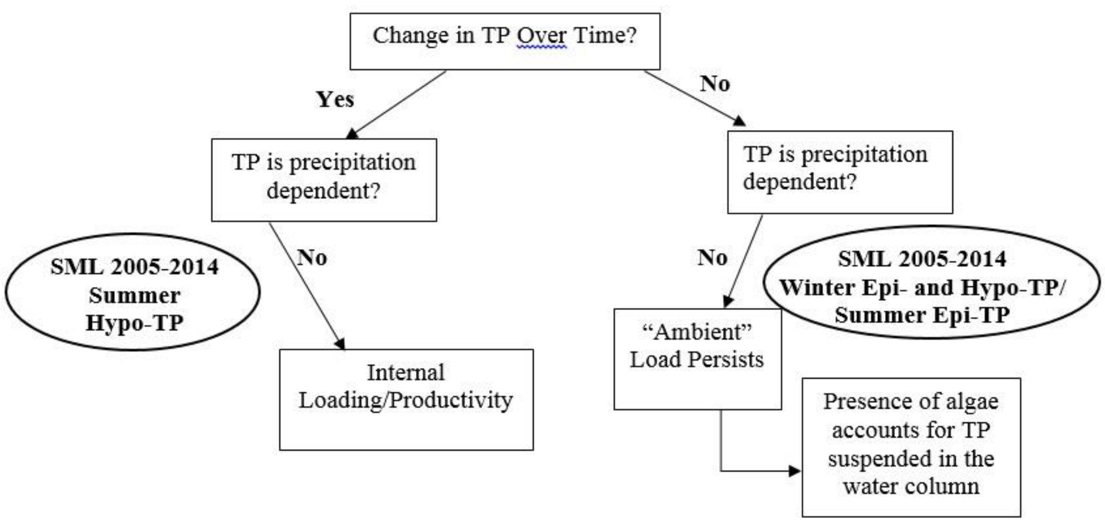


Figure 3.17 St. Mary Lake (SML) causal load flow chart. Primary cause of total phosphorus (TP) loads in SML are shown based on the flow chart.

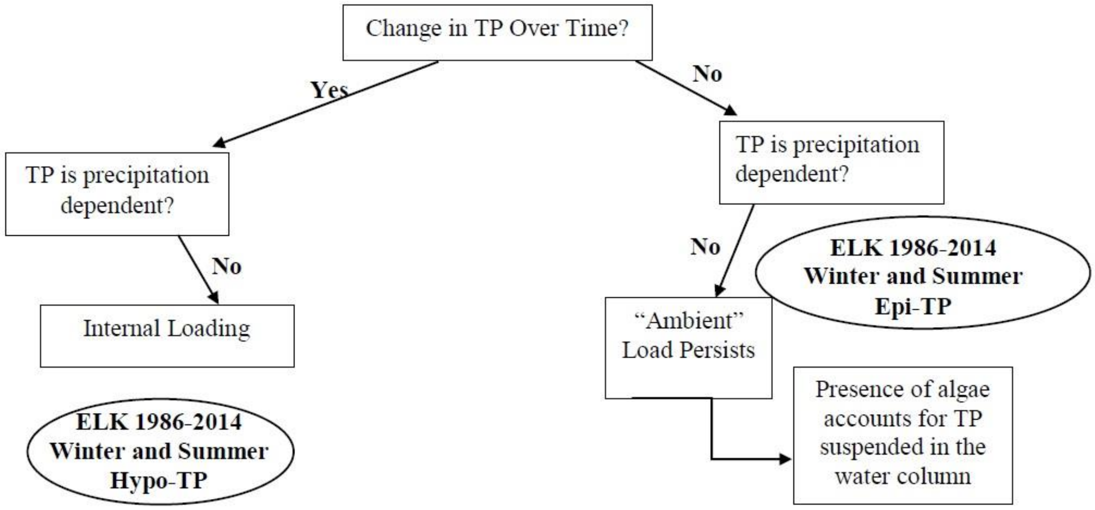


Figure 3.18 Elk Lake (ELK) causal load flow chart. Primary cause of total phosphorus (TP) loads in ELK are shown based on the flow chart.

Chapter 4 Validation of the MapShed© Model Outputs for Sooke, Shawnigan, and St. Mary Lakes

Abstract

MapShed©, a GIS-based software program, was used to predict external nutrient transport from the watersheds by integrating specific combinations of vegetation (land cover) and soil characteristics, in addition to, local topography, hydrology, and climate. Mass balances were produced from nutrient concentration data, estimates of atmospheric inputs, and estimates of mass outflow for Sooke, Shawnigan, and St. Mary Lake in order to determine whether MapShed© would be sufficient to evaluate the effect of external loading on in-lake water quality in the absence of monthly sample data. It was determined that when mass outflow estimates and sedimentation rates from literature were subtracted from external loading estimates the general consequences of external loading can be concluded from MapShed© load estimates. The effective nutrient concentrations from the load estimates were lower than the average in-lake concentrations, and indicate that a decline in nutrient concentrations can be expected. MapShed© transport rates generally agreed with those in relevant literature. Potential nutrient load reductions were evaluated by revising land-use scenarios for Shawnigan, St. Mary, and Elk Lakes, and found that some nutrient load reductions are possible. However, the majority of reforestation options would require transforming a significant portion of open, pasture, and cropland into forested areas, and it is not likely that land-owner approval could be obtained. Given that in-lake processes are primarily responsible for variations nutrient concentrations, land-based restoration may not be the most effective solution for reducing in-lake nitrogen and phosphorus concentrations.

Introduction

Numerous mathematical models have been developed and applied in attempts to predict the affect external loading has on in-lake TP concentrations (e.g. Vollenweider 1975; Dillon, 1975; Reckhow, 1980). The mathematical models were created because water quality monitoring is often too sporadic to gain a complete understanding of how lakes are trending. Many models are very complicated and difficult for watershed managers to use. Many still have been shown to be insufficient when applied to watersheds other than those for which they were created (Nurnberg, 1984). These models also typically do not include variables to account for specific watershed characteristics such as land cover and soil types, and often require extensive storm water monitoring in order to external nutrient loads. Like in-lake datasets, storm water datasets are usually incomplete, and external loading estimates are made based on sporadic data which may not be representative of annual fluctuations in external loading.

The development of geographical information systems (GIS) and the availability of spatial datasets provide an opportunity to model nutrient transport (external loading) in watersheds using multivariate modeling that simultaneously calculates the cumulative loading that results from specific combinations of vegetation (land cover) and soil characteristics, in addition to, local topography, hydrology, and climate. MapShed© (PennState 2013) is a user-friendly open-source (free) software program used to predict non-point nutrient transport to streams and lakes from the surrounding land area. Many of the required input datasets are available as free downloads or are readily obtained from local governments. If MapShed© produces accurate results, the software could help small communities to determine which remedial efforts are best suited to a specific lake system even if existing data are minimal.

Vollenweider (1975) accounted for the ambient nutrient mass by proposing that external loads entering a lake minus mass outflow can predict the change in mass concentration from year to year. As proposed by Vollenweider (1975), the balance of any non-conservative substance would be:

Change in mass = inflow mass - outflow mass - sedimentation (1)

The mass balances and transport coefficient results, as presented below, support the applicability of MapShed© to coastal BC watersheds. Although the model outputs cannot be directly matched to in-lake mass on a month to month basis, the annual TP and TN loads estimated by MapShed©, along with annual outflow volumes and phosphorus sedimentation rates (Nurnberg, 1984) indicate that the model predicts land-based loads with sufficient accuracy to determine the effect of external loading on in-lake nutrient concentrations. Linear regression results show that nutrient concentrations have mostly declined or have remained steady since the 1970's. In addition to the mass P in rainy season lake outflows, which would eliminate a fraction of the nutrient loads estimated by land-based loading coefficients, the natural biological cycles in lakes are able to utilize and recycle the mass nutrients within the water column itself, such that nutrients are generally not accumulating over time, hence the recurring annual stable-state or declining nutrient mass determined by the analyses in Chapter 3. Mass balances produced using measured nitrogen and phosphorus concentrations coupled with MapShed© external load estimates reflect this stable state condition when mass-out is accurately quantified.

Further, transport coefficients produced by MapShed© were compared with scientifically accepted transport coefficients for N and P based on land cover type. External loading coefficients (estimates by land cover) are precipitation-based because they are based on measurements of nutrients in storm water flows (stream flow). Thus, they are very specific to each watershed, and will vary each season and year depending on the amount of precipitation the watershed receives, and the resulting dilution. General load estimates (coefficients), usually expressed in kg/ha or kg/km^2, such as those summarized by Lin (2004) cannot be extrapolated generally across years even for the same watershed. Exact measurements are needed for many years to determine the nutrient load from storm water as related to precipitation, and then, possibly nutrient loads can be estimated based on the annual precipitation in years where storm water loads are not measured. Consistent and continuous storm water monitoring data is minimal and sporadic in the subject watersheds. Thus, the external loading model (MapShed©) can provide some necessary external loading estimates.

Finally, the MapShed© models are applied to each watershed to determine the feasibility of reducing external loading through re-forestation of open, disturbed, cropland, and pasture lands. Lake nutrient concentration in watersheds with low transport capacity (i.e., smaller with fewer inflows) is most influenced (in order of descending importance) by lake water residence time, watershed to lake area ratio, ratio of lake area to lake volume, percent cover by agriculture, and percent cover by forest (Fraterrigo and Downing 2008). The contribution of nutrients from developed areas is also demonstrated by comparing hypothetical 100% forested conditions with scenarios that include only forest and developed areas. According to model outputs best-case scenarios for watershed re-forestation nutrient load reductions would be minimal, and would likely not affect in-lake water quality concentrations. Nutrient inputs from developed areas contribute the majority of nutrients to the lakes.

In the follow study, I evaluated whether the MapShed© model was useful in estimating the external loading to the Sooke (SOL), Shawnigan (SHL), and St. Mary (SML) watersheds in regard to the influence of external loading on in-lake water quality. I determined the relative accuracy of MapShed© transport rates by comparing model outputs to those found in scientific literature. I also entered multiple land-use scenarios into MapShed to determine whether reductions in in-lake nutrient concentrations could result if areas with minimal vegetation are reforested.

Atmospheric Deposition

Research on atmospheric deposition of nitrogen and phosphorus is limited for coastal temperate rainforest ecosystems. Research conducted throughout North America emphasizes the need to evaluate both wet and dry deposition of N and P due to significant differences in both the quantity and form of N and P in rain and dustfall (Anderson and Downing 2006; Boehme et al., 2009). In western North America, dry nitrogen deposition from airborne particulates is a major contributor to nitrogen deposition rates particularly in forested areas near urban centers (Fenn et al. 2013). Studies largely agree that the majority of nitrogen from the atmosphere is deposited by rain, and that nearly all nitrogen in rain is in dissolved form. Conversely, dry atmospheric TP may be 2.8 times greater than TP in rain, and the soluble reactive P fraction has been shown to be 62% in dustfall compared to 7% in rain (Anderson and Downing, 2006). Atmospheric deposition has been shown to contribute 8-50% of the total phosphorus load to freshwater lakes (Boehme et al. 2009), and is a necessary element of lake nutrient budgets.

