An Analysis of Phosphorus Loading and Trophic State in Fletchers Lake, Nova Scotia

by

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"It always seems impossible until it's done."

- Nelson Mandela

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ABSTRACT

Fletchers Lake, Fall River, Nova Scotia, is located within a diverse watershed, and is influenced by urban stormwater runoff, wastewater discharge and on-site septic systems. Due to the lake's multi-uses and more planned development within the watershed, the Halifax Regional Municipality has identified accelerated eutrophication as a critical issue. With limited historical data on Fletchers Lake trophic parameters, there is a need for baseline data to identify future changes. A mass balance model identified the largest phosphorus (P) sources: headwaters and on-site septic systems. Baseline P export from the lake's tributaries was measured, and P export coefficients were calculated for forested and low-density residential land use. Forested export coefficients varied between subwatersheds. The low-density residential coefficient was lower than an urban export coefficient widely applied in P prediction models. Trophic parameters, total phosphorus, nitrogen, chlorophyll *a*, measured as oligotrophic. Secchi depth measured as mesotrophic likely due to high humic contents.

LIST OF ABBREVIATIONS AND SYMBOLS USED

%	Percent
<	Less Than
σ	Sedimentation rate /yr
µg/L	Micrograms per litre
mg/L	Milligrams per litre
A	Land area m^2
c	Phosphorus export coefficient $mg/m^2/yr$
$CaCO_3$	Calcium Carbonate
CCME	Canadian Council of the Ministers of the Environment
Chl *a*	Chlorophyll *a*
C_{vw}	Volume-weighted concentration mass/L
CWRS	Centre for Water Resources Studies
d	Day
DO	Dissolved Oxygen
DOC	Dissolved Organic Carbon
EU	European Union
Fe	Iron
g	Gram
GIS	Geographic Information Systems
ha	Hectare
hr	Hour
HRM	Halifax Regional Municipality
kg	Kilogram
km	Kilometre
L	Annual P loading to lake mg
m^2	Metre Squared
mg	Milligram
mL	Millilitre
N	Nitrogen
NTU	Nephelometric Turbidity Units
^{o}C	Degrees Celsius
OECD	Organization for Economic Cooperation and Development
ORP	Oxidation-Reduction potential
P	Phosphorus
p	Hydraulic flushing rate /yr
q_s	Hydraulic Load
R_{sp}	Soil P retention coefficient
R_p	Theoretical sedimentation coefficient
SD	Secchi Depth
TN	Total Nitrogen
TOC	Total Organic Carbon
TP	Total Phosphorus mg/m^3
TSS	Total Suspended Solids
v	Apparent settling velocity m/yr
WFD	Water Framework Directive

WRT	Water retention time
WWTF	Wastewater treatment facility
yr	Year
z	Mean lake depth m

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CHAPTER 1 INTRODUCTION

1.1 Phosphorus and Eutrophication

Phosphorus (P) is most commonly the limiting nutrient in freshwater systems, and limits the growth of aquatic organisms such as algae. Anthropogenic activities have increased P loading into freshwater lakes and rivers leading to elevated P concentrations. Such increases in this limiting nutrient have resulted in accelerated algal growth and, in severe cases, algal blooms. Algal blooms are a concern as they can make freshwater un-usable for purposes such as a drinking water source, recreation, as well as agriculture (Codd et al., 1999; Codd et al., 2005). Cyanobacteria, or blue-green algae, are one type of algae that have produced major annual blooms in the well-documented case of Lake Erie (Burns et al., 2005; Watson et al., 2016). Microcystin is a toxin produced by blue-green algae that can have lethal implications for animals and serious health implications for humans through exposure by skin-contact or direct ingestion of contaminated water (Codd et al., 2005). Algal blooms also contribute to the accelerated rate of deposition of organic matter to a lake's sediment, leading to increased decomposition by microbes and a resulting depletion of dissolved oxygen (DO) causing fish kills as well as shifts in other freshwater species abundance and diversity (Watson, et al., 2016).

1.2 Motivation for Research

The focus on Fletchers Lake in Fall River, Nova Scotia, is prompted by extensive proposed residential and commercial development, designated as the "Fall River Growth Center", surrounding the lake within its watershed (Jacques-Whitford, 2009). Such development would cause a temporary increase in P export due to construction, and would also increase sediment loading into the lake and its supplying tributaries. Additionally, development would lead to increased long-term export of P into the lake by permanently altering land cover characteristics. Increased long-term P export could also result from additional wastewater associated with new development, depending on how the wastewater is managed. As Fletchers Lake is a drinking water source for private residences in the Fall River area, and for recreational activities such as boating, fishing and swimming, the negative effects of increased phosphorus loading could severely impact these water uses. With limited knowledge on the current trophic state of Fletchers Lake, and with limited historical data on trophic state parameters, the risk which development could pose is unclear. Jacques-Whitford (2009) have predicted increases in lake P concentrations and shifts in trophic status for various development scenarios with lakeshore capacity models. Therefore, P loading and accelerated eutrophication are identified by the Halifax Regional Municipality (HRM) as critical issues. It is important to identify the current baseline trophic conditions in the lake as well as current baseline P export from tributaries, so that the pre-development state of the lake can later be referenced for indication of P loading changes and trophic state changes.

1.2.1 Research Objectives

- Build a mass-balance of P in Fletchers Lake over the 2015 - 2016 year
- Measure baseline P export within the Fletchers Lake watershed and calculate P export coefficients for dominant land uses to compare with previously used literature values
- Characterize Fletchers Lake baseline water quality and corresponding trophic conditions

1.2.2 Research Questions

1. What is the largest source of P to Fletchers Lake? (Internal loading from sediment, WWTF discharge, individual tributaries, headwaters)

2. Are literature-derived P export coefficients an appropriate estimate for P export in the Fletchers Lake watershed?

3. What is the current trophic state of Fletchers Lake according to literature-identified trophic state indicators: phosphorus, nitrogen, chlorophyll *a*, secchi depth?

CHAPTER 2 LITERATURE REVIEW

2.1 General Background

2.1.1 Eutrophication

Eutrophication describes the increase in productivity of any water system, freshwater or marine. More specifically, it is the increase in primary productivity of freshwater systems as a result of increased nutrient inputs: nitrogen (N) and phosphorus (P). In the early 1960s eutrophication of freshwaters was becoming a concern in North America and in Europe (Kalff, 2002). Shortly afterward, relationships between lake P concentrations and phytoplankton biomass were being developed (Sakamoto, 1966), and later between P export from watersheds and algal biomass (Vollenweider, 1968). Some of the most well-known and impactful research, which proved P to be the major culprit for eutrophication, was that conducted by David W. Schindler on the Experiment Great Lakes Area in Ontario (Schindler et al., 1973). By sectioning off lakes to create closed-systems and adding combinations of nutrients to the different systems, Schindler et al., (1973) were able to prove P limitation by creating eutrophic conditions with cyanobacterial blooms resulting from P addition. Reference lakes with only Carbon (C), Nitrogen (N), and C and N together did not create eutrophic conditions, therefore disproving speculations of other nutrient being limiting to algal blooms. N limited lakes typically occur at low N:P molar ratios, from < 10 to < 20 (Sakamoto, 1966; Nurnberg, 1996; Guildford and Hecky, 2000).

Although eutrophication is a natural process, which lakes undergo over geological timescales, anthropogenic activities have accelerated this process by increasing fluxes of nutrients to freshwaters on much smaller timescales. This anthropogenic-caused eutrophication has been coined “cultural eutrophication”. Activities such as fertilizer application, mass-farming operations producing excessive amounts of manure, large-scale watershed development, and P-containing detergent use have all increased P loading to lakes and rivers historically and in the present. The largest consequence of eutrophication is excessive algal growth that can lead to algal blooms, which in certain cases, produce toxins such as microcystin. The specific type of algae which can produce microcystin is cyanobacteria, also called blue-green algae, and are responsible for the major annual blooms in the case of Lake Erie, as well as around the world (Codd et al., 2005; Watson et al., 2016). Aside from the potentially lethal effects of microcystin on wild life and livestock, algal blooms can make freshwater un-usable for purposes such as a drinking water source, recreation, as well as agriculture (Codd et al., 1999; Codd et al., 2005). Microcystin can also have serious health implications for humans through exposure by skin-contact or direct ingestion of contaminated water (Watson et al., 2016). The accelerated rate of deposition of organic matter to the lake's sediment, and resulting increased decomposition can cause anoxic water conditions which can lead to fish kills as well as shifts in other freshwater species abundance and diversity (Watson, et al., 2016).

2.1.2 Trophic State

As previously mentioned, trophic state refers to the primary productivity of a water system. Water bodies with low productivity and nutrient levels are classified at a lower trophic level than water bodies with high levels of algae and nutrients. Trophic levels start with oligotrophic as the lowest trophic state, meaning "poorly nourished", and increase to the highest trophic level, "eutrophic", meaning "well nourished" (Brylinsky, 2004). Historically, indirect measures for trophic state have included the following water quality parameters: P, total nitrogen (TN), chlorophyll *a* (Chl *a*), and secchi depth (SD) (Carlson, 1977; Vollenweider and Kerekes 1982; Nurnberg, 1996; Dodds, 1998). Furthermore, recent work by Webster et al., (2008), founded on previous studies showing strong relationships between colour (or Dissolved Organic Carbon (DOC) for which colour is a surrogate measure) and lake productivity (Carpenter et al., 1998; Nurnberg and Shaw, 1999), has supported the inclusion of DOC into the traditional trophic state evaluation. Lakes with high levels of humic matter, which makes up the majority of DOC and contributes to colour, are labelled dystrophic, and tend to deviate from the typical correlations between Chl *a*, TP and Secchi disk (Webster et al., 2008; Nurnberg and Shaw, 1999). Colour limits light penetration through the water column, and therefore directly impacts SD, a measure of light transparency. Through light limitation, colour can also limit algal growth, therefore directly influencing Chl a (Webster et al., 2008).

