INVESTIGATIVE SURVEY OF STORMWATER LOADING – NITROGEN AND PHOSPHORUS IMPACTS ON URBAN EUTROPHIC LAKE: LONG LAKE, LACEY, WASHINGTON

by

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ABSTRACT

Investigative Survey of Stormwater Loading – Nitrogen and Phosphorus Impacts on Urban Eutrophic Lake: Long Lake, Lacey, Washington

Jessica K. Converse

Harmful algal blooms are increasing in prevalence and duration worldwide as a result of global environmental perturbations including nutrient enrichment and climatic changes. Such is the case for Long Lake, one in a chain of three lakes located in Lacey, Washington. This urban, eutrophic lake experiences summer long closures due to cyanotoxins released from bloom-forming cyanobacteria. As a result, the Long Lake Management District ordered an investigative study be completed to assess the nutrient contribution of stormwater flowing into Long Lake. Grab samples were collected from multiple sites around the lake during three storm events and two non-storm events between November 2020 and March 2021. Samples were analyzed for total nitrogen (TN), total phosphorus (TP), and soluble reactive phosphorus (SRP) due to the influence these nutrients have on primary production and cyanobacterial blooms. Results indicated that the earlier seasonal storm events (November, December) transported higher concentrations of nutrients into the lake rather than later seasonal storms (February). Additionally, the highest concentrations of TP and SRP were delivered during storm events from the storm drain outfalls. Measured between January and March 2021, dissolved oxygen (DO) readings were highest at the surface of the lake. However, several meters below the surface DO levels were near 0 mg/L demonstrating that Long Lake was stratified. It is uncommon for the lake to be stratified during the winter months and could suggest that the lake’s surface temperature is warming earlier in the year. Conductivity readings differed between Long Lake’s two basins with the south basin reading higher in conductivity than the north. The Long Lake Management District would benefit from further study of stormwater nutrients in the form of nitrates and nitrites as those nutrients were not analyzed for during this study. Furthermore, DO and temperature readings should be conducted throughout the year to better understand Long Lake’s nutrient turnover rate.

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I would not be here without the loving support of all the friends who believe in me, (and Jennifer, thank you for feeding this college student!) I aim to give back to the world what has been so freely given to me.

Thank you.

1. INTRODUCTION

Due to the regular and increasing presence of cyanobacterial caused harmful algal blooms (henceforth referred to as cyanoHABs) in Long Lake (Lacey, WA), a research study was conducted to see if the limiting nutrients for these bacteria are being deposited through stormwater runoff. The nutrients to be analyzed, nitrogen and phosphorus, are regularly occurring within this environment. Long Lake is a eutrophic and highly productive lake (Thurston County Environmental Health Division, 2019). High concentrations of nutrients are present as a result of the historical dumping of logging pulp into the lake (Cracknell, P., personal communication, September 2020) as well as from the subsequent effects of urban development around the lake. Increased sediment transport due to deforestation, construction and impervious surfaces contribute nutrients into the lake that would otherwise be sequestered on land (Lathrop et al., 1998; Smith et al., 2020; Yang & Lusk, 2018).

Since 1989, Long Lake's water quality has been visibly deteriorating. Largely as a result of aquatic plant growth, the lake has become increasingly colonized by native and invasive species (Entranco, 1994). Additionally, increased nutrients, warmer temperatures, sunlight, and reduced water flow provide the optimal growing conditions for freshwater cyanobacteria (Ho & Michalak, 2015; Jacoby et al., 2000). Cyanobacteria, also known as "blue-green algae" have been attributed to the toxic algal blooms in Long Lake based on the cyanotoxins present including *microcystin*, *anatoxin-a*, *cylindrospermopsin*, and *saxitoxin* (Washington State Freshwater Algae Control Program, 2020). While not every algal bloom is toxic, data collected from Long Lake over the past decade show that toxic algal blooms are becoming increasingly regular with six years out of the past decade (2010-2020) testing above recommended limits for the cyanotoxin, *microcystin* and one year (2016) testing above recommended limits for *anatoxin-a* (WADOE 2020).

Microcystin is a type of cyanotoxin which can impair liver function and may be carcinogenic (Carmichael, 1991; Corbel et al., 2014). *Anatoxin-a* is a neurotoxin which disrupts signaling at nervous and neuromuscular junctions, causing paralysis which can lead to respiratory failure and death (Aronstam & Witkopt, 1981). Cyanotoxins can be lethal to livestock and pets that drink affected waters, and numerous fish and bird kills have been attributed to toxic bacteria (US EPA, 2013). The algal blooms themselves look like a green "scum" or paint spilled across the surface. When extremely dense, as it was last summer on Long Lake (2020), the blooms can become thick enough to block sunlight from reaching aquatic plants below the water's surface (Wehr et al., 2015).

For example, this past summer (2020), cyanobacteria proliferated the lake's warm, nutrient rich waters producing thick mats of malodourous, green "scum" on its surface (Cracknell, 2020). Whether lake residents would choose to swim in such conditions or not, Long Lake residents were prohibited from entering the lake due to the presence of the cyanobacterial hepatotoxin, *microcystin*, the same toxin that has precluded recreation at some point in Long Lake for most summers over the past decade (Ecology, 2020). It is still unknown precisely under what conditions cyanobacteria will produce their toxins, however warmer temperatures and high levels of nutrients, specifically nitrogen (N) and phosphorus (P), have been noted as precursors to the blooms themselves (Ho & Michalak, 2015; Jacoby et al., 2000). Understanding how to adjust to and possibly ameliorate cyanoHABs is a priority for those concerned with water quality and has become the top priority for Long Lake's Lake Management District (LLMD).

The process of nutrient cycling within this waterbody is complex, and the LLMD has prioritized known areas of concern for nutrient enrichment including invasive species management, septic system upkeep, and lawn fertilizer runoff (Lake Management District #21, 2017). However, stormwater outfalls have yet to be taken into account. This study aims to answer whether or not stormwater is a contributing source of total nitrogen (TN), total phosphorus (TP), or orthophosphate (PO_4^{3-}) for the Long Lake system during the winter storm period. These nutrients were selected due to their influence on cyanobacteria growth (Beversdorf et al., 2017; Bhateria & Jain, 2016). This investigative survey will compare water samples collected from stormwater outfalls around the basin to those collected in the lake to understand how significant an issue stormwater pollution may have on this waterbody.

The subsequent literature review will address nitrogen and phosphorus cycling in lake ecosystems, discuss some of the impacts land-use has on nutrient cycling in urban-eutrophic lakes, the role of stormwater outfalls as nutrient inputs, what is currently known about the health risks associated with cyanoHAB exposure, as well as how the LLMD has treated toxic blooms in the past. Additionally, watershed management practices that show promise for curbing cyanobacteria growth will be posited for their utility to the Long Lake drainage basin.

2. LITERATURE REVIEW

2.1 Introduction

Long Lake (47.02°N, 122.77°W) is the third and largest basin in a chain of four lakes located in Lacey, Washington and provides recreational, aesthetic, and groundwater filtration benefits to the residents throughout its drainage basin. Initially formed through glaciation, the lakes are connected by extensive wetlands draining to Woodland Creek, one of five major tributaries to Henderson Inlet at the southern terminus of the Puget Sound estuary (Thurston County, 1995a). The lakes are hydraulically connected by surface and subsurface flow ways beginning at the headwaters of Hicks Lake which flows into Pattison Lake, then to Long Lake through a series of wetlands, and finally to Lake Lois via Woodland Creek, formed at the outlet of Long Lake. All four lakes lie between 40 – 48 meters above sea level (United States Geological Survey, 1989). The surface area of Long Lake is approximately 8.25 square miles with mean and maximum depths of 3.7 meters (12 feet) and 6.4 meters (21 feet), respectively (Thurston County Environmental Health Division, 2019).

As much of the wetland structure of this area has been replaced by fill soil displaced by urban development, the natural ability of this chain of lakes to minimize nutrient concentrations has been substantially altered over time (Thurston County, 1995a). Wetlands provide essential ecosystem services to the area including the removal of sediment and pollution from surface water, groundwater filtration, and habitat for birds, insects, and spawning salmonoids (Thurston County, 1995a). Suspended sediment transported through the wetlands bring essential minerals into the lake ecosystem. This sediment either settles on the lake bottom or remains suspended in the water column, sustaining organismal growth and nutrient cycling within the lake system (Feng et al., 2020). Anthropogenic activities can influence nutrient cycling most notably through land-use practices. By covering landscapes with impervious surfacing or diverting and channeling water for industrial and agricultural purposes, the natural hydrologic processes of filtration are impacted. This allows associated runoff to flow directly into our waterways increasing their overall nutrient load (Conway & Lathrop, 2005; Dodson et al., 2007). The excess nutrients encourage plant and algal productivity which inevitably reduces clear water for drinking, swimming, boating, and fishing.

Trophic State Indicators (TSI) identify parameters which correlate with a lake's algal biomass, endemic or invasive, although invasive plants are cause for concern due to their propensity to overwhelm and dominate new territory. Invasive plant species have flourished in Long Lake, brought unknowingly into the lake by boat recreators and thrown away haphazardly by bygone aquarium aficionados (EnviroVision Corp., 2004). Around the basin, impervious surfaces increase as asphalt is laid for new construction, and oak forests are replaced by manicured lawns which likely are accompanied by the heavy use of fertilizers. Nutrients flow into Long Lake through multiple inputs including groundwater seepage, lawn overflow, stormwater conveyance, diffuse non-point source pollution, and from the lake's own biotic activity. The following sections will provide a basic overview of limnological processes needed to be understood, which then informs future nutrient mitigation options.

2.2 Thermal Stratification of Lakes

Lake productivity is largely determined by the basin's depth, light requirements and nutrient supply. If the lake is deep enough, *thermal stratification* will occur in which distinct layers or "strata" are formed and separated by differences in temperature. The warm upper strata is known as the *epilimnion*. It is where most photosynthetic activity occurs as the surface receives the greatest amount of solar radiation. The sun's light may be reflected, absorbed or transmitted through the water column based on the light's wavelength and water clarity as suspended solids, partially decomposed organic matter, and plankton can block the light's path. Turbid waters, rich in suspended clays, can prevent the sun's radiation from reaching even a few centimeters below the surface. Conversely, light traveling through clear water can penetrate meters below the surface (Vallentyne, 1974).

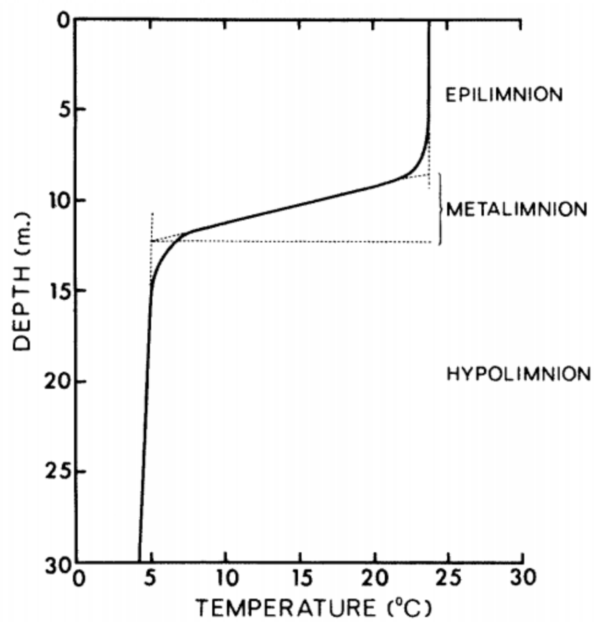
The epilimnion is the warmest lake layer as it receives the most radiation from sunlight. As a lake's depth increases, the energy of the sun's visible light decreases; 50% of which is absorbed by water within the first 10 meters if unimpeded by the lake's clarity. *Light attenuation* is the gradual decrease in light intensity as it travels through matter. Light attenuation also affects the colors that permeate water at depth with the blue range of visible light traveling the farthest (University of Hawai'i, 2021). With diminished energy, the temperature of water decreases. As water (H_2O) molecules cool, they slow down, get slightly closer to one another and occupy a smaller volume resulting in an increase in density (American Chemical Society, 2021).

Water is an interesting compound as it actually becomes lighter when in a solid state. As a general rule however, the change in the density of water per degree change in temperature increases as the temperature departs from 4° C, waters maximum density, both above and below (Vallentyne, 1974). As such, density increases as temperature decreases below freezing.

Thus, below the epilimnion lies the colder and denser waters of the *hypolimnion*. As organisms die, detritus sinks from the epilimnion to the lake bottom. Sedimentation of nutrients occurs here, storing macronutrients (carbon, nitrogen, phosphorus, hydrogen, and sulfur) and micronutrients (silicon, magnesium, copper, and zinc) for future use. Decomposition dominates over photosynthesis here, consuming oxygen and releasing nutrients back into the water column at a rate determined both by seasonality and biotic activity (Tundisi & Tundisi, 2011).

Between the epilimnion and the hypolimnion lies a zone of transition where the water's temperature changes rapidly. A layer is formed known as the metalimnion or *thermocline* which actually creates a physical barrier between the epilimnetic and hypolimnetic strata. This occurs due to the density gradient between strata which prevents their vertical mixing. It takes physical work to mix layered masses of water of varying density, similar to the work it takes to mix oil and vinegar. Additionally, the thermocline acts as a biological barrier creating niches for aquatic plant and animal life based on their own requirements for life. The epilimnion provides a warmer, more illuminated environment for many species of green plants, insects, plankton, and fish to thrive.

Figure 1 *Thermal Stratification in Lakes*



Note. Typical orientation in lakes at temperate latitudes during summer months (Vallentyne, 1974).

Cold-water species prefer the hypolimnion, and still others find their requirements met in the metalimnion. Some aquatic species may even migrate between the layers (Vallentyne, 1974).

By converting sunlight into calories of edible food through photosynthesis, autotrophs (i.e., algae, higher plants, and some protists and bacteria) form the base of the food chain as primary producers. The abundance and rate of production at this trophic level is the foremost determinant of productivity at all higher levels of the food chain. In lake ecosystems, much of the production is done by algae suspended in the water column, known collectively as *phytoplankton,* and larger flora. The term "algae" is an informal term used for a very large and diverse group of photosynthetic, mostly eukaryotic aquatic organisms. Types of phytoplankton include diatoms, chrysophytes, cryptophytes, dinoflagellates, green algae (chlorophytes), and cyanobacteria (Allan

& Castillo, 2007; Schindler & Vallentyne, 2008). Many species of cyanobacteria possess gas vacuoles which allow them to move throughout the water column as their needs require . This includes the bacterium *Microcystis aeruginosa*, the most common bloom-forming and frequently toxic cyanobacterium found in lakes (United States Environmental Protection Agency, 2021b). By floating in the upper strata of stratified lakes, they take advantage of the euphotic zone, an area with sufficient light for photosynthesis to occur. Cyanobacteria are then capable of "dipping" into the hypolimnion to obtain nutrients before returning back to this zone of production (Cottingham et al., 2015).

Other physical and chemical consequences occur in the water as a result of the thermal stratification. In addition to biotic life, the vertical distribution of gases and nutrients may vary throughout the column or they may become concentrated in individual strata. Strong winds and other climactic forces may interrupt this stratification, contributing the work necessary for mixing. When this occurs, nutrient-rich hypolimnetic waters are brought up toward the surface. Algal blooms may be observed after strong wind activity as both benthic nutrients and organisms are brought toward the surface and photic zone (Carrick et al., 1993). Seasonal shifts in temperature also cause a lake to "turn-over," redistributing nutrients at that time. This turn-over occurs as a result of surface waters becoming colder and denser than the hypolimnion at which point the epilimnion falls below and replaces it (Schindler & Vallentyne, 2008).

Dissolved oxygen (DO) produced by photosynthetic organisms in the epilimnion is often transported to the hypolimnion during these seasonal and climatic events. Consequently, the amount of oxygen available in the hypolimnion is finite and is consumed at a rate proportional to the amount of biotic activity occurring within the strata. Once a lake turns over in the spring, seasonal warming initiates the layered stratification that inevitably locks in dissolved oxygen levels. Bacteria gradually deplete dissolved oxygen levels as they decompose dead plant and animal matter falling from the epilimnion. The greater the supply of organic matter from the epilimnion, the more rapid the oxygen depletion will be in the hypolimnion. Highly productive eutrophic lakes with smaller hypolimnetic volumes can lose their dissolved oxygen in a matter of weeks. The opposite is true of low productive oligotrophic lakes with larger hypolimnetic volumes which can retain oxygen levels throughout the year (Schindler & Vallentyne, 2008). Increasing temperatures and seasonal shifts occurring as a result of climate change compound this issue.

Warming surface waters are the most direct response to global temperature increases, and unfortunately for Long Lake, lakes located within temperate regions are the most responsive to those changes (Piccolroaz et al., 2020). Furthermore, cyanobacteria taxa *Anabaena* and *Microcystis,* both prevalent species in the lake, are responsive to concurrent increases in both temperature and nutrients (Rigosi et al., 2014). Lake surface water temperature trends have been increasing in excess of ambient air temperatures worldwide, and are projected to accelerate (Adrian et al., 2009; O'Reilly et al., 2015; Paerl & Paul, 2011). High-latitude regions that experience ice cover in wintertime are having shorter periods of icing or none at all. This leads to stronger vertical temperature stratification for longer periods of time, and fish die-off events increase due to the hypoxic water conditions (Carmichael, 1991). Alterations in top predator populations in lakes can in turn alter the balance of productivity, predation, and energy flow throughout the food web, a concept also known as a "trophic cascade".

While not the subject of this thesis, exploring the cascading trophic interactions of Long Lake may lead to a better understanding of the dominance of cyanobacteria there as well. Zooplankton largely graze on algae, but do not graze on cyanobacteria due to their size and toxicity (Entranco, 1994). Research has shown that in addition to the harmful effects microcystin poses to animals and humans it can even inhibit the ingestion process by *Daphnia galeata*, a typical filter-feeding grazer in eutrophic lakes (Rohrlack et al., 1999). Trophic cascade researchers believe trophic interactions may explain the differences in productivity among lakes with similar nutrient supplies but contrasting food webs (Carpenter et al., 1985; Ripple et al., 2016).

2.3 Nutrient Cycling in Lakes

The profusion of primary producers in lakes is related to the availability of nutrients, in particular macronutrients: phosphorus (P) and nitrogen (N). Phosphorus and nitrogen are two of the six principal elements (hydrogen, oxygen, carbon, nitrogen, phosphorus and sulfur) that form the backbone of life on Earth, and are also important due to their effect on eutrophication (S.R. Carpenter et al., 1998; Dean, 1999; Pick & Lean, 2010; Vallentyne, 1974). Together, N and P limit rates of primary production in most ecosystems on this planet including inland waters. The ratio of algal demand relative to supply for phosphorus is, on average, higher than for other elements found in plant tissue. Phosphorus does not have a gaseous phase so the atmosphere is not a significant source, unlike nitrogen and carbon (Schindler & Vallentyne, 2008; Welch et al., 2005). Human activity inhibits lake ecosystem resilience through its contribution of synthetic fertilizers, fossil fuel emissions, and altered watersheds ultimately contributing to the flux of nitrogen and phosphorus into aquatic ecosystems.

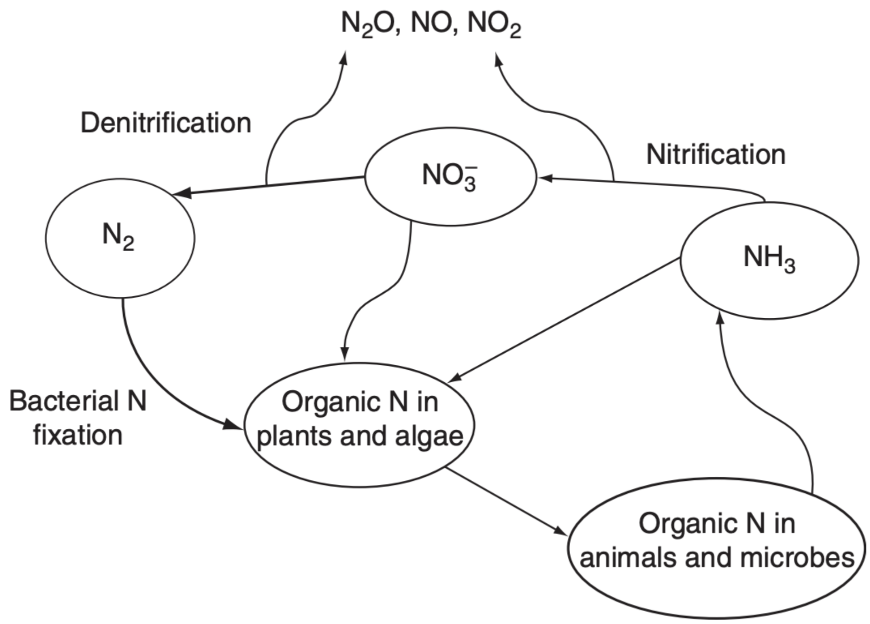
2.3.1 Nitrogen Cycle

Nitrogen (N) is the fourth most abundant element found in living biomass after hydrogen, carbon, and oxygen (R. Howarth, 2009; Stein & Klotz, 2016). Due to its very active oxidation-reduction cycle, nitrogen transitions between forms and is recycled within ecosystems at a higher rate than phosphorus (R. Howarth, 2009). Aquatic organisms use N primarily to synthesize proteins and amino acids (Tundisi & Tundisi, 2011). Nitrogen occurs in aquatic ecosystems in both organic and inorganic forms and has multiple fates once it enters surface waters. “Reactive N,” that which supports or are products of cellular metabolism and growth, may be permanently removed via denitrification, stored in the sediment, or stored temporarily in biomass (Stein & Klotz, 2016). Organic forms of N are supplied by particulate and dissolved nitrogen found in living biomass and detritus.

Inorganic forms include dissolved N_2 gas, oxidized ions such as nitrate (NO_3^-) and nitrite (NO_2^-), the reduced ammonium ion (NH_4+), and the reduced ammonia gas (NH_3) (Howarth, 2009). Not surprisingly, many commercial fertilizers include a combination of these forms (Mattson et al., 2009). Additionally, household sewage contributes forms of inorganic N to waterways via sewage treatment effluent and leaking septic tanks. Even functioning systems contribute nitrogen to groundwater. They are generally considered nonpoint source pollution unless the effluent reaches stormwater infrastructure that is covered by a general permit (Washington State Department of Ecology, 2015). In 2015, the City of Lacey reported a moderate to high level of nutrient, pathogen, and toxin delivery as a result of urban land uses including septic tanks, fertilization, and impervious surfaces (City of Lacey, 2015). The latter is consistently a topic of discussion for the Long Lake community and is brought up in their monthly newsletters with LLMD members urging their neighbors to stay on top of septic maintenance (Lake Management District #21, 2017; Long Lake Management District #21, 2019).