Raymond et al. (2010) found that average nitrogen (as $NH_4 + NO_3^-$) in southwestern BC coastal precipitation ranges from ~1.00 mg/L measured in Victoria and on Saturna Island to 1.28 mg/L measured in downtown Vancouver. Atmospheric nitrogen deposition (wet + dry) was found to increase with urban density from 2.64 to 4.82 to ~8.55 kg/ha/yr on Saturna Island, in Victoria, and in Vancouver respectively (Raymond et al. 2010). The main sources of atmospheric N in urban areas are emissions from vehicles and shipping operations, and the elevated N deposition measured on Saturna Island, compared to the “pristine” values presented below for Washington and Oregon, would likely be due to shipping operations (Raymond, Bassingthwaighte, and Shaw 2010). Atmospheric nitrogen deposition rates are very low on the BC coast due to small, dispersed human settlements and coastal winds as evidenced by the extremely high nitrogen deposition rates of 25.7 kg/ha/yr found by (Vingarzan et al. 2000) in the Fraser Valley transportation corridor (Putt 2014). Nitrogen deposition rates are much lower in sparsely populated areas dominated by forests. (Martin and Harr 1988) found that wet nitrogen deposition in the H.J. Andrews Research Forest in the western Cascade Mountains (72 miles west of Eugene, OR) totalled 2.0 kg/ha/yr. A study conducted over 2 years in the Olympic, Mount Rainier, and North Cascades national parks (Washington State) found that total atmospheric nitrogen deposition (wet + dry) in these relatively pristine forests averaged 1.58 kg/ha/yr (Fenn et al. 2013).

Fenn et al. (2013) evaluated the differences between ambient atmospheric nitrogen deposition and the resulting through-fall (i.e., on the ground below the forest canopy) nitrogen deposition. Compared to ambient nitrogen of 1.58 kg/ha/yr, through-fall nitrogen deposition averaged only 0.65 kg/ha/yr. Fenn et al. (2013) also showed that significant reductions between ambient and through-fall nitrogen deposition have been observed during numerous studies conducted in Washington State and throughout the world. Ambient atmospheric nitrogen deposition rates were shown to be reduced by 42-64% in the summer, and up to 92% in the winter, presumably due to the retention of NO_3^- by foliage, twigs, and bark. It is important to note that NO_3^--N was 2.5 times higher than NH_4-N at atmospheric gauging stations, but NH_4-N was 5.2 times higher than NO_3^--N when measured below the canopy (i.e., through-fall), and numerous studies support canopy uptake of NO_3^--N (Fenn et al. 2013). NH_4 is available for uptake by plant roots, and excess NH_4 on the ground surface is easily transformed back into NO_3^- in rain-water for transport in overland flows.

Based on these studies, ambient atmospheric nitrogen deposition into the subject lakes likely ranges from 1.58 to 2.64 kg/ha/yr, and is added to the nutrient budgets based on the area of the lake surface only because deposition is significantly reduced before reaching the ground in forested areas (particularly in the winter months when nutrient loads to lakes have been shown to the be highest).

There are no published studies evaluating atmospheric P deposition on Vancouver Island or in the Gulf Islands. As reported by Putt (2014), TP deposition rates in the H.J. Andrews Research Forest in the western Cascade Mountains of Oregon have been measured since 1969, and the most recent 10-year (2000-2009) average TP deposition rate is 0.02 kg-P/ha/yr. For comparison, Vingarzan (2000) reported TP deposition rates of 0.15 kg-P/ha/yr in the Fraser Valley (Putt 2014), and Dr. Richard Nordin reported 0.136

kg/ha/yr TP in precipitation at SOL (Nordin 2015). The Nordin (2015) estimate is used for the mass balances in this report.

Nutrient Retention

Water residence time is very positively correlated with the retention of both nitrogen and phosphorus by lakes (Saunders and Kalff 2001; Brett 2008). Regarding nitrogen, the use of the term "retention" is misleading because, unlike phosphorus, studies show that the denitrification process is responsible for 63% of nitrogen "retention" in lakes, and sedimentation 37% (Saunders and Kalff 2001). During denitrification, bacteria transform water soluble nitrogen (typically nitrate NO^{3}-) into nitrogen gas (N^{2}). This means that, although the concentration of nitrogen that could potentially be transported out of the lake decreases with increased retention time, 63% of the nitrogen is also not retained in the lake water.

Phosphorus, on the other hand, has been widely proved to be retained in the lake system through particle settling and sedimentation where it is either permanently sequestered in the lake sediment, or cycled in and out of the sediment and water column. Carey and Rydin (2011) found that phosphorus sequestration in lake sediments varies predictably between eutrophic, mesotrophic, and oligotrophic lakes. Total phosphorus measured in the sediment of 94 lakes (to 30 cm depth) showed that eutrophic lakes exhibit high TP in surface sediment, with a rapid decrease in TP along the sediment depth gradient. Oligotrophic lakes exhibit low TP in surface sediment, with a moderate rate of increase in TP with depth. Mesotrophic lakes exhibit relatively constant TP with depth. The study found that the TP trend within the sediment corresponded with the water column TP concentrations that characterize lake trophic status (Table 5) (Carey and

Rydin 2011). Nurnberg (1996) positively correlated “hard” water with lake productivity, and stated that the nutrient-rich sediments that result from the external loading of nutrient-rich soils release more phosphorus under anoxic conditions, and have low capacity for sequestering phosphorus in the sediment. Phosphorus sequestration is driven by the availability of soil minerals such as calcium, iron (Fe-III), and aluminum (Al). Calcium is positively correlated with phosphorus loading (external and internal) (Nurnberg 1996), whereas Fe-III and Al are fundamental to phosphorus sequestration by soils and lake sediment (Carey & Rydin, 2011). Low hypolimnic oxygen levels, which are typical during summer stratification, hinder the ability of iron and aluminum to sequester phosphorus (due to resulting molecular state changes, e.g., Fe-III to Fe-II). This results in high internal phosphorus loading due to the release of phosphorus from the lake sediment. Unlike that of oligotrophic lakes, the sediment in eutrophic lakes has very limited phosphorus sequestration capacity, and it is this factor, rather than increased external loading, that determines the availability of phosphorus to cycle back to the water column (internal cycling) (Carey and Rydin 2011).

The phosphorus sedimentation rate is the estimated fraction of incoming mass TP that settles to the lake bottom and is presumably adsorbed to lake sediments. There is agreement in the literature that the sedimentation rate of TP is a function of the annual areal water load (outflow/surface area) of a lake. Thus, sedimentation will vary according to the volume of water that flows out of the lake, as the outflow volume is understood to represent the linear movement of water from inflows to outflow. Several formulas for estimating the rate of sedimentation (also referred to as “retention”) have been developed, but sedimentation rate formulae typically only apply to the lakes from which they were derived, and do not transfer arbitrarily to other lakes (Nurnberg 1984). Nurnberg (1984) found that her formula for phosphorus sedimentation ($R=15/(18+qs)$) was applicable over a large range of lake depths, residence times, and areal water loads. In general, with greater outflow volume the sedimentation rate will decrease.

Watershed Restoration

Fraterrigo and Downing (2008) determined that differences in watershed size and hydrologic character (i.e., number of inflow pathways) account for differences in the way land use affects nutrient transport in watersheds. They found that the proximity of commercial/residential and agricultural lands to lakes influences nutrient concentrations in watersheds with low transport capacity, but is unrelated to nutrient concentrations in watersheds with high transport capacity where fluctuations in nutrient concentrations are governed by inputs into streams at the scale of the entire watershed. The study suggests that restoration and/or nutrient reduction efforts for lakes in larger watersheds should be focused farther up in the watershed and adjacent to in-flow streams, whereas restoration along the lake shore is more effective in smaller watersheds (Fraterrigo and Downing 2008). Duarte et al. (2009) present evidence that ecosystems that have been disturbed by changes in land use and human activity will not return to their original state regardless of the elimination of known stressors. Instead, the ecosystems are more likely to experience a “regime shift” and operate from a new baseline for decades after the environmental stressors have been removed. A study of the recovery time for 35 lakes following the implementation of nutrient reduction strategies found that new equilibriums for nitrogen could typically be observed in less than 5 years, whereas phosphorus concentrations took 10-15 years to reach a post-restoration stable state (Jeppesen et al. 2005). Forested watersheds that are subjected to logging operations contribute increased nitrogen to streams and receiving waters for up to 15 years after harvest (Zhu et al. 2005).

Lakes Used for Model Validation

In order to determine whether the MapShed© nutrient transport modeling software can accurately predict in-lake TP trends, it was necessary to choose lakes where nearly continuous monthly concentration data have been collected. In addition to the extensive linear regression analyses presented in Chapter 3, the annual Nov. to Nov. net loads from water quality data indicate whether a lake has gained or lost mass TP. Nearly continuous in-lake TP and TN data are available for Sooke (SOL), Shawnigan (SHL) Lakes, and TP data are available for St. Mary (SML) since 2007 (and prior), so these three lakes were evaluated to establish the accuracy of MapShed© estimated external loading.

SOL is managed by the Victoria Capital Regional District (CRD) and SHL is managed by the Cowichan Valley Regional District (CVRD). These lakes are located within adjacent watersheds approximately 35 km north of Victoria, BC on Vancouver Island. SML is managed primarily by the North Salt Spring Water District (NSSWD), and is located on Salt Spring Island, 32 km northwest of the SOL/SHL divide. SOL lies within an undeveloped and protected watershed, and provides 90% of the drinking water for the city of Victoria. SHL lies within a partially developed watershed where logging operations are ongoing, and is the drinking water source for the SHL community. SML lies within a watershed subdivided into rural residential parcels, and provides drinking water for much of north Salt Spring Island. All three watersheds share very similar climate and geology, and are within the Coastal Douglas-fir biogeoclimatic zone.

Land cover in the SOL watershed is primarily forest (91%) with some open area (8.5%). The remaining cover is disturbed (i.e., lacks vegetative cover; 0.3%) and wetland (0.2%). The SHL watershed consists of forest (44%), open land (37%), low-density residential (12%), medium-density residential (3.7%), and disturbed lands (2.6%), with very small areas of wetland (0.6%) and urban area (0.03%). The SML watershed is primarily forested (69%) with low-density development (20%). There are some pasture areas (8.1%) and wetlands (1.5%), and very small areas of disturbed (0.7%), open (0.4%), medium-density residential (0.2%), and roadway (0.2%). The ELK watershed consists of forest (45%), low-density residential (23%), pasture (9.4%), open lands (6%), annual cropland (4.4%), medium-density residential (5%), commercial (4%), and disturbed areas (1.5%). Wetlands (1.6%) and ponds (0.1%) constitute the smallest fraction of land cover.