There has also been a shift toward the measurement of biological indicators to classify and monitor trophic state such as in The European Union (EU) Water Framework Directive (WFD) Common Implementation Strategy (2003; Carvalho et al., 2006). The EU WFD (2003) requires all countries in the EU to use biological indicators to monitor trophic state in freshwater systems (Carvalho et al., 2006). Indicators such as phytoplankton and macrophyte biomass are direct measures of trophic state and reflect water quality over longer timeframes than indirect water quality parameters, which can fluctuate over short time frames (Rakocevic-Nedovic & Hollert, 2005; Dudley et al., 2013). For these reasons, biological indicators are more accurate trophic parameters. An issue arising with the use of biological indicators is that many species are only found in certain geographic locations (Nurnberg, 1996). Thus, multiple biological indices would have to be developed for different geographic regions containing different species. Additionally, accurate taxonomical identification is costly and time-consuming (Carvalho et al., 2013).

Currently in Canada, and in most of North America, trophic state is mainly classified by Total Phosphorus (TP) concentrations. The Canadian Council of Ministers of the Environment (CCME) have adopted the Organization for Economic Co-operation and Development (OECD) P trigger ranges to guide provinces and municipalities on P lake management (Table 1). The trigger ranges originate from Vollenweider and Kerekes (1982), and serve as a guide for provinces and municipalities on the management of lakes and lakeshore development, depending on the intended uses for examined water bodies. As lakes across Canada naturally differ in trophic state depending on many factors such as age, geographic location, and surrounding geology and vegetation, there is no one standard trophic state or P concentration appropriate for all lakes. As a result, the trigger ranges require the knowledge of baseline conditions of the particular lake in question, to assess whether the lake has changed and requires further evaluation. This can be a limitation if no previous baseline measurements were taken. In terms of their application for lake management, the CCME (2004) advises "if the current concentration of phosphorus in a given lake is greater than the upper limit of its trigger range, or 50 % higher than its baseline, further assessment is recommended" depending on its intended uses.

Table 1 CCME (2004) P trigger ranges for Canadian lakes and rivers

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

2.1.3 Physical Factors Affecting Trophic State

Although phosphorus (P) has become the most widely used indicator for trophic state, recent literature is showing that other factors can significantly affect the water quality and biological productivity in lakes, and in some cases, more so than P levels.

Climate change is slowly increasing global temperatures and resultantly ocean and freshwater temperatures are also rising (Levitus et al., 2000; Schindler et al., 1996; McCormick and Fahnensteil, 1999; Burns et al., 2005). Rigosi et al (2015) have shown through modelling lake temperature increases (by 0.08 deg C from 24 deg C), risk of harmful cyanobacteria blooms can increase by 5 %; the same increase in bloom risk was found for a P increase from 10 μg/L to 20 μg/L. The Rigosi et al (2015) model used to estimate cyanobacteria bloom risk showed a much greater sensitivity to temperature changes (20.3 %) than to P changes (0.12 %), indicating that rising temperatures may be a more important factor influencing lake trophic state than P. Furthermore, as temperature directly influenced cyanobacterial bloom risk, it could be one of the more important factors affecting recreational use of water bodies, due to the associated health hazard of cyanobacterial toxins (Rigosi et al., 2015).

Water retention time (WRT), also defined as a lake's flushing rate, can be a strong determining factor of nutrient levels, and has shown negative correlations with P and colour (Webster et al., 2008). Higher flushing rates shorten the time that P is available to be assimilated by algae, and the time that algae have to establish communities, therefore, shorter WRTs are also negatively correlated with algal blooms (Londe et al., 2015). However, P and Chl *a* still show strong positive correlation over vast ranges of WRT and many studies report that nutrient enrichment to flowing rivers and streams can still result in a measurable increase in benthic algae concentrations (Hoyer and Jones, 1983; Smith, 1999). This emphasizes that a short WRT cannot entirely prevent algal blooms from occurring, although it is certainly an important factor that can affect trophic state.

Hydromodification, or the alteration of land use, water ways, and water withdrawal, affects natural water pathways by changing stream and river courses, water flows, and water drainage from and into land (Mohamoud et al., 2009). Fluxes of nutrients and sediments into waterways are also affected through the process of development and permanent alterations to land permeability. Aside from increased nutrient concentrations in water bodies affected by hydromodification, which can directly lead to changes in trophic state, elevated levels of suspended sediments and higher water turbidity may also result. Increased turbidity can be perceived as an aesthetic nuisance from a recreational water-use perspective. Reduced land permeability and removal of natural vegetation also limits soil and vegetative filtration of fecal coliforms from runoff, leading to elevated levels of pathogenic bacteria in water bodies (Mallin et al., 2001). Through increased nutrient, sediment and coliform runoff, hydromodification can directly affect trophic state, and in particular, the recreational use of water due to elevated bacteria levels.

Another factor that has shown to influence trophic state is water colour, also measured by the surrogate parameter dissolved organic carbon (DOC). The main contributor of color, and the main component of DOC, is humic matter, which absorbs light and limits its penetration into the water column, thus having a negative relationship with Secchi depth, the measure of water transparency (Webster et al., 2008). Humic matter is also a carbon source for heterotrophic organisms and can affect lake metabolism and levels of other nutrients. Studies have shown that colour can directly influence algal growth through its strong correlations with chlorophyll *a* (Webster et al., 2008). By its absorption of light, high colour can potentially limit algal and macrophyte growth. Contrastingly, colour has been shown to have a positive relationship with chlorophyll *a* in some studies such as Webster et al. (2008) and Nurnberg and Shaw (1999). It is hypothesized that this could be a result of higher numbers of motile algae, and higher concentrations of chlorophyll *a* produced by individual algal cells, as well as the increased carbon food source from humic matter (Webster et al., 2008; Nurnberg and Shaw, 1999). Due to its strong effects on chlorophyll *a* and secchi depth, two widely used trophic state indicators, as well as on overall dynamics like lake metabolism, colour is an important parameter to consider when evaluating lake trophic state in coloured lakes.

2.2 P Biogeochemical Cycle

As summarized by K.C. Ruttenberg in Holland & Turekian (2004), the global phosphorus (P) cycle is composed of four major fluxes: uplift of bedrock containing P-rich minerals, physical and chemical weathering of this exposed rock, riverine transport of P in the form of dissolved and particulate P, and the sedimentation of particulate P within minerals and organic compounds to the bottom of the ocean. The uplift of bedrock occurs on geological timescales (on the order of tens of millions to billions of years) whereas the rest of the P cycle fluxes occur on a much shorter timescale (Pierrou, 1976; Filippelli, 2008). This also means that sediments and bedrock are the largest reservoir for P and have the longest residence time for storing P (Pierrou, 1976).

The largest terrestrial source of P is the weathering of exposed bedrock (Pierrou, 1976; Lajtha & Schlesinger, 1988). Physical weathering produces particulate phosphorus bound to minerals, mainly apatite as previously mentioned, which can erode directly into rivers or become part of the soil P reservoir, the second largest P reservoir (defined as soil < 60 cm deep) (Syers et al. 1967; Lajtha & Schlesinger, 1988). Chemical weathering produces dissolved phosphorus, which can directly flow into the river system or enter the soil reservoir to be taken up by land biota (Maybeck, 1982; Pierrou, 1976). The land biota reservoir P can then eventually make its way into the river system as detritus, or it can internally cycle between the soil and land biota reservoirs as organisms die, decompose and the resulting nutrients get taken up again by new biota (Schlesinger, 2013).

The two main fluxes that contribute P to aquatic reservoirs are deposition of atmospheric phosphorus and the riverine transport of P (Pierrou, 1976; Holland & Turekian, 2004). Rivers transport both dissolved and particulate P to lakes and oceans; however, it is estimated that 89 % of riverine P is in the organic particulate form (Maybeck, 1982).

The third largest P reservoir is the ocean reservoir, which is broken down into two compartments: the surface ocean, spanning 0-300 m below the surface, and the deep ocean, spanning below 300 m (Schlesinger, 2013; Holland & Turekian, 2004). The residence time for P in the surface ocean is short, lasting around 2.5 – 4 years, while the residence time in the deep ocean is estimated at around 1500 years (Holland & Turekian, 2004). Within the total ocean reservoir, an internal P cycling occurs between dissolved P and the P within biota, which take up the dissolved form (Pierrou, 1976). A sedimentation flux occurs between the deep ocean and the sediment reservoir, by which particulate P within detritus, and P bound within minerals such as hydroxyapatite, settles to the bottom of the ocean (Pierrou, 1976; Filippelli, 2008). The cycle closes when millions to billions of years later, the sediment P re-emerges as bedrock uplifts above the ground or ocean surface

(Pierrou, 1976). P associated with sediments in the ocean can become re-suspended or re-dissolved within the water column, which is a minor flux (Pierrou, 1976).

Smaller components of the phosphorus cycle include the atmospheric reservoir and atmospheric transport of P in the form of particulate in dust and in the form of phosphine gas (PH_3) (Maybeck, 1982; Glindemann et al., 2005). Phosphine gas is present in minute quantities, and requires unique conditions to form, therefore, P does not have a significant gaseous phase in the global P cycle (Glindemann et al., 2005).