The most common forms of N taken up by algae, rooted plants, fungi and bacteria are ammonium (NH_4^+), nitrate (NO_3^-), nitrite (NO_2^-), and urea ($(NH_2)_2CO$) with nitrate and nitrite being most available in freshwater (Howarth, 2009; Stein & Klotz, 2016; Tundisi & Tundisi, 2011). Ammonium, whether taken up directly, hydrolyzed (chemically broken down by its reaction with water), or reductively assimilated in the organism, is used to produce organic nitrogen compounds. Nitrogen flows throughout the food web through predation and decomposition and is mineralized by its conversion from organic to inorganic forms (Howarth, 2009). Bacteria play a pivotal role in the conversion of nitrogen in aquatic ecosystems and are labeled by their participation in the cycle (e.g., “nitrifiers,” “denitrifiers,” and “nitrogen fixers”).

Figure 2 *Nitrogen Cycle*



Note. A simplified diagram of the nitrogen cycle in aquatic ecosystems (RW Howarth, 2002).

Nitrogen fixation is the process by which atmospheric N_2 is converted to NH_4^+. Nitrogen gas (N_2) constitutes approximately 78% of the gaseous composition of our earth's atmosphere (Wallace & Hobbs, 2006), although only organisms that can fix N in its gaseous form are able to benefit from its abundance. As nitrogen is evident in the composition of most life on Earth, "nitrogen fixers" are essential for its provision in biologically available forms. Cyanobacteria are well adapted for this uptake as many cyanobacterial strains, both with and without a heterocyst - a specialized nitrogen-fixing cell, are capable of nitrogenase activity. Additionally, these bacterium contain protein polymers, cyanophycin and phycocyanin, endowing them with the ability to store nitrogen (Watzer & Forchhammer, 2018; B.A. Whitton & Carr, 1982). Interestingly enough, phycocyanin is the blue pigment-protein also responsible for cyanobacteria's earlier classification as a "blue-green algae" (Brian A. Whitton & Potts, 2000).

Nitrogen fixation used to be exclusive to heterotrophic bacteria, cyanobacteria and archaea until the early 20th century. It was during this time that the Haber-Bosch process was invented and allowed for the industrial conversion of N_2 to ammonia, NH_3 (Gold et al., 2019; Stein & Klotz, 2016). The advent of the Haber-Bosch process is a double-edged sword as nitrogen has become both more plentiful and toxic. Atmospheric nitrogen converted to nitrogen-based fertilizers has enabled agricultural practices to expand to industrial levels of production as well. The resulting fertilizer runoff has had disastrous effects downstream and will be discussed later in this thesis.

Nitrification is the process by which bacteria sequentially oxidize ammonia into nitrite (1), then nitrite to nitrate (2) (American Water Works Association & Economic and Engineering Services, 2002):

$$(1)\quad NH_3 + O_2 \rightarrow NO_2^{-} + 3H^{+} + 2e^{-}$$

$$(2)\quad NO_2^{-} + H_2O \rightarrow NO_3^{-} + 2H^{+} + 2e^{-}$$

There are two primary genera of autotrophic bacteria, *Nitrosomonas* and *Nitrobacter*, that preform nitrification, although other autotrophic and heterotrophic bacteria as well as some fungi can carry out this process also (American Water Works Association & Economic and Engineering Services, 2002). Nitrification provides the energy necessary for chemosynthesis, a process in which carbon dioxide is fixed to produce biomass. The growth of nitrifying bacteria is relatively slow compared with other chemosynthetic processes (e.g., oxidizing sulfur or iron compounds), and in combination with their predation by other trophic grazers, can result in a slower population growth rate (R. Howarth, 2009). Due to their slow growth rate, ammonia is allowed to accumulate and is made available for use by other organisms. Nitrifiers are obligate aerobes and require oxygen for the denitrification process to occur. As dissolved oxygen levels drop, denitrification is made possible (Tundisi & Tundisi, 2011).

Denitrification is the process by which nitrite and nitrate are reduced to the inert gas, N_2. This process requires adequate carbon and nitrate sources, the latter produced primarily in lake sediments through the nitrification of ammonium. However, nitrate may also be diffused through the overlying water (Eyre et al., 2013; Seitzinger et al., 2006). Denitrification is catalyzed by heterotrophic bacteria which decompose detritus, and use nitrogen as an electron acceptor much like organisms use oxygen as an electron acceptor to perform respiration (Stein & Klotz, 2016). This typically occurs in oxygen reduced or anoxic environments as is frequently the case in aquatic sediments and stratified lakes (R. Howarth, 2009; Illinois State Environmental Protection Agency, n.d.). Denitrification provides a mechanism by which nitrogen can be reduced and released back into the atmosphere with lakes and reservoirs accounting for 33% of the total global nitrogen removal (Harrison et al., 2009).

Denitrification can be expressed as a redox reaction (1; Boundless, 2021):

$$(1)\quad 2\ NO_3^{2-} + 10\ e^- + 12\ H^+ \rightarrow N_2 + 6\ H_2O$$

Additionally, the bacterium *Thiobacillus denitrificans* is an example of a denitrifying organism. Widely distributed in soil and aquatic environments, *T.denitrificans* also links the geochemical cycles of sulfur and nitrogen (Tundisi & Tundisi, 2011):

$$(2)\quad 5\ S + 6\ NO_3 + 2\ H_2O \rightarrow 5\ SO_4 + 3\ N_2 + 4\ H + \text{energy}$$

Denitrification and nitrogen fixation perform an essential role together in that these processes regulate the balance of nitrogen within aquatic ecosystems. While nitrogen fixation makes nitrogen available to the aquatic environment, denitrification releases nitrogen back into the atmosphere from laden benthic sediments and anoxic lake strata, acting as a control on system level primary productivity (Eyre et al., 2013). In general, the rate of nitrogen removal has been observed to correlate positively with the rate of nitrogen loading, water residence time, and negatively correlated with lake mean depth (Kelly et al., 1987; Saunders & Kalff, 2001).

The samples collected for this study were analyzed for their concentrations of total nitrogen (TN) utilizing the kjeldahl 4500-N (organic) C method. This method determines nitrogen in the tri-negative state. It fails to account for nitrogen in the form of azide, azine, azo, hydrazone, nitrate, nitrite, nitrile, nitro, nitroso, oxime, and semi-carbazone. "Kjeldahl nitrogen" is the sum of organic nitrogen and ammonia nitrogen (Baird & Eaton, 2017).

2.3.2 Phosphorus Cycle

As the 12th most abundant element in the Earth's crust, phosphorus is widely distributed throughout the globe. Phosphorus is essential for the growth and maintenance of living organisms, and is a component of nucleic acids, adenosine triphosphate (ATP), and phospholipids (Tundisi & Tundisi, 2011). Phosphorus is highly reactive and as such does not occur as a free element on Earth. This means that it is regularly bonded with other elements as phosphates (PO_4^{3-}) (Déry & Anderson, 2007). The most common forms of organic phosphorus are biological in origin, although dissolved phosphates are also delivered through the weathering of rock phosphate. While phosphorus moves quickly through plants and animals, the phosphorus cycle is one of the slowest biogeochemical cycles on Earth. Phosphorus moves slowly through the soil and ocean until it reaches both its beginning and end in the subduction zones of the Earth's crust (Turner et al., 2005).

In natural waters, phosphorus is categorized into three component parts: soluble reactive phosphorus (SRP), soluble unreactive or soluble organic phosphorus (SUP), and particulate phosphorus (PP) (Rigler, 1973). SRP consists of the soluble inorganic orthophosphate (PO_4) which is the primary source of phosphate for aquatic plants and phytoplankton (Carlson & Simpson, 1996; Tundisi & Tundisi, 2011). For this reason, SRP is one of the phosphorus tests being analyzed in this study. The concentration of SRP would be indicative of the amount of P immediately available for algal uptake. The difference between TP and SRP is the measurement of orthophosphate pre- and post- filtering as TP includes P attached to particulate matter and SRP measures solely dissolved P.

The accumulation of phosphorus in lake sediments is an important component of the phosphorus cycle. Phosphorus is often attached to sediment particles which leads to its accumulation through sedimentation, forming an important nutrient reservoir (Tundisi & Tundisi, 2011; Wetzel, 2001). During oxic periods within the hypolimnion, phosphorus undergoes a complexation process and is bound to metal ions making it biologically unavailable (Tundisi &

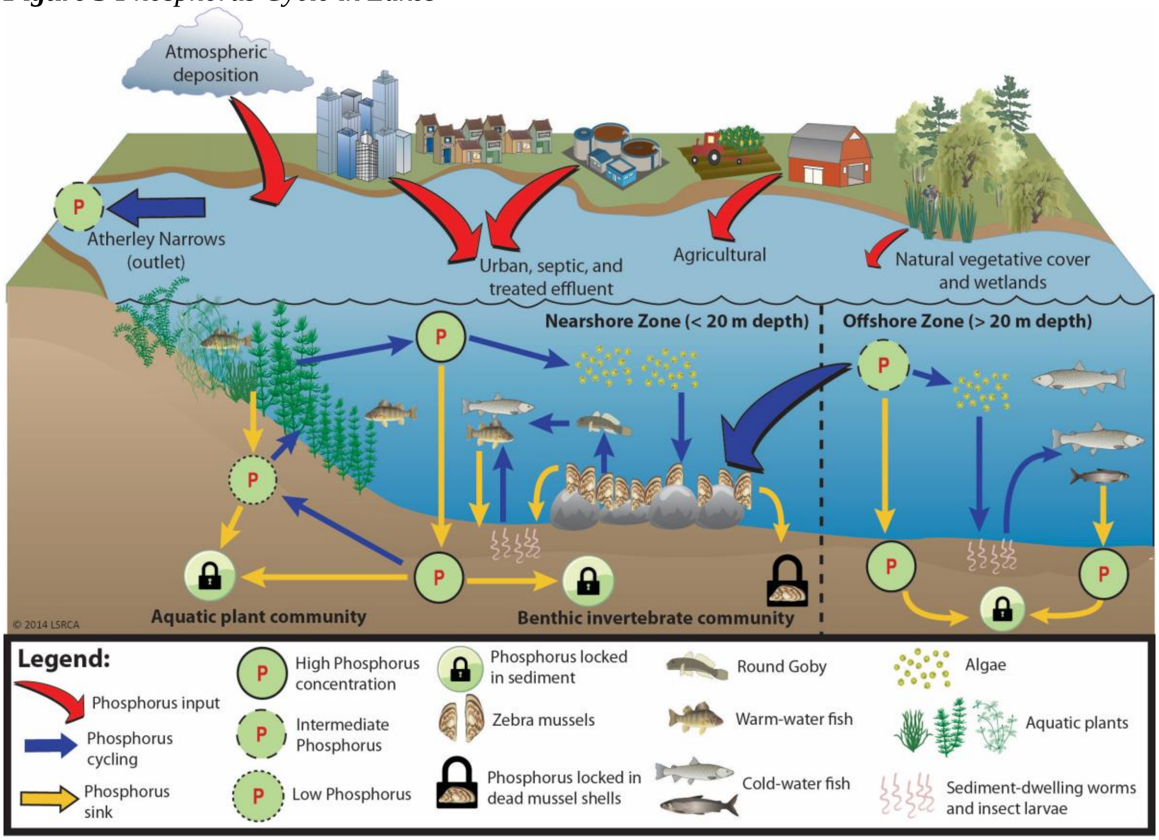
Tundisi, 2011). However, as the hypolimnion loses oxygen to biological processes (i.e., decomposition and productivity) and becomes anoxic, phosphorus is released from the sediment into the lake's water column causing further eutrophication (Niirnberg, 1994; Song et al., 2017; Spears et al., 2007; Yuan et al., 2019).

The condition of phosphate release is widely accepted, yet still not precisely understood. Under certain conditions, phosphates are retained in lake sediments due to a micro-layer of ferric acid [FeO(OH)] at the sediment-water interface which absorbs phosphorus in oxic conditions. However, under anoxic conditions, this micro-layer loses its ability to absorb phosphorus and both elements are released to the water column (Mortimer, 1941). While this is a widely accepted theory for phosphate release from sediments, others oppose the simplicity of this explanation and its universal application (Golterman, 2001; Hupfer & Lewandowski, 2008). They insist that it is not merely the lack of oxygen that allows for the dissolution of phosphates into the water column, rather a combination of dynamic factors contribute to its release.

Biotic respiration and bacterial decomposition in lakes releases carbon dioxide into the water column. The release of carbon dioxide (CO_2) often occurs concomitantly with acidification as CO_2 trapped in the hypolimnion will lower the water's pH. Researchers suggests the solubilization of apatite, a phosphate loaded mineral, would occur in such conditions leaching minerals into the water column (Golterman, 2001). Microbial activity by sediment bacteria (Kleeberg & Dudel, 1997) and other redox conditions (e.g., dissolution of calcium-bound P) could also contribute to the internal phosphorus loading mechanisms within a lake (Hupfer & Lewandowski, 2008). These hypotheses posit the need for greater study on phosphorus release mechanisms rather than relying solely on measuring a lakes anoxic factor. It is possible that this simplification is affecting modeling capabilities which will have far-reaching consequences on lake management.

Macroscopic animals (e.g., fish, insects) play major roles in nutrient cycling within lakes. Sediment disturbance by benthic dwelling animals resuspends sedimented phosphorus into the water column. Their excretion and the decay of their carcasses remineralize significant quantities of nutrients that then become available for algal uptake (Welch et al., 2005). The common carp, *Cyprinus carpio*, a non-native bottom-feeder, was found to release phosphorus at rates similar to external loading of phosphorus into a series of Minnesotan ponds (Lamarra, 1975). Figure 3 depicts many aspects of the phosphorus cycle described courtesy of the Lake Simcoe Region Conservation Authority.

Figure 3 *Phosphorus Cycle in Lakes*



Note. A simplified diagram of the phosphorus cycle in lakes (*Phosphorus Cycle*, 2016).

2.3.3 Hydraulic Residence Time

The amount of time it takes for all water to flow through a lake system completely is known as its hydraulic residence time (HRT). The HRT includes all water flowing into the lake from river, groundwater, and rainfall inputs which takes approximately two years to occur for Long Lake (P. Cracknell, personal communication, 2021). The HRT:

> …affects the chemical composition of lake waters by controlling the time available for biogeochemical and photochemical processes to operate, the extent of accumulation, loss of dissolved and particulate materials and the duration of biogeochemical interactions with the lake sediments and littoral zone. (Bhateria & Jain, 2016).

Shifts in precipitation and evaporation alter a lake's water budget and HRT, and shallow lakes can be severely affected by changes in climate due to their large surface area to volume ratios (Adrian et al., 2009; Bhateria & Jain, 2016; Zhang et al., 2016). A prolonged residence time caused by reduced precipitation and inflows could result in amplified phosphorus accumulation and eutrophication. In lakes that experience anoxic hypolimnetic conditions, nutrients released from benthic sediments leads to increased internal phosphorus loading (Bhateria & Jain, 2016). On the other hand, an increase in precipitation might decrease a lake's HRT but the precipitation might also bring with it more frequent concentrations of nutrients through stormwater (Wu & Malmström, 2015).

The samples collected for this study were analyzed for total phosphorus (TP) which includes orthophosphate, condensed phosphate, and organic phosphate. Additionally, samples were analyzed for their concentrations of soluble reactive phosphorus (SRP), the most biologically available form of phosphorus.

2.4 Trophic State Indicators

Long Lake is considered to be a eutrophic lake based on the Trophic State Indicators/Index (TSI) utilized by lake managers and limnologists (Bell-McKinnon, 2010; Butkus, 2004). The TSI, as developed by Robert Carlson, use algal biomass as the basis of a waterbody's trophic state classification, and notes Secchi depth/transparency (SD), chlorophyll-a (Chl), total nitrogen (TN) and/or total phosphorus (TP) considerations. While this is a rough estimate of the trophic condition of a waterbody, water quality managers can assess the trophic state based on: changes in nutrient levels (measured by total phosphorus and total nitrogen) that may cause changes in algal biomass (measured by chlorophyll-*a*) which in turn can result in changes in lake clarity (measured by Secchi disk transparency) (Pavluk & De Vaate, 2018).

Table 1 *Classes of TSI values and their ecological attributes.*

TSI	*Chl* ($\mu g\ L^{-1}$)	*SD (m)*	*TP* ($\mu g\ L^{-1}$)	*Ecological attributes*
<30	<0.95	>8	<6	Oligotrophy: Clear water, oxygen throughout the year in the entire hypolimnion
30–40	0.95–2.6	8–4	6–12	Hypolimnia of shallower lakes may become anoxic
40–50	2.6–7.3	4–2	12–24	Mesotrophy: Water moderately clear; increasing probability of hypolimnetic anoxia during summer
50–60	7.3–20	2–1	24–48	Eutrophy: Anoxic hypolimnia, macrophyte problems possible
60–70	20–56	0.5–1	48–96	Blue-green algae dominate, algal scums and macrophyte problems
70–80	56–155	0.25–0.5	96–192	Hypereutrophy (light-limited productivity): Dense algae and macrophytes, algal blooms possible throughout summer
>80	>155	<0.25	192–384	Algal scums, few macrophytes

Adapted from Carlson, R.E., Simpson, J., 1996. A coordinator's guide to volunteer lake monitoring methods. USA: North American Lake Management Society Madison (Pavluk & De Vaate, 2018)

TSI can be formulated based on the above influencing factors and are presented below (Pavluk & De Vaate, 2018):

TSI (SD) = 60 – 14.41 ln Secchi disk depth (meters)

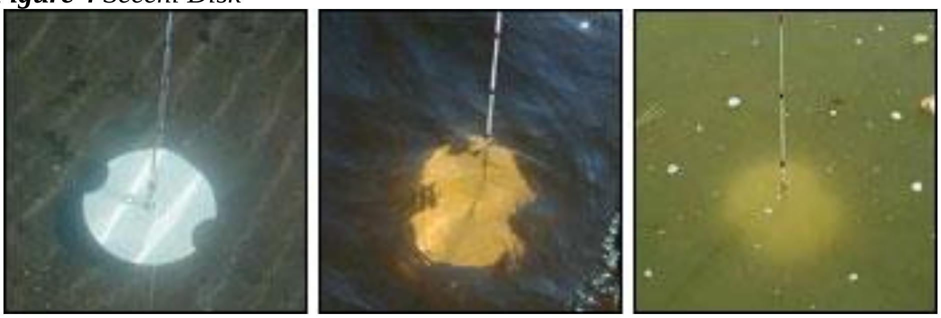
TSI (Chl) = 9.81 ln chlorophyll-*a* (μg/L) + 30.6
TSI (TP) = 14.42 ln total phosphorus (μg/L) + 4.15

Each of these values can be used to classify a waterbody as they are interrelated by linear regression. If the TSI values are not similar, it may suggest that algal growth is limited by something else including light or other necessary elements. Secchi transparency can be affected also by variables other than algae such as erosional silt or construction runoff (Pavluk & De Vaate, 2018). Because the nutrients nitrogen and phosphorus were described in depth earlier, the following sections will describe only the Secchi disk and chlorophyll-*a* measurements.

2.4.1 Secchi Disk

The Secchi disk, named for papal scientific advisor Pietro Angelo Secchi, has been employed since the 19th century to measure the transparency of the water. While developed for use primarily in lakes, the Secchi disk can be used in riverine and marine environments. The information provided by Secchi readings can be discounted based on the subjective nature of observer measurements. However, the Secchi disk is easy to use and provides useful information to those engaged in water quality studies (Carlson & Simpson, 1996). The Secchi disk is a contrast instrument. As the disk is lowered into the water, an observer watches to discern at what depth the disk disappears behind the ambient background.

Figure 4 *Secchi Disk*



Note. Photo courtesy of NASA Earth Observatory (*Plumbing the Depths*, 2002)

This contrast relationship is represented by the simplified equation (Carlson & Simpson, 1996):

$$Contrast = \frac{Object\ luminance - Background\ luminance}{Background\ luminance}$$

And based on the theoretical equation by (Preisendorfer & Duntly, 1952):

$$C_R = C_O e^{-(\alpha - K)z}$$

C_R is the apparent contrast; C_O, the inherent contrast; z, the depth of disk disappearance (Secchi depth); α, the beam attenuation coefficient; and K, the vertical attenuation coefficient.

This equation can be rearranged to examine the factors affecting the depth at which the Secchi disk disappears (Z_{SD}):

$$Z_{SD} = \frac{\ln\left(\frac{C_O}{C_R}\right)}{(\alpha + K)}$$

The depth of disappearance will depend on the contrast of the disk relative to the background. In theory, the Secchi disk should disappear into a light consuming black background. This type of background might exist in the open ocean but is not typical of lakes as they are often filled with suspended silts and clays. Measuring Secchi depth is a subjective art and limited to the sensitivity of the human eye itself (Carlson & Simpson, 1996).

2.4.2 Chlorophyll-a

Chlorophyll-*a* is the principal pigment that captures light for photosynthesis within the chloroplast. Absorbing light in the 429 (violet-blue) and 659 (orange-red) nanometer range, chlorophyll-*a* reflects blue-green in color and is present in all plants, algae, and cyanobacteria (Panawala, 2017). Chlorophyll-*a* is used to measure photosynthetic activity and algal biomass in lakes. There are several ways to measure chlorophyll-*a* including: water sampling, filtration, and analysis using electromagnetic spectroscopy methods in the laboratory, and/or satellite photos can be referenced to measure the intensity of colors in the area of interest (United States Environmental Protection Agency, 2021a).

2.5 Impacts of Land-use on Nutrient Cycling in Urbanized Eutrophic Lakes

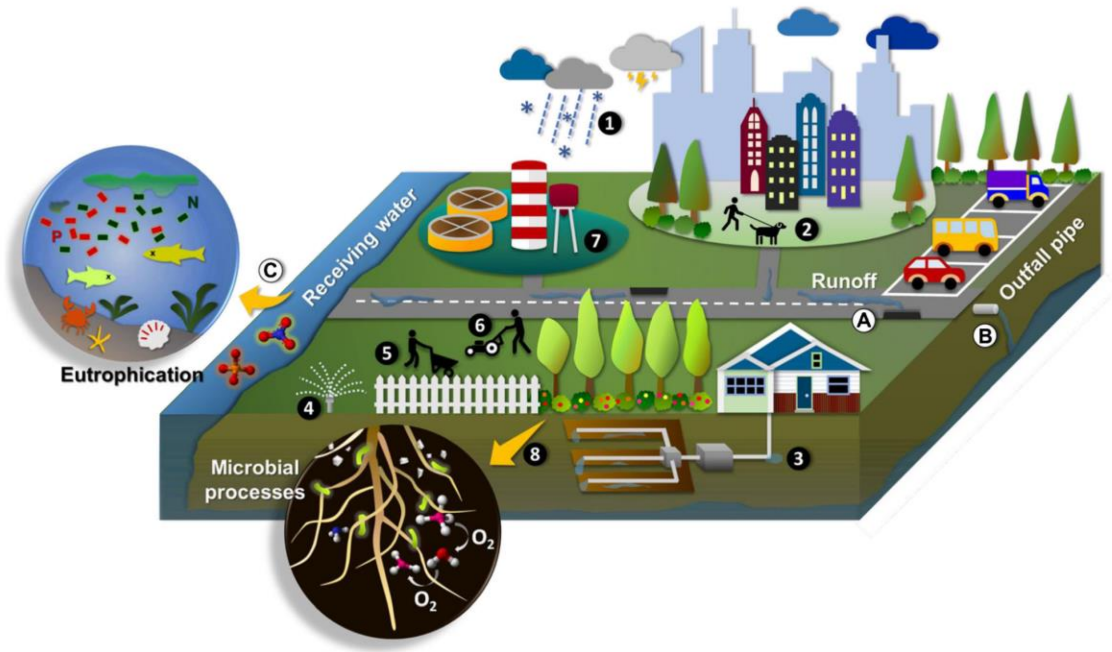
In natural conditions, those unaffected by anthropogenic pressures, ammonium can be relatively low in the lake's epilimnion. The process of nutrient enrichment is known as *eutrophication*, and often parallels the senescence of a lake waterbody. Over time, a lake's trophic state transitions from clear "young" waters, relatively devoid of nutrients, to waterbodies less clear and filled with biotic activity. Eventually, the lake becomes shallower, filling in with sediment and detritus from decomposing plant matter (Tundisi & Tundisi, 2011). This transition, commonly known as *succession*, occurs over the course of centuries, if not millennia (Bell-McKinnon, 2010). However, human activities accelerate eutrophication, henceforth known as *cultural eutrophication*, through increased nutrient loading. Inputs may include nitrogen, phosphorus and other pollutants via wastewater discharges, septic systems, construction, agricultural runoff, and stormwater.