Methods

MapShed© Inputs

GIS data were acquired from DataBC (Province of British Columbia, 2015), GeoGratis (Natural Resources Canada, 2015), and local governments. Data were created and edited using ESRI ArcInfo 10.0 (Environmental Systems Research Institute (ESRI), 2014). Basins and land cover files were delineated manually using topographic overlays created from the elevation (DEM) files, and the aerial basemap downloaded from the ESRI library. Weather data were acquired from the Environment Canada Climate database (Government of Canada, 2015), and formatted using Microsoft Excel. SOL was sub-divided into 17 sub-basins in order to perform MapShed© analyses. SHL was sub-divided into seven sub-basins in order to perform MapShed© analyses (Figure 4.1). The topography and drainage features in the SML watershed made it possible to include several drainages within each sub-basin, and this watershed was sub-divided into five basins for MapShed© analyses (Figure 4.2). Although the frequency of nutrient sampling in ELK was not sufficient to perform a mass balance for model validation, land-based nitrogen and phosphorus exports were estimated using the MapShed© model. Watershed topography and drainage features made it possible to include several drainages within each ELK sub-basin, and the watershed was subdivided into three basins (Figure 4.3).

Mass Balance

SOL monthly TP and TN concentrations were recorded in the north basin of SOL (SOL-04); however, the average Epi-TP concentrations at a sample point near the SOL outlet (SOL-00), where data were recorded from 2009-2013, show a similar average TP concentration (SOL-00=3.84 ug/L; SOL-04=3.89 ug/L). Similarly, SML monthly TP concentrations were recorded near the north end, but lake-perimeter sampling conducted in 2014 determined no difference in Epi-TP at various locations around the lake (Hodgins, 2014). Mass balances were determined for each lake by converting monthly concentrations (ug/L) into kilograms (kg) in the lake using the lake volume relevant to the sample data. For instance, in SOL, samples from the hypolimnion were limited to 2006-2008, and the difference in the averages of the available hypolimnion data and the larger epilimnion data set were minimal (i.e., 4.42 and 4.12 respectively), thus the entire lake volume (160.3×10^6 m^3) was used to determine in-lake mass (kg) based on monthly Epi-TP and Epi-TN concentrations. Epi-TP and Epi-TN concentrations were also multiplied by the entire SHL lake volume (71.9×10^6 m^3). Conversely, in SML, sufficient data were available to separate the epilimnion and the hypolimnion, and the volume of the hypolimnion was calculated based on the sample depth of 10-12 m, such that the volume applied to the Hypo-TP data was two times the lake surface area. The product was subtracted from the entire lake volume (15.9 x 10^6 m^3), and the remaining volume was used to determine the TP mass in the epilimnion. The resulting Epi-TP and Hypo-TP kilograms were then combined to produce a TP mass for the entire lake. TN data are not available for SML. The differences in lake TP and TN mass in each month were calculated (e.g., Diff.=T_2-T_1 where T_2=Feb. and T_1= Jan.; T_2=Mar. and T_1= Feb., etc.) in order to compare changes in lake mass with external loading estimated by MapShed©. The net TP and TN mass (also the difference from November to November) were then calculated for each year, and for the entire sample period. Lake outflow data were acquired from the CRD for SOL and the NSSWD for SML. The outflow volume from SHL was estimated using a lake level line graph produced by the CVRD for 2006-2009 estimates. The 2015 lake level line graph (Cowichan Valley Regional District, 2015) shows the outflow rate (L/s) that corresponds to the lake level, and this was used to estimate lake outflow volume based upon lake levels in the 2006-07 and 2007-08 water years. Outflow volumes for 2010-11 through 2012-2013 water years were estimated by calculating the percent precipitation in those years verses 2006-2007 and applying that to the outflow. Dam overflow months were recorded by CRD for SOL. The precipitation fraction in these months was used to weight each month’s contribution to total overflow (Q). SOL withdrawals for consumption (W) were recorded monthly. SHL outflow volume was weighted for winter months based upon known lake overflow months as documented for SOL, and SML outflow volumes were recorded monthly for both Q and W. The monthly Epi-TP concentrations were then multiplied by each month’s outflow volumes and combined to determine the annual mass TP that flows out of each lake. SHL

Epi-TP data were not recorded consistently in 2009, so the mass balance for 2008-2009 and 2009-2010 was not completed. Average TP data and outflow volumes are shown in Table 2 for general comparison between lakes, but all calculations were made based upon monthly data as presented in Tables 5, 6, and 8.

Model Load Evaluation

After determining the net change in lake mass using the monthly TP concentration data, the external load estimated by MapShed© was combined with the estimated atmospheric deposition (TP=13.6 kg/km^2 (Nordin, 2015)) on the lake surface, to determine the monthly and annual external loading to the lake. The TP mass-out was subtracted from the annual external load (kg), and the remainder was assumed to remain in the lake when the net mass (Nov.-Nov.) was positive, or to have settled to the lake bottom when the net mass was negative. Sedimentation rates were evaluated based on the equation developed by (Nurnberg 1984) and found by that study to best represent TP sedimentation in lakes of various areas and depths. Nurnberg (1984) found that the equation:

$$R=15/(18+qs) \quad (2)$$

where R = sedimentation rate

qs = ((Q+W)/surface area),

adequately quantifies TP sedimentation in unstratified oligotrophic lakes, but is not as accurate for more productive lakes. Other attempts to quantify TP sedimentation have had similar limitations (e.g., (Dillon and Kirchner 1975)). TP transport coefficients by land cover type were determined from MapShed outputs (kg) and the area of each land cover type by watershed.

The nitrogen balances for SOL and SHL reflect the general assumption that the denitrification process is responsible for 63% of nitrogen "retention" in lakes, and sedimentation 37% (Saunders and Kalff 2001). After TN outflow, 63% of the remaining MapShed© estimated net mass TN is assumed to have been lost through denitrification and 37% retained for sediment/water exchange.

Results

Sooke and Shawnigan Lakes

Previous linear regression analyses conducted on the change in TP in SOL from 2006-2013 and SHL from 2006-2014 showed a definite decline in TP (Table 4.2). With the exception of SHL winter Epi-TP, TP concentrations for this time period were not precipitation dependent, and this indicates that the primary cause of the change in TP was due to internal mechanisms, and that a net loss of TP to outflow and lake sediments is occurring in both SOL and SHL.

Tables 4.3, 4.5, 4.7, and 4.9 show the annual net nutrient loads based on monthly TP and TN concentrations (Net Mass TP, TN), the mass balance derived from MapShed© TP and TN loading, atmospheric deposition, and lake-outflow volume.

The TP mass balance for SOL 2007-2013 (Table 4.3) estimates an average land-based TP load of 509 kg/yr, and an atmospheric load of 79.6 kg/yr, for a total 588 kg/yr in external loading. The average outflow mass, estimated based on Epi-TP and outflow volumes was 303 kg/yr. The Nurnberg (1984) equation resulted in a sedimentation rate of 0.49. Annual calculations of in-lake mass determined by the measured TP concentration from November to November each year resulted in net zero loading for 2007-2013, and this is congruent with the linear regression analyses. Assuming all TP that was not lost to outflow is sequestered in the lake sediments (i.e., sedimentation), the average percentage of total external TP lost to sediment was determined to be 48%. For SOL the MapShed© estimates are supported by outflow and sedimentation estimates, and it could be determined that external loading is not contributing to increasing in-lake TP in the absence of monthly in-lake samples. Measured outflow volumes and concentrations are, however, necessary.

As demonstrated in Chapter 2, TP concentrations in SHL are higher than those measured in SOL. The mass balance for SHL 2007-2013 estimates an average external TP load (land-based+atmospheric) of 844 kg/yr TP (Table 4.5). This is 1.5 times higher than the 588 kg/yr estimated for SOL, and is consistent with the difference in TP concentrations described in Chapter 2. The average outflow mass was estimated to be 154 kg/yr. The Nurnberg (1984) equation resulted in a sedimentation rate of 0.69. Annual calculations of in-lake mass determined by the measured TP concentration from November to November each year resulted in negative net loading (-566 kg) for 2007-2013, and this is congruent with the linear regression analyses which showed declining TP due to reductions in external loading and reduced lake productivity (Chapter 3). The estimated average loss to sediment is 82% of the total external load; however, it is likely that the outflows were underestimated, and additional loss of TP during winter lake-overflow would reduce the TP remaining in the lake. It is not unlikely that outflow mass was underestimated by an average 12% (or more).

The TN mass balance for SOL 2007-2013 (Table 4.6) estimates an average external TN load (land-based+atmospheric) of 17,110 kg/yr. Outflow volumes indicate that 37% of nitrogen flows out of the lake. The remaining nitrogen is “retained” through denitrification and sedimentation which are estimated to be 63% and 37% respectively (Saunders and Kalff 2001), and resulted in 40% of the total external load lost to denitrification, and an average 23% retained in the lake for sediment/water exchange and biological productivity. Measured TN concentration data yielded a 6-year negative net mass TN (-4251), as expected due to the significant decline found by linear regression analyses (Chapter 3). The estimated percentage of nitrogen lost via outflow (37%) exceeds the estimated percentage retained (23%), and consistent with findings from measured concentrations, no inter-annual accumulation of TN would be expected.

The TN mass balance for SHL 2007-2013 (Table 4.9) estimates an average external TN load (land-based+atmospheric) of 19,537 kg/yr. Outflow volumes indicate that 22% of nitrogen flows out of the lake, again this is likely underestimated due to underestimation of outflow volume. The remaining nitrogen is “retained” through denitrification and sedimentation which are estimated to be 63% and 37% respectively (Saunders and Kalff 2001), and resulted in 49% of the total external load lost to denitrification, and an average 29% retained in the lake for sediment/water exchange and biological productivity. Measured TN concentration data yielded a 6-year negative net mass TN (-4251), as expected due to the significant decline found by linear regression analyses (Chapter 3).

Tables 4.4, 4.6, 4.8, and 4.10 show the effective nutrient concentrations that correlate to the change in TP and TN mass. The 6-year average measured TP in SOL is

3.91 ug/L, and the effective net TP concentration from samples is 0.25 ug/L, which is essentially equal to zero (Table 4.4). The effective TP concentration from the MapShed© net load (external-losses) is 1.78 ug/L. This indicates that external loading would result in declining TP since the estimated annual effective concentration is half of the average in-lake TP concentration. Similarly the 6-year average effective TN concentration derived from MapShed© is lower (24.9 ug/L) than the average measured TN (81.5 ug/L), and also indicates declining TN.

The SHL effective TP concentration comparison (Table 4.6) resulted in higher values for the MapShed© net concentration (9.60 ug/L) than the 6-year average measured concentration (6.53 ug/L). Both external loading estimates and the MapShed© net concentration would indicate increasing TP in SHL, while the measured net indicates an extreme decrease (-7.87 ug/L). Without accurate outflow volume data, an accurate mass balance and effective TP concentrations cannot be determined, and this is evident in the results for SHL, which would indicate that TP concentrations should increase due to external loading. TN estimates of effective concentrations better correlated with what is known to be occurring in SHL. The effective TN concentration from MapShed© estimates is 77.7 ug/L, and the average measured TN is 199 ug/L. The difference indicates that TN is in decline, as supported by linear regression analyses.