Human activities that have increased natural P fluxes include mining, clear-cutting, sewage discharge (industrial, agricultural, domestic), and fertilizer application (Pierrou, 1976; Holland & Turekian, 2004). Impervious surfaces, which dominate urban landscapes, can contribute to P flux to freshwater environments due to the accumulation of anthropogenically-sourced P-containing matter between rainfall events, examples of which include construction sediment, lawn fertilizer, solid waste, as well as animal fecal matter (Waller, 1977). In this way, anthropogenic activities either directly or eventually increase the flux of P first into river systems, lakes and oceans. Additionally, since the flux of weathering is slow and cannot replenish soil P at the same rate that it is being eroded and depleted through agricultural activities and deforestation, there is concern for the complete depletion of soil and P worldwide (Elser and Bennett, 2011; Sverdrup, 2014).

2.2.1 P Geochemistry in Bedrock, Water and Soil

The main P reservoirs that participate in the above-mentioned fluxes include bedrock, soils, biota, and freshwater and marine P. There are two main minerals that contain significant amounts of P in bedrock: carbonate fluorapatite (CFA) in sedimentary bedrock, and fluorapatite in igneous bedrock (Holland & Turekian, 2004; Fillipelli, 2008). In soils P is present in organic and inorganic forms. Inorganic forms include apatite and various ferric, aluminum and manganese oxides and oxyhydroxides, as well as orthophosphate dissolved in pores within the soil (Syers et al., 1967). Organic soil P includes that within organic matter and P sorbed to organic molecules (Crews et al., 1995). Within biota, P is used for building genetic material such as DNA, within cell membranes as the phospholipid bilayer, and functionally in energy transfer reactions with ATP (Pierrou, 1976).

In freshwater and marine environments P is present in two forms: dissolved and particulate P. Dissolved P includes orthophosphate (PO_4^{3-}), as well as inorganic and organic-P that is dissolved. Particulate P refers to any organic and inorganic P that is present in particulate form (Maybeck, 1982). Once in the water column, dissolved orthophosphate is typically quickly assimilated by algae and converted to organic-P, which eventually settles to the sediments as autochthonous organic-P when the algae die (Bostrom et al., 1988). It can also react with inorganic and organic compounds such as Fe oxyhydroxides, clays and carbonates, in which case it precipitates, then settles to the sediments along with allochthonous particulate P (Bostrom et al., 1988; Sondergaard et al., 2003).

Once settled at the sediments, many different chemical and physical factors influence the stability and release of P from the sediments back into the water column. Such factors include microbial activity, redox conditions, and pH (Lijklema, 1980; Sondergaard et al., 2003). Decay of organic matter by microbial decomposition can also directly release phosphate and organic-P from the sediments (Bostrom et al., 1988; Marsden, 1989). Since temperature can increase microbial activity, it may indirectly increase P release via this pathway (Bostrom et al., 1988; Kalff, 2002). Microbial respiration at the sediment-water interface can also indirectly increase P release by creating an anaerobic environment, conducive to lower redox conditions (Marsden, 1989). Such conditions facilitate the reduction of ferric iron (Fe^{3+}) to dissolved ferrous iron (Fe^{2+}), effectively releasing P bound to iron oxyhydroxides (Bostrom et al., 1988; Sondergaard et al., 2003). Contrastingly, a higher redox potential at the sediment layer is associated with aerobic conditions and prevents P release from the sediments (Bostrom et al., 1988; Marsden, 1989). This facilitates the adsorption or precipitation of P with ferric iron as well as the incorporation of detrital P into microbial biomass (Lijklema, 1980; Bostrom et al., 1988). Elevated pH due to higher primary productivity can be associated with replacement of sediment phosphate with OH in ferric oxyhydroxides, thereby releasing phosphate back into the water column (Lijklema, 1980; Bostrom et al., 1988). Higher pH in lakes that have abundant $CaCO_3$ can cause the opposite effect, by inducing the precipitation of $CaCO_3$, with which PO_4^{3-} can co-precipitate, and by inducing precipitation of hydroxyapatite, onto which P can adsorb, thereby removing it from the water column (Bostrom et al., 1988; Marsden, 1989). Additional physical factors that influence phosphorus release into the water column include wind turbulence and bioturbation caused by macroinvertebrate activity (Lijklema, 1980; Marsden, 1989; Sondergaard et al., 2003; Cyr et al., 2009).

Within soils, P is mainly adsorbed to other particles, but also exists in smaller quantities as dissolved P within soil pore water (Holland & Turekian, 2004). Dissolved P within soils can adsorb to clays, oxides and oxyhydroxides (Frossard et al., 1995). Factors affecting P sorption in soils are similar to those affecting P sorption to sediments in marine and freshwater environments. Chien et al. (1982) found that increasing temperatures increased sorption in two types of acidic soils. As previously discussed on P dynamics in water environments, lower redox state and higher pH is similarly associated with more desorbed P, or P that is dissolved within the soil (Roy & Datta, 1985).

2.3 P Export Coefficients

Phosphorus export coefficients represent the mass of P exported from a unit area of land over time, and are typically expressed in units of $mg/m^2/y$ and kg/ha/yr. These export coefficients are assigned to different land use types such as forested, urban or agricultural land, as different land use types can export different amounts of P (Reckhow, 1980**)**. Export coefficients can also vary by climate, as different levels of annual precipitation and seasons can affect the dynamics of soil and water pathways (Esbroeck et al., 2016). Geology can also affect P export as different types of bedrock contain different amounts of phosphorus-containing minerals, produce different kinds of soil and can be variably prone to weathering (Syers et al. 1967; Dillon and Rigler, 1975; Crews et al., 1995). It has been found that P export coefficients widely vary in the literature, even among similar land use types, which can greatly affect the accuracy of predicted P loading, and the accuracy of models which use these export coefficients. In order to examine the literature for the applicability to this project, only export coefficients reported for North America were reviewed to maintain similarities between climate, geology and land-use history.

One of the most widely used resources for P export coefficients is Reckhow (1980). This paper compiles an extensive summary of P export coefficients measured primarily in North America (some studies from Europe and South Africa were also summarized) for various land use types. Reckhow's (1980) residential export coefficients were measured from watersheds that also contained secondary land use types such as agricultural and industrial, which exported much higher amounts of P compared with the watersheds listed as "low-density". Additionally, many watersheds listed do not specify whether they are low, medium, or high-density residential development, contributing to the wide range of P export listed for this land type. The forested watersheds summarized by Reckhow (1980) also exhibited a wide range in P export due to the stark difference in export between hardwood (1 – 90 mg/m^2/yr) and softwood (36 – 830 mg/m^2/yr). This is likely a result of increased soil acidity from coniferous tree needles, which increases P leaching from soils (Tamm & Hallbäcken, 1988; SanClements et al., 2010). It should also be noted that the studies summarized ranged in geographic location, resulting in different climates and geology between measured watersheds, which can lead to different levels of P export (Dillon and Rigler, 1975; Esbroeck et al., 2016). Dillon and Rigler (1975) reported that P export for watersheds off the Pre-Cambrian shield (mainly composed of igneous plutonic bedrock) can be double that for watersheds on the shield. Atmospheric P deposition can differ regionally, as well as P loading from regional differences in overland flow, which can contribute to the wide ranges in P export documented for the various land use types (Reckhow, 1980; Paterson et al., 2006). A further limitation is that the coefficients do not account for P uptake in streams or by riparian vegetation, which can vary by location, but has been shown to be significant at certain times of year (Mulholland et al., 1997). Lastly, sampling methods tend to vary from study to study, with some studies focusing on rain and snow melt events (Winter and Duthie, 2000) and others following weekly or bi-weekly schedules (Scott et al., 2000; Molot and Dillon, 1993; Dillon and Kirchner, 1975). Such variations in sampling design can introduce further variability in P export measurements.

Table 2 shows a comparison of the P export coefficient ranges reported in Reckhow (1980) with literature-review ranges compiled by AECOM (2013), and values used in locally produced P modelling reports, AECOM (2013) and Scott and Hart (2004). Both reports utilized P export coefficients for estimating P export in a modified version of the Dillon-Rigler (1975) P prediction model to asses the P concentrations in 21 lakes within the Bedford West Planning Area in HRM. It is observed that all of the P export coefficients used in both reports fall into ranges reported in the literature. However, some ranges for P export coefficients in the literature are wide with large variability such as those for forested, industrial, commercial, residential and roadway land uses (AECOM, 2013; Reckhow, 1980). It should also be noted, as is common for many Nova Scotia (NS) studies utilizing P prediction models, that many P export coefficients are taken from literature values measured in Ontario (Waller and Hart, 1985; HESL and MOE, 2011), as there is a lack of measured values for this province. Both local P modelling studies shown in Table 2 used industrial, institutional and light commercial P export coefficients from Waller and Hart (1985), and high and low-density residential, as well as open space coefficients from HESL and MOE (2011), both of which conducted their measurements in Ontario. The high variability in literature export values as well as the extensive use of export coefficients measured in Ontario creates uncertainty as to whether such values would be appropriate for NS. A study by Scott et al. (2000) on forested P export in NS found that aside from igneous soils, for which phosphorus export was about 45 percent higher, P export in NS is about "20-50 percent lower than that reported in similar bedrock-land use categories in Ontario and the United States." This suggests that non-locally-measured P export coefficients can significantly decrease the accuracy of predicting local P loading.