Cultural eutrophication is a common and growing problem in lakes. In 1998, 45% of lakes in the United States were impaired by eutrophic conditions and considered not clean enough to support fishing or swimming (United States Environmental Protection Agency, 1998). In 2004, 64% of lakes were impacted (United States Environmental Protection Agency, 2004). As of 2012, national lake conditions have worsened in regard to: cyanobacteria density, *microcystin* concentrations, and biological condition (i.e., degraded benthic macroinvertebrate communities). The National Lakes Assessment determined that one in three lakes (35%) have excess nitrogen and two out of five lakes (40%) have excess phosphorus. Lakes with high levels of phosphorus are 2.2 times as likely to have degraded benthic macroinvertebrate communities, and 1.6 times as likely when in excess of nitrogen (United States Environmental Protection Agency, 2012).

Eutrophic conditions can remain stable due to several mechanisms at play within them including: the internal loading and recycling of phosphorus, the loss of rooted macrophytes which results in the destabilization and resuspension of nutrient laden sediments, and changes in the food web that reduce grazing of nuisance algae (Stephen R. Carpenter & Cottingham, 1997). Eutrophication is a factor in the loss of aquatic biodiversity (Seehausen et al., 1997), and the toxic cyanobacterial blooms that flourish in such ecosystems contribute to a wide range of water related issues. CyanoHABs contribute to summer fish kills (Carmichael, 1991), foul odors (Paerl et al., 1985), unpalatable drinking water (Hardy et al., 2015; Li et al., 2011; Weirich & Miller, 2014), and the formation of trihalomethane, a known carcinogen, when cyanobacteria are treated with chlorine in water treatment facilities (Palmstrom et al., 2009).

Agricultural practices are infamous for their excessive nutrient runoff leading to such events as the record-setting algal bloom on Lake Erie, Michigan in 2011, three times greater than ever recorded (Michalak et al., 2013). Land-use practices are significant drivers of ecosystem imbalances due to alteration of the natural landscape and connected hydrology, disrupting water flow and overloading nutrient budgets in surface waters (Dodson et al., 2007; Mohamedali et al., 2011; O'Driscoll et al., 2010). Land-use practices that replace vegetation and soil with impervious surfaces are often associated with degraded water quality due to stormwater runoff (McGrane, 2016). Impervious surfaces allow water to flow overland picking up a slurry of contaminants including sediment, chemicals and bacteria. Stormwater is then transported to our waterways via storm drains and treatment structures (e.g., ponds, swales, waste effluent), and eventually discharged to lakes, groundwater, streams, rivers, and estuaries (Fig.4) (Yang & Lusk, 2018).

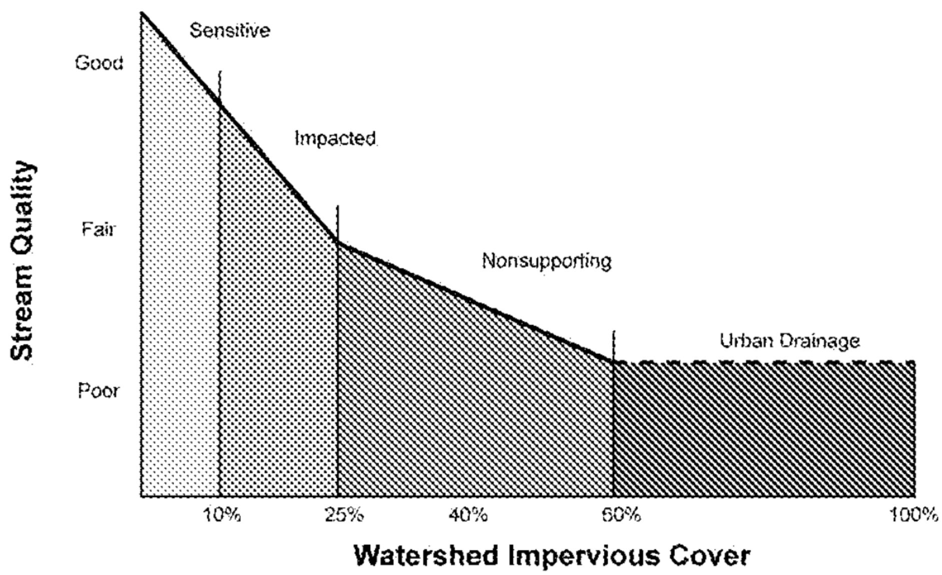
Figure 5 *Overview of pathways and sources of nutrients in urban environment.*



Note. "(A) Urban stormwater runoff is generated when precipitation from rain/snowmelt events over impervious surfaces. (B) Runoff water then makes its way into storm drains and discharges into streams, rivers, and estuaries untreated. (C) Excessive amounts of nutrients in water bodies can cause eutrophication, often leading to fish kills. The potential nutrient sources in urban stormwater runoff include (1) atmospheric deposition, (2) pet waste, (3) improperly functioning septic systems, (4) landscape irrigation, (5) use of chemical fertilizers on lawns, (6) soil and decomposition plant materials, (7) leaking sanitary sewers, and (8) microbial sources" (Yang & Lusk, 2018).

The degree to which an area is covered by impervious surfaces can determine the health and function of associated waterways. In 1994, the Impervious Cover Model (ICM) was developed to predict the behavior of urban stream indicators based on the percent impervious cover in their contributing subwatershed. A subwatershed being defined as an area of land that water passes through before draining into a larger body of water. The researchers' hypothesis was that increasing the amount of impervious surface will degrade stream quality along a reasonably predictable gradient. Schueler (2009) followed up on their model by conducting a meta-analysis of 61 research studies which employed the ICM and found that the majority of studies either confirmed or reinforced the ICM hypothesis. It was concluded that stream health became severely impacted once the subwatershed became 10% covered with impervious surfaces (Fig. 5).

Figure 6 *Impervious Cover Model.*



Note. Courtesy of Schueler et al., 2009.

The model was updated in 2012 to make it more accessible to watershed planners, stormwater engineers, water quality regulators, economists, and policy makers. The improvement came with the implication that managers should test their ability to apply a multiple management strategy toward improving the gradient of stream degradation rather than reinforcing it (Schueler et al., 2009).

2.6 Urban Stormwater Pollution

Nonpoint source pollution is undoubtedly a major contributor of pollutants to aquatic ecosystems (Abrams & Jarrell, 1995; Mohamedali et al., 2011; Washington State Department of Ecology, 2012; Welch & Jacoby, 2001). As one type of *diffuse pollution*, stormwater is capable of amassing pollutants at the catchment scale and has a very significant effect on receiving waters (Kim et al., 2007; Scottish Environment Protection Agency (SEPA), 2021). At a micro level, pollutants can be relatively innocuous individually, but their collection over a larger area concentrates their potency. As the name suggests, treating nonpoint source pollution is made more complicated due to the inability to directly identify its source.

Depending on the area, stormwater effluent may contain petroleum products, agricultural residues, organic matter, nutrients and anything else in its path (Burton & Pitt, 2001; Lee & Bang, 2000; O'Driscoll et al., 2010; Walsh et al., 2005; Washington Department of Ecology, 2010). Several stormwater models have attributed higher magnitudes of pollutant loading to the amount of impervious cover within a studied watershed (Lee & Bang, 2000; Schueler et al., 2009). Lee and Bang (2000) characterized stormwater loading in nine urban watersheds and found that both particulate and dissolved nutrient concentrations were highest in residential watersheds. Higher concentrations of pollutants in areas with reduced impervious cover exceeded areas with more impervious cover only when combined sewer systems (CSOs), which collect rainwater runoff, domestic sewage, and industrial wastewater in the same pipe, were present. Although heavy metals were in higher concentrations most often in industrial areas, this research suggests that the relative ease of transport (impervious cover) and proximity to human developments largely determines the concentration of pollutants downstream. It was recently discovered by University of Washington researchers that a highly ubiquitous chemical component of tire rubber [*N*-(1,3-dimethylbutyl)-*N′*-phenyl-p-phenylenediamine (6PPD)] was the cause of regular acute Coho salmon mortality events. These events were tied specifically to their return to streams within urban and suburban areas of the Pacific Northwest. Analysis of roadway runoff and stormwater-affected creeks in these areas determined a widespread occurrence of the toxic 6PPD quinone transformation (Tian et al., 2021).

It is widely accepted that diffuse pollution has pervasive effects on the environments receiving its effluent (Lusk et al., 2020; T. L. C. Moore et al., 2011; O'Driscoll et al., 2010; Rodríguez-Rojas et al., 2018; United States Environmental Protection Agency, 2020a). Many argue that it is resulting in the eutrophication of lakes, a status that remains relatively stable, unless active intervention is taken (S.R. Carpenter et al., 1998; Paerl, 2017; Silva et al., 2019; Smith et al., 2020; Wium-Andersen et al., 2013; Wu & Malmström, 2015). Researchers within the field of civil engineering are even pressing the necessity and capability of Life Cycle Assessments (LCA), models that estimate the environmental impacts associated with all stages of a products life (i.e., raw material extraction, material processing, manufacture, distribution, and use), to include stormwater impacts (Phillips et al., 2018). Similar to the LCA's ability to quantify the impact land-use has on soil erosion, *urban stormwater pollution* can also become a quantifiable unit to predict and mitigate. The LCA model Phillips et al. (2018) produced demonstrated the influence pollutants in urban stormwater has throughout human, freshwater, marine, and terrestrial ecosystems. Their results determined that "urban stormwater pollution has the highest relative contribution to the eutrophication potentials" (Phillips et al., 2018).

2.7 Stormwater Management

Since the Clean Water Act (CWA) of 1970, the United States has made a concerted effort to "restore and maintain the chemical, physical, and biological integrity of the Nation's waterways" (Federal Water Pollution Control Act, 2002). Remediation of point sources of pollution has been relatively successful considering that today U.S. states recognize non-point pollution to be the leading remaining sources of water pollution (US Environmental Protection Agency, n.d.). Management of these pollutants is an evolving field of possibilities. Currently, stormwater discharges are authorized under the CWA and regulated by the EPA through a permitting system known as the National Pollutant Discharge Elimination System (NPDES). The NPDES permit program authorizes states to regulate stormwater discharge in compliance with total maximum daily load (TMDL) parameters based on water quality standards determined by the state. NPDES permits allow monitored quantities of stormwater to be discharged to the municipal storm sewer system (MS4) or discharged into the ‘waters of the state.’

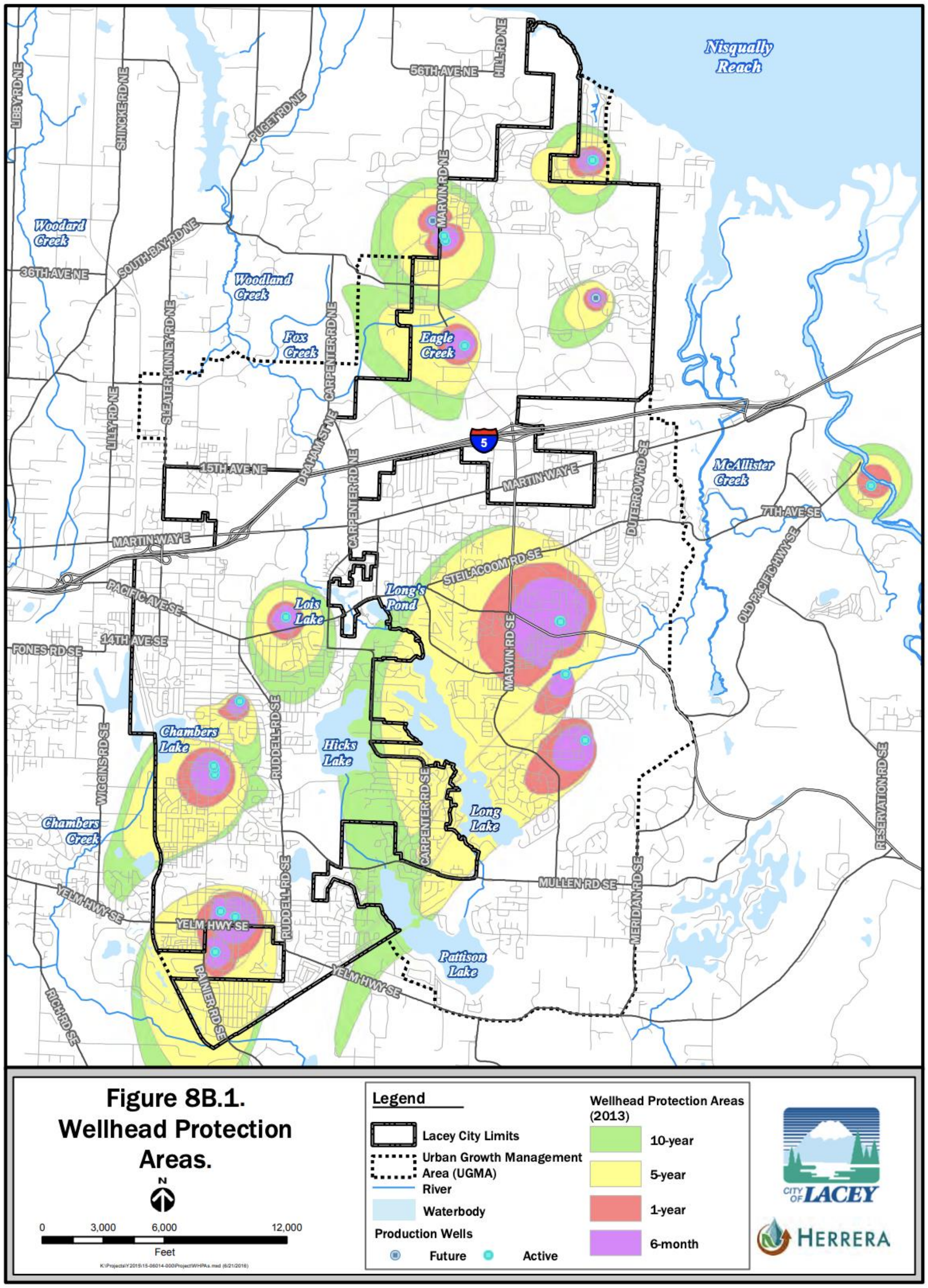
Even though the City of Lacey was not incorporated until 1966, New American settlers have altered the landscape since the 1840s. Farming, logging, and housing development has altered the Henderson Inlet watershed to the point that migrating salmon can no longer return to these natal waterways. Even after the passage of the CWA, wetlands continued to be filled to make way for new construction (Thurston County, 1995b). The Shoreline Management Act passed close behind the CWA (1971) and is another statute implemented for the protection of Washington state shorelines from further development. County designed Shoreline Master Programs are regularly updated to help rein in urban development and in tandem with municipal Stormwater Design Manuals (SDM) can help mitigate future impacts.

The City of Lacey Department of Public Works updates their SDM with improved management practices based on the best available science. The department manages stormwater conveyance and discharges throughout the city, and under their jurisdiction, stormwater catchments, drains, and outfalls are designed for allowable quantities of stormwater to enter local waterways. Yet improvements to stormwater conveyance, retention, and treatment apply primarily to new development which has the opportunity to incorporate such changes.

Long Lake is protected for multiple designated uses under Washington’s Water Quality Standards for Surface Waters (WAC 173-201A-600, 1992) including for the protection of wildlife habitat, boating, and aesthetic values. Equally important to consider are the groundwater contributions the chain of lakes provide to the critical aquifer recharge areas surrounding them. Long Lake dominates the maps of the wellhead protection and critical aquifer recharge areas (Fig. 7 & 8). These critical sites are essential for the replenishment of local drinking water. The water flowing through these basins seeps into the groundwater table where it is eventually extracted for drinking water from the nearby McCallister Park Well (Kevin Hansen, Thurston County Hydrologist, personal communication, November 5, 2020).

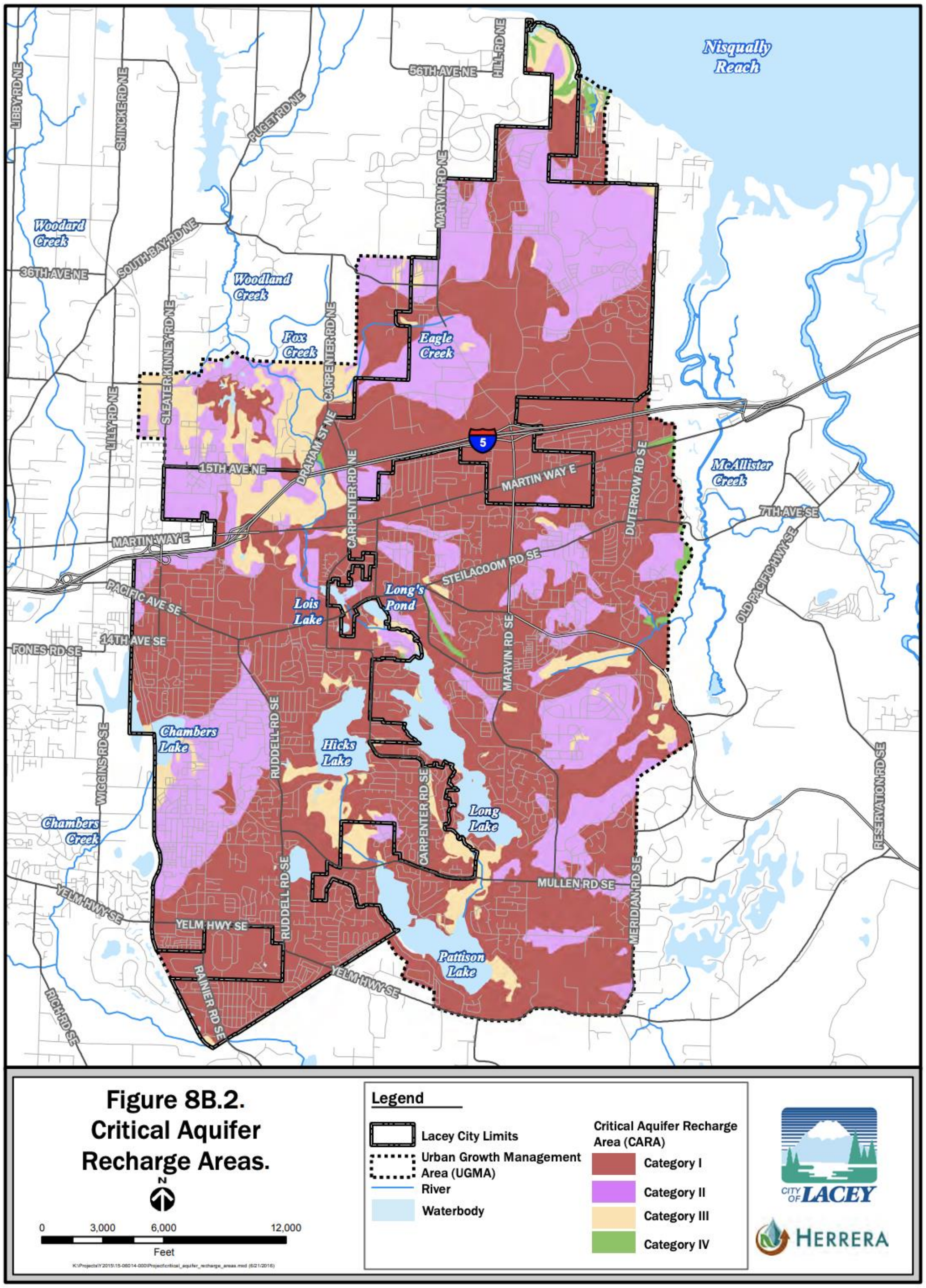
On the Wellhead Protection Areas map (Fig. 7), the colors represent the area that could drain through the soil underground to a water source well during the time interval. For example, the purple area around a well is the area where a contaminant could reach the well within a 6-month time interval. The wells are generally at several hundred feet depth below the ground surface, so the travel time is downward as well as laterally through the soil and rock. The Critical Aquifer Recharge Areas (Fig.8) are in different categories based on how permeable the soils are. Soils that drain quickest and provide the least amount of natural protection for aquifers are Category I, which would require the highest level of stormwater runoff treatment and protection from contamination (Doug Christenson, City of Lacey Water Resources Engineer, personal communication, March 31, 2021).

Figure 7 *Wellhead Protection Areas within the City of Lacey.*



Note. Map courtesy of the City of Lacey's Stormwater Design Manual (City of Lacey Department of Public Works, 2016).

Figure 8 *Critical Aquifer Recharge Areas within the City of Lacey.*



Note. Map courtesy of the City of Lacey's Stormwater Design Manual (City of Lacey Department of Public Works, 2016).

As will be discussed later in this thesis, the research on the residual effects of cyanoHABs to groundwater is lacking. However, there is research to support the need for focused attention and decisive action to curb the overwhelming presence of cyanoHABs throughout freshwater systems. In the following section, a brief overview of some stormwater management practices will be presented.

2.8 Stormwater Mitigation

Stormwater control measures (SCMs) take many forms including but not limited to stormwater treatment ponds, bioretention cells ("rain gardens"), permeable pavers, vegetated buffer strips, natural riparian buffer zones, and wetlands. Stormwater channeled to SCMs can mitigate the negative impacts of pollutants on receiving waters by detaining water, allowing particle sedimentation, and increasing soil infiltration capacity (Welch et al., 2005; Wium-Andersen et al., 2013; Yang & Lusk, 2018). SCM treatment efficacy varies due to the variable amount of pollutant loading, flow volume, site location, and climate all of which make it difficult to compare treatments. Nonetheless, SCMs should be considered when designing water quality management programs as the removal of nitrogen and phosphorus does occur within these systems, and sometimes to great effect.