St. Mary Lake

Previous linear regression analyses on the change in TP in SML from 2006-2014 showed no change in TP over time, with the exception of a decrease in summer Hypo-TP which is likely the result of aerator operation during the summers of 2009-2011 (Table 4.11). TP concentrations from 2006-2014 were not precipitation dependent. It was determined that the fluctuations in SML TP are primarily due to the internal loading of TP by the sediments which provides phosphorus to support an abundance of algae nearly all year. Increased water temperature and decreased dissolved oxygen conditions have allowed algae to persist for longer periods in the water column and this results in elevated "ambient" TP, as reflected in the increasing in-lake TP mass shown in Table 4.11.

The sedimentation rates (Nurnberg, 1984) for SML varied between 0.005 and 0.06, and very little sedimentation is expected for SML (Table 4.12). It is possible that atmospheric TP inputs are overestimated for SML because a reduction in external load would result in lower estimated percent loss to sediments. Known annual net algal cell concentrations are also provided in Table 4.12. Unfortunately, only three years of algae data were available, but the high annual net algal concentration (2,434 cells/mL) during 2010-2011 likely explains the elevated TP mass (644 kg) that remained in the water column during that year.

MapShed© external loading estimated 48.1 kg/yr from land-based sources and atmospheric loading was estimated to be 24.7 kg/yr, for an average external load of 72.9 kg/yr. Outflow volumes indicate that 66 kg/yr is lost via the Duck Creek outflow resulting in a net external load of 6.6 kg/yr. This is in agreement with the linear regression analyses which indicate no significant change in TP over time, and variations in TP due to internal loading and in-lake productivity. The effective concentration from load estimates is 0.42 ug/L compared to 6-year average measured TP of 28.3 ug/L. The effective loading from gross external load (prior to losses) is 4.58 ug/L and further illustrates the negligible effect of external loading on water quality in SML.

Transport Rates

The TP and TN transport rates estimated by MapShed© are presented for each land cover type, and the average by watershed, in Tables 4.14 and 4.15. As expected, overall estimated transport rates were lowest in SOL (8.74 kg/km^2 TP and 279 kg/km^2 TN), which has the most natural watershed. Transport rates estimated for the SHL watershed were 21.4 kg/km^2 TP and 357 kg/km^2 TN, and are higher than those in the SML watershed which were 13.1 kg/km^2 TP and 343 kg/km^2 TN. As discussed below, transport rates by land cover type are within the range of those found in the literature.

Reductions from Watershed Reforestation

Potential reductions in external loading were evaluated by altering the land cover used by the MapShed© model. SOL was not included in these analyses because the watershed is already in a relatively pristine condition, and was used as a control to evaluate the relative loading from the other watersheds. The 6-year average load estimates for SOL were 17,095 kg/yr of nitrogen and 509 kg/yr of phosphorus.

The SHL watershed was sub-divided into seven different sub-basins for MapShed© analysis based topography and drainage features (Figure 4.1). The MapShed© model was based on actual precipitation in 2004-2013 for each land cover scenario. The 10-year average load estimates for SHL based on current land cover were 18,345 kg/yr of nitrogen and 716 kg/yr of phosphorus. If all disturbed and open areas were reforested, TN loading would decrease by 2%, and TP loads would decrease by 3%. Evaluating the historic condition, in which the watershed was completely forested, with the precipitation amounts from 2004-2013 showed that anthropogenic disturbances in the watershed have increased TN loading by 6% and TP loading by 27%. Table 4.17 demonstrates the percent reductions in TN and TP in each sub-basin that would result from re-forestation of disturbed and open areas, and the hypothetical reductions that could be expected if the watershed were reverted to forest cover only. Only the reforestation of all disturbed and open areas is feasible as reverted private residential and commercial lands to forest is unlikely. Under this scenario, 0-3% reductions in TN per sub-basin could be expected. Anthropogenic activities have resulted in 24% and 11% increases in nitrogen loading in sub-basins SHL-3 and SHL-4 which contain the highest densities of urban development. Minimal reductions in TP would occur if all disturbed and open areas were reforested. Only sub-basin SHL-1 may benefit from reforestation which could reduced TP inputs by 8%. Anthropogenic activities have resulted in 43% and 66% increases in TP in sub-basins SHL-3 and SHL-4. The greatest opportunity for reforestation is available in sub-basin SHL-1, the headwaters of Shawnigan Creek; however, reforestation of this sub-basin would result in a 1.7% reduction in TN and 0.07% reduction relative to the entire watershed.

The SML watershed was sub-divided into five sub-basins for MapShed© analyses based topography and drainage features (Figure 4.2). The MapShed© model was based on actual precipitation in 2004-2014 for each land cover scenario. The 11-year average load estimates for SML based on current land cover were 1,695 kg/yr of nitrogen and 51 kg/yr of phosphorus. If the watershed were reforested such that only wetland, forest, and developed areas remained, TN loading would decrease by 34%, and TP loads would decrease by 14%. Evaluating the historic condition, in which the watershed was completely forested, with the precipitation amounts from 2004-2014 showed that anthropogenic disturbances in the watershed have increased TN loading by 44% and TP loading by 37%. Table 4.18 demonstrates the percent reductions in TN and TP in each sub-basin that would result if only wetland, forest, and developed areas remained in the watershed, and the hypothetical reductions that could be expected if the watershed were reverted to forest cover only. The majority of pasture, open, and disturbed lands are within SML-3. If these areas were reforested, TN loading would be reduced by 284 kg/yr, a 17% watershed-wide reduction. TP loading would be reduced by 3 kg/yr, a 9% watershed-wide reduction.

The ELK watershed was sub-divided into three sub-basins for MapShed© analyses. The MapShed© model was based on actual precipitation in 2004-2015 for each land cover scenario. The 12-year average load estimates for ELK based on current land cover were 1,321 kg/yr of nitrogen and 40 kg/yr of phosphorus. If all cropland in the watershed was reforested TN loading would decrease by 13%, and TP loads would decrease by 15%. Evaluating the historic condition, in which the watershed was completely forested, with the precipitation amounts from 2004-2015 showed that anthropogenic disturbances in the watershed have increased TN loading by 60% and TP loading by 38%. ELK-1 encompasses the O'Donnel Creek watershed, ELK-2 is the east side of the lake, and ELK-3 is the west side of the lake. ELK-1 contains the majority of cropland and pasture in the watershed, and ELK-3 contains the majority of urban areas. Comparisons of loading rates indicate that most nitrogen originates from sub-basin ELK-1, whereas most phosphorus originates from sub-basin ELK-2. This is in agreement with Glandon et al. (1981) and Wickham (2002) who determined that agricultural areas are the source of excessive nitrogen transport, whereas urban areas tend to produce excessive phosphorus. The best-case restoration scenario would be the reforestation of cropland and pasture in ELK-1, because developed areas cannot likely be modified. Reforestation of cropland alone could reduce TN and TP loading rates from Z1 by 13%. Reforestation of both cropland and pasture could reduce TN loading rates from Z1 by 60% and TP by 33%. Reforestation in Z1 only would result in TN reductions of 6-28% and TP reductions of 5-11% for the entire watershed (Table 4.19).

Discussion

Phosphorus Sedimentation Rates

Table 4.3 shows excellent consistency between the predicted TP sedimentation rate (Nurnberg, 1984) and the estimated loss to sediment in SOL. The estimated loss to sediment was calculated by assuming that all of the MapShed© net mass TP is retained in lake sediments. This supposition is supported by Carey and Rydin (2011) who found an increase in TP with depth in oligotrophic lake sediments, and also by the linear regression results presented in Table 4.2 where a definite decrease in TP is evident since 2006. Applying the sedimentation rate to the MapShed© net model-load (after subtracting the outflow mass) appears to work very well for an oligotrophic watershed with accurate outflow volume records, such as SOL. Even with the low sedimentation result for 2007-2008, the 6-year average result was 48% sedimentation to a predicted sedimentation rate of 0.49.

The discrepancy between the estimated loss to sediment and the sedimentation rates for SHL are likely due to underestimation of the annual outflow volumes. As described in the methods, winter outflow data were not available for SHL, and the outflows were estimated using only a graph of lake levels. If the outflow volumes were underestimated, then a greater mass of TP left the lake through Shawnigan Creek than is shown in Table 4.5. Since the percentage of phosphorus that leaves SHL is likely higher, the sedimentation percentage is lower, and would match the predicted sedimentation rates more closely. The Nurnberg (1984) equation for sedimentation rates (Equation 2) results in a rate of 0.83 for SHL when qs=0. This further supports the idea that the outflow volume for SHL is likely underestimated, especially in years when percent precipitation was used to adjust the final outflow volume, as outflow volume does not directly relate to precipitation when a linear regression test is run.

Equations used to predict sedimentation rates in lakes have been proved to be less accurate for eutrophic lakes (Nurnberg, 1984). Most meso/eutrophic lakes experience water temperature stratification in the summer when the deep water is colder than the surface, and becomes isolated from the oxygen exchange that occurs in the upper stratum. The effects of this stratification period are very complex and vary from year to year depending on the period of stratification, and the extent of productivity in the lake. However, the sedimentation rates (Nurnberg 1984) for SML varied between 0.005 and 0.06, which correspond well with MapShed© model rates between 2009 and 2013 (Table 4.12).

Nitrogen Balance

The nitrogen balances for SOL and SHL reflect the general assumption that the denitrification process is responsible for 63% of nitrogen “retention” in lakes, and sedimentation 37% (Saunders and Kalff 2001). After TN outflow, 63% of the remaining MapShed© estimated net mass TN is assumed to have been lost through denitrification and 37% retained for sediment/water exchange. For SOL, calculations resulted in an average 37% loss to outflow, 40% loss to denitrification, and 23% retained for sediment/water exchange (Table 4.7). The percent of TN lost to denitrification is higher (49%) in SHL (Table 4.9), but this factor is again influenced by the likely underestimation of lake outflows. Proportionally, however, because the in-lake TN concentration is known to be higher in SHL, we would expect a higher TN retention rate, and 29% retention seems congruent when compared to SOL.

Effective Nutrient Concentrations

The effective nutrient concentrations represent the effect that the incoming nutrient loads have on the overall water quality of the lakes. The effective concentrations from the MapShed© net TP and TN are higher than those derived from measured TP and TN net because they do not account for losses to sediment or denitrification. In SOL where TN and TP are decreasing (Table 4.2), the effective TP concentration calculated from the MapShed© net (Table 4.4) is approximately half of the average measured TP (1.78 and 3.91 ug/L respectively). As expected, this indicates that the incoming TP load will not contribute to increasing in-lake TP. Similarly for SOL TN, the effective TN concentration from the MapShed© net TN is 1/3 the average measured TN (24.9 and 81.5 ug/L respectively) (Table 4.8), and contributes to declining TN.

For SHL, again the underestimation of lake outflow data has influenced the outcome of the analyses. Linear regression statistics demonstrate a decline in TP since 2006 (Table 4.2), but the effective TP concentration from the MapShed© net (Table 4.6) is greater than the average measured TP. If the estimated loss to outflow is increased from an average 18% to 40%, the effective TP concentration is similar to the average measured TP, and would result in stable or declining TP. This emphasizes the need for accurate outflow data to accompany the MapShed© model. Interestingly, the average effective TN concentration calculated from the MapShed© net TN (77.7 ug/L) (Table

4.10) would indicate declining TN, although not at the same magnitude reflected in the measured net (-134 ug/L).