Export coefficients measured in NS are most abundant for the forested land use category, with many P modelling studies using values measured by Scott et al., (2000) for forested land on igneous and sedimentary bedrock, with additional smaller land categories such as “>15 % cleared”. The land use descriptions for these export coefficients are specific for geology and additional smaller land use components within the measured subwatersheds such as wetland and cleared forest percentages, which helps improve the accuracy in their application to varieties of forested land. There is, however, further room for improvement with estimating forested P export, in terms of characterizing dominant tree species. As previously mentioned, export coefficients summarized in Reckhow (1980) have shown a wide range between hardwood forests (1 – 90 $mg/m^2/yr$) and softwood forests (36 – 830 $mg/m^2/yr$). The use of P export coefficients specific to forest type, or dominant tree species, would therefore improve accuracy of predicting P export from forested land in modelling studies.

Lastly, there is a lack of studies conducted in NS regarding P export from developed land uses, which are most commonly used in P prediction models to estimate resulting, post-development changes, in lake P concentrations. Examples include industrial, commercial and different levels of residential development, which identifies a literature gap for P export from developed land uses in the province.

Table Comparison of P export coefficient literature ranges to export coefficients used in AECOM (2013) and Scott and Hart (2004) reports. Values are in $mg/m^2/y$.

Land Use	Literature Ranges		AECOM (2013)		Scott and Hart (2004)	
	AECOM (2013)	Reckhow (1980)	Land Use	Coefficient	Land Use	Coefficient
Atmospheric Deposition	16.7-25		Water	17.3[a]	Precipitation	17.3[a]
Forest	2-20	1-830	Forest	6.9[b]	Forest	6.9[b]
			Forest-meadow	8.3[b]	Forest + >15 % cleared	8.3[b]
Wetland	16-24.7		Wetland	8.3[b]		
Industrial	149.1-534.7	75-417	Industrial	202[c]	Industrial	202[c]
Institutional	42		Institutional	42[c]	Institutional	42[c]
Commercial	40-398	66-485	Commercial	202[c]	Commercial (light)	40[c]
			Commercial and	167[d]		
Residential	0.5-220.8	19-270	Residential: *High density*	132[e]	Urban (residential)	52[f]
			Medium density	52[f]		
			Low density	13[e]		
			Open space	13[e]		
Quarry	0.4-10.8		Quarry	8[e]		
Roadway	83-350		Roadway	202[g]		

a) Underwood (1984)
b) Scott et al., (2000)
c) Waller and Hart (1985)
d) An average of commercial and high-density residential land export coefficients AECOM (2013)
e) HESL and MOE (2011)
f) Waller (1977)
g) Scott and Hart (2004)

2.4 P Prediction Models

The use of mass balance models for lake systems has helped identify and quantify mechanisms governing P cycling, like sedimentation and internal loading, as well as identify dominant sources of P to a lake system. One of the earliest and most popular mass balance models was developed by Vollenweider (1969) and is shown is steady-state form in Equation 2.4 below (Jones and Bachman, 1976).

$$TP = \frac{L}{z(\sigma + p)} \qquad [2.4]$$

TP = lake P concentration in mg/m^3
L = annual P loading to lake in mg/m^2
z = mean lake depth
σ = sedimentation rate /yr
p = hydraulic flushing rate /yr

The model assumes:

- Rate of P loading, the flushing rate, and the sedimentation rate are constant
- The lake is well-mixed (it is a Continuously Stirred Tank Reactor or CSTR)
- Outflow concentration of P is representative of the P concentration within the lake
- The sedimentation of P is proportional to the in-lake P concentration
- The lake is a conservative system (Dillon 1974; Jones and Bachman 1976)

The Vollenweider (1969) model was further developed by Dillon and Rigler (1975) to predict the concentration of P in a lake with the use of P export coefficients for different land use types. The P export coefficients are applied to areas of different land use types within the lake's watershed, to estimate the variations of P export from each land category. The total load of P to the lake is a sum of the P contributed from land use types within the watershed, atmospheric deposition, on-site septic systems within 300 m of waterways, wastewater treatment facilities (WWTF), and in-flow from headwaters if applicable (Scott et al., 2003). Many studies have applied the simple mass balance model to compare loading sources (Winter, 2002) and to estimate P concentrations in lakes at risk for, or already experiencing algal blooms (Nurnberg, 2013; Zhang et al., 2013; Lathrop et al., 1998).

Limitations of the simple mass balance model include:

- Many lakes are stratified in the summer, and are therefore not well-mixed
- The model assumes constant P inputs; however, with seasonal changes in flows, the inputs and outputs to a lake system are not entirely constant throughout the year
- The model runs on an annual time-step, so it cannot capture seasonal or monthly changes in P that can result in short-term algal bloom responses (Zhang et al., 2013).

The Dillon and Rigler (1975) model has undergone many modifications since its original version (Paterson et al., 2006), and the most recent version has been published for use in NS by Brylinsky (2004) for Nova Scotia Environment. This version of the model has been applied in many studies within HRM in order to manage lakeshore development and lake trophic conditions in a development context (Scott and Hart, 2004; Jacques-Whitford, 2009; AECOM 2013). Lakes within the Paper Mill Lake watershed (Scott and Hart, 2004; AECOM, 2013) and the Shubenacadie Grande Lake watershed are some of those that have been assessed (Jacques-Whitford, 2009).

Additional limitations to the Brylinsky (2004) model are listed below, in addition to those summarized earlier for simple mass balance models:

- The model assumes all inputs contribute equally to annual mean P concentration, regardless of where they are located along the lake (for example, a source located closer to the outlet is modelled to contribute the same amount of P as a source located near the inlet).
- All septic tanks within 300 m of the lake, and within 300 m of waterways flowing to the lake, contribute some fraction of their P load (a soil P retention

factor is applied) to the lake system. The 300 m distance was arbitrarily chosen (Brylinsky, 2004; Scott et al., 2003).

- All sources of P contribute to the lake system within the annual time-step, although the time for certain sources to make their way to the lake system could be on a much longer time-scale, such as on the order of decades (Scott et al., 2003)
- The model is not applicable to shallow lakes (Brylinsky, 2004; Scott et al., 2003).
- The model may not be applicable to lakes that have dystrophic qualities such as higher color or humic matter (Brylinsky, 2004; Scott et al., 2003).
- The model was calibrated on lakes in Ontario, on the Canadian shield and therefore may not be applicable to lakes in other geographical regions due to differing climate and geology (Paterson et al., 2006).

2.5 Assessment of Lakes in Nova Scotia

Lakes in NS are characterized by their high levels of organic acids, or humic matter, low pH levels, as well as low P concentrations and low productivity, putting most lakes on the oligotrophic-dystrophic spectrum (Clair et al., 2007; Freedman et al., 1989; Underwood, 1986; Webster et al., 2008). As a result of both natural acidity and atmospheric sulphate deposition, NS's surface waters have some of the lowest reported surface-water pH in North America (Ginn et al., 2007; Underwood 1986). Additional natural contributors to higher lake acidity include sea salt spray and discharge from peatlands, which also contribute high levels of DOC (Underwood, 1986).

2.6 Research Gaps

Currently, the CCME (2004) P trigger ranges are the main guideline used in NS with regard to lake management. As NS lakes are characterized by high levels of DOC (Underwood, 1986), which tend to deviate from trends seen with traditional trophic indicators (Webster et al., 2008), it may be more appropriate for lake management guidelines used in NS to incorporate trophic measures such as DOC, or

Chl *a* into their assessments. More research on this matter would identify what parameters would be most appropriate and cost-effective for assessing the province's unique lake characteristics.

There also exists a literature gap for P export from developed landscapes such as industrial, commercial and residential land in NS, therefore, export coefficients are taken from studies conducted in Ontario and the United States for these land use types (Scott and Hart 2004; AECOM, 2013). However, this creates an uncertainty surrounding the P export estimates, as a different climate and geology can result in different levels of export. As was reported by Scott et al. (2000), P export in NS can differ by 20-50 % from export measured in Ontario and the United States, suggesting the use of non-locally-measured P export coefficients in P prediction models can decrease the accuracy of model estimates. In addition, there is a lack of research on differing P export from different dominant tree species. Large differences in P export were reported in Reckhow (1980) for softwood forests versus hardwood forests, suggesting that more detailed export coefficients for particular tree-species would greatly improve estimates of P loading.

Lastly, there is a lack of studies validating the use of the Dillon and Rigler (1975) model in NS. The model has undergone many modifications since its original version; however, most calibration studies were conducted in Ontario (Paterson et al., 2006). There is a research gap with regard to the model's applicability in NS, especially with the abundance of dystrophic lakes in the province, which have been shown to deviate from traditional P and Chl *a* trends (Webster et al., 2008)

CHAPTER 3 METHODOLOGY

3.1 Study site: Fletchers Lake, Fall River

The study site for this project, Fletchers Lake, is located in Fall River, NS (Lat. 44.82, Long. -63.61; Figure 3.1). Fall River is situated on the mainland portion of the province, within Halifax County, located on the mid-Western coast of Nova Scotia. The milder climate in mainland Nova Scotia, compared with the rest of Canada, is largely a result of the province's location on the Atlantic Ocean (Mudroch, 1987). In Fall River, the daily average temperature in June (1981-2010) is recorded as 15.1 °C with an average precipitation of 96.2 mm. In January the daily average temperature is 5.9 °C with an average precipitation of 134.3 mm (Environment Canada, 2011). Average annual precipitation for the 1981-2010 period is recorded as 1396.2 mm (Environment Canada, 2011).