In the research study of riparian buffers conducted by Polyakov et al. (2005) they found phosphorus removal rates as high as 93% although more often within 60-90%. Removal rates were closely associated with the retention time of P in the SCM and sediment particle size. The removal mechanisms largely depended on the form of P entering the buffer with soluble P being most available for plant uptake. Buffers were assessed to be most effective at trapping sediment bound P with soluble P passing through over time. Riparian zones are effective at removing nitrogen from shallow subsurface water and most effective when groundwater has increased interaction with vegetation. Another study found these areas reduced nitrogen in groundwater by 95% with 65-75% attributed to denitrification (Polyakov et al., 2005). The variability of riparian buffer efficacy was reiterated in Yang & Lusk's (2018) study as well as the recommendation that further study on the types of plants most suitable for long term nutrient sequestration and removal be conducted (Yang & Lusk, 2018).

Stormwater ponds (dry and wet) were found to reduce total nitrogen by 27$\pm$23 and 40$\pm$31% respectively and phosphorus by 19 – 50% when studied in wet ponds in Bellevue, Washington (Comings et al., 2000). As a result of their higher residence times, wet ponds were credited with a higher N and P removal capacity. This allowed for sediment-bound P time to settle and interact with anaerobic zones promoting denitrification of N and stabilization of P and an overall improved performance (Bettez & Groffman, 2012; Comings et al., 2000). In the study conducted by Bettez & Groffman (2012) near Baltimore, Maryland, they found that the denitrification potential of SCMs were comparable if not higher in some cases than natural riparian zones in this urbanized area. They deduced that the higher rate of denitrification [1.2 mg N kg^{-1} hr^{-1} (SCM) to 0.4 mg N kg^{-1} hr^{-1} (natural buffer)] was likely due to the interaction opportunity nutrient-laden stormwater had with denitrifying sediments as they were engineered to reduce flow of peak stormwater discharge (Bettez & Groffman, 2012). While wet ponds are a form of nutrient mitigation, they can succumb to eutrophication at higher rates due to the high nutrient input to pond volume, age, and the lack of biodiversity within them (Dodson et al., 2007; Wium-Andersen et al., 2013). Song et al., (2017) found stormwater ponds to be unsatisfactory phosphorus retainers as minerals were resuspended back into the water column following sedimentation due to the high rate of decomposition occurring within these ecosystems. Even so, the use of stormwater retention ponds is increasing as individual states continue to construct them in relation to an increase in urban development (Siewicki et al., 2007).

Whether or not cyanoHABs will occur as a direct result of this year's stormwater contribution waits to be seen, knowing full well that stormwater is not the only contributing source of nutrients to this system. Phosphorus stored in the benthic sediment is the supply of nearly half the amount of phosphorus loading to Long Lake (Entranco, 1994; EnviroVision Corp., 2004). Even if all phosphorus inputs were to completely stop today, the amount of phosphorus stored in the sediment may be enough to fuel cyanoHABs for decades (Lathrop et al., 1998). Nevertheless, the County and LLMD should be aware of all inputs which may be contributing to these blooms as well as where to direct mitigation efforts.

2.9 Public Health Risks Associated with CyanoHAB Exposure

Analogous to the flaming Cuyahoga River of 1969, harmful algal blooms (HABs) may well prove to be as fiery an indicator of the cumulative impacts of humans on freshwater ecosystems (A. W. Griffith & Gobler, 2020; Hallegraeff, 2010). The United States responded to the challenge of industrial waste disposal in public waterways by restructuring the Federal Water Pollution Control Act (1948) into what is recognized today as the Clean Water Act (1972). The Clean Water Act regulates pollution control programs and allows for controlled amounts of polluting effluent to be discharged based on permitted allowances (United States Environmental Protection Agency, 2020b). Such allowances are set by researched parameters for water quality safety. When such standards do not exist, the phrase “free from toxic substances in toxic amounts” may be used as an equivalent directive. Nonetheless, the water quality criteria by which U.S. states and tribes protect aquatic life or human health do not exist for algal toxins, those which are produced by harmful algal blooms (United States Environmental Protection Agency, 2020c).

Researchers argue that without increased assessment and monitoring of algal toxins, the appropriate safety standards regarding cyanoHAB exposure cannot be determined (Hudnell, 2009; Weirich & Miller, 2014). Methodologies for addressing cyanoHABs vary across state, tribe or territory although many do not have formal algal toxin management programs for surface waters (Brooks et al., 2016). HABs may even be the greatest threat to inland water quality due to the magnitude, frequency, and duration of the blooms (Brooks et al., 2016). Moreover, greater risk may be attributed to unclear safety risks associated with various levels of exposure to cyanotoxins (Hudnell, 2009; Weirich & Miller, 2014). Cyanotoxins are not present in every algal bloom, however it is theorized that cyanobacteria produce these non-essential secondary metabolites to provide some benefit to the organism (Carmichael, 1991; Dziallas & Grossart, 2011; Percival et al., 2014; Rohrlack et al., 1999).

2.12 Summary

The purpose of this literature review was to familiarize the reader with some basic concepts of limnology that are important for lake management. Additionally, specific attention to the impact land-use, impervious cover, and the subsequent stormwater runoff have on lake ecosystems was presented as well as their influence on cyanoHAB occurrence. The adaptive qualities and toxic possibilities of cyanobacteria dominance were presented to demonstrate the need for equally adaptive human management. Overall, lake productivity is largely determined by its basin size, light availability, and its aquatic plant community. A lake’s resilience and ability to remain balanced depends on limiting nutrient concentrations within its water column. Long Lake has required adaptive management methods to make improvements to its water quality, and the community remains committed to identifying best management practices as exemplified by this investigative study.

3. METHODS

3.1 Introduction

Located within the Lacey (Washington) city limits, Long Lake is a part of the Henderson Inlet watershed. Long Lake consists of two basins which together measure 8.25 square miles. The mean depth of the lake is 3.7 meters (12 feet) with a maximum depth of 6.4 meters (21 feet). The volume of the lake is ≊ 4,810,600 cubic meters and 3,900 acre-feet, qualifying Long Lake as a shoreline of statewide significance (lakes or reservoirs covering at least 1,000 surface acres) under the Shoreline Management Act of 1971 (RCW 90.58). The land around Long Lake is primarily used for residential property with a small percentage remaining forested or in use for agricultural purposes. The shoreline is heavily developed with dense residential property. The lake itself is primarily used for fishing, swimming, boating, and other water sports. Overall, the general topography of the lake is flat with extensive wetlands between the chain of lakes (Thurston County Environmental Health Division, 2019).

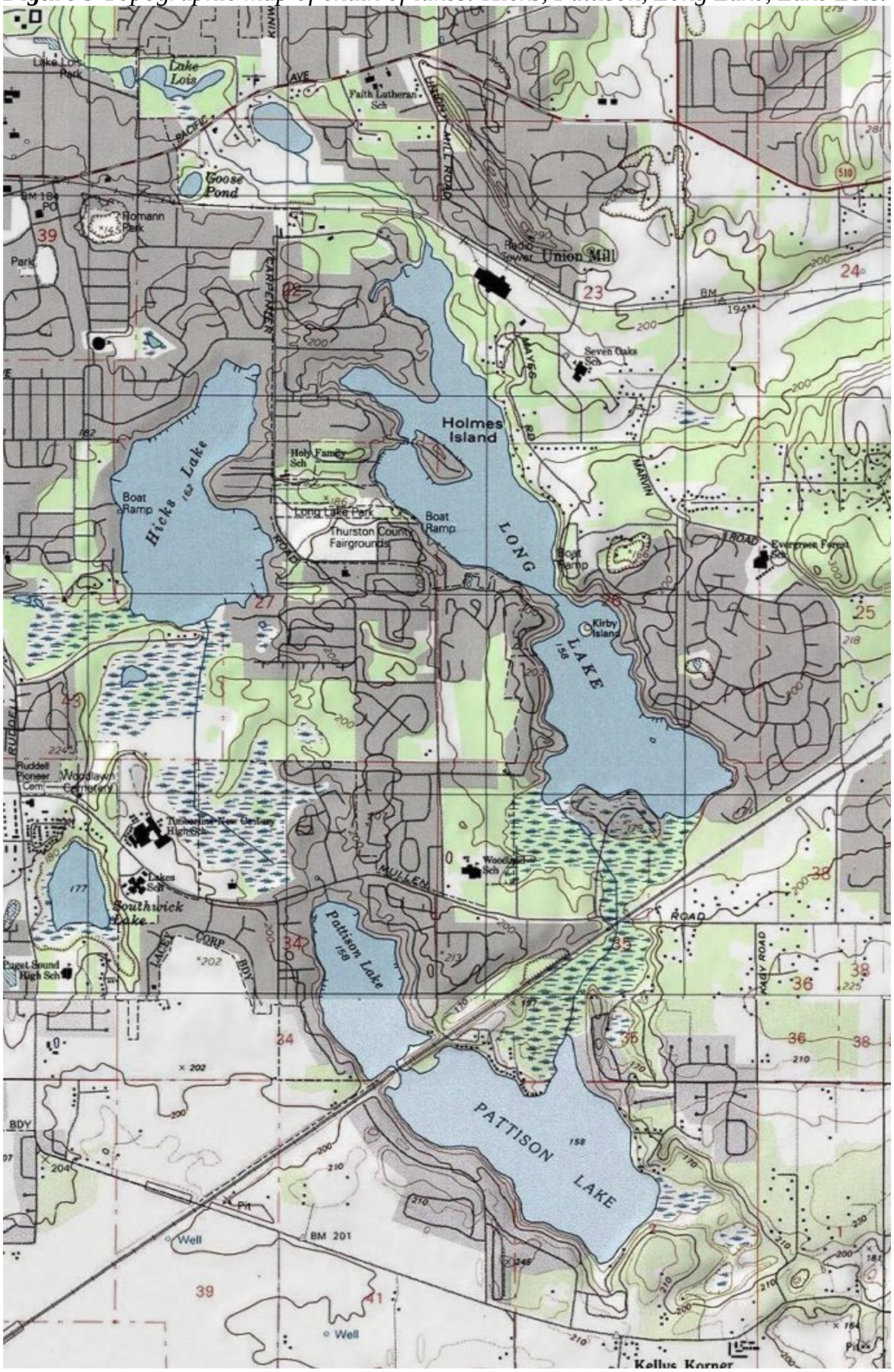
Long Lake is the third in a chain of four lakes that largely comprise the Woodland Creek basin. The headwaters of this basin begin at Hicks Lake where water enters the system through groundwater seepage and surface flow. It is eventually discharged into Pattison Lake through a 38-acre palustrine wetland located at the southwest border of Hicks Lake. Pattison Lake flows into Long Lake through a 119-acre palustrine wetland located between them. While this wetland floods seasonally and connects the two lakes during flooding, a ditch constructed to float logs from one lake to the other connects the lakes permanently. Water leaves the Long Lake system through a surface outlet in the north basin becoming Woodland Creek. The creek flows into Lake Lois, completing the chain, and finally discharges into Henderson Inlet located in north Thurston County (Thurston County, 1995a).

Long Lake can be divided up into two basins based on volume characteristics and morphology: The North and South basins. For the last year that data was available (2019), thermal stratification of Long Lake began in May and was apparent in the north basin until September. In September, fall turnover was evident in the north basin and almost completely mixed in the south basin due to its shallower depth. The north basin (LO3) which contains Holmes Island was classified as eutrophic (TSI > 50) from 2016 to 2019, with a higher trend of productivity and reduced transparency from 2008 to 2018. The south basin (LO4) has been considered eutrophic intermittently over the past decade (2010-2020). In 2019, average surface TP concentrations ranged from 0.030 to 0.032 mg/L at LO3 and 0.030 mg/L to 0.035 mg/L at LO4 testing above the action level (0.020 mg/L) for the Puget Lowlands (Thurston County Environmental Health Division, 2019). According to Washington State's water quality standards for surface waters:

> If TP values are higher than the ecoregional action value, then the lake would be placed on the 303(d) list of water bodies with water quality limitations. Lakes placed on the 303(d) list receive priority status for lake-specific studies (A. Moore & Hicks, 2004).

An "ecoregion" being classified as a major ecosystem defined by distinctive geography and receiving uniform solar radiation and moisture. In 2018, the average total nitrogen levels in both basins ranged from 0.5 to 0.6 mg/L with a significant upward trend (p-value < 0.05) from July to October (Thurston County Environmental Health Division, 2019). While nitrogen is known to be a contributing factor for algal growth, the State of Washington does not have established action or cleanup levels for surface total nitrogen (Thurston County Environmental Health Division, 2019).

Figure 9 *Topographic map of chain of lakes: Hicks, Pattison, Long Lake, Lake Lois.*



Note. Water flows between the chain of lakes in the following order: Hicks, Pattison, Long, and Lois. Map courtesy of (United States Geological Survey, 2020).

3.2 Research Methods

The methods for this investigative study were based on the Department of Ecology's standard operating procedures (SOP) for collecting grab samples from stormwater discharges (State Department of Ecology, 2018). The larger study designed by Thurston County encompasses the entire 2021 water year (October 2020 to September 2021). For the purposes of this thesis however, only storm events (i.e., periods of heavy rainfall with daily precipitation 6.35 millimeters or greater) from November 2020 to April 2021 were included. Sampling of stormwater effluent was collected from catchment outfalls around Long Lake during these events. Storm events were selected because they might contain typical "first flush effect" concentrations (Bach et al., 2010; Lee & Bang, 2000). The first flush effect can be defined as:

> a phenomenon in which a greater proportion of pollutant loads are washed off during the beginning of a rainfall event than other periods…[and] is more likely to occur in a smaller catchment with more impervious land surfaces (Qin et al., 2016).

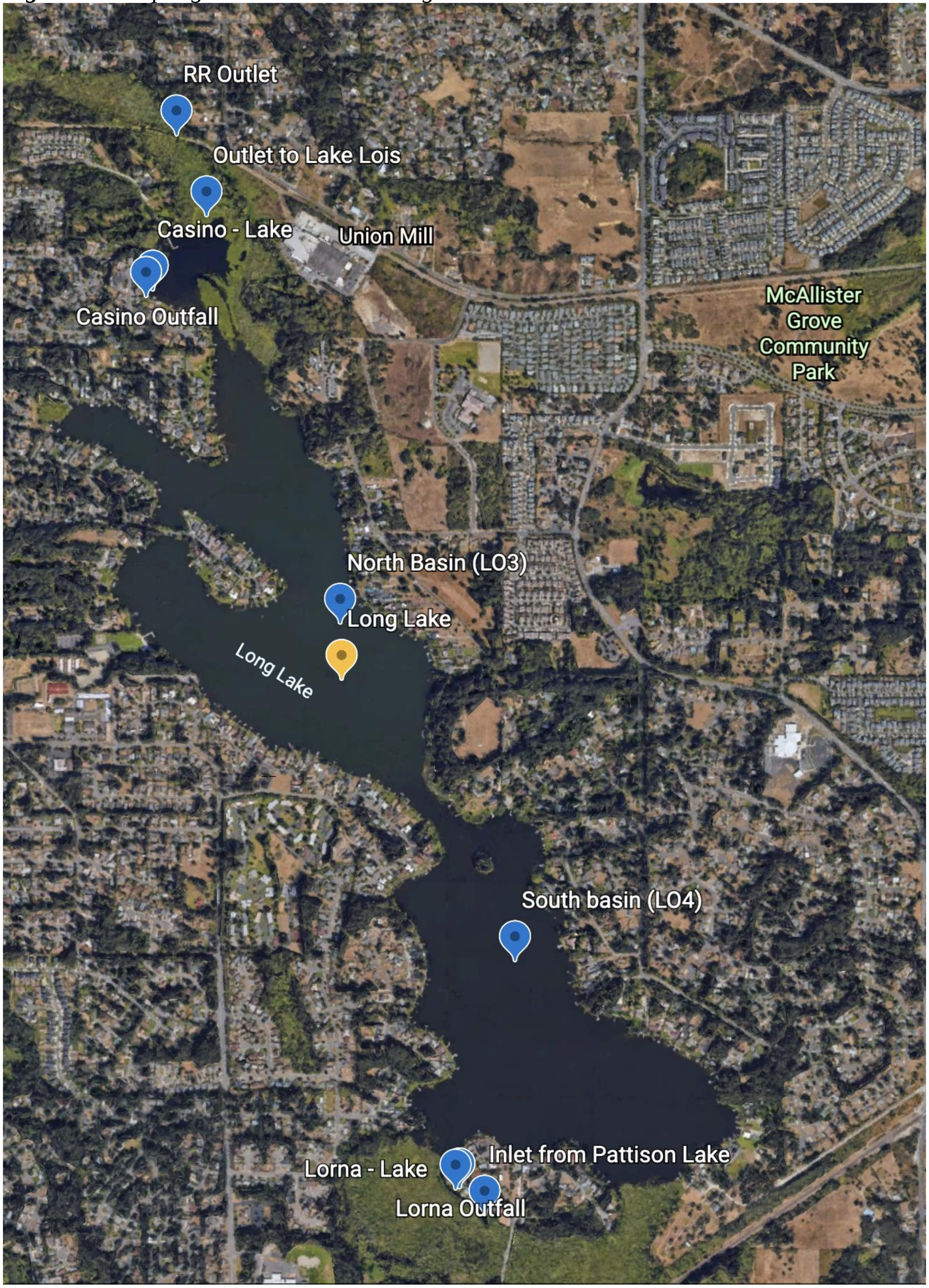
Meteorological information was obtained from the regional office of the National Oceanic and Atmospheric Administration (https://forecast.weather.gov/). All water sampling was done manually using *grab sampling* techniques. Grab samples are single discrete samples collected during a very short time period at a single location (United States Environmental Protection Agency, 2013). Samples were collected in 250 ml narrow-mouth polyethylene bottles, cleaned with a non-phosphate detergent (i.e., Alconox) that were rinsed thoroughly with tap then deionized water and air dried.

3.3 Site Selection

Per the Department of Ecology protocol, stormwater outfalls were selected based on their representational qualities of the urban area. Stormwater catchments, areas designed to channel neighborhood ditches to county installed outfalls, were identified and visited to assess accessibility. Sampling sites were determined to be free-flowing and unaffected by stagnated water. These sites were also appropriate representations of mixing due to the catchment design which pools multiple neighborhood ditch channels together before discharging into the lake. The locations of the two stormwater outfalls known as “Casino Drain” and “Lorna Drain” are depicted in Figures 10 and 11. Casino Drain is located in the north basin, whereas Lorna Drain is located in the south basin. These represent two of the nine stormwater outfalls present around Long Lake.

Also depicted in the figure are sampling points at Long Lake’s inlet “Pattison Inlet”, and two outlet sampling locations “Lake Lois” and “RR Outlet” (Fig. 10-13). Note that the inlet sampling does not represent water upstream of Long Lake; rather, the sampling location is in Long Lake, close to the inlet. As such, it represents the portion of Long Lake most influenced by the inlet from Pattison Lake. Similarly, the Lake Lois outlet represents water quality leaving the Long Lake system. The Lake Lois site is located within Long Lake opposite the RR Outlet. Finally, the RR Outlet site represents water quality exiting Long Lake after it has passed through the wetland structure which separates Long Lake from the beginning flow of Woodland Creek. The deepest points of each basin, north “LO3” and south “LO4,” were also sampled (Fig. 10). These sites were selected as they are relatively central within their respective basins representing where the confluence and mixing of water across the epilimnion in that basin would be maximized. Samples were collected at sites other than the stormwater outfalls to better understand water quality throughout the lake system: as water entered the lake, mixed between basins, and exited the lake.

Figure 10 *Sampling Locations across Long Lake.*



Note. A single sample was taken at each location. LO3 (North basin) and LO4 (South basin) are located above the lake's maximum depth in each basin. Image courtesy of Google Earth.

Figure 11 *Sampling sites in the southern basin sans LO4.*

Note. The Pattison Inlet sampling area is approximately eight meters in width. Image courtesy of Google Earth.

Figure 12 *Sampling sites in the northern basin sans LO3.*

Note. Image courtesy of Google Earth.

Figure 13 *Outlet sampling sites.*

Note. Image courtesy of Google Earth.

3.4 Field Sampling Methods

Sampling on storm event days included: one sample collected three to five meters from the outfall at five to ten centimeters depth from the lake surface. Another sample was collected mid-stream of stormwater effluent. Lake samples taken at this distance from the outfall might illustrate the gradient of nutrient concentrations between the drain and lake sites, as well as the influence of stormwater drains on the near-shore environment. A single sample was taken at both the inlet and outlet of the lake at the surface (5-10 cm depth) and within each basin at the surface above their deepest point respectively. A total of eight field samples were taken during each rainfall event in addition to one field blank for accuracy.

Storm Event Sites:

Two stormwater outfalls (Casino, Lorna) + two near-outfall (Casino Lake, Lorna Lake) + one inlet (Pattison) + one outlet (RR Outlet) + one north basin (LO3) + one south basin (LO4) + one field blank = eight samples plus one field blank.

On non-storm event (baseline) days, seven samples plus one field blank were collected. Stormwater outfalls were excluded from this sampling event as they were not running at the time. Additionally, the "outlet" site, Lois Lake, was taken during baseline events to monitor nutrients exiting the Long Lake system within the boundary of the lake.

Baseline Sites:

Two near-outfall (Casino-lake, Lorna-lake) + one inlet (Pattison) + two outlet (Lake Lois, RR Outlet) + one north basin (LO3) + one south basin (LO4) + one field blank = seven samples plus one field blank.

Samples were taken on the following dates: November 3, 2020; December 16, 2020; February 17, 2021; February 22, 2021; March 23, 2021. Of these dates, November 3, December 16, and February 22 were storm events, indicating three storm events sampled and two baseline events occurred.

Table 2 *Sampling dates, event type, and weather on sample day.*

Date	Weather on Sample Day	Temperature (°C) Monthly Average (Low/High)
November 03, 2020 (storm event)	Mostly cloudy, raining 6.35 mm precipitation over 24 hours 0.2 - 4.4 mph ESE wind	10.5 (0/15)
December 16, 2020 (storm event)	Partly cloudy, raining 13.208 mm precipitation over 24 hours	6 (-3/10)

	0.2 – 4.5 mph ESE wind	
February 17, 2021 (baseline)	Partly sunny 0.00 mm precipitation over 24 hours 0.3 – 4.4 mph ESE wind	6 (-3/13)
February 22, 2021 (storm event)	Mostly cloudy, raining 10.16 mm precipitation over 24 hours 0.4 – 4.4 mph ENE wind	7 (3/14)
March 19, 2021 (baseline)	Cloudy 3.556 mm precipitation over 24 hours 0-6 mph ENE wind	12 (2/13)
March 23, 2021 (baseline)	Partly sunny 1.778 mm precipitation over 24 hours 2-3 mph SSW wind	11 (2/13)

Samples were immediately stored on re-freezable ice packs in a portable cooler during collection and transport to the laboratory. All samples were analyzed within 24 hours of collection. IEH Analytical Laboratories located in Lake Forest Park, Washington performed the analysis. Water samples were analyzed for total nitrogen (TN), total phosphorus (TP), and soluble reactive phosphorus (SRP). TP and SRP were analyzed using the 365.1 method by semi-automated colorimetry. TN was analyzed using the 4500-N(Organic)- C. Semi-Micro-Kjeldahl method.