SML effective TP concentrations demonstrate that external loading is contributing a very small fraction of phosphorus to the lake when the outflow mass is considered (Table 4.12). This is confirmed by the linear regression data (Table 4.11) which show no precipitation dependence, and which indicate internal and ambient loading as the primary factors influencing water quality in SML.

Transport Rates

Nutrient transport rates have been estimated generally by land cover type, and specifically for SOL and Cusheon Lake near SML. Lin 2004 conducted a comprehensive review of nitrogen and phosphorus transport coefficients and found wide variation among studies conducted in various geographical areas. Beaulac and Reckhow (1982) summarized transport coefficients by very specific land cover types based on their previous published studies (Beaulac and Reckhow 1982). Zhu and Mazumder (2007) modeled nitrogen transport in the SOL watershed based on soil type and land cover. Table 16 summarizes the transport coefficients for the land cover types found in the SOL, SHL, and SML watersheds as referenced in the literature.

MapShed© estimated TP transport in SOL, SHL, and SML to be 8.74, 21.4, and 13.1 kg/km2 respectively (Table 4.14). TN transport in SOL, SHL, and SML was estimated to be 279, 357, and 343 kg/km^2 respectively (Table 4.15). The Cusheon Lake phosphorus estimate (Sprague 2007) was developed using the Dillon-Rigler-Hutchinson Cottage Country model from Ontario (Dillon et al. 1986). The MapShed© TP estimates are in agreement with the Sprague (2007) estimate of 10.6 kg/km^2 TP for the whole watershed given that compared to the nearly pristine SOL watershed, the Cusheon watershed has residential developments along the lakeshore, but is less densely developed than SHL, and Cusheon Lake has slightly higher forest cover (73% of the watershed). The MapShed© TN estimates are within the range of other estimates for forest land cover (286-400 kg/km^2), and that of the Hood Canal, WA watershed (Steinberg et al. 2011).

Land-Based Nutrient Reductions

Anthropogenic influences have increased TN loading in the SHL watershed by 6%, but much more in SML and ELK at 44% and 60% respectively. TP loading from anthropogenic influences was similar in SML and ELK at 37-38%, and somewhat less in SHL at 27%. Alteration of land cover in the watersheds yielded different potential reductions for each watershed. The net benefit of the most feasible restoration scenario in SHL would only reduce TN loading by 1.7% and TP loading by 0.07%. By comparison, additional loading reductions may be possible in SML. If pasture, open, and disturbed lands on the west side of the lake were reforested, TN loads from the watershed could be reduced by 17% and TP 9%; however, all lands are privately owned, and there is an active horse operation and large family farm occupying the non-forested lands. It is unlikely that complete reforestation would be possible. Similarly in the ELK watershed, a 28% reduction in TN loading and an 11% reduction in TP loading is possible only if all crop and pasture lands in ELK-1 were reforested. Again these lands are privately owned, and complete reforestation is not likely.

Required Model Inputs

MapShed© requires spatial (GIS) files for watershed basins, land use/cover, elevation (DEM), soils, streams, and weather station locations. Weather data files are required to be specifically formatted and saved in .csv format. Watershed basins can be delineated using a topographic overlay to establish drainage boundaries. The locations of weather stations can be obtained from Provincial databases, and data for the corresponding station ID can be found in the Government of Canada historical weather database. Land use/cover can be obtained from local governments and refined using aerial photography. Elevation, soils, and streams files can be obtained from online GIS archives. The MapShed© User Guide, available from the MapShed© website, specifies file attributes and formats, and explains how to further refine inputs and interpret outputs.

Accurate winter overflow volume data are important when using MapShed© to predict the effects of external loading on lake water quality trends. As discussed above, TP sedimentation predicted by the MapShed© mass balance is very close to that which was predicted by the Nurnberg (1984) formula in SOL where monthly outflow volumes were concise. Outflow nutrient concentration data is also necessary, but average monthly in-lake concentration data from one deep water point did provide a sufficient estimate of outflow mass in this study. Estimates of atmospheric inputs of TP by area are also needed since atmospheric loading accounted for around 15% of TP into SOL, 10% into SHL, and 34% into SML. Atmospheric TN is negligible on the rural coast, and is not necessary for the mass balance, but would account for a greater contribution nearer to urban/industrial areas.

Conclusion

The mass balances developed from MapShed© model estimates agreed with what is known about the trends of nutrients in SOL and SML. When all additional TP remaining in the lake after loss by outflow is assumed to be retained in the sediment, the sedimentation rates are in agreement with Nurnberg (1984) in most years. The effective nutrient concentrations indicate that external loading is not contributing to increased nutrient mass in the lakes as was demonstrated in Chapter 3. The error shown for SHL demonstrates the importance of monitoring lake-outflow volumes and nutrient concentrations in order to develop an accurate mass balance. If the SHL winter overflow volume is indeed higher than could be estimated in this study, the sedimentation rates and effective nutrient concentrations would most likely coincide with the known water quality trends and sedimentation rates. The MapShed© model, coupled with accurate outflow data gives a good indication as to the influence of external loading on in-lake nutrient concentrations.

MapShed© estimates indicate that watershed reforestation could reduce nutrient loading into the watersheds. The extent to which this would affect in-lake water quality is unknown. Given that most of the causal loads were determined to be due to in-lake processes (Chapter 3), and that there are land ownership constraints on possible reforestation efforts, land-based restoration may not be the most effective solution for reducing in-lake nitrogen and phosphorus concentrations.

Tables

Table 4.1. Summary of mass balance inputs for Sooke (SOL), Shawnigan (SHL), and St. Mary (SML) lakes. The date range of available data, lake volume (m^3), average total phosphorus (TP), average total nitrogen (TN), and average annual outflow (dam3) are shown for each lake for which a mass balance was calculated.

Lake	Data Range	Lake Volume for Mass	Avg. TP (ug/L)	Avg. TN (ug/L)	Avg. Annual Outflow (dam^3)
SOL	11/2007-11/2013	160.3 x 10^6 m^3	4.5	90.4	77,275
SHL	11/2006-11/2008 11/2010-11/2013	71.9 x 10^6 m^3	6.4	194	22,049
SML	11/2007-11/2014	Epi - 12.26 x 10^6 m^3 Hypo - 3.64 x 10^6 m^3	21.7 71.8	ND	2,694

Table 4.2. Summary of Linear Regression Results for Sooke (SOL) and Shawnigan (SHL) total phosphorus (TP) and total nitrogen (TN) trends. The change in TP is shown as decreasing or no change along with the p-value, R^2 value, and linear equation produced by the statistical analysis. For precipitation, linear regression analyses resulted in $p>0.05$ when TP or TN were not correlated to precipitation (N), and $p<0.05$ when TP or TN were correlated to precipitation (Y). The "Causal Load" is the result of analyses presented in Chapter 2.

Lake	Sample Years	Metric	Season	Change in TP over Time				Precipitation		Causal Load
				TP Status	P-Value	R^2	Linear Equation	Dependent?	R^2	
SOL	2006-13	Epi-TP	Winter	Decrease	p << 0.001	0.42	y=-0.03x+6	N	0.00	in-lake
			Summer	Decrease	p << 0.001	0.38	y=-0.03x+5	N	0.00	in-lake
	2006-08	Hypo-TP	Winter	Decrease	p = 0.041	0.31	y=-0.08x+6	N	0.00	in-lake
			Summer	No change	p = 0.445	0.04	y=-0.02x+4	N	0.03	ambient
	2006-13	Epi-TN	Winter	Decrease	p <<0.001	0.40	y=-0.6x+121	N	0.00	in-lake
			Summer	Decrease	p <<0.001	0.52	y=-0.6x+113	N	0.00	in-lake
SHL	2006-14	Epi-TP	Winter	Decrease	p <<0.001	0.45	y=-0.04x+8	Y	0.15	external
			Summer	Decrease	p = 0.003	0.27	y=-0.08x+9	N	0.03	in-lake
		Hypo-TP	Winter	No change	p = 0.191	0.05	y=-0.01x+7	N	0.00	ambient
			Summer	Decrease	p = 0.004	0.23	y=-0.04x+7	N	0.01	in-lake
	2006-13	Epi-TN	Winter	No change	p = 0.153	0.06	y=-0.4x+226	N	0.08	ambient
			Summer	No change	p = 0.145	0.06	y=-0.3x+193	N	0.01	ambient

Table 4.3. Sooke Lake annual mass balances from monthly total phosphorus (TP) concentration (ug/L) evaluated with MapShed© model outputs for estimated TP loading from the watershed. Net in-lake mass TP (kg/year) was calculated by subtracting the mass (kg) in November of year 2 (ex. 2008) from that of November of year 1 (ex. 2007). MapShed estimated load (kg/yr) was obtained from the model outputs, and atmospheric (Atm.) TP (kg/yr) was estimated at 13.6 kg/km^2 (Nordin 2015) and applied to the area of the lake surface (5.90 km^2). Outflow mass TP (kg/yr) was estimated based on in-lake TP concentrations and the outflow volume (dam^3). The predicted sedimentation rate derived from Equation 2 (Nurnberg 1984) represents the estimated percent loss to sediment. Loss to sediment (%) represents the percentage of TP that remains after the percent loss in outflow (based on Outflow Mass TP (kg/yr).

Year	**Ann. Avg. Precip. (mm)**	**Net In-Lake Mass TP (kg/yr)**	**MapShed Est. Load TP (kg/yr)**	**Total Ext. Load (kg/yr) (+79.6 kg/yr Atm. TP)**	**Outflow Mass TP (kg/yr)**	**MapShed Est. Net Mass TP[A] (kg/yr)**	**Outflow Vol. (dam^3)**	**Loss to Sed. %[C]**	**Predicted Sed. Rate (=15/(18+qs))**	**Loss in Outflow %**
2007-08	1,185	-68.7	416	496	350	146	66,761	29%[D]	0.51	71%
2008-09	1,211	106	377	456	221	236	53,982	52%	0.55	48%
2009-10	1,202	-60.8	585	664	309	355	80,070	54%	0.48	46%
2010-11	1,296	-177	614	694	380	313	97,640	45%	0.43	55%
2011-12	1,174	-188	582	661	323	338	85,559	51%	0.46	49%
2012-13	989	428	479	558	236	323	79,638	58%	0.48	42%
6-year Avg.	**1,176**	**39.5[B]**	**509**	**588**	**303**	**285**	**77,275**	**48%**	**0.49**	**52%**

[A]Calculated with measured outflow; MapShed Est. Net Mass TP = (MapShed Est. Load TP + Atmosph. TP) - Outflow Mass TP.
[B] Value represents 0.25 ug/L and with standard error of 0.1-0.5 for n=12, the value can be assumed equal to 0.
[C] %Loss to Sediment = Net Mass TP/Total Ext. Load.
[D] Given that there was a net negative mass in 2007-2008, if TP concentration were unknown, applying a 0.51 sedimentation rate would still yield a net loss, although the downstream loading affects could be underestimated.