In 2006, a population of about 1887 was recorded in Fall River (Statistics Canada, 2012). The urban land uses within the town comprise primarily of low-density residential land, and of some commercial and institutional land. Around 300 Ha within and around the Fall River area are forested and designated as potential developable lands (Jacques-Whitford, 2009). Fletchers Lake serves as a drinking water source to approximately 83 homes and businesses in the Fall River area and also acts as the receiving water to two wastewater treatment facilities (WWTFs): Lockview-MacPherson WWTF and Wellington WWTF (Halifax Water, 2017).

Figure 3.1 The province of Nova Scotia (top right), Fletchers Lake in relation to Halifax peninsula (main frame).

Figure 3.2 Satellite image (2015; 0.6 m, MDA Corporation) of Fall River and Fletchers Lake watershed delineated in black.

3.1.1 Physiographic Characteristics

The Fletchers Lake watershed spans an area of 16.9 km^2, and is underlain by two types of bedrock: granite, and the Halifax Formation consisting of slate, silt, and sandstone (Keppie, 2000). The watershed comprises mainly of the Wolfville soil series on the North, East and South portions, which is characterized as sandy clay loam, and rockland on the North-West portion, where >60 % of the land is exposed bedrock (MacDougall et al., 1963). Tree species are a mix of hardwood and softwood with dominant hardwood species being Sugar Maple, Yellow Birch, Beech, Oak and Red Maple, and with Black spruce, Red Spruce and Balsam Fir as the dominant hardwood species (Government of Nova Scotia, 2015a).

3.1.2 Hydrologic Characteristics

Fletchers Lake has an area of 1.01 km^2 and a length of about 4.2 km from inlet to outlet. The Fletchers Lake watershed is delineated in Figure 3.2. The lake has a residence time of 10.41 days (flushing rate of 35.06 times/year) and an average depth of 3.72 m (Hart et al., 1978). The two deepest points in the lake are found at the two basins: the inlet basin is 6 m deep and the outlet basin is 11 m deep (Figure 3.2). Historically, Fletchers Lake has classified under the oligotrophic category (CCME, 2004) with a phosphorus concentration between 6 μg/L and 9.3 μg/L (Mudroch et al., 1987; Hart et al., 1978; Jacques-Whitford, 2009; Stantec, 2012). As a dimictic lake, Fletchers Lake experiences two yearly turnovers, the first occurring in early spring, and the second in early fall. The lake undergoes stratification in mid summer (June - August).

Fletchers Lake is a component of the headwaters to the Shubenacadie River watershed, the largest watershed in NS (Mudroch et al., 1987). The main chain of lakes flowing into Fletchers Lake includes (from the start of direction of flow): Lake Charles, Lake William, and Lake Thomas (Figure 3.3). Fletchers Lake discharges into Shubenacadie Grand Lake, which then discharges into Shubenacadie River, and finally Cobequid Bay.

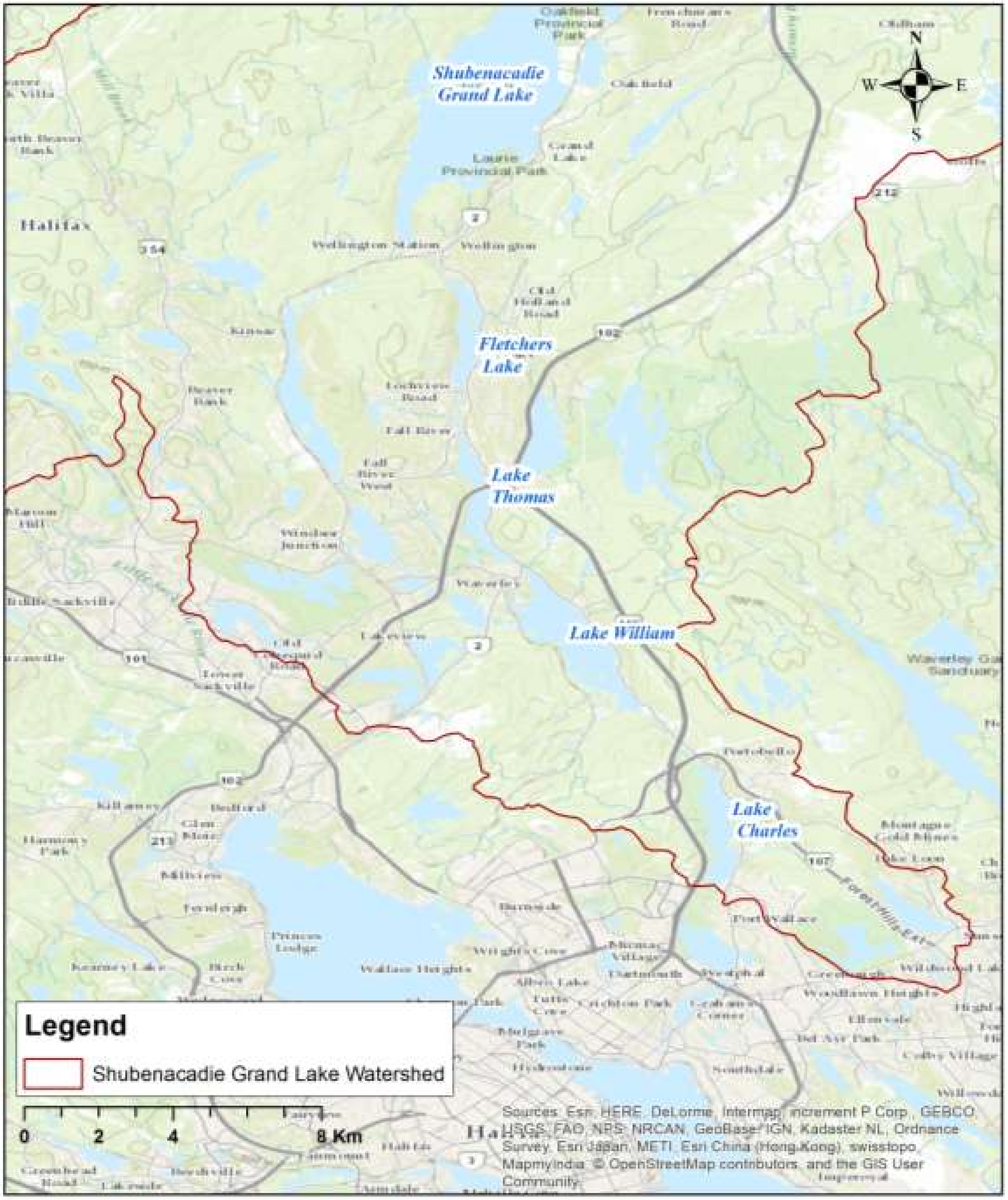


Figure 3.3 Map of main lakes within Shubenacadie Grand lake watershed. Lakes comprising Fletchers Lake headwaters are labelled: Lake Charles, Lake William, and Lake Thomas.

3.2 Data Collection

3.2.1 Introduction

A full-year monitoring program was conducted from June 2015 until August 2016 to collect water quality data and flow measurements from the main tributaries supplying the lake as well as the outlet and inlet. In-lake sampling was conducted in the summer period to measure trophic state parameters as well as to obtain oxygen and temperature profiles.

3.2.2 Stream Flow Measurement and Water Sampling

Bi-weekly flow measurements were obtained with a Model 625A Pygmy Meter (Gurley Precision Instruments, Troy, NY, USA) attached to a Model 1100 Digital Flow Indicator read-out and 1.5 m depth rod in all tributaries (Figure 3.4), as well as the inlet and outlet. Tributary sampling locations were chosen to be consistent with previous Halifax Water sampling locations. Cross sections of each site were chosen with a uniform width and depth and any woody debris or large rocks were cleared from the cross section before measurements were taken. Points of measurement along the cross-section were chosen so that the total number of measurements was 10 or more, and the distance between measurements ranged between 15-100 cm depending on the total width of the cross section. Depth was measured at each flow measurement point and the velocity-area Six-Tenths-Depth method was used (Dingman, 2002). Water samples were also taken from the streams for water quality testing and a YSI sonde Model 600 (Yellow Springs, OH, USA) was used to obtain dissolved oxygen (DO), pH, conductivity, and temperature measurements. Samples were collected in new polyethylene bottles, which were rinsed with distilled water and/or sampling water 3 times prior to sample collection. Wastewater effluent samples were collected from the Lockview-MacPherson WWTF when access was available from December 2015 – May 2016.

Two Onset HOBO pressure loggers (Onset Computer Corporation, Bourne, MA, USA) were installed to collect continuous water level data. One was installed in a 0.3 m PVC pipe fastened with two bungee chords to a mount embedded in the bottom of Holland Brook (FBB in Figure 3.4), and another was installed in a 1 m tall wooden box fastened to a bridge at the lake inlet (FI in Figure 3.4). The continuous hourly pressure data was used to calculate continuous flows, based on a pressure-discharge relationship developed for each location.

In June of 2016, the bridge at the lake inlet underwent construction to be replaced. Before construction began, the pressure logger was moved further downstream, away from the construction site, and the water pressure just before the move was recorded, as well as the water pressure at the new location after the move. The pressures post-move were adjusted by the difference in water pressure between the two locations.