3.4.1 Water Quality and Depth Profile Measurements

Dissolved oxygen (DO), temperature, conductivity, and pH were measured at the same time as stormwater sampling using portable meters (Oakton Ion 6 Acorn Series for pH and YSI Pro2030 for the other measurements). These measurements of DO, pH, conductivity, and temperature were collected on February 17, February 22, and March 23, 2021. These are regular variables measured to understand water quality and overall lake health. Furthermore, low pH is indicative of anoxic conditions that can lead to the internal phosphorus loading Long Lake's water quality program hopes to address. Water quality measurements were collected at the surface of the LO3, LO4, Lorna Lake, Pattison Inlet, Casino Lake, Lake Lois, and RR Outlet sites on February 17. A more extensive depth profile (1 to 10 meters) was completed at LO3 and LO4 on February 22, and complete depth profile measurements (1 to 10 meters) were taken at Casino Lake, Lake Lois, LO3 and LO4 on March 23. Surface measurements at RR Outlet, Pattison Inlet, and Lorna Lake were taken on March 23 as well due to the shallow depth of these sites. Due to restrictions pertaining to the length of the Oakton pH meter cord, pH could only be taken at surface on all occasions.

3.4.2 Sample Analysis

Total phosphorus was measured following EPA's methodology 365.1 method analysis which is composed of two general procedural steps: (1) conversion of total phosphorus (orthophosphate, condensed phosphate, and organic phosphate) to dissolved orthophosphate, and (2) colorimetric determination of dissolved orthophosphate.

All samples are first mixed with a sulfuric acid solution, which a solution of ammonium persulfate is then added to. The sample is then boiled down and acidified using an acid wash solution composed of sulfuric acid and reagent water which converts all phosphorus forms (i.e., condensed and organic P) to orthophosphate. (Note: the acid wash is only applied to samples tested for TP and not SRP. Testing for SRP includes a filtration step before the colorimetric test. This biologically available, dissolved form of phosphorus is what is considered present in the filtrate of a sample filtered through a phosphorus-free filter of 0.45-micron pore size.) The test measures both dissolved and particulate orthophosphate before filtration. Ammonium molybdate and antimony potassium tartrate are then added to the treated sample. These compounds react in an acid medium with the dilute solutions of phosphorus to form an antimony-phosphomolybdate complex. This complex is reduced to an intensely blue-colored complex by ascorbic acid. The color is proportional to the phosphorus concentration which can then be measured by colorimetry (United States Environmental Protection Agency, 1993).

Only orthophosphate can be directly measured, as such other phosphorus compounds must be converted to this reactive form by various sample pretreatments described in the method (United States Environmental Protection Agency, 1993). The samples are then measured using colorimetry, a technique used to determine the number of colored compounds in a solution. This is accomplished using a colorimeter which can test the concentration of a solution by measuring its absorbance of a specific wavelength of light (Housecroft, 2006). A colorimetric test takes place at 650 to 660 or 880 nm in a 15-mm or 50-mm tubular flow cell. Higher concentrations can be determined by diluting the sample. IEH Analytical Laboratory uses the Astoria Pacific Segmented Flow instrument for their analysis which has a detection limit of 0.002 mg/L for TP and 0.001 mg/L for SRP.

Total nitrogen was measured following the 4500-N C. semi-micro Kjeldahl method as described in Standard Methods for the Examination of Water and Wastewater 23rd edition (2017). This protocol consists of three main steps: sample digestion, distillation, and ammonia determination. Using sulfuric acid, a variety of catalysts, and salts, this method converts organically bound nitrogen in samples to ammonium and is subsequently measured. This is the standard method for determining organic and ammonia nitrogen from water, and is applicable to samples containing high concentrations of organic nitrogen with the sum of organic N plus ammonia nitrogen (Baird & Eaton, 2017). IEH Analytical Laboratory uses the Alpkem Rapid Flow instrument for their analysis. The detection limit for total nitrogen was 0.050 mg/L.

3.5 Statistics

3.5.1 Nutrient Data

Two-sample t-test analyses were performed to identify whether drain nutrient concentrations were significantly different from lake nutrient concentrations using the RStudio Statistical Environment, Version 1.4.1106 (R Core Team, 2021). Prior to running tests, the data was assessed for normality. For each sampling event, sample location was the independent variable, whereas the nutrient concentration (TP, SRP, and TN) were the dependent variables. Variables were deemed significant when the p value was less than 0.05. TP and SRP were log transformed to achieve normal distributions. Relationships between the sample locations were assessed using the Wilcoxon Signed-Rank test.

Additionally, median nutrient (TP, SRP, TN) concentrations were compared with Washington State's nutrient load allocation limits and/or action levels as designated by reporting from the Puget lowlands ecoregion (Table 3). While various forms of nitrogen have suggested nutrient load limits, there is no definite limit for TN. Furthermore, sample analysis of TN was inclusive of ammonia and organic nitrogen only. As such, the sum of ammonia (0.034 mg/L) and organic nitrogen (0.007 mg/L) for a total of 0.041 mg/L was used as the comparative action limit. Additionally, the action level for TP (0.02 mg/L) was used for SRP as the Washington Administrative Code for surface water does not differentiate between the different forms of phosphorus. Nutrient concentrations were then analyzed for their distribution from this allocation level using the one-sided nonparametric Wilcoxon Rank-Sum test also known as the Mann-Whitney U-test.

Table 3 *Washington state nutrient load allocation guidance.*

Washington State Nutrient Load Allocation	Water Type	Nutrient Load Allocation / Action Limit
WAC-173-200-040 – Washington State Legislature	Groundwater	10 mg/L - Nitrate as Nitrogen
WAC 246-290-310 – Washington State Legislature	Maximum Contaminant Level (MCL) for drinking water	10 mg/L – Nitrate as Nitrogen 1 mg/L – Nitrite as Nitrogen

Deschutes River Total Maximum Daily Load: (Washington State Department of Ecology, 2015)	Groundwater	0.054 mg/L - organic phosphorus 0.052 mg/L - inorganic phosphorus 0.616 mg/L - nitrate 0.034 mg/L – ammonia 0.007 mg/L - organic nitrogen
WAC 173-201A-230 – Washington State Legislature	Surface waters	**0.02 mg/L – ambient total phosphorus**

Note. Nutrient load allocation limits in bold are the levels used for subsequent statistical analysis i.e., TP and SRP.

3.5.2 Storm Drain to Lake Comparisons

The Wilcoxon signed-rank test uses the ranks of the data measurements to test whether the frequency distributions of the two sample groups are the same. If the distribution of the two groups has the same shape, the test compares the location of the sample medians or means of the two groups (Whitlock & Schluter, 2015). The Wilcoxon signed-rank test was used to compare "drain" to "lake" nutrient concentrations, and to assess whether storm drains were a significant contributor of nutrients over the sampling period.

3.5.3 Storm Event to Baseline Comparisons

As the Wilcoxon signed-rank test requires equally comparable datasets (i.e., equal number of data points), the November 3 storm event was removed to refine the data to meet these parameters. This storm event was selected for removal as it was least similar to the other sampling events based on the time of year and season in which it was collected. The remaining sampling dates were closer in temporal and seasonal range.

3.5.4 Nutrient Load Allocation Limits

The median nutrient concentration was evaluated against Washington State's nutrient load allocation limits (Table 3) using the Wilcoxon rank-sum test. This test evaluates how far from the hypothesized median (i.e. nutrient load allocation limit) each data point lies. For the purposes of this study, the null hypothesis was that the median nutrient concentration was less than or equal to 0.041 mg/L (TN), 0.02 mg/L (TP), and 0.02 mg/L (SRP). The alternative hypothesis is that the median nutrient concentration is higher.

4. RESULTS

4.1 Total Nitrogen

Total nitrogen ranged from being below the detection limit of the instrument (< 0.05 mg/L) to 1.74 mg/L across all sites between November 3, 2020 to March 23, 2021. The average concentration was 0.74 mg/L with a standard deviation of 0.49 mg/L. Minimum values were found at the following sites: Lorna Drain and Pattison Inlet, whereas high values were found at these sites: Casino Drain, Casino Lake, Lorna lake, Lake Lois, and RR Outlet.

4.1.1 Values Relative to Action Limit

TN does not have specified nutrient criteria in Washington State, so there is no concrete action limit from which to compare the sample concentrations. However, due to the fact that sample analysis of TN for this study includes only the sum total of ammonia and organic nitrogen, the various forms of nitrogen that are delimited for the Budd Inlet, Capitol Lake, and Deschutes River watershed (Table 3) can be used as a comparative limit. The nitrogen values for ammonia (0.034 mg/L) and organic nitrogen (0.007 mg/L) from Table 3 were used to compare sample concentrations to an extrapolated action limit (0.041 mg/L) for the purposes of this study.

4.1.2 Storm Verses Baseline Sampling Events

Storm TN concentrations were not significantly higher than baseline values ($V = 50$, p value = 0.9), with the one exception being Casino Lake, where all storm events had higher concentrations. The December 16 storm event consistently had the highest TN concentrations across all sites, except for the Inlet, Outlet and Casino Drain values. Out of the three storm events where sampling occurred, the later season event (Feb. 22) almost always had lower TN concentrations relative to the other sites.

4.1.3 Storm Drain Verses Lake TN Concentrations

At the two sites (Casino Drain and Lorna Drain) where comparative measurements of the storm drain TN concentrations could be made with lake measurements, no significant difference was observed in storm drain concentrations relative to the lake near the storm drain. This accounts for the two dates where storm events happened, and lake samples were concurrently collected. Unfortunately, for the 2/22 storm event, lake data was unable to be collected.

4.1.4 Inlet and Outlet Comparisons

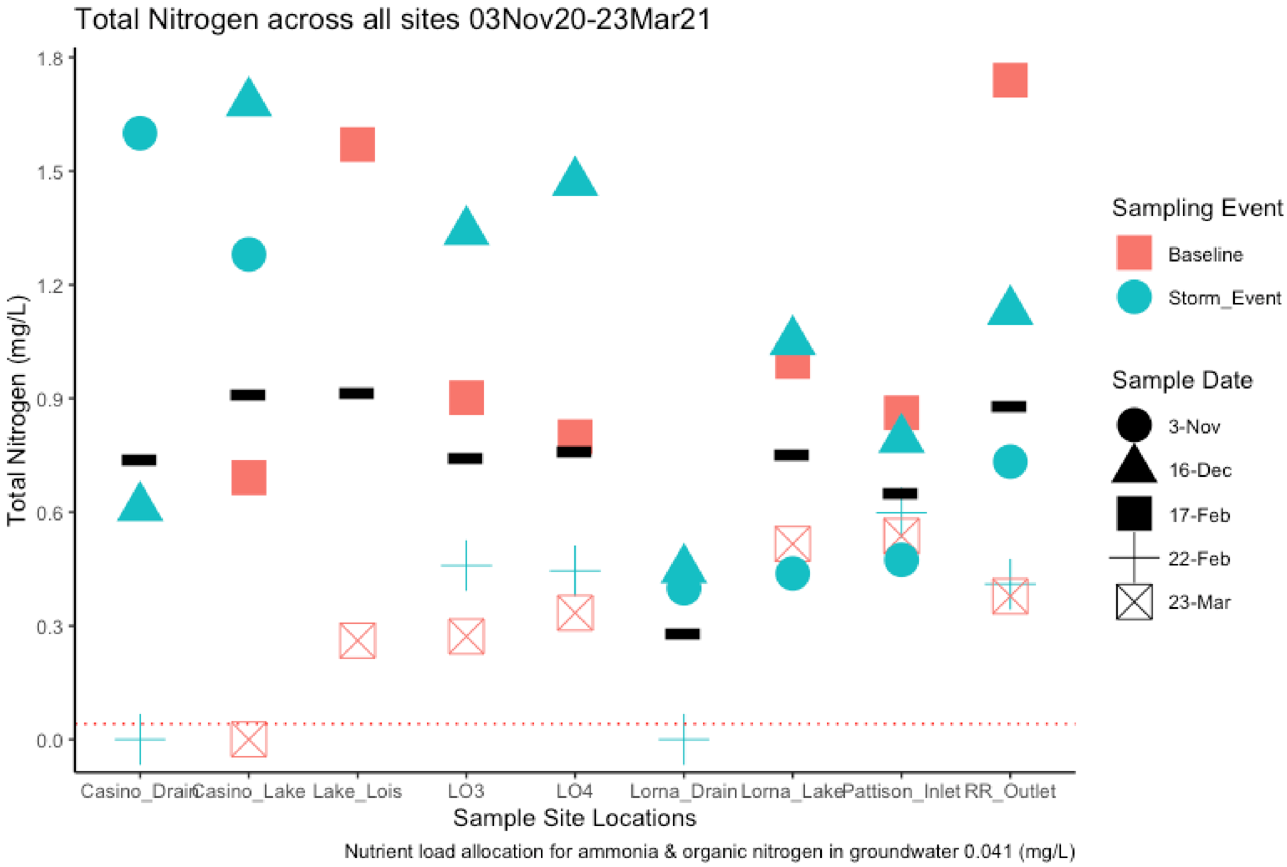
Pattison Inlet had much less variability in TN concentrations relative to the outlet of the lake (Fig. 17). Interestingly, TN concentrations were lower at Pattison Inlet for the first three sampling events relative to the outlet, including the two storm events. This trend did not hold for the 3rd late season storm event, with higher concentrations of nutrients at the inlet. There was no significant difference between Inlet and Outlet concentrations (i.e., RR Outlet) when all the data was combined.

Table 4 *Summary statistics for site sample concentrations of total nitrogen.*

Site Location	**Total Nitrogen (mg/L)**
Casino Drain	Range: 0.00 – 1.60 Average: 0.74 Standard deviation: 0.81
Casino Lake	Range: 0.00 – 1.68 Average: 0.91 Standard deviation: 0.73
Lorna Drain	Range: 0.00 – 0.45 Average: 0.28 Standard deviation: 0.25

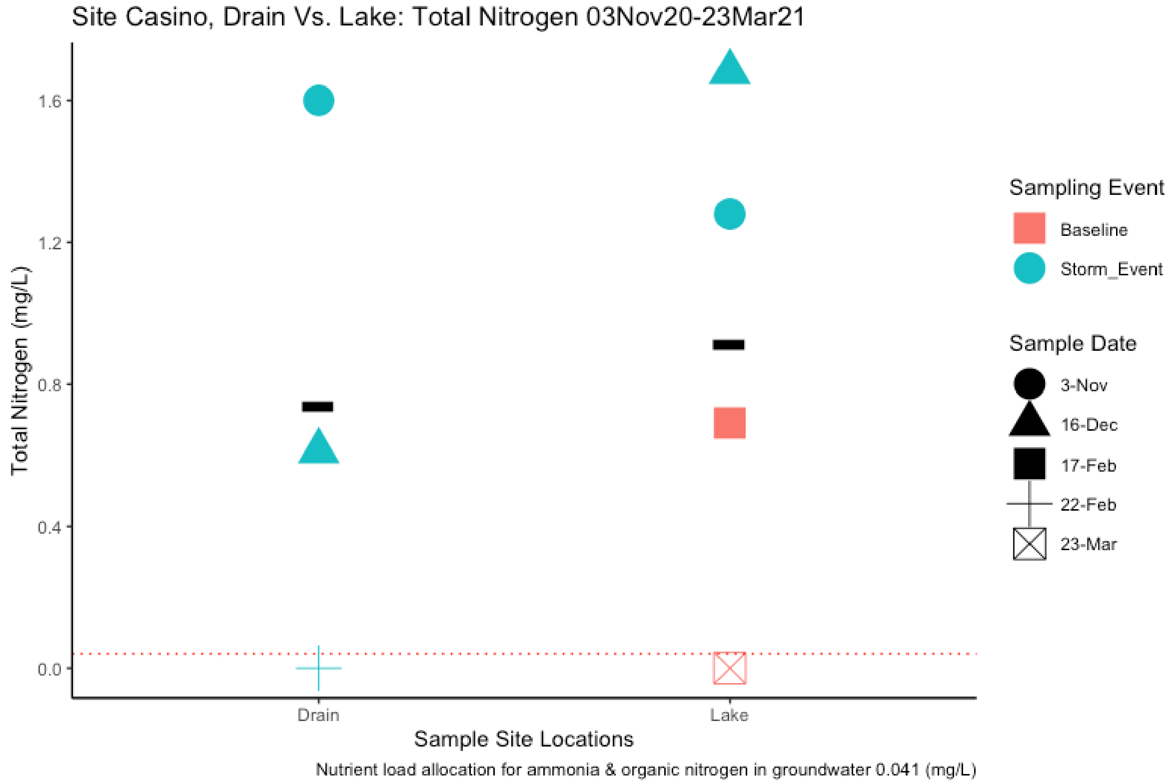
Lorna Lake	Range: 0.44 – 1.05 Average: 0.75 Standard deviation: 0.32
Pattison Inlet	Range: 0.47 – 0.86 Average: 0.65 Standard deviation: 0.17
Lake Lois Outlet	Range: 0.26 – 1.57 Average: 0.92 Standard deviation: 0.93
RR Outlet	Range: 0.38 – 1.74 Average: 0.88 Standard deviation: 0.57

Figure 14



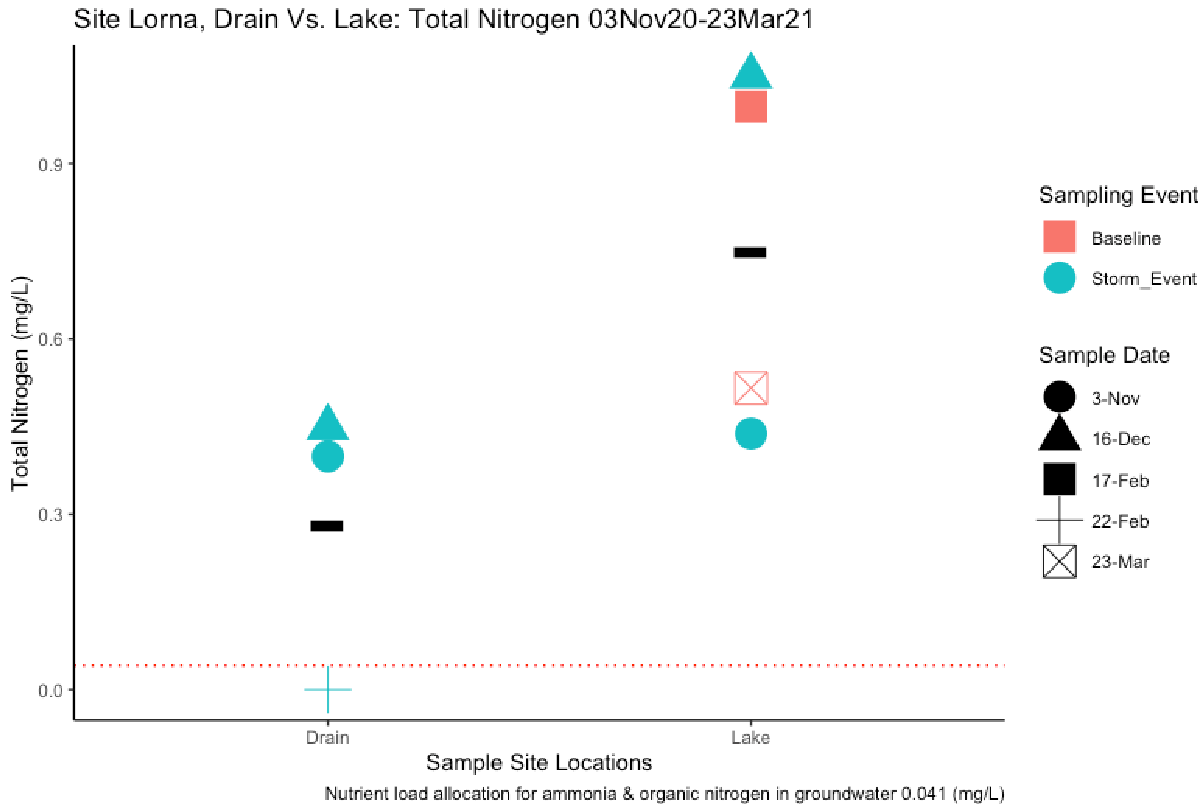
Note. The solid black dash represents the mean TN concentration at each site.

Figure 15



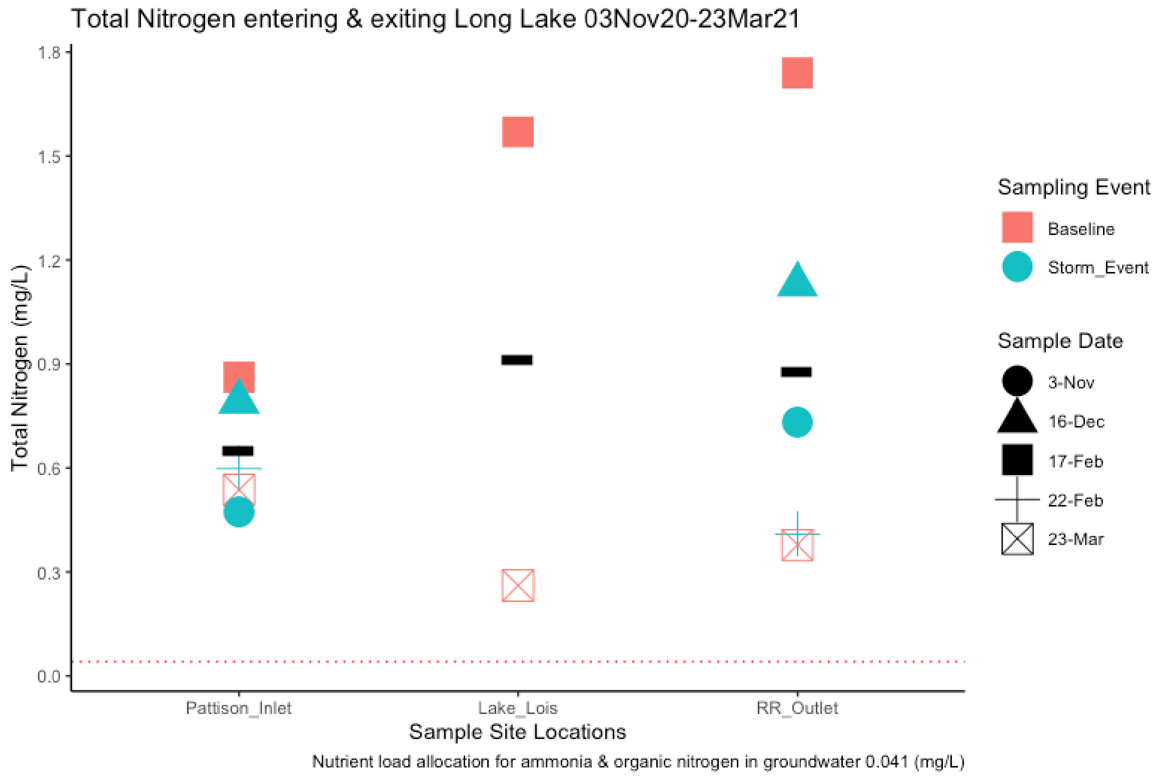
Note. The solid black dash represents the mean TN concentration at each site.

Figure 16



Note. The solid black dash represents the mean TN concentration at each site.

Figure 17



Note. The solid black dash represents the mean TN concentration at each site.