Table 4.4. Sooke Lake effective total phosphorus (TP) concentrations. Effective TP concentrations (ug/L) were calculated using the annual net mass TP (kg) from measured in-lake concentrations, MapShed© external load estimates, and the MapShed© net (MapShed© TP (kg) in - Loss in Outflow) (from Table 4.3) divided by the lake volume to determine the resulting concentration in ug/L.

Year	Avg. Measured TP (ug/L)	Effective TP Concentration (ug/L)		
		From Measured Net	From MapShed© External Load	From MapShed© Net
2007-08	5.04	-0.43	3.09	0.91
2008-09	4.29	0.66	2.84	1.47
2009-10	3.93	-0.38	4.14	2.21
2010-11	3.30	-1.10	4.32	1.95
2011-12	2.11	-1.17	4.12	2.11
2012-13	4.78	2.67	3.48	2.01
6-year Avg.	**3.91**	**0.25**	**3.67**	**1.78**

Table 4.5. Shawnigan Lake annual mass balances from monthly total phosphorus (TP) concentration (ug/L) evaluated with MapShed© model outputs for estimated TP loading from the watershed. Net in-lake mass TP (kg/year) was calculated by subtracting the mass (kg) in November of year 2 (ex. 2008) from that of November of year 1 (ex. 2007). MapShed estimated load (kg/yr) was obtained from the model outputs, and atmospheric (Atm.) TP (kg/yr) was estimated at 13.6 kg/km^2 (Nordin 2015) and applied to the area of the lake surface (5.50 km^2). Outflow mass TP (kg/yr) was estimated based on in-lake TP concentrations and the outflow volume (dam^3). The predicted sedimentation rate derived from Equation 2 (Nurnberg 1984) represents the estimated percent loss to sediment. Loss to sediment (%) represents the percentage of TP that remains after the percent loss in outflow (based on Outflow Mass TP (kg/yr).

Year	**Ann. Avg. Precip. (mm)**	**Net In-Lake Mass TP (kg)**	**MapShed© Est. Load TP (kg)**	**Total Ext. Load (kg) (+74.8 kg Atm. TP)**	**Outflow Mass TP (kg)**	**MapShed© Est. Net Mass TP[A] (kg)**	**Outflow Vol. (dam^3)**	**Loss to Sed. % [B]**	**Predicted Sed. Rate (=15/(18+qs))**	**Loss in Outflow %**
2006-07	1,270	-60.4	865	940	218	722	24,322	77%	0.69	23%
2007-08	939	-300	616	691	142	549	20,269	79%	0.69	21%
Not Enough Continuous Data 2008-09/2009-2010										
2010-11	1,225	-76.6	879	954	158	796	23,471	83%	0.67	17%
2011-12	1,174	0	816	891	151	740	22,483	83%	0.68	17%
2012-13	1,025	-129	669	744	99	645	19,701	87%	0.70	13%
5-year	**1,127**	**-566**	**769**	**844**	**154**	**690**	**22,049**	**82%**	**0.69**	**18%**

[A]Calculated with measured outflow; MapShed Est. Net Mass TP = (MapShed Est. Load TP + Atmosph. TP) - Outflow Mass TP.
[B] %Loss to Sediment = Net Mass TP/Total Ext. Load.

Table 4.6. Shawnigan Lake effective total phosphorus (TP) concentrations. Effective TP concentrations (ug/L) were calculated using the annual net mass TP (kg) from measured in-lake concentrations, MapShed© external load estimates, and the MapShed© net (MapShed© TP (kg) in - Loss in Outflow) (from Table 4.3) divided by the lake volume to determine the resulting concentration in ug/L. It is likely that the outflow volumes were underestimated, and an additional 40% loss to outflow would result in an effective TP concentration similar to the measured average, which is expected since the water quality is not changing significantly over time.

Year	Avg. Measured TP (ug/L)	Effective TP Concentration (ug/L)			
		From Measured Net	From MapShed External Load	From MapShed Net	Assuming 40% Loss to Outflow
2006-07	8.97	-0.84	13.1	10.0	7.84
2007-08	6.68	-4.17	9.61	7.63	5.77
Not Enough Continuous Data 2008-09/2009-2010					
2010-11	6.37	-1.07	13.3	11.1	8.0
2011-12	5.72	0	12.4	10.3	7.44
2012-13	4.91	-1.79	10.3	8.97	4.14
6-year Avg.	**6.53**	**-7.87**	**11.7**	**9.60**	**6.64**

Table 4.7. Sooke Lake (SOL) annual mass balances from monthly total nitrogen (TN) concentration (ug/L) evaluated with MapShed© model outputs for estimated TP loading from the watershed. Net in-lake mass TN (kg/year) was calculated by subtracting the mass (kg) in November of year 2 (ex. 2008) from that of November of year 1 (ex. 2007). MapShed estimated load (kg/yr) was obtained from the model outputs, and atmospheric (Atm.) TN (kg/yr) was estimated at 2.64 kg/km^2 (Raymond et al. 2010) and applied to the area of the lake surface (5.90 km^2). Outflow mass TN (kg/yr) was estimated based on in-lake TN concentrations and the outflow volume (dam^3). The percentage of TN estimated to be retained for sediment (Sed.) water (H_2O) exchange is that which remains after the percent loss in outflow, and 63% is assumed to be lost due to denitrification (N_2) (Saunders and Kalff 2001).

Year	**Ann. Avg. Precip. (mm)**	**Net In-Lake Mass TN (kg)**	**MapShed © Est. Load TN (kg)**	**Total Ext. Load (kg) (+15.6 kg Atm. TN)**	**Outflow Mass TN (kg)**	**Loss to N_2 (kg)**	**MapShed© Est. Net Mass TN[A] (kg)**	**Outflow Vol. (dam^3)**	**Loss in Outflow %**	**Loss to N_2 %**	**% Retained for Sed./H_2O exchange**
2007-08	1,185	-5333	13856	13872	6604	4578	2689	66,761	48%	33%	19%
2008-09	1,211	2516	12393	12408	4133	5213	3062	53,982	33%	42%	25%
2009-10	1,202	-2222	19828	19843	7027	8074	4742	80,070	35%	41%	24%
2010-11	1,296	-485	20587	20602	8373	7705	4525	97,640	41%	37%	22%
2011-12	1,174	1328	19707	19722	6804	8139	4780	85,559	34%	41%	24%
2012-13	989	-55	16196	16211	5121	6987	4104	79,638	32%	43%	25%
6-year Avg.	**1,176**	**-4251**	**17095**	**17110**	**6344**	**6783**	**3984**	**77275**	**37%**	**40%**	**23%**

[A]Calculated with measured outflow; MapShed Est. Net Mass TN = (MapShed Est. Load TN + Atmosph. TN) - (Outflow Mass TN+Denitrification)
[B] Value represents 0.25 ug/L and with standard error of 0.1-0.5 for n=12, the value can be assumed equal to 0.
[C] %Loss to Sediment = Net Mass TP/Total Ext. Load.
[D] Given that there was a net negative mass in 2007-2008, if TP concentration were unknown, applying a 0.51 sedimentation rate would still yield a net loss, although the downstream loading affects could be underestimated.

Table 4.8. Sooke Lake effective total nitrogen (TN) concentrations. Effective TN concentrations (ug/L) were calculated using the annual net mass TN (kg) from measured in-lake concentrations, MapShed© external load estimates, and the MapShed© net (MapShed© TN (kg) in - Loss in Outflow) (from Table 4.3) divided by the lake volume to determine the resulting concentration in ug/L.

Year	Avg. Measured TN (ug/L)	Effective TN Concentration (ug/L)		
		From Measured Net	From MapShed External Load	From MapShed Net
2007-08	95.6	-33.3	86.4	16.8
2008-09	76.7	15.7	77.3	19.1
2009-10	84.7	-13.9	124	29.6
2010-11	82.0	-3.0	128	28.2
2011-12	77.8	8.3	123	29.8
2012-13	72.2	-0.34	101	25.6
6-year Avg.	81.5	-26.5	107	24.9

Table 4.9. Shawnigan Lake (SHL) annual mass balances from monthly total nitrogen (TN) concentration (ug/L) evaluated with MapShed© model outputs for estimated TP loading from the watershed. Net in-lake mass TN (kg/year) was calculated by subtracting the mass (kg) in November of year 2 (ex. 2008) from that of November of year 1 (ex. 2007). MapShed estimated load (kg/yr) was obtained from the model outputs, and atmospheric (Atm.) TN (kg/yr) was estimated at 2.64 kg/km^2 (Raymond et al. 2010) and applied to the area of the lake surface (5.90 km^2). Outflow mass TN (kg/yr) was estimated based on in-lake TN concentrations and the outflow volume (dam^3). The percentage of TN estimated to be retained for sediment (Sed.) water (H_2O) exchange is that which remains after the percent loss in outflow, and 63% is assumed to be lost due to denitrification (N_2) (Saunders and Kalff 2001).

Year	Ann. Avg. Precip. (mm)	Net In-Lake Mass TN (kg)	MapShed Est. Load TN (kg)	Total Ext. Load (kg) (+15.6 kg Atm. TN)	Outflow Mass TN (kg)	Loss to N_2 (kg)	MapShed Est. Net Mass TN[A] (kg)	Outflow Vol. (dam^3)	Loss in Outflow %	Loss to N_2 %	% Retained for Sed./H_2O exchange
2006-07	1270	-5173	22759	22774	6167	10463	6145	24322	27%	46%	27%
2007-08	939	-6870	14904	14919	2848	7604	4466	20269	19%	51%	30%
Not Enough Continuous Data 2008-09/2009-2010											
2010-11	1225	1458	22141	22155	5124	10730	6302	23471	23%	48%	28%
2011-12	1174	766	20924	20938	4527	10339	6072	22483	22%	49%	29%
2012-13	1025	156	16883	16897	3505	8437	4955	19701	21%	50%	29%
6-year Avg.	**1127**	**-9663**	**19522**	**19537**	**4434**	**9515**	**5588**	**22049**	**22%**	**49%**	**29%**

[A]Calculated with measured outflow; MapShed Est. Net Mass TN = (MapShed Est. Load TN + Atmosph. TN) - (Outflow Mass TN+Denitrification)

[B] Value represents 0.25 ug/L and with standard error of 0.1-0.5 for n=12, the value can be assumed equal to 0.

[C] %Loss to Sediment = Net Mass TP/Total Ext. Load.

[D] Given that there was a net negative mass in 2007-2008, if TP concentration were unknown, applying a 0.51 sedimentation rate would still yield a net loss, although the downstream loading affects could be underestimated.

Table 4.10. Shawnigan Lake effective total nitrogen (TN) concentrations. Effective TN concentrations (ug/L) were calculated using the annual net mass TN (kg) from measured in-lake concentrations, MapShed© external load estimates, and the MapShed© net (MapShed© TN (kg) in - Loss in Outflow) (from Table 4.3) divided by the lake volume to determine the resulting concentration in ug/L.