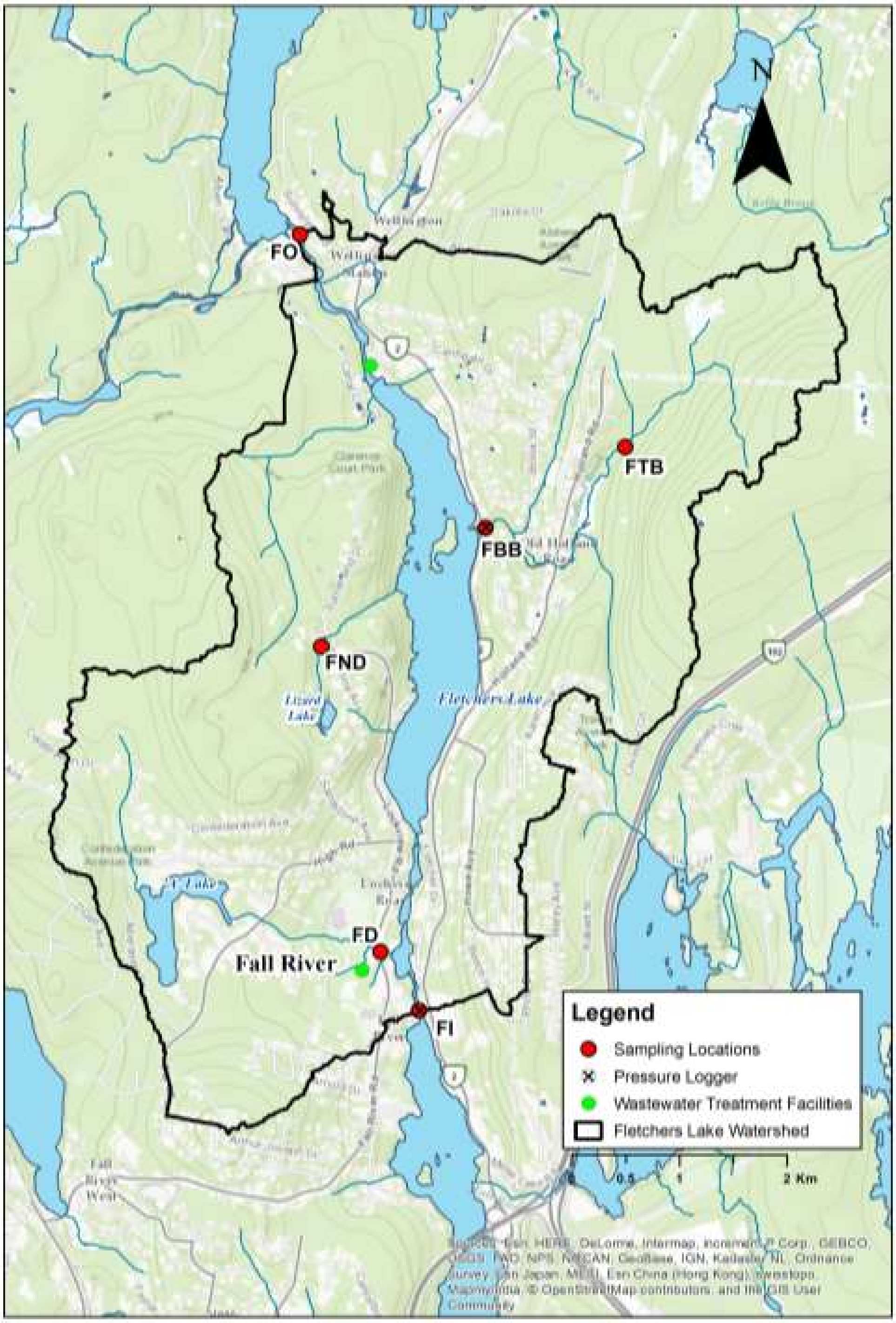


Figure 3.4 Map of the stream, inlet and outlet sampling sites (red) as well as the WWTFs (Lockview-MacPherson WWTF near FD, and Wellington WWTF near FO) in green. Pressure logger locations are marked with a red X. 1. FO – Fletchers Lake outlet 2. FTB- top of Holland Brook 3. FBB- bottom of Holland Brook 4. FI – Fletchers Lake Inlet 5. FD –Developed subwatershed 6. FND – New Development subwatershed stream running from Lizard Lake.

3.2.3 Calculation of Total and Average Daily Flows

Pressure data was collected at 30-minute intervals over the course of a year (July 2015 to August 2016) with HOBO data loggers installed in the bottom of Holland Brook (FBB) and the lake inlet (FI). The pressures were then corrected for atmospheric pressure with pressure data from the Government of Canada weather station at the Halifax Stanfield International Airport ("Halifax Intl A" weather station, Lat. 44.88, Long. -63.51), simply by subtracting the atmospheric pressure from the measured water pressure for each day. After the correction they were regressed against measured flows, in m^3/s, at FBB and FI (Figure 3.5). This regression was used to calculate hourly flows (flows in m^3/s multiplied by 3600) for the entire year at those locations, from the hourly measured HOBO logger pressures. The hourly flows (m^3/hr) were totalled for each day, and these total daily flows were then used to calculate daily P export in the mass balance calculation in Sections 3.4.1. The hourly flows, in m^3/s, were also averaged per day to obtain average daily flows in m^3/s to produce hydrographs.

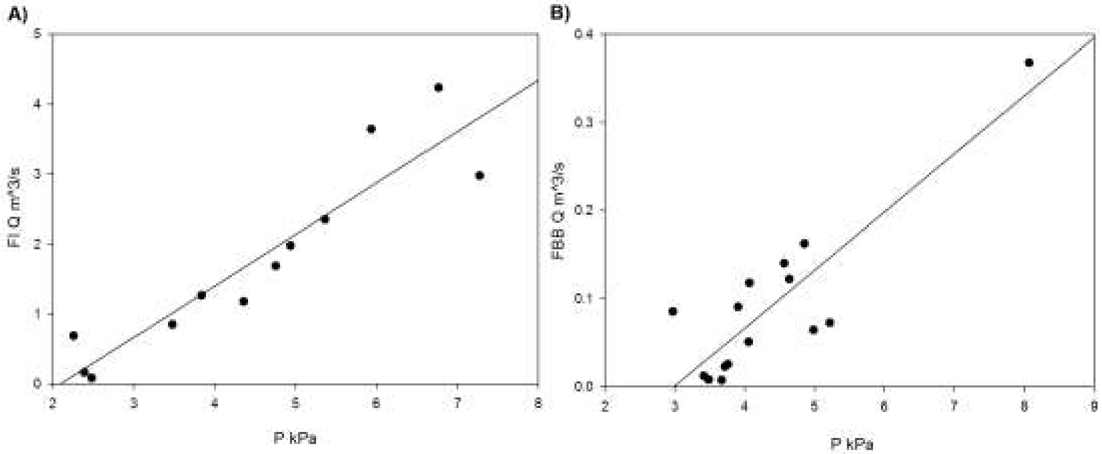


Figure 3.5 A) Q vs P FI regression equation: $y=0.7383x-1.5584$, $R^2 = 0.8602$, B) Q vs P FBB regression equation: $y = 0.0659x - 0.1971$, $R^2 = 0.75618$

To calculate total daily flows for the non-data logger locations, the totalled daily flows for FI (m^3/day) were regressed against measured flows for FO (measured in m^3/s and converted to m^3/d). This regression equation was then used to calculate total daily flows at FO from the totalled daily flows at FI. The same procedure was followed for calculating total daily flows for FTB, FD, and FND from the totalled daily flows at FBB. All regressions used to calculate total daily flows at locations with no data loggers are shown below in Figure 3.6. The total daily flows at FTB, FD, and FND were then used to calculate daily P export from these locations for use in the mass balance calculation and for calculating export coefficients.

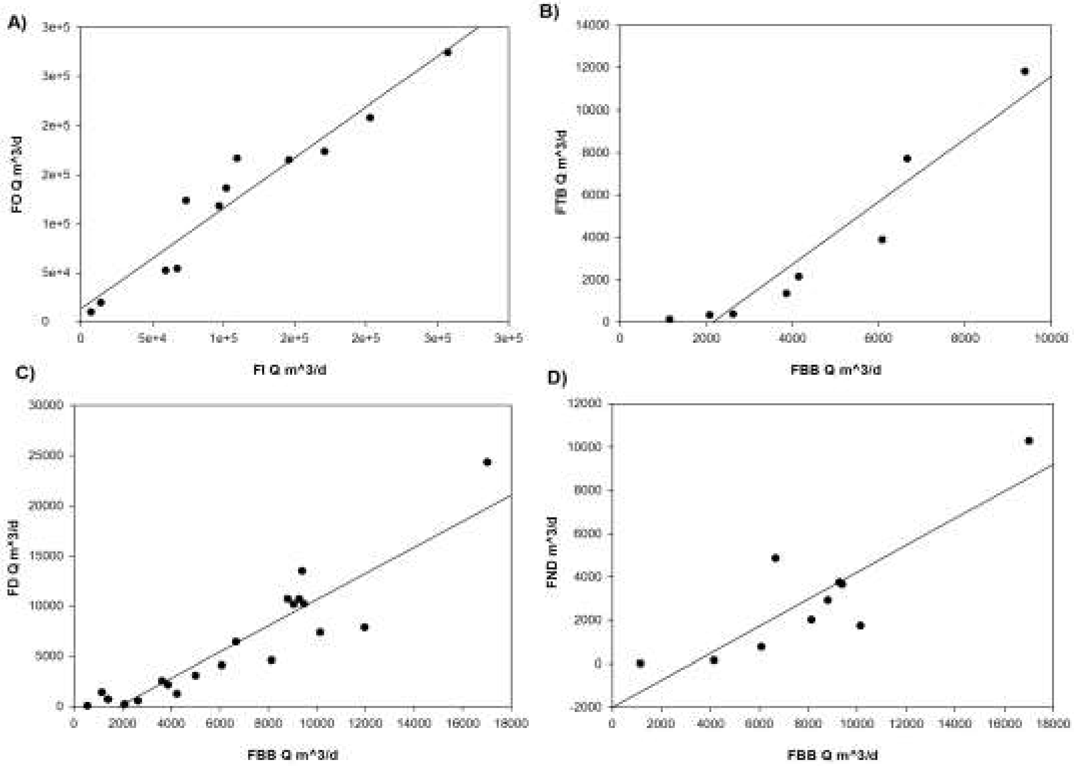


Figure 3.6 Regressions used to calculate total daily flows in m^3/d. A) FO vs FI (m^3/d) regression equation: $y = 1.0287x + 13222$, $R^2 = 0.92604$, B) FTB vs FBB (m^3/d) regression equation: $y = 1.4756x - 3161.8$, $R^2 = 0.91236$, C) FD vs FBB (m^3/d) regression equation: $y = 1.3019x - 2387.1$, $R^2 = 0.84116$, D) FND vs FBB (m^3/d) regression equation: $y = 0.6226x - 2007.9$, $R^2 = 0.74296$.