4.2 Total Phosphorus

Total phosphorus ranged from 0.01 mg/L to 0.41 mg/L across all sites between November 3, 2020 to March 23, 2021. The average concentration was 0.05 mg/L with a standard deviation of 0.07 mg/L. Minimum values were found at the following sites: Lake Lois, Lorna Lake, Pattison Inlet, and RR Outlet, whereas high values were found at these sites: Casino Drain, Casino Lake, and Lorna Drain.

4.2.1 Values Relative to Action Limit

The action level for total phosphorus as determined by Washington State surface water nutrient criteria was set at 0.02 mg/L (Table 3) (WAC 173-201A-230, 2006). Using all sites together, values were significantly higher than the action level ($V = 469$, p-value = 6.49e-05).

Values were higher than the action level at all sites and on all sampling occasions except for Casino Lake, LO3, Lorna Lake, Pattison Inlet (February 17, 2021) and Lake Lois, LO3, Lorna Lake, and Pattison Inlet (March 23, 2021) which were also baseline sampling events (Fig. 18). Values fell above 0.02 mg/L at every site except Lake Lois, which had the lowest concentration of TP values.

4.2.2 Storm Verses Baseline Sampling Events

Storm TP concentrations were significantly higher than baseline (V = 11, p value = 0.01). The November 3 storm event consistently had the highest TP concentrations across all sites. Out of the three storm events where sampling occurred, the later season event (Feb. 22) almost always had lower TP concentrations relative to the other sites.

4.2.3 Storm Drain Verses Lake TP Concentrations

At the two sites (Casino Drain and Lorna Drain) where comparative measurements of the storm drain TP concentrations could be made with lake measurements, no significant difference was observed in storm drain concentrations relative to the lake near the storm drain. Nonetheless, discharge from the storm drains did contain higher concentrations of TP compared to the lake samples. This accounts for the two dates where storm events happened, and lake samples were concurrently collected. Unfortunately, for the February 22 storm event, lake data was unable to be collected.

4.2.4 Inlet and Outlet Comparisons

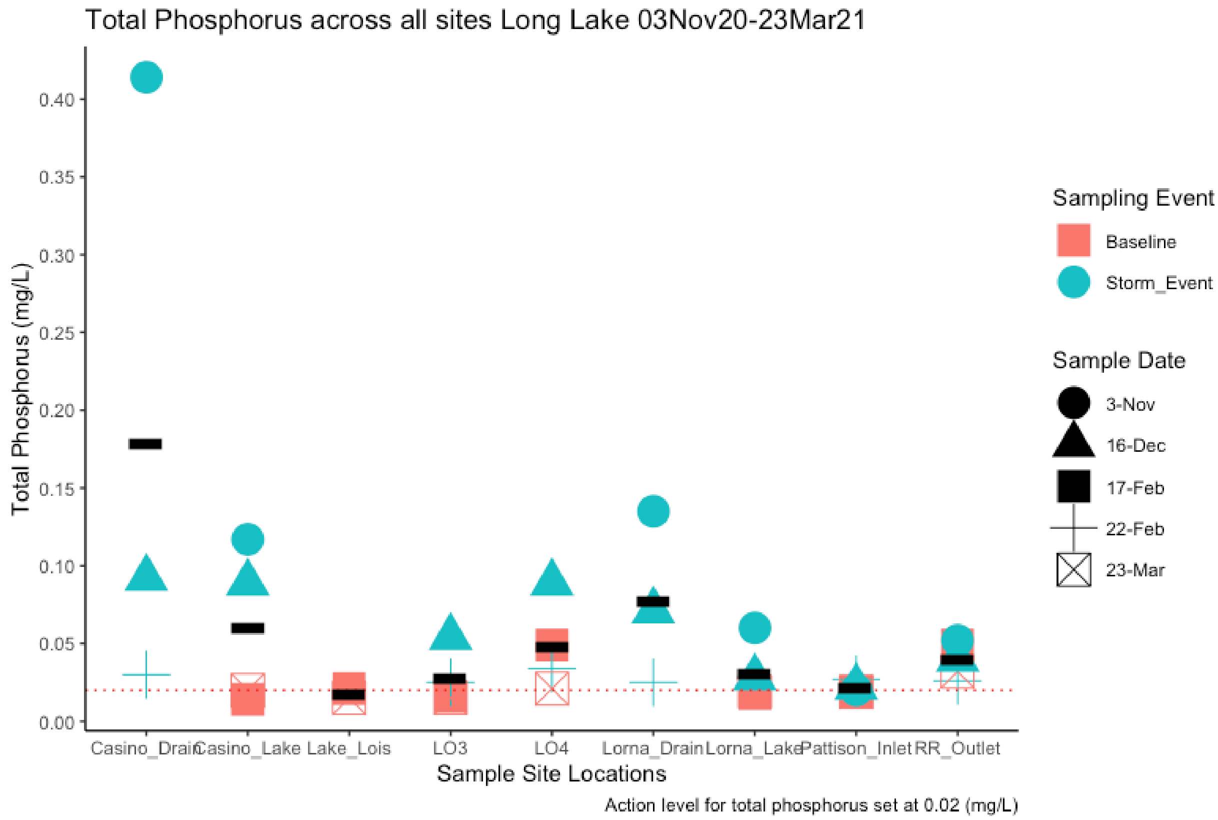
Similar to TN, Pattison Inlet had much less variability in TP concentrations relative to the outlet of the lake (Fig. 21). Interestingly, TP concentrations were lower at Pattison Inlet for the first three sampling events relative to RR Outlet, including the two storm events. This trend did not hold for the 3rd late season storm event, with concentrations higher at the inlet. There was no significant difference between Inlet and Outlet concentrations (i.e., RR Outlet) when all the data was combined.

Table 5 *Summary statistics for site sample concentrations of total phosphorus.*

Site Location	**Total Phosphorus (mg/L)**
Casino Drain	R: 0.03 – 0.41 Avg: 0.18 SD: 0.21
Casino Lake	R: 0.01 – 0.12 Avg: 0.06 SD: 0.05
Lorna Drain	R: 0.03 – 0.14 Avg: 0.08 SD: 0.06
Lorna Lake	R: 0.02 – 0.06 Avg: 0.03 SD: 0.02
Pattison Inlet	R: 0.02 – 0.03 Avg: 0.02 SD: 0.003
Lake Lois Outlet	R: 0.02 – 0.02 Avg: 0.02 SD: 0.004

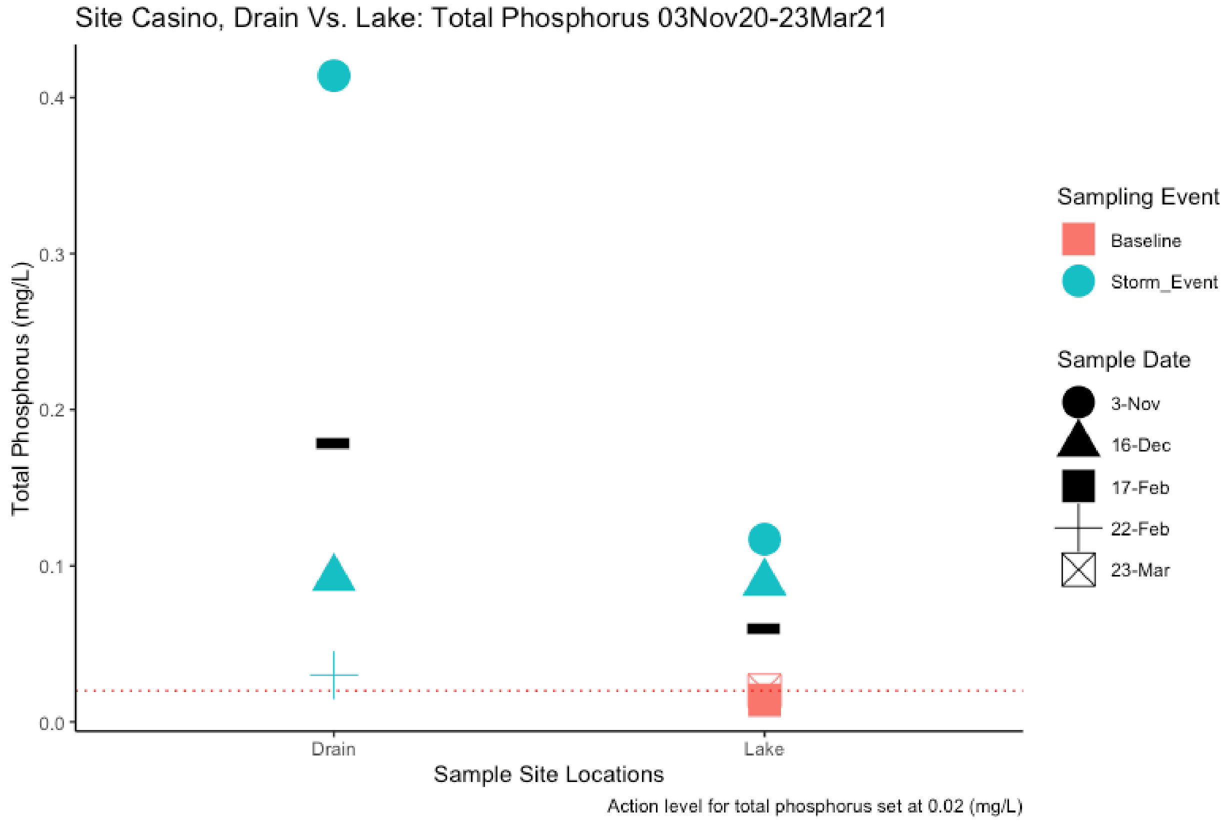
RR Outlet	R: 0.03 – 0.05 Avg: 0.04 SD: 0.01

Figure 18



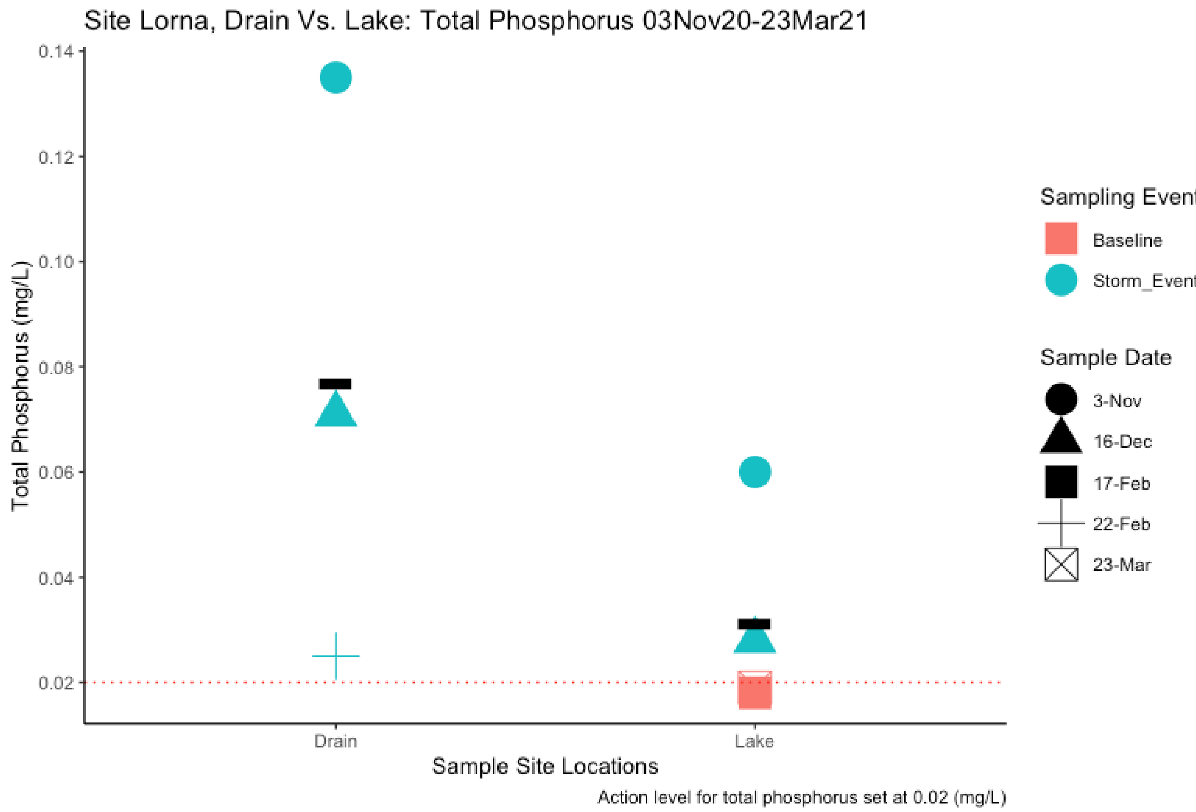
Note. The solid black dash represents the mean TP concentration at each site.

Figure 19



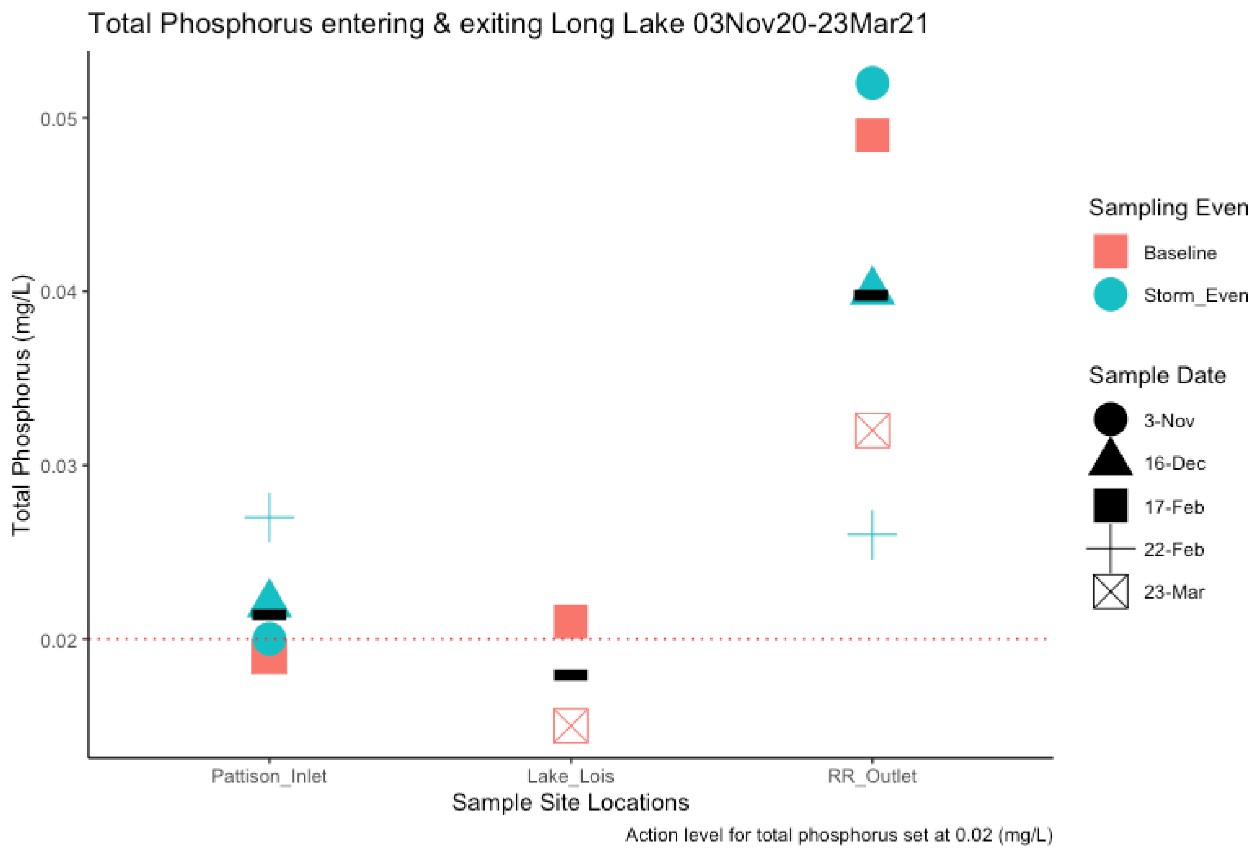
Note. Solid black dash represents the mean TP concentration at each site.

Figure 20



Note. Solid black dash represents the mean TP concentration at each site.

Figure 21



Note. Solid black dash represents the mean TP concentration at each site.

4.3 Soluble Reactive Phosphorus

Soluble reactive phosphorus ranged from being below the detection limit of the instrument (< 0.001 mg/L) to 0.05 mg/L across all sites between November 3, 2020 to March 23, 2021. The average concentration was 0.01 mg/L with a standard deviation of 0.01 mg/L. Minimum values were found at the following sites: Lake Lois, RR Outlet, Lorna Lake, Casino Lake, and Pattison Inlet, whereas high values were found at these sites: Casino Drain and Lorna Drain.

4.3.1 Values Relative to Action Limit

The action level used for SRP was the same action level for TP (0.02 mg/L) as the Washington Administrative Code for surface water does not differentiate between the different forms of phosphorus. Using all sites together, values were not significantly higher than the action level. Values rarely rose above 0.02 mg/L except during the November 3 storm event from each drain.

4.3.2 Storm Verses Baseline Sampling Events

Storm SRP concentrations were not significantly higher than baseline. However, drain sites consistently contributed higher concentrations during storm events. The November 3 storm event had the highest SRP concentrations across all sites, except at the Inlet and Outlet. Out of the three storm events where sampling occurred the later season event (Feb. 22) almost always had lower SRP concentrations relative to the other sites.

4.3.3 Storm Drain Verses Lake SRP Concentrations

At the two sites (Casino Drain and Lorna Drain) where comparative measurements of the storm drain SRP concentrations could be made with lake measurements, no significant difference was observed in storm drain concentrations relative to the lake near the storm drain. Nonetheless, the storm events transported higher concentrations of SRP into Long Lake. This accounts for the two dates where storm events happened, and lake samples were concurrently collected. Unfortunately, for the February 22 storm event, lake data was unable to be collected.

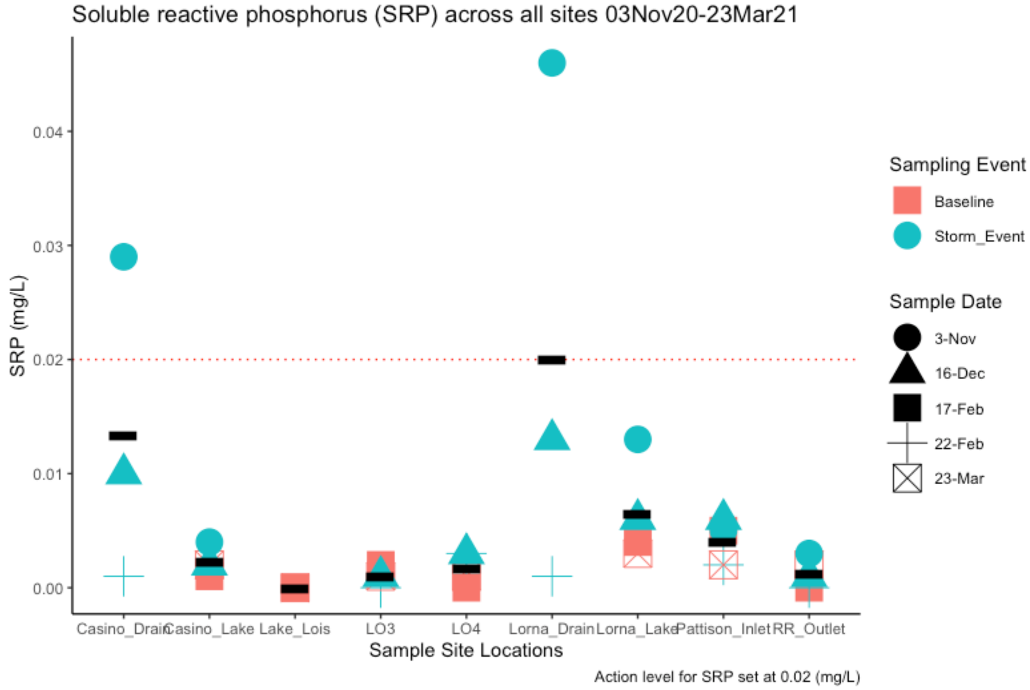
4.3.4 Inlet and Outlet Comparisons

The variability in SRP concentrations amongst Pattison Inlet, Lake Lois, and RR Outlet were relative similar (Fig. 25). SRP concentrations were highest at Pattison Inlet for the first three sampling events relative to the inlet, including the three storm events. There was no significant difference between Inlet and Outlet concentrations (i.e., RR Outlet) when all the data was combined (V = 10, p-value = 0.1).