Year	Avg. Measured TN (ug/L)	Effective Concentration (ug/L)		
		From Measured Net	From MapShed External Load	From MapShed Net
2006-07	247	-71.9	317	85.5
2007-08	183	-95.5	207	62.1
Not Enough Continuous Data 2008-09/2009-2010				
2010-11	204	20.3	308	87.6
2011-12	205	10.7	291	84.5
2012-13	156	2.17	235	68.9
6-year Avg.	**199**	**-134**	**271**	**77.7**

Table 4.11. Summary of Linear Regression Results for St. Mary (SML) total phosphorus (TP) trends. The change in TP is shown as decreasing or no change along with the p-value, R^2 value, and linear equation produced by the statistical analysis. For precipitation, linear regression analyses resulted in $p>0.05$ when TP was not correlated to precipitation (N), and $p<0.05$ when TP was correlated to precipitation (Y). The "Causal Load" is the result of analyses presented in Chapter 2.

Lake	**Sample Years**	**Metric**	**Season**	**Change in TP over Time**				**Precipitation**		**Causal Load**
				TP Status	**P-Value**	**R^2**	**Linear Equation**	**Dependent?**	**R^2**	
SML	2005-14	Epi-TP	Winter	No change	p = 0.711	0.00	y=0.03x+33	N	0.04	ambient
			Summer	Increase	p = 0.035	0.08	y=0.14x+14	N	0.01	internal
	2006-14	Hypo-TP	Winter	No change	p = 0.130	0.05	y=0.13x+26	N	0.00	ambient
			Summer	Decrease	p = 0.001	0.17	y=-1.7x+222	N	0.00	internal

Table 4.12. St. Mary Lake annual mass balances from monthly total phosphorus (TP) concentration (ug/L) evaluated with MapShed© model outputs for estimated TP loading from the watershed. Net in-lake mass TP (kg/year) was calculated by subtracting the mass (kg) in November of year 2 (ex. 2008) from that of November of year 1 (ex. 2007). MapShed estimated load (kg/yr) was obtained from the model outputs, and atmospheric (Atm.) TP (kg/yr) was estimated at 13.6 kg/km^2 (Nordin 2015) and applied to the area of the lake surface (5.90 km^2). Outflow mass TP (kg/yr) was estimated based on in-lake TP concentrations and the outflow volume (dam^3). The predicted sedimentation rate derived from Equation 2 (Nurnberg 1984) represents the estimated percent loss to sediment. Loss to sediment (%) represents the percentage of TP that remains after the percent loss in outflow (based on Outflow Mass TP (kg/yr).

Year	Prec. (mm)	Annual Average		Net In-Lake Mass TP[A] (kg)	MapShed Est. Load TP (kg)	Total Ext. Load (kg) (+24.7 kg Atm. TP)	Outflow Mass TP (kg)	MapShed Est. Net Mass TP[B] (kg)	Outflow Vol. (dam^3)	Net Algae cells/mL	Ann. Ext. Load %	
		Epi-TP (ug/L)	Hypo-TP (ug/L)								Loss to Sed.[C]	Loss in Outflow
2007-08	791	21.7	71.8	15	43.4	68.2	38.2	30.0	1,920	ND	22%	56%
2008-09	909	20.2	20.6	**-168**	47.7	72.5	32.3	40.2	1,736	145	55%	45%
2009-10	973	14.5	20.8	**-102**	49.7	74.5	65.5	8.9	4,984	232	12%	88%
2010-11	904	31.2	41.6	**644**	56.1	80.8	83.4	-2.6	3,370	**2,434**	0%	100%
2011-12	907	48.1	69.7	72	47.8	72.6	80.5	-9.0	1,630	-1,128	0%	100%
2012-13	896	40.1	80.1	162	51.2	76.0	119.6	-43.6	3,076	ND	0%	100%
2013-14	827	22.2	53.9	-446	40.7	65.4	43.2	22.3	2,144	ND	34%	66%
7-year	887	28.3	51.2	**177**	48.1	72.9	66.1	6.6	2,694	ND	18%	79%

[A]Net Mass TP = Net Mass Epi-TP+Net Mass Hypo-TP, from Nov.-Nov., where Epi volume=12.26x10^6 m^3 and Hypo volume=3.64x10^6 m^3.
[B]Calculated with measured outflow; MapShed Est. Net Mass TP = (MapShed Est. Load TP + Atmosph. TP) - Outflow Mass TP.
[C] %Loss to Sediment = Net Mass TP/Total Ext. Load.

Table 4.13. St. Mary Lake effective total phosphorus (TP) concentrations. Effective TP concentrations (ug/L) were calculated using the annual net mass TP (kg) from measured in-lake concentrations, MapShed© external load estimates, and the MapShed© net (MapShed© TP (kg) in - Loss in Outflow) (from Table 4.3) divided by the lake volume to determine the resulting concentration in ug/L.

Year	**Avg. Measured TP (ug/L)**	**Effective TP Concentration (ug/L)**		
		From Measured Net	**From MapShed External Load**	**From MapShed Net**
2007-08	21.7	0.96	4.29	1.89
2008-09	20.2	-10.6	4.56	2.53
2009-10	14.4	-6.42	4.69	0.56
2010-11	31.2	40.5	5.08	-0.16
2011-12	48.1	4.53	4.57	-0.57
2012-13	40.1	10.2	4.78	-2.74
2013-14	22.2	-28.1	4.11	1.40
6-year Avg.	**28.3**	**11.1**	**4.58**	**0.42**

Table 4.14. MapShed© 2007-2014 average total phosphorus transport rates by watershed and land cover type. The percent cover by area and TP transport rates (in kg/km^2 and kg/hectare (HA)) utilized by the model are shown for Sooke (SOL), Shawnigan (SHL), and St. Mary (SML) lakes.

	SOL			SHL			SML		
Land Use	% Cover	Kg/Km^2	Kg/HA	% Cover	Kg/Km^2	Kg/HA	% Cover	Kg/Km^2	Kg/HA
Hay/Pasture	---	---	---	---	---	---	8.1%	13.5	0.135
Forest	91%	7.65	0.077	44%	7.3	0.073	69%	7.2	0.072
Wetland	0.1%	8.82	0.088	0.6%	8.5	0.085	1.5%	7.1	0.071
Disturbed	0.3%	9.40	0.094	2.6%	9.4	0.094	0.7%	8.7	0.087
Open Land	8.5%	9.07	0.091	37%	10.4	0.104	0.4%	11.0	0.110
Med./High Density Mixed	---	---	---	0.03%	60.8	0.608	0.2%	31.3	0.313
Low Density Residential	---	---	---	12%	16.7	0.167	20%	13.2	0.132
Med. Density Residential	---	---	---	3.7%	36.6	0.366	0.2%	---	---
Average Total Phosphorus Load		**8.74**	**0.087**		**21.4**	**0.214**		**13.1**	**0.131**

Table 4.15. MapShed© 2007-2014 average total nitrogen transport rates by watershed and land cover type. The percent cover by area and TN transport rates (in kg/km^2 and kg/hectare (HA)) utilized by the model are shown for Sooke (SOL), Shawnigan (SHL), and St. Mary (SML) lakes.

	SOL			SHL			SML		
Land Use	% Cover	Kg/Km^2	Kg/HA	% Cover	Kg/Km^2	Kg/HA	% Cover	Kg/Km^2	Kg/HA
Hay/Pasture	---	---	---	---	---	---	8.1%	305	3.05
Forest	91%	254	2.54	44%	225	2.25	69%	282	2.82
Wetland	0.1%	276	2.76	0.6%	248	2.48	1.5%	294	2.94
Disturbed	0.3%	251	2.51	2.6%	223	2.23	0.7%	285	2.85
Open Land	8.5%	333	3.33	37%	311	3.11	0.4%	363	3.63
Med./High Density Mixed	---	---	---	0.03%	731	7.31	0.2%	526	5.26
Low Density Residential	---	---	---	12%	309	3.09	20%	344	3.44
Med. Density Residential	---	---	---	3.7%	455	4.55	0.2%	---	---
Average Total Nitrogen Load		**279**	**2.79**		**357**	**3.57**		**343**	**3.43**

Table 4.16. Nutrient transport rates from literature. Transport rates for total nitrogen (TN) and total phosphorus (TP) found in scientific literature area summarized by land cover type.

Land Cover Type	**TN kg/km^2**	**TP kg/km^2**	**Source**
Hay/Pasture	152	25	(Beaulac and Reckhow 1982)
Forest - 0-20 years	400	ND	(Zhu and Mazumder 2008)
Forest (general)	286	24	(Lin 2004)
Wetland-forested	460	ND	(Zhu and Mazumder 2008)
Wetland-herbaceous	96	4.3	(Glandon et al. 1981)
Disturbed	126	ND	(Zhu and Mazumder 2008)
Urban (general)	997	191	(Lin 2004)
Cultus Lake, BC watershed	576	24	(Putt 2014)
Cusheon Lake, BC watershed	ND	10.6	(Sprague 2007)
Hood Canal, WA watershed	330	ND	(Steinberg et al. 2011)

Table 4.17 Percent reduction in nitrogen and phosphorus loading (kg/year) by land cover scenario for the Shawnigan watershed. Reduction percentages were estimated for each sub-basin by comparing the model outputs of current land use conditions to that of a reforested watershed with no disturbed (Dist.) or Open areas, and a hypothetical completely reforested condition (Forest Only).

	C1	C2	C3	C4	C5	C6	C7
Nitrogen Reductions							
No Dist./Open	-3%	-2%	0%	-1%	-1%	-3%	-2%
Forest Only	-1%	-4%	-24%	-11%	-2%	-4%	-3%
Phosphorus Reductions							
No Dist./Open	-8%	-3%	+1%	-0%	-0%	+2%	-1%
Forest Only	-9%	-26%	-66%	-43%	-11%	-26%	-20%

Table 4.18 Percent reduction in nitrogen and phosphorus loading (kg/year) by land cover scenario for the St. Mary watershed. Reduction percentages were estimated for each sub-basin by comparing the model outputs of current land use conditions to that of a reforested watershed with no disturbed (Dist.) or Open areas, and a hypothetical completely reforested condition (Forest Only).

	C1	C2	C3	C4	C5
Nitrogen Reductions					
No Dist./Open	17%	34%	58%	0%	10%
Forest Only	8%	12%	16%	36%	30%
Phosphorus Reductions					
No Dist./Open	8%	19%	29%	0%	2%
Forest Only	14%	17%	29%	62%	42%

Table 4.19 Percent reduction in total nitrogen (TN) loading (kg/yr) by land cover scenario for the Elk watershed if sub-basin ELK-1 were reforested. Reduction percentages were estimated based on the reforestation of crop and crop + pasture in sub-basin ELK-1, and estimating the resulting watershed-wide reduction.