To calculate average daily flows (in m^3/s) in the non-data logger locations, the measured flows at FI (in m^3/s), were regressed against measured flows at FO (m^3/s), and the measured flows at FBB (m^3/s) were regressed against measured flows at FTB, FD, and FND. These regressions were then used to calculate average daily flows (in m³/s) for non-data logger locations from the average daily flows at FI and FBB (m³/s), calculated from the data logger data. The hourly flows, in m³/s, were then averaged over each day to obtain average daily flows in m³/s for non-data logger locations, in order to compile the hydrographs in Section 4.4 All regressions used to calculate average daily flows in m³/s are shown below in Figure 3.7.

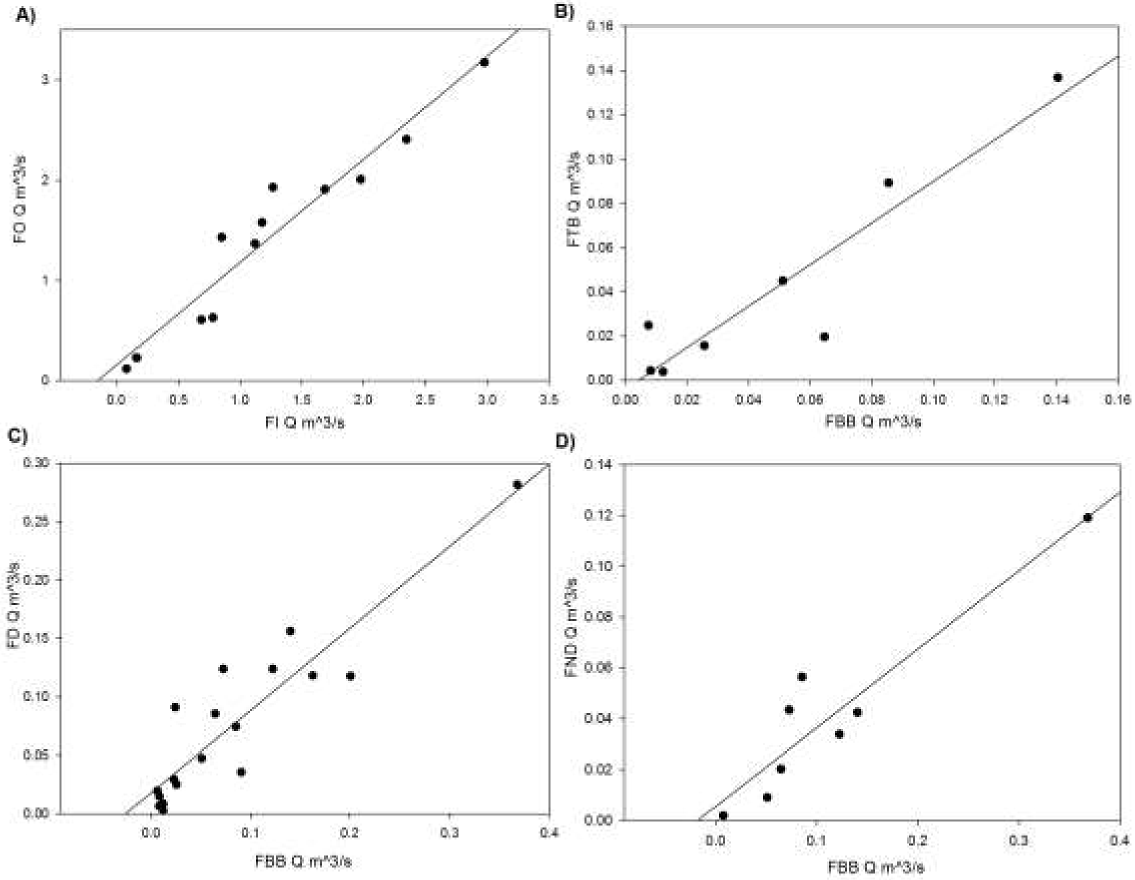


Figure 3.7 Regressions used to calculate average daily flows in m^3/s. FO Q vs FI Q regression equation: y = 1.0287x + 0.153, R² = 0.92604, FTB Q vs FBB Q regression equation: y = 0.9413x - 0.0041, R² = 0.86321, FD Q vs FBB Q regression equation: y = 0.7039x + 0.0178, R² = 0.83612, FND vs FBB regression equation: y = 0.3093x + 0.0055, R² = 0.87376

3.2.4 Lake Sampling

In-lake water sampling was conducted once in August of 2015, and 3 times in 2016; after spring turnover (May), during late-summer stratification (September), and after fall turnover (November). Water samples were obtained for water quality analysis, and Secchi depth, as well as *in-situ* YSI sonde Model 600 (with 15 m cable; Yellow Springs, OH, USA) measurements were taken at each of the two basins, shown below in Figure 3.8. The lake sampling locations were chosen to be consistent with historical sampling locations by Hart et al., (1978) and by the HRM Lakes Water Quality Monitoring Program (Stantec, 2012). The YSI sonde was used to measure DO, pH, cond., T, and ORP throughout the water column. Water samples and sonde measurements were taken every 1-3 meters depending on the depth of each basin, and it was ensured that the epilimnion, metalimnion and hypolimnion were sampled according to the temperature and DO profiles. A 2.2 L PVC Kemmerer was used to obtain water samples and a depth sounder was used to locate the basins.

Sources: Esri, HERE, DeLorme, Intermap, increment P Corp., GEBCO, USGS, FAO, NPS, NRCAN, GeoBase, IGN, Kadaster NL, Ordnance Survey, Esri Japan, METI, Esri China (Hong Kong), swisstopo, Mapmyindia, © OpenStreetMap contributors, and the GIS User Community

Figure 3.8 Map of Fletchers Lake Watershed and lake sampling locations, F Stn 1 (6 m deep) and F Stn 2 (11 m deep). Lake Inlet (South end of lake) and Outlet (North end of lake) are labelled.

3.2.5 Water Quality Analysis

Water samples were stored in a cooler from the time of sampling until arrival at the laboratory where they were stored in a 4 °C fridge until analysis. All tests were performed within holding times indicated in Standard Methods of Water and Wastewater (APHA, 1998).

Stream water samples and wastewater effluent samples from the Lockview-Macpherson WWTF were analyzed for trophic state indicators: total phosphorus (TP), dissolved phosphorus (DP), soluble reactive phosphorus (SRP), total organic carbon (TOC) and dissolved organic carbon (DOC), and total nitrogen (TN). Standard quality parameters were also measured: turbidity and total suspended solids (TSS). Lake samples were analyzed for the same parameters, excluding TSS, and also for color and chlorophyll *a*.

Phosphorus samples were measured with the ascorbic acid method by Murphy and Riley (1962) on a 100 mm pathlength cell with a detection limit of 1 µg/L P. Total and dissolved phosphorus samples were digested first according to Menzel and Corwin, (1965), and soluble reactive and dissolved phosphorus samples were filtered through a 0.45 µm membrane before processing. Turbidity was measured according to Standard Method 2130 B on a HACH Turbidimeter 2100AN (HACH, Loveland, CO, USA), TSS was measured according to Standard Method 2540 D, and true color (filtered through a 0.45 µm cellulose membrane filter was measured according to Standard Method 2120 C on a DR 5000 spectrophotometer using the in-built color program at a 455 nm wavelength (APHA, 1998). The TOC, DOC and TN samples were analyzed on the TOC-V_{CPH} Total Organic Carbon Analyzer by Shimadzu (Shimadzu, Boston, MA, USA). DOC was filtered through a 0.45 µm polyethersulfone (PES) filter prior to analysis. The detection limit for TOC and DOC was 0.250 mg/L and for TN was 0.120 mg/L. Chlorophyll *a* was measured using a fluorometric acidification method by Yentsch & Menzel (1963), modified by Holm-Hansen et al. (1965) and recommended by Strickland & Parsons (1968) with a detection limit of 0.05 µg/L. Chlorophyll *a* samples were immediately filtered (within 6 hours of samples being obtained) through Whatman GF/C filters (1-3 drops of a 1 % $MgCO_3$ solution was placed on the filters before filtering). The filters were folded and frozen in aluminum foil-sealed petri dishes until they were analyzed.