Table 6 *Summary statistics for site sample concentrations of soluble reactive phosphorus (SRP).*

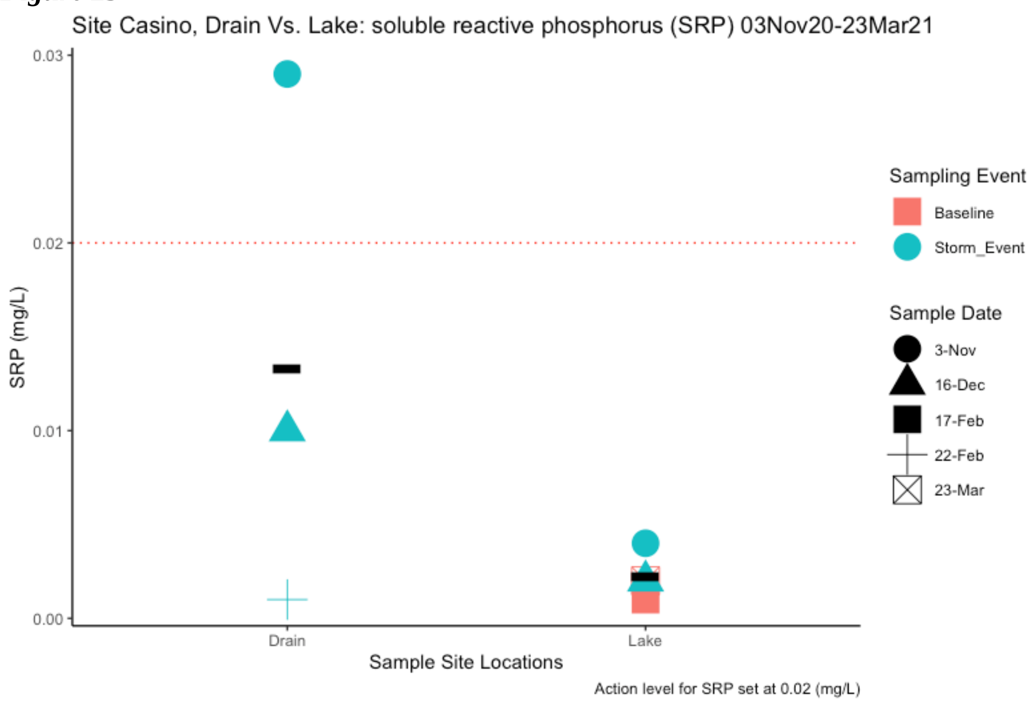
Site Location	**Soluble Reactive Phosphorus (mg/L)**
Casino Drain	R: 0.001 – 0.029 Avg: 0.013 SD: 0.014
Casino Lake	R: 0.001 – 0.004 Avg: 0.002 SD: 0.001
Lorna Drain	R: 0.001 – 0.046 Avg: 0.020 SD: 0.023
Lorna Lake	R: 0.003 – 0.013 Avg: 0.002 SD: 0.005
Pattison Inlet	R: 0.002 – 0.006 Avg: 0.004 SD: 0.002
Lake Lois Outlet	R: 0.00 – 0.00 Avg: 0.00 SD: 0.00
RR Outlet	R: 0.00 – 0.003 Avg: 0.001 SD: 0.001

Figure 22



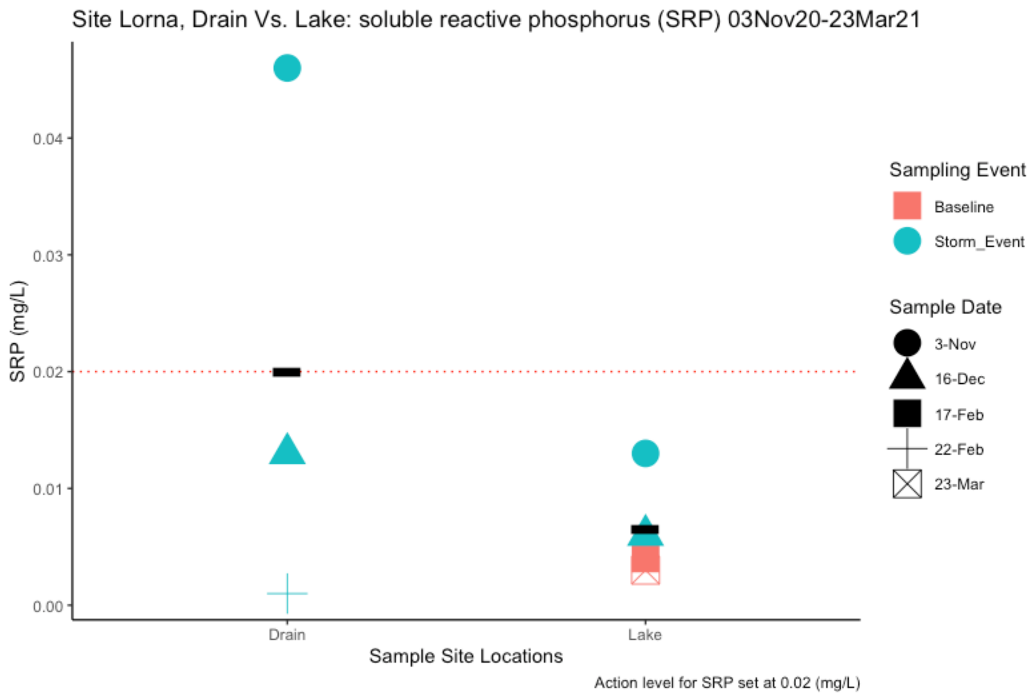
Note. The solid black dash represents the mean SRP concentration at each site.

Figure 23



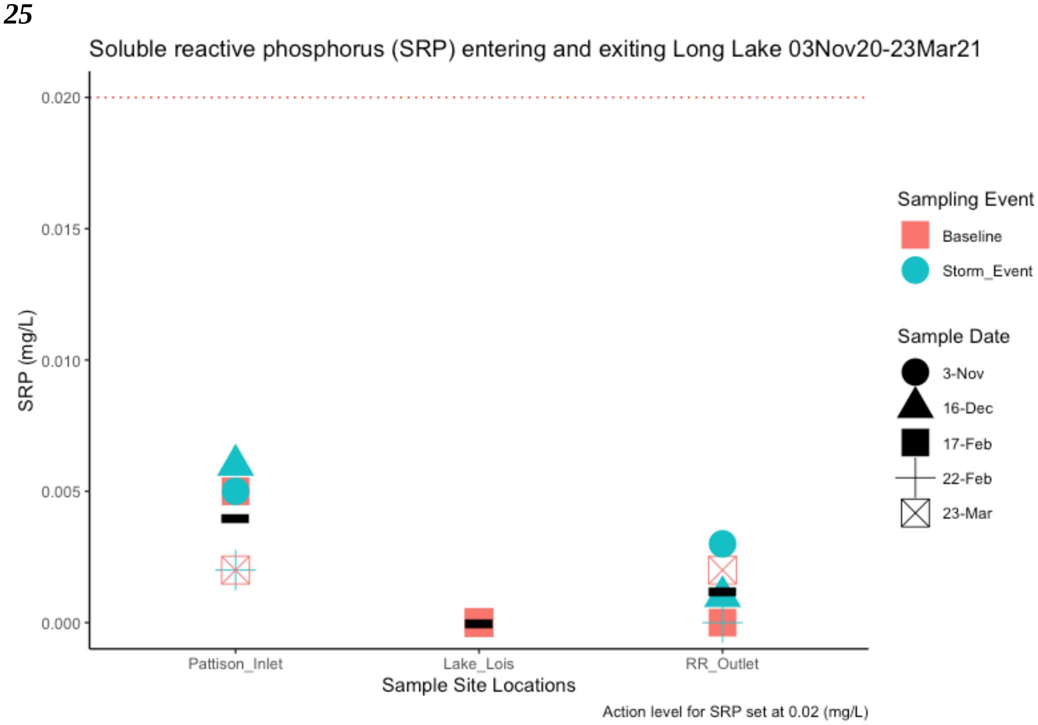
Note. The solid black dash represents the mean SRP concentration at each site.

Figure 24



Note. The solid black dash represents the mean SRP concentration at each site.

Figure 25



Note. The solid black dash represents the mean SRP concentration at each site

4.4 Depth Profiles

Depth profiles for dissolved oxygen (DO), conductivity, and temperature were constructed from data collected by the Oakton Ion 6 Acorn Series and YSI Pro2030 portable meters on February 17 (9 data points collected), February 22 (12 data points collected), and March 23, 2021 (26 data points collected).

4.4.1 Dissolved Oxygen

Dissolved oxygen ranged from 0.17 mg/L to 13.66 mg/L between 1- and 10-meters depth across all sites on February 17, February 22, and March 23, 2021. The total average concentration was 6.65 mg/L with a standard deviation of 5.69 mg/L when all dissolved oxygen readings were included. Due to the rapid decrease in DO, it was surmised that the lake was stratified. As such, DO readings taken near the surface, and before DO concentrations dropped massively, were considered to be within the epilimnion. Readings below 1 mg/L were then considered to be hypolimnetic. Average DO at all sites and dates where values were greater than 1 mg/L was 11.12 mg/L and 0.303 mg/L throughout the hypolimnion. DO was at its highest throughout the water column on February 22 (average epilimnion: 13.52 mg/L, average hypolimnion: 0.4 mg/L) (Table 10) and lowest during the March 23 readings (average epilimnion: 9.58 mg/L, average hypolimnion: 0.29 mg/L) (Table 11).

The two deepest points in Long Lake, LO3 (North basin) and LO4 (South basin), were graphed to display the changes in oxygen with depth (Fig. 26-28). Over the course of the three monitoring events, DO decreases in the epilimnion. There is an apparent decrease in the depth of the epilimnion as DO readings below 1 mg/L occur closer to the surface of Long Lake. Additionally, DO readings taken from the Lorna Lake and Pattison Inlet sites are especially low compared, which is further surprising given that data was collected near the surface.

Table 7 *Dissolved oxygen readings: February 17, 2021.*

Location	**Date**	**DO (mg/L)**	**DO%**
LO4 (Island): 10 m	17-Feb	11.29	87.8
1m/surface		13.06	101.7
Pattison Inlet		6.42	49
Lorna Lake		5.93	45.1
LO3: 10 m		0.27	2.1
0.3 m/surface		12.16	95.6
Lake Lois Outlet		12.74	100.6
Casino Lake		12.84	101
RR Outlet		12.38	98.5

Figure 26 *Changes in oxygen with depth at LO3 & LO4, 2/17.*

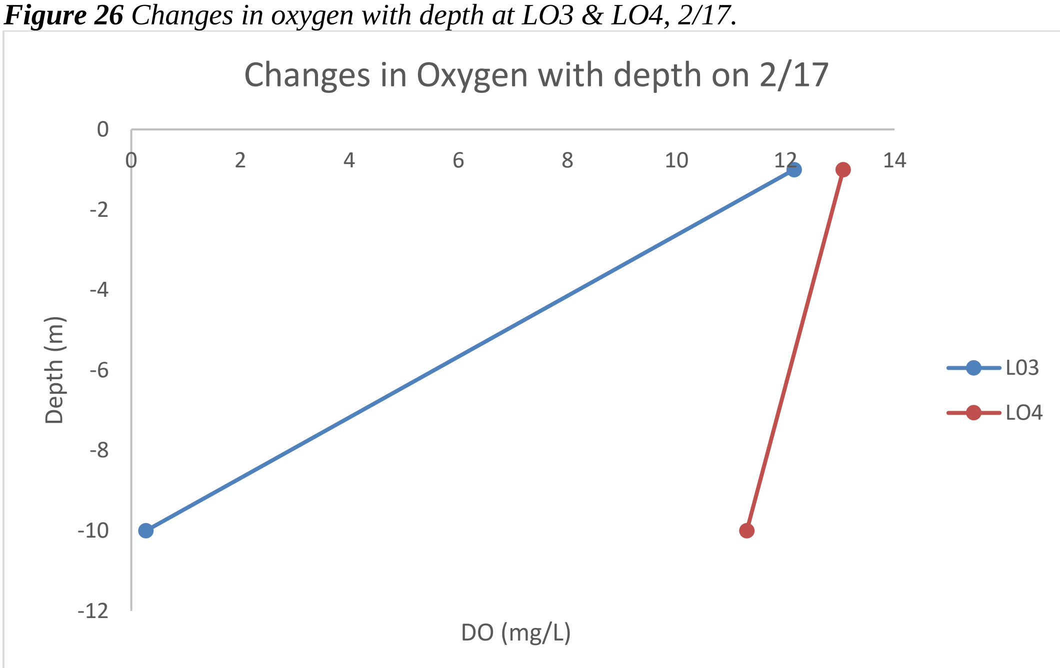


Figure 27 *Changes in oxygen with depth at LO3 & LO4, 2/22.*

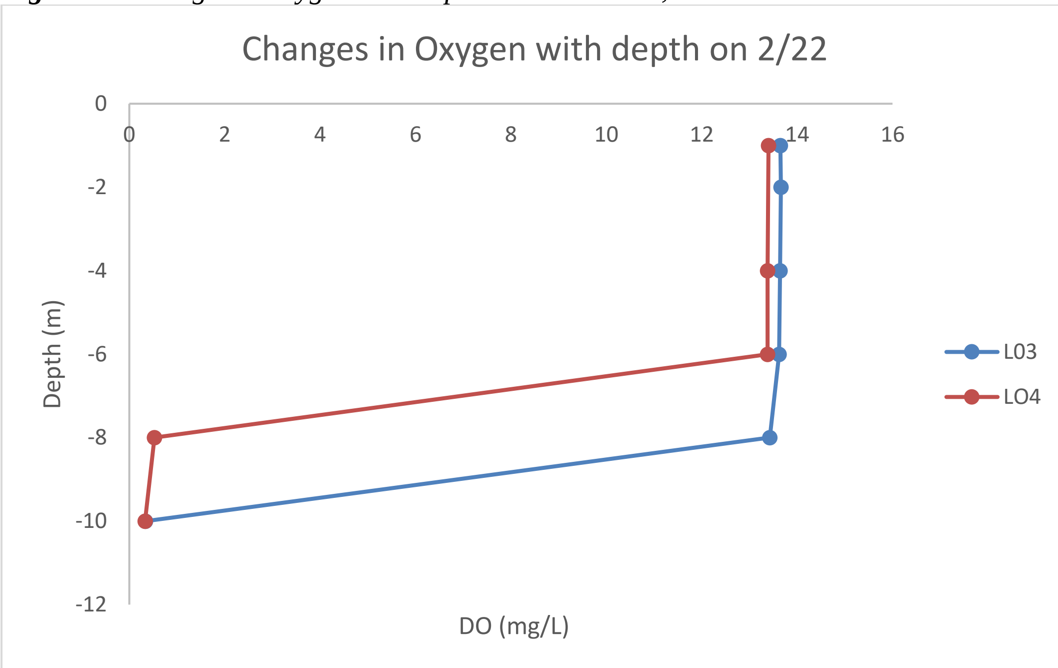
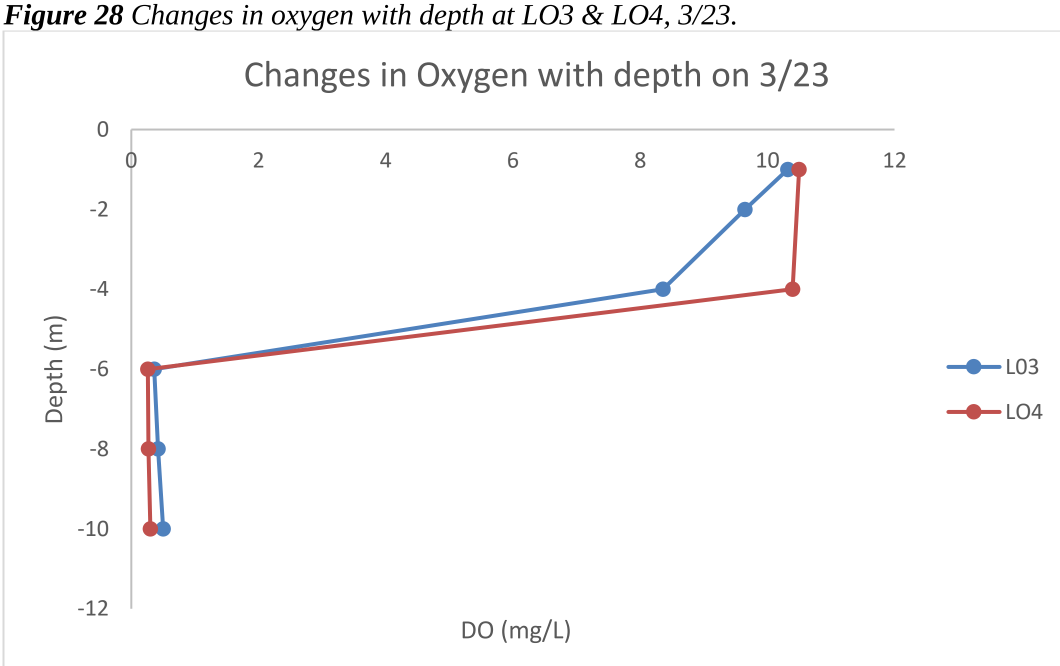


Figure 28 *Changes in oxygen with depth at LO3 & LO4, 3/23.*



4.4.2 Conductivity

Conductivity ranged from 113.7 uS/cm to 190.7 uS/cm between 1- and 10-meters depth across all sites on February 17, February 22, and March 23, 2021. The total average conductivity was 131.02 uS/cm with a standard deviation of 18.1 uS/cm when all conductivity readings were included. The average conductivity within the epilimnion across all sites and dates was 124.73 uS/cm and 140.28 uS/cm within the hypolimnion. Conductivity readings were at their highest and lowest points throughout the water column on March 23 (epilimnion: 113.7 uS/cm, LO3; hypolimnion: 190.7 uS/cm, Casino Lake) (Table 14) and lowest on average during the February 17 readings (average epilimnion: 126.425 uS/cm, average hypolimnion: 118.8 uS/cm) (Table 12). Conductivity was consistently higher in the South basin (LO4) except 10 meters below the surface within the hypolimnion of LO3 on February 22 (Fig. 29 – 31).

Figure 29 *Changes in conductivity with depth, 2/17.*

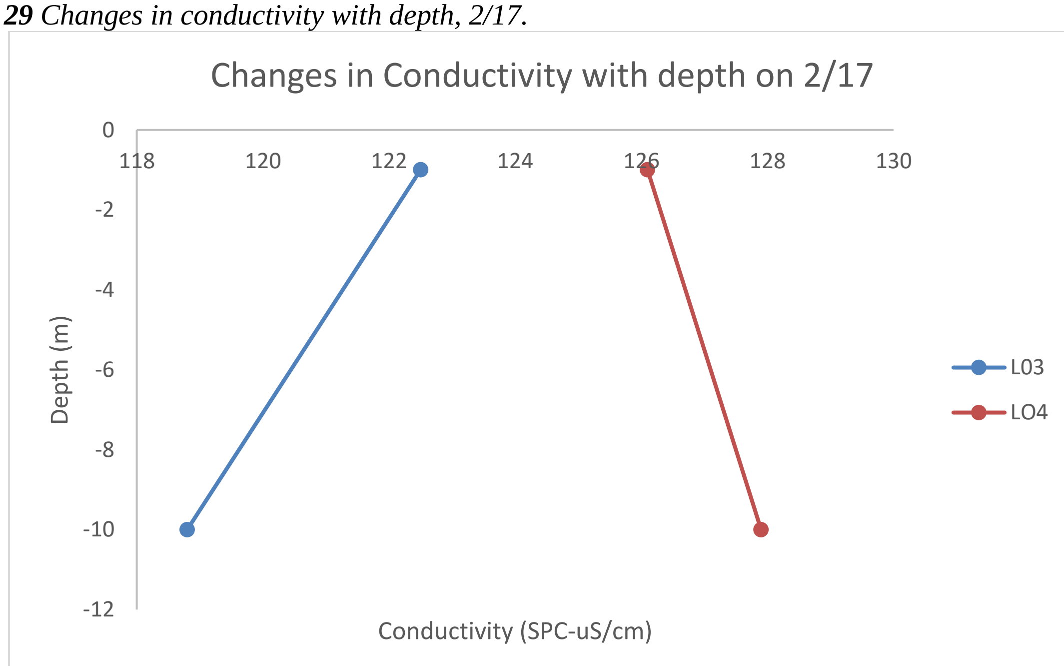


Figure 30 *Changes in conductivity with depth, 2/22.*

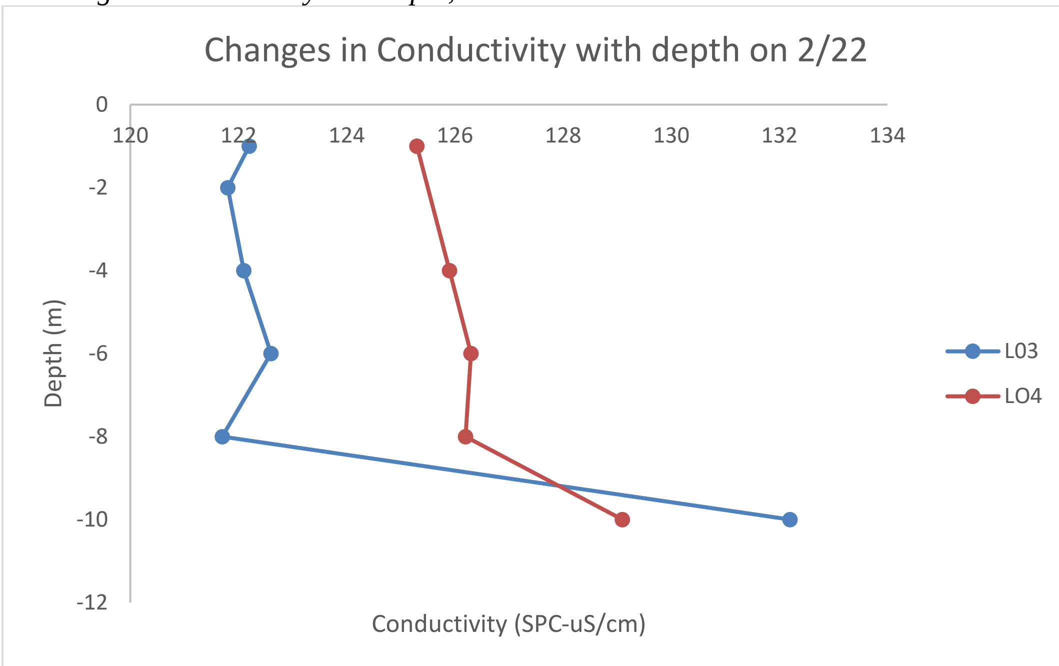
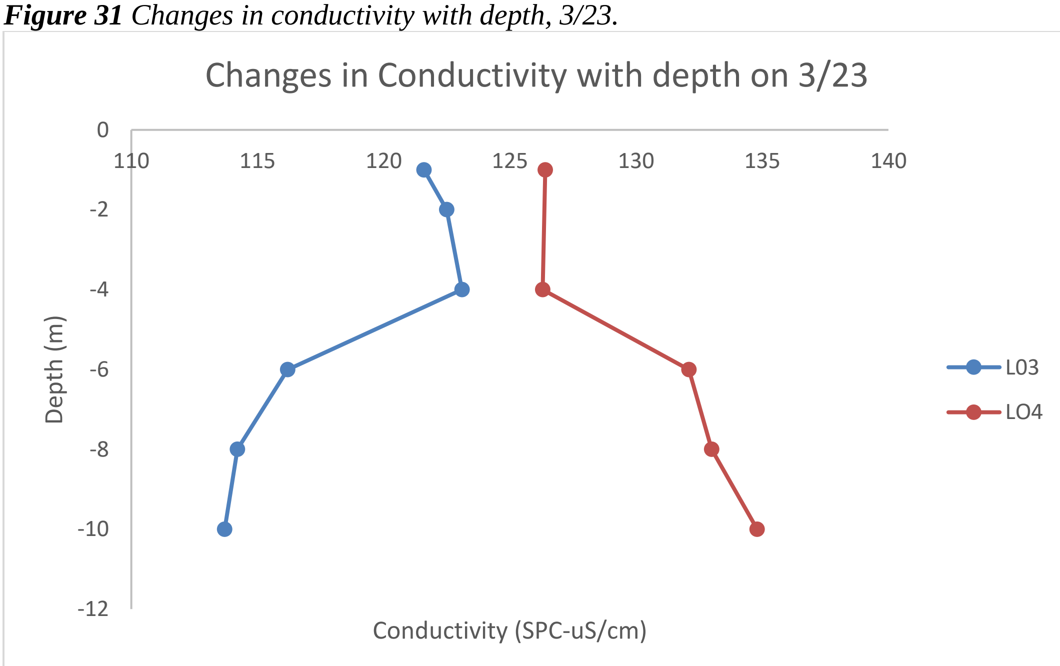


Figure 31 *Changes in conductivity with depth, 3/23.*



4.4.3 Temperature and pH

Temperature ranged from 3.7 °C and 11 °C between 1- and 10-meters depth across all sites on February 17, February 22, and March 23, 2021. The total average temperature was 7.4 °C with a standard deviation of 1.9 °C when all temperature readings were included. The average temperature on each date was as follows: February 17 (4.8 °C), February 22 (5.9 °C) and March 23 (9 °C). The average temperature within the epilimnion across all sites and dates was 6.8 °C and 8.3 °C within the hypolimnion. Temperature was at its highest on March 23 (Table 17: 11 °C, RR Outlet) and lowest on February 17 (Table 15: 3.7 °C, Lorna Lake). The temperature was consistently higher in the North basin (LO3) except within the hypolimnion of LO4 (approximately 8 – 10 meters depth) on March 23 (Fig. 32 – 34).