	TN kg/yr	% TN Reduction	TP kg/yr	% TP Reduction
Current Land Cover	2775	---	132	---
No Crop	2597	6%	126	5%
No Crop or Pasture	1987	28%	117	11%

Figures

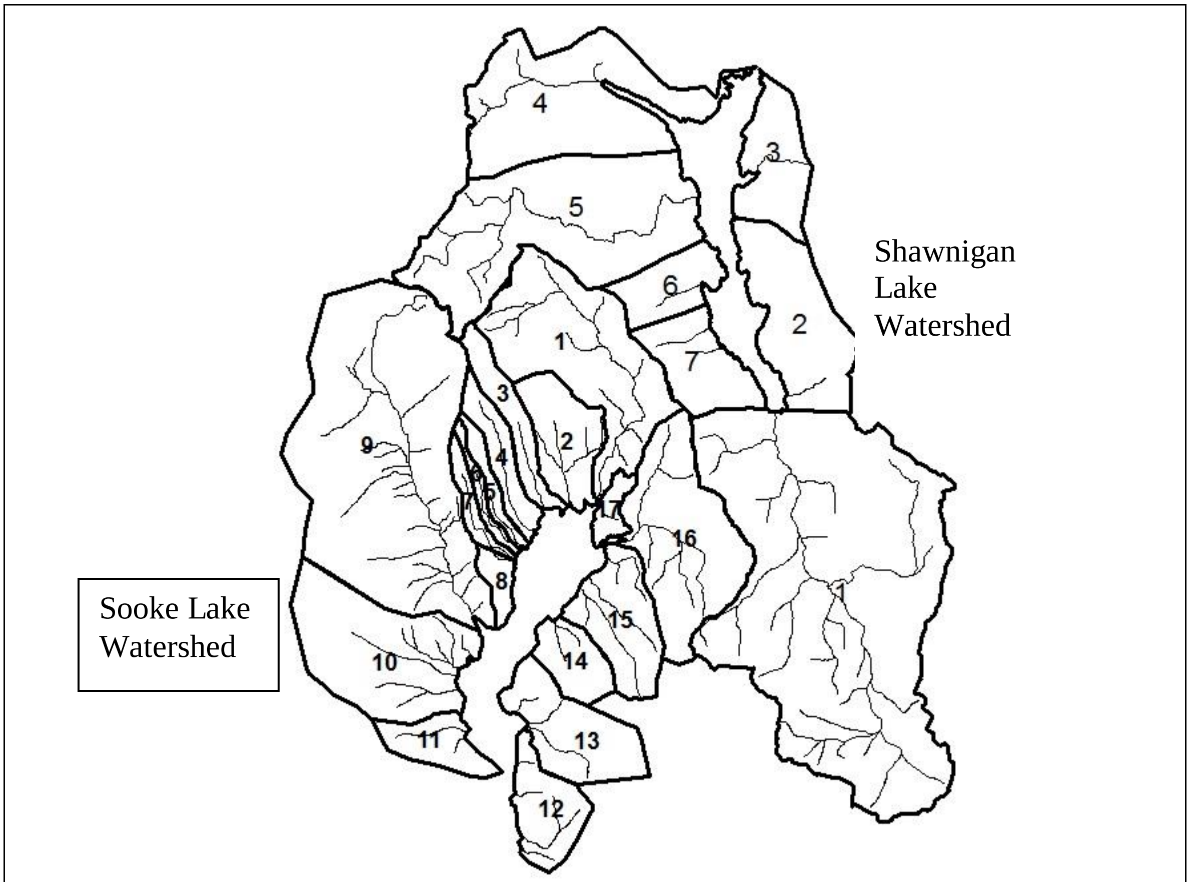


Figure 4.1 Sooke (SOL) and Shawnigan (SHL) watersheds and sub-basins. The SOL watershed was divided into 17 sub-basins, and the SHL watershed was divided into 7 sub-basins for analyses based on topography and drainage patterns.

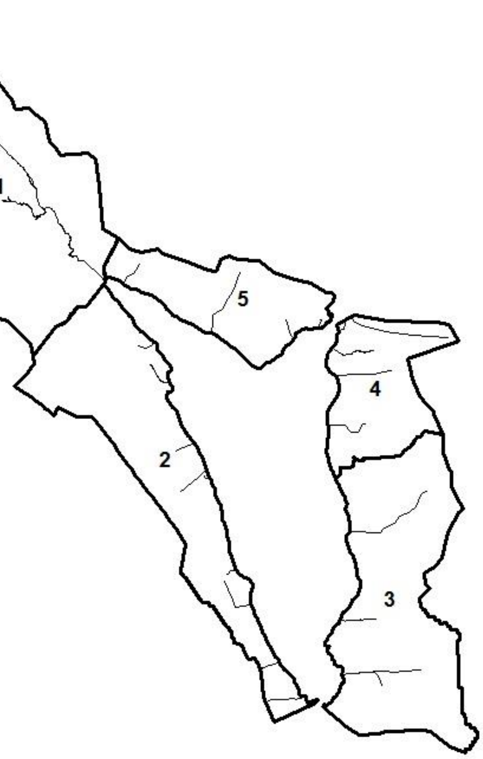


Figure 4.2 St. Mary Lake (SML) watershed and sub-basins. The St. Mary watershed was divided into 5 sub-basins for analyses based on topography and drainage patterns.

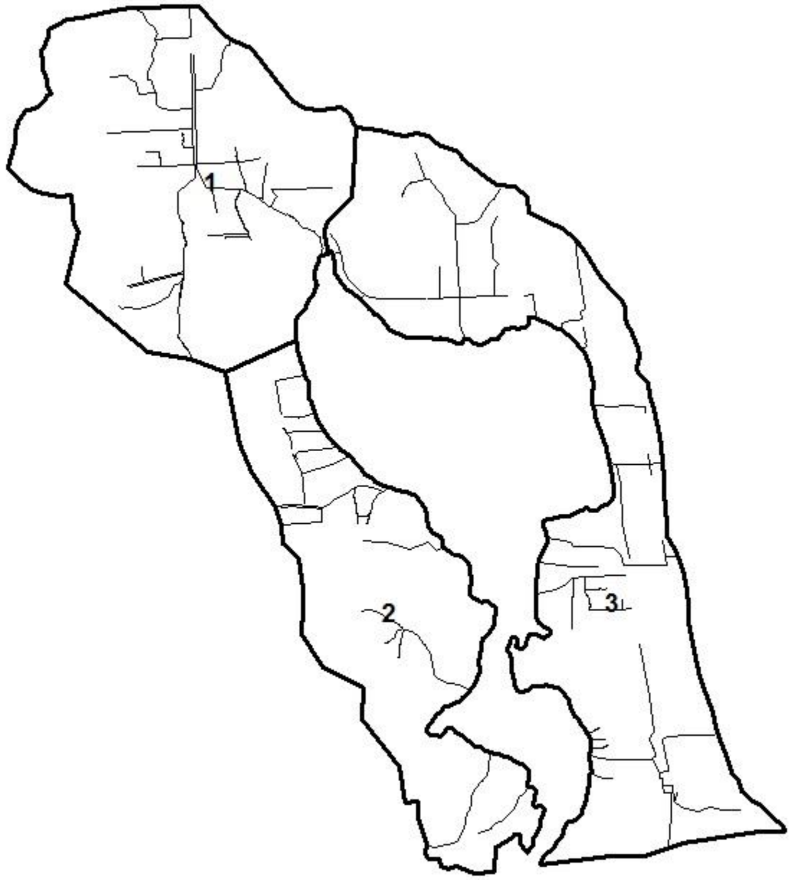


Figure 4.3 Elk Lake (ELK) watershed and sub-basins. The Elk watershed was divided into 3 sub-basins for analyses based on topography and drainage patterns. For this watershed, the MapShed© program required larger basins. This may be due to very minor changes in elevation around Elk Lake.

Chapter 5: Conclusions - summary and synthesis

Research Objectives

The objectives of my research were to evaluate the trends of nutrient concentrations in the subject watersheds, to determine the impacts of watershed land-uses, and to evaluate the benefits that land-based restoration may have for lake water quality. In *Chapter 2*, I compared nitrogen and phosphorus for Sooke and Shawnigan Lakes, and qualitatively evaluated watershed characteristics to determine if natural components (i.e., precipitation, topography, and soils) could account for differences in lake nutrient concentrations. In *Chapter 3*, I performed linear regression analyses to determine changes in lake nutrient concentrations over time, and nutrient-precipitation dependence. In *Chapter 4*, I evaluated the results of watershed external load estimates from MapShed© model outputs for applicability to watersheds with minimal in-lake samples. I modeled the most feasible Land-use scenarios to determine potential benefits from land-based restoration efforts.

Research Conclusions

My research confirmed that changes in land use have affected the water quality in Shawnigan, St. Mary, and Elk Lakes. Watershed development has increased the transport of both nitrogen and phosphorus into these lakes over the estimated historic (forested) condition. Declines in new development since at least the 1970’s have enabled the lakes to experience phases of decline, and to maintain stable-state conditions over the past 10 to 30 years. The significant decline in both nitrogen and phosphorus, and the increasing N:P trends in Sooke and Shawnigan Lakes indicate that the watersheds are still experiencing some recovery from past commercial forestry disturbances, and are trending toward declining lake productivity. Phosphorus reserves in St. Mary Lake, coupled with warmer year-round water temperatures are contributing to greater variation in winter phosphorus levels, which is perpetuated by algal productivity even during the winter.

Operation of the aeration system in the summer has reduced sediment-phosphorus loading to the hypolimnion. Internal cycling of phosphorus reserves is the primary contributor to increased phosphorus in the water columns in both St. Mary and Elk Lakes, and this perpetuates a cycle of in-lake productivity. Lake management in these two productive lakes should focus on reducing internal cycling through actions which reduce internal phosphorus loading. Measures which would balance the in-lake food-web to reduce algal abundance, coupled with maintaining the lowest in-lake phosphorus concentration possible will reduce the frequency and toxicity of algae blooms in these lakes.

The mass balances developed from MapShed© model estimates agreed with what is known about the trends of nutrients in SOL and SML. When all additional TP remaining in the lake after loss by outflow is assumed to be retained in the sediment, the sedimentation rates are in agreement with Nurnberg (1984) in most years. The effective nutrient concentrations indicate that external loading is not contributing to increased nutrient mass in the lakes as was demonstrated in Chapter 3. The error shown for SHL demonstrates the importance of monitoring lake-outflow volumes and nutrient concentrations in order to develop an accurate mass balance. If the SHL winter overflow volume is indeed higher than could be estimated in this study, the sedimentation rates and effective nutrient concentrations would most likely coincide with the known water quality trends and sedimentation rates. The MapShed© model, coupled with accurate outflow data gives a good indication as to the influence of external loading on in-lake nutrient concentrations.

For SOL and SHL, I determined that the lake is likely to maintain only moderate productivity given the current trends in N:P ratios. Other water quality concerns such as turbidity and fecal contamination should therefore be the focus of future research. The results of my research agreed with the work of others addressing the same questions for SML and ELK. Because external TP load reductions would not likely substantially affect in-lake TP concentrations, internal mechanisms need to be addressed if the frequency and duration of algal blooms in these lakes is to be reduced. Future research should focus on the potential for in-lake aeration to reduce internal phosphorus loading over the long-term, and what other in-lake restoration options are available. The ecology of freshwater algae is still not well-understood, and additional research into physiology and population dynamics would be useful for the long-term management of algal blooms.

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