The pH meter itself was less than a meter long (approximately 0.6 m). As such, pH readings were logged only at the lake surface. The average pH on each date was as follows: February 17 (7.5), February 22 (7.3) and March 23 (6.9). The average pH across the three dates was 7.2. The lowest reading was at the RR Outlet site on March 23 (Table 17: 6.6 pH), and the highest reading was logged at Lorna Lake on February 17 (Table 15: 7.3 pH).

Table 8 *Temperature and pH readings, 2/17.*

Location	pH	Temp (°C)
LO4 (Island): 0.3 m/surface	7.65	4.6
10 m		4.7
Lorna Lake/inlet	7.46	3.8
Lorna Lake	7.26	3.7
LO3: 0.3 m/surface	7.76	4.9
10 m		5.4

Lois Lake Outlet	7.62	5.1
Casino	7.67	5.2
RR	7.46	5.6

Figure 32 *Changes in temperature with depth, 2/17.*

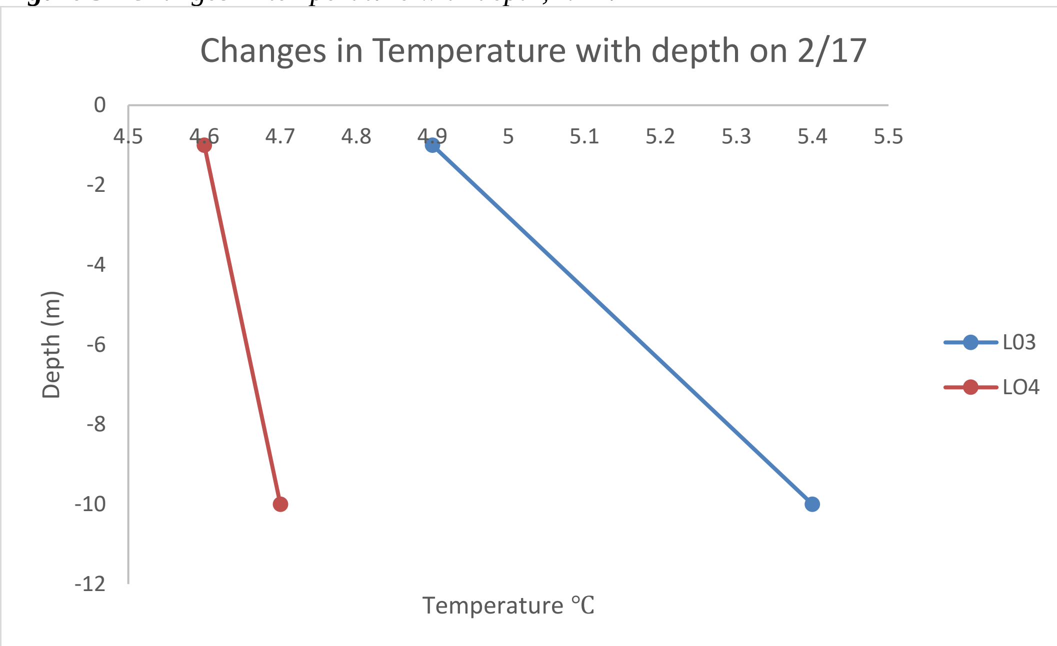


Table 9 *Temperature and pH readings, 2/22.*

Location	**pH**	**Temp (°C)**
LO4 (Island): 0.3 m/surface	7.65	5.9
2 m		5.9
6 m		5.9
8 m		5.9
10 m		5.9
LO3: 0.3 m/surface	7	6
1 m		6
2 m		6

4 m		5.9
6 m		5.9
8 m		5.9
10 m		6

Figure 33 *Changes in temperature with depth, 2/22.*

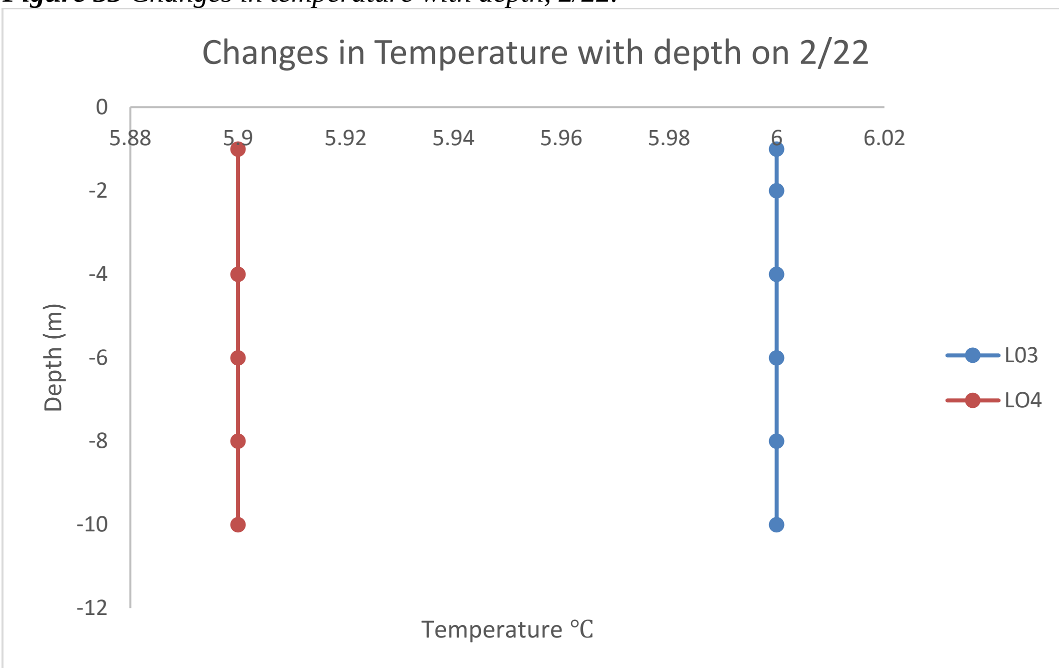
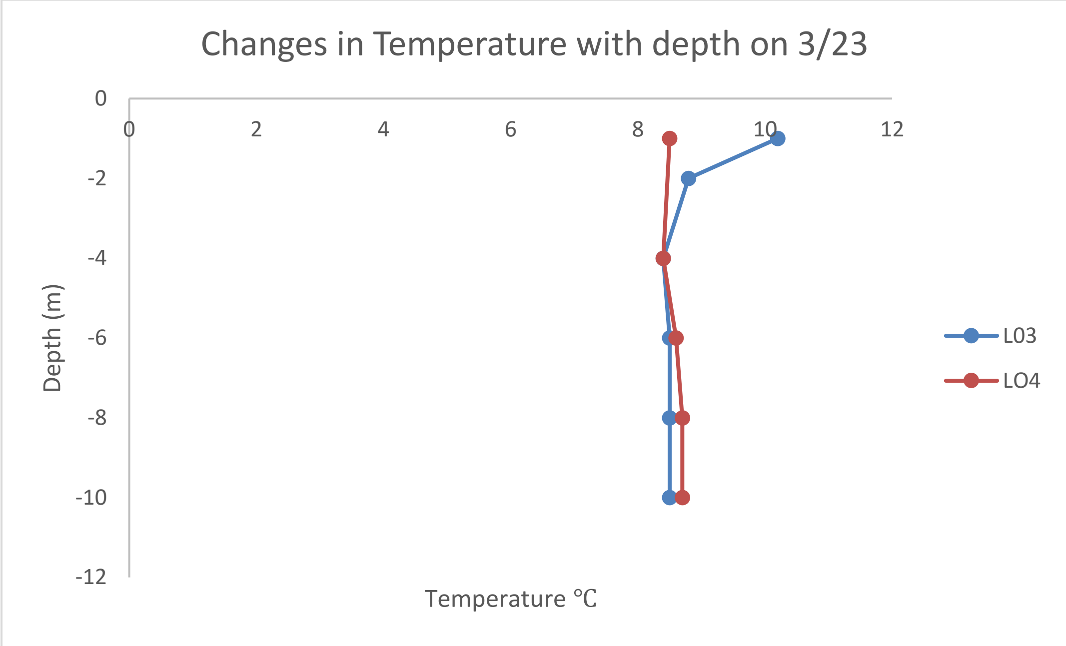


Table 10 *Temperature and pH readings, 3/23.*

Location	**pH**	**Temp (°C)**
LO4 (Island): 10 m		8.7
7 m		8.7
5 m		8.6
2 m		8.4
0.3 m/surface	6.95	8.5
Pattison Inlet	7.35	7.9
Lorna Lake	6.74	7.8

LO3: 10 m		8.5
8 m		8.5
6 m		8.5
4 m		8.4
2 m		8.8
0.3 m	6.83	10.2
Lois Lake Outlet: 10 m		8.8
7 m		8.7
5 m		8.7
3 m		8.8
0.3 m/surface	6.56	9.5
Casino Lake: 10 m		9.3
8 m		9.3
6 m		9.2
4 m		9.3
2 m		9.4
1 m		9.4
0.3 m/surface	7.32	10.2
RR	6.6	11

Figure 34 *Changes in temperature with depth, 3/23.*



5. DISCUSSION

5.1 Dissolved Oxygen

An important outcome of this study is that it has demonstrated that dissolved oxygen concentrations at depth were remarkably low. This is consistent with previous work at Long Lake (Thurston County Environmental Health Division, 2017, 2019). How do these low oxygen concentrations potentially impact the organisms found in the water? According to the Water Quality Standards for Surface Waters of the State of Washington (2004), dissolved oxygen criteria for aquatic life are as follows:

Table 11 *Dissolved Oxygen Criteria for Aquatic Life in Fresh Water*

Category	**Lowest 1-Day Minimum**
Char Spawning and Rearing	9.5 mg/L
Core Summer Salmonid Habitat	9.5 mg/L
Salmonid Spawning, Rearing, and Migration	8.0 mg/L
Salmonid Rearing and Migration **Only**	6.5 mg/L
Non-anadromous Interior Redband Trout	8.0 mg/L
Indigenous Warm Water Species	6.5 mg/L

Note. Table adapted from Water Quality Standards for Surface Waters of the State of Washington, Table 200 (1)(d) (2004).

According to Washington State water quality standards, DO levels should not decrease more than 0.2 mg/L from the standards set in Table 18 in a single day to maintain habitat viable for the aforementioned fish. Furthermore, DO levels less than 4 mg/L would not support many fish at all (Wetzel, 2001). The dissolved oxygen standards above give appropriate levels to consider. As such, the lowest DO levels recorded in the epilimnion measured during February and March ranged from 5.93 – 7.35 mg/L, which was supportive of life, but not for all types of fish listed in the above table. Furthermore, while the surface waters tended to have acceptable DO concentrations, it is worth noting that two locations (Lorna Lake and Casino Lake) consistently had lower DO concentrations. The Lorna sites are located in a channel leading to the south basin, and the Casino sites are located in a cove on the far north side of Long Lake. As such, both areas do not receive much wind action as they are relatively sheltered. As a result, both of these areas are less aerated than the majority of the lake which may explain their lower DO levels (personal communication with Paula Cracknell, Aquatic Resource Specialist, 2021). However, the proximity of Lorna and Casino Lake sites to a stormwater outfall might also be a contributing factor. It is interesting that DO dropped significantly near the surface at the Casino Lake site. Future site-specific study as well as the impact stormwater has on near-shore environments is warranted.

DO levels were 0.27 mg/L (LO3) on February 17, 0.33 mg/L (LO4) on February 22, and 0.2 mg/L (Casino Lake and Lake Lois) on March 23 within the hypolimnion. The lowest DO concentrations were recorded concurrently with warmer temperature readings. It is possible that increased biotic activity utilized more DO as a result. Low DO levels are not uncommon in Long Lake. Water quality reporting for Long Lake has noted DO approaching 0 mg/L near the bottom of the lake during periods of stratification (Thurston County Environmental Health Division, 2017). Low levels of DO are expected as oxygen is utilized for redox processes occurring within the hypolimnion. Long Lake is eutrophic, thus a highly productive system (Thurston County Environmental Health Division, 2017, 2018, 2019). Invasive plants, algal blooms, and native species contribute to the abundance of organic matter which decays in the hypolimnion consuming oxygen.

Anoxic conditions cause phosphorus to be released from benthic and littoral sediments due to the diminished effect of a micro-layer of ferric acid [FeO(OH)] at the sediment-water interface which absorbs phosphorus in oxic conditions (Kleeberg & Dudel, 1997; Mortimer, 1941; Tammeorg et al., 2017). Additionally, absorption of CO_2 from abundant macrophyte growth can alter water chemistry by raising the pH and affecting nutrient dynamics as well (Welch et al., 2005). For example, an increase in pH from 8.0 to 9.0 at least doubles the rate of P released from oxic littoral sediments (Barko & James, 1998). This pH change can easily occur in actively growing macrophyte beds. Furthermore, anoxic conditions caused by night respiration can enhance sediment P release as discussed above (Welch et al., 2005). The impact abundant macrophyte growth has on lake water quality can be an important reason for their control as exhibited by the efforts of Long Lake's LMD. The Long Lake LMD is actively engaged in aquatic plant removal due to the overabundance of both invasive and native species (Lake Management District #21, 2017). It is recommended that future studies incorporate DO and pH readings at depth to further investigate nutrient dynamics in conjunction with aquatic plant removal in Long Lake.

5.2 Actionable Nutrient Load Allocation Limits

There was not enough data to conclude that the nutrient concentrations of either storm drains or lake samples were statistically significant when compared with the action level for nutrients. Nonetheless, it can be observed that the samples analyzed were often above designated action levels or nutrient allocation limits set for TN, TP, and SRP. Furthermore, the nutrient concentrations of TP were consistently much higher than its designated action level specifically during storm events. Baseline concentrations of TP dropped below the action level suggestive of its dilution in the larger lake water body. Future mitigation options targeting stormwater can hopefully reduce TP inputs to make TP reductions and other mitigation efforts (i.e., alum treatment) most effective.

Nutrient criteria do not exist for TN and SRP in Washington State (personal communication, Washington Department of Ecology, 2021). Nutrient load allocation limits for TN (ammonia and organic nitrogen in this study) was derived from TMDL reporting for a nearby watershed (Washington State Department of Ecology, 2012). Additionally, the action level for TP (0.02 mg/L) was used for SRP as the Washington Administrative Code for surface water does not differentiate between the different forms of phosphorus. It is recommended that future studies continue analyzing for nutrients and include all inorganic forms of nitrogen (ammonia, nitrite, nitrate).

5.3 Temporal Variability of Storm Events

A total of three storm events were sampled over the course of the study period (November 2020 – March 2021). Nutrient concentrations varied considerably between these three events suggesting a temporal difference in nutrient accumulation and discharge. When compared to each other, it is apparent that the November 3 storm event was the greatest contributor of TP and SRP across all sites sampled. TP and SRP concentrations decreased by the December 16 storm event and further still by the February 22 storm event (Table 19). Nevertheless, TP concentrations were greater than or equal to the 0.02 mg/L nutrient action level across all sites during each storm event. TN does not distinctly follow this trend as the December 16 storm contributed more TN than the November 3 storm event, followed by the February 22 event (Table 19).

Higher nutrient concentrations in November could have resulted from fall leaf litter. Soil and organic material contribute significant concentrations of nutrients in urban runoff in areas with high tree or vegetation cover. It was estimated that 50% of the annual export of N and P from an urban watershed in Saint Paul, Minnesota was due to leaf litter transported through winter snowmelt (Bratt et al., 2017). However, while there are trees and vegetation amongst the houses surrounding Long Lake, the area around Long Lake is heavily urbanized, and no natural shoreline exists. It may be more likely that the summer accumulation of pollutants on impervious surfaces played a more significant role than leaf litter. In order to test this idea fully, it is recommended that sampling occur earlier in the fall season, as the first sample taken for this study was done on November 3, 2020. The size and shape of lawns, connectivity of impervious surfaces, and type of stormwater infrastructure can be more predictive of water quality considering this specific watershed than that of an area densely forested (Yang & Lusk, 2018).

5.4 Storm Drain Effluent

Nutrient concentrations from storm drain runoff were noticeably higher than samples taken from the lake sites during storm events. Casino Drain appeared to be the greatest contributor of nutrients overall with higher levels of TN, TP and SRP during the November 3 and December 16 storm events. Lorna Drain also contained higher levels of SRP on November 3 and December 16. SRP concentrations are noticeably higher than baseline levels during the November 3 and December 16 storm events suggestive of the influence storm events and stormwater outfalls have on transporting this nutrient.

5.5 Inlet vs. Outlet

Nutrient concentrations of TP and TN appeared to increase as water passed through the Long Lake system with equal variability in concentrations between baseline and storm event sampling. However, further study is necessary to achieve the statistic power to assess the significance of this hypothesis. TN concentrations increase from inlet to outlet during most periods with the exception of the February 22 storm event and March 23 baseline sampling. While both Lake Lois and RR Outlet were both considered "outlet" sites for Long Lake, the Lake Lois site was located within Long Lake whereas RR Outlet is truly the beginning of Woodland Creek. It is possible that nutrient concentrations increased as a result of water flowing through the wetland structure between these two sites. Primary production and decomposition occurring in the wetland could result in nutrients being carried downstream unless otherwise absorbed. The opposite trend was true for SRP concentrations as it was highest entering the Long Lake system from Pattison Inlet and decreased as it exited the lake. This may be due to its form and availability which is ready for uptake by organisms within the lake as well as the wetland itself.

Water leaving Long Lake forms the Woodland Creek tributary and eventually flows into Henderson Inlet. The residential influence on this waterway is significant as a Washington Department of Ecology study found the nitrate and nitrite concentrations in Woodland Creek to be some of the highest of any creeks that discharge to Henderson Inlet, and attributed the excess nutrients to stormwater discharge from a local community development (Woodland Creek Estates) (Hempleman, 2006). It is important to analyze for nitrites and nitrates in future Long Lake studies, to understand possible residential effects on N export into the lake.

5.6 Study Issues

The storm drains and lake sites were visited in both the wet and dry seasons; nonetheless, on the sampling date February 22 it was discovered that the Lorna Drain site was submerged as a result of increased precipitation and the basin's rising water level. As a result, the sample was collected from the stormwater catchment at street level.

There are several issues with extrapolating load allocation limits for TN as these limits were calculated for the specific ecosystem interactions of Budd Inlet not Henderson Inlet. Nonetheless, these hypothetical limits were used as a reference mark to possibly detrimental levels of incoming nutrients. It is recommended that future water samples be analyzed for nitrate and nitrite as they were lacking in this study. Such samples would aid water quality mangers in more conclusive efforts to identify and mitigate nitrogen inputs entering Long Lake. Finally, it is unfortunate that more data points were not collected over the duration of this study which would have aided in giving the study greater statistical power. Capturing more storm events from more storm drains would provide a bigger picture of the storm drain nutrient influence on Long Lake.

6. CONCLUSION & FUTURE WORK

6.1 CyanoHAB Assessment & Management

The increasing presence of freshwater HABs globally has provoked researchers to petition for greater comprehensive analysis, monitoring, and management of blooms (Anderson et al., 2002; Weirich & Miller, 2014). Brooks et al. and Hudnell suggest that a national directive is needed to adequately prioritize the mounting danger of freshwater HABs to water quality. National prioritization would make more funding available for the research necessary to understand cyanoHAB causes and consequences. Both articles encourage collaboration amongst the sciences as the environmental, toxicological, and medical effects of cyanoHABs on water quality requires a multifaceted scientific approach to assess and manage this risk more effectively. Prioritizing research on freshwater HABs equally to those in marine waters would sanction broader analytical and testing standards to be made available. Understanding both environments is necessary to determine guidance levels considered safe during acute, short term, sub-chronic, and chronic exposure periods to HABs.

There is still a lot to learn and understand about cyanoHABs, and a major point of argument from Brooks et al. (2016) is how limited our understanding is of these blooms are across spatiotemporal scales. Research published by the Environmental Protection Agency (EPA) suggests that it may be possible to produce guidance for the purpose of operating at multiple scales and curbing HAB incidences nationally (King et al., 2019). King et al. found in their study that lakes, wetlands, and streams showed similar responses to nutrient and biotic properties at a national scale. Their research suggests freshwater ecosystems may also respond similarly to future global changes, and while climate was not analyzed in this study, it is another variable of HAB growth that is predicted to have an overarching effect (Yindong et al., 2021; Zhang et al., 2016).

6.2 Long Lake Management

The effects cyanoHABs pose to the water quality experienced by Lacey residents are felt acutely. Brooks et al. call for national prescription, but local officials and lake management districts are limited by financial constraints. One type of treatment that has been utilized in the past and is the projected future treatment is the application of Aluminum sulfate to the lake surface. An Aluminum (alum) sulfate treatment removes phosphates through precipitation when added to a lake's surface. The bound particulates, known as a *floc*, become heavier in the water forcing the phosphates to settle on the lake bottom while also creating a barrier that retards sediment phosphorus release (North American Lake Management Society, 2004).

This type of treatment is effective at absorbing phosphates, but may only last between 5-15 years before the cyanobacteria spores present in the benthic sediment are released from their torpid state (Huser et al., 2016; North American Lake Management Society, 2004). A costly re-application every 10 or so years would be required for long-term maintenance. This approach may be sustainable for those with thousands of dollars on hand but is not the resolute answer to excess nutrients also. Thus, a multifaceted approach is needed. One that includes continued algae treatments, decreasing nutrient inputs and helpful behavioral changes by local residents. It is imperative that external loading of nutrients be reduced to make any impact on reducing internal loading within the lake.

Cyanobacterial harmful algal blooms have consistently presented Long Lake residents with interrupted use of their precious lake. As such, their lake management district has resolved to address this issue through their own water quality management program. Such a program includes aquatic vegetation control, aluminum sulfate treatments, and now stormwater monitoring, of which this study examined for the first time. The data collected this year will contribute to the creation of a more robust phosphorus model of this system. With the stormwater monitoring program goal in mind, the LLMD and Thurston County can look at the 2020-2021 water year to assess which sampling sites will be best suited for permanent monitoring locations.

The grab sampling technique is reproduceable by the residents themselves, so more frequent sampling can be done at little to no cost. Analysis of the samples will be a continuous cost, but the LLMD is willing and able to make this project a priority through sustained funding. Longer and more frequent rainstorms led to many of the stormwater outfalls becoming submerged under water. As such, it is likely that additional equipment will be installed to gather data when the outfalls become unusable.

The premise and drive behind the LLMD have been and will continue to be grounded in the community engagement that leads it. The Lake Management District contract is based on the community's commitment to fund raising and mutual management of this communal resource with Thurston County aquatic resource specialists. Additionally, LMDs can authorize private consultants and vendors to provide services to meet their needs through contract with Thurston County. The funding process includes petitions which are mailed to the LLMD community. If the potential action is agreed to, the action is authorized. Monthly meetings keep Long Lake residents informed of the progress on current projects and present relevant science as it is understood and uncovered by Long Lake's own scientific studies.

In conclusion, this research has highlighted the importance of storm drain transport of nutrients (especially SRP) and has also highlighted the low dissolved oxygen levels present in the lake during the winter. This information is critical towards developing an effective management plan to restoring Long Lake back to a period where the general public can once again recreate in it.

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Appendices