3.3 Data Analysis

3.3.1 Mass Balance Calculation

Total daily P export (in µg/day) was calculated by multiplying total daily flows (m^3/day) by measured P concentrations (µg/L) from the same day in sampled tributaries, the lake outlet and inlet. The flows were a set of continuous data, as the data loggers installed at FBB and the lake inlet (FI) provided continuous water pressure data, which was used through a series of regressions (Section 3.3.3) to obtain total daily flows at all locations. The measured P concentrations were a set of non-continuous data, as water samples were only taken every two weeks. As such, linear interpolation was used to calculate phosphorus concentrations on days that the streams, inlet and outlet were not sampled. The linear interpolation equation is shown below (Equation 3.1). For each non-sampled day, the TP (y) was calculated using the last previous measured TP concentration (y1), the next measured TP concentration (y1), the number of days between the last measured concentration and the non-sampled day (x – xo), and the difference in days between the two measured concentrations (x1 – xo).

$$y = yo + (x - xo)\frac{y1 - yo}{x1 - xo} \qquad [3.4.1]$$

The total daily export amounts (mg P) were summed over the year for each sampling location to obtain yearly export (mg P/yr). Phosphorus mass loading from wastewater treatment plants was also included in the mass balance using discharge volumes, Lockview-MacPherson WWTF- 455 m^3/d, and Wellington WWTF- 68 m^3/d (Halifax Water, 2017) and effluent phosphorus concentrations (P concentrations were measured in effluent samples from the Lockview-MacPherson WWTF, and values reported in compliance summaries as quarterly averages for the Halifax Regional Water Commission (HWRC) Board were used for the Wellington WWTF. The inputs to the system included the export measured from FI, FTB, FBB, FD FND, land from the rest of the watershed including septic tanks, WWTFs, and atmospheric deposition onto the lake area. The output from the system was the export from FO, the lake outlet. Mass of P lost to sedimentation to the lakebed was calculated as the difference between total inputs and measured output through the outlet. The largest source of P to Fletchers Lake was identified from the P mass balance calculation.

3.3.2 P Export: Rest of Watershed

To calculate P export outside of the measured subwatersheds to drainage points at sampling locations in Figure 3.4 (sampling points on streams of order 1), the land use categories in the rest of the watershed were characterized by interpreting an orthorectified satellite image (0.6 m, MDA Corporation) acquired by GeoEye-1 on July 14, 2015 projected in UTM Zone 20 and coordinated in North American Datum 1983 (NAD83), combined with Government of Nova Scotia (2015a) Forestry Data (Table 3). Land use types were categorized in ArcGIS according to NS Forestry Data “Fornon” codes into more general land classes specified in the first column of Table 3. The new land classes were given unique codes, from 0 for cleared land through to 8 for hardwood, and they were grouped together using the Dissolve tool. Areas of each land class were then calculated in the Attribute Table.

Table 2 Forestry Data land classes used to categorize land use types within the watershed, and NS Forestry Data land classes grouped in each chosen general land class

Land use class	**Forestry Data Land Classes** (examples not coded in forestry data are shown in brackets)
Cleared	dead stand, partially cleared stand, wind throw, rock barren, powerline corridor, rail corridor, brush, sugar bush,barren, clear cut, gravel pit
Low density residential	road corridors, urban
Institutional *	urban (school, church, fire station, WWTF)
Commercial *	urban (Stores, bakeries, grocery store, drugstore)
Wetland	wetland, beaver flowage, treed bogs, lake wetland, open bogs
Inland water	Inland water (lake, pond)
Mixedwood	74-26 % softwood by basal area
Softwood	75 % softwood by basal area
Hardwood	<25 % softwood species by basal area

* identified by examining forestry data over-laid on aerial image, examples in brackets

The P export from land use types outside of the subwatersheds that were measured, was then calculated using the measured export coefficient measured in this study for forested land and literature coefficients for the other land classes listed in Table 2. Equation 3.2 was used to calculate P export (L), in mg, from different land use types using areas of those land types, A in m^2 and P export coefficients, c in mg/m^2/yr, for i individual land use types (Soranno, 1996).

$$L = \sum_{i=1}^{m} c_i A_i \qquad [3.2]$$

The P export from septic tanks (outside the boundaries of the measured subwatersheds) was calculated and included as part of the total P export from the rest of the watershed. Residences with septic tanks were identified using the satellite image (0.6 m, MDA Corporation) and property lines data from the Dalhousie GIS Center. The buffering tool was used on HRM waterlines data (GeoNOVA, 2016) in ArcGIS, to identify 300 m buffer zones. The property data was then clipped with the 300 m buffer layer to display properties within the buffer zone. This new property layer was overlaid on the 2015 aerial image to identify properties with residences. The total number of properties with residences (703) were then enumerated using the Attribute Table and the number of residences serviced by the Wellington and Lockview-MacPherson WWTFs (82) (Blair Titus, personal communication, February 10, 2017) was deducted from this total to obtain the number of residences with septic tanks that are within 300 m of waterways.

The loading from septic tanks was estimated using average persons per household, 2.4 (total Halifax population by private household; Statistics Canada, 2012), a P production of 0.8 kg P/capita/yr (Dillon et al., 1986) and a soil P retention coefficient (R_{sp}). Two different R_{sp}s of 0.5 (Hart et al., 1978; Jacques-Whitford, 2009) and 0.8 (Sinclair et al., 2014) were applied to the septic tank loading and compared. 0.5 has previously been used in P models in HRM (Jacques-Whitford, 2009; Scott and Hart, 2004) and 0.8 is a measured R_{sp} for systems under 30 years old. The resulting proportion of P settled to the lakebed (difference between inputs and output) for each soil retention coefficient was compared to a theoretical lake P sedimentation coefficient (R_p) calculated with the Kirchner and Dillon (1975) formula shown below (3.3). The apparent settling velocity (v) for an oxic hypolimnion, 12.4, was used, and the areal hydraulic load (q_s) was calculated as shown in equation 3.4.

$$R_p = \frac{v}{v - q_s} \qquad [3.3]$$

$$q_s = \frac{hydraulic\ load\ (\mathrm{m}^3/\mathrm{yr})}{lake\ surface\ area\ (m^2)} \qquad [3.4]$$

Hydraulic load was calculated by summing annual hydraulic load measured from tributaries and the inlet, precipitation onto the lake minus evaporation (0.552 m; Jacques-Whitford, 2009), as well the rest of the watershed. Hydraulic load from the rest of the watershed was estimated by calculating runoff coefficients for developed and non-developed land using measured flows from a developed subwatershed (FD) and from a non-developed subwatershed (FTB). These runoff coefficients were then applied to the runoff (precipitation x area) from the developed and non-developed land in the rest of the watershed to account for infiltration and evapotranspiration.

Additionally, the P loading from septic tanks within the FBB subwatershed was estimated (2.4 person per household; Census 2011; 0.8 kg/capita/yr; Scott and Hart, 2004) with the two different soil retention coefficients (0.5 and 0.8) and the P export value was compared with the actual measured export from this subwatershed. The soil retention coefficient giving the P export closest to the measured export at FBB, and also giving the closest in-lake P sedimentation to the theoretical sedimentation, was chosen for use in the mass balance calculation.

3.3.3 P Export Coefficients

Phosphorus export coefficients for the drainage sub-basins supplying the measured tributaries were calculated using the P concentrations and flow data measured in those tributaries. Phosphorus export coefficients (mg P/m^2/yr) were calculated for three main land–use types that were prominent in each subwatershed: deciduous forested, coniferous forested, and low density residential. The yearly P mass exported from each tributary was calculated as explained in Section 3.4.1. This yearly export in mg was divided by the contributing subwatershed area (found in the Attribute table for each subwatershed polygon in ArcGIS) to obtain export coefficients in mg/m^2/yr.

Subwatersheds and the overall lake watershed were delineated using the Arc Hydro toolset in ArcGIS. A LIDAR Digital Elevation Model (DEM) (2 x 2 m resolution) covering the watershed was obtained from the Dalhousie GIS Centre (Dalhousie University, Halifax, Nova Scotia, Canada). A sequence of terrain pre-processing steps were completed on the DEM layer using Arc Hydro (Maidment, 2002) to generate the watershed delineations. This included DEM reconditioning and filling of sinks, defining flow direction, flow accumulation, and streams within the DEM grid, stream segmentation, catchment grid and polygon processing, drainage line processing, adjoint catchment processing and drainage point processing.

The Point Delineation feature was used to create the watershed. Sampling points for the streams (FD, FND, FBB and FTB in Figure 3.4) were selected as pour points for the subwatersheds, and the lake outlet sampling location (FO) was selected as the pour point to create the lake's watershed. The land use types within the subwatersheds were characterized in finer detail through satellite image (0.6 m, MDA Corporation) interpretation as described in Section 3.4.2, in order to calculate percentages of other land types aside from the primary land types. The percentages of different land use classes, summarized in Table 3, were calculated for each subwatershed. Polygons were drawn around these land use types, and areas were found in each polygon's Attribute Table.

3.3.4 Precipitation Data

Daily precipitation data for the year was obtained from the nearest Government of Canada weather station at the Halifax Stanfield International Airport ("Halifax Intl A" weather station, Lat. 44.88, Long. -63.51) located about 7 km away from Fletchers Lake.

3.3.5 Volume-Weighted In-Lake Parameters

All parameters (TP, DP, SRP, Chl *a*, TN TOC, color) measured within the lake basins, Fstn-1 and Fstn-2 in Figure 3.8, were calculated as volume-weighted concentrations using lake volume at different depth sections based on a bathymetric map of the lake. The calculation was completed as per Equation 3.5 shown below. The total concentration for each basin (C_{vw}) was calculated as a sum of the weighted concentrations measured at each sampling depth ($c_1 \dots c_i$). Weights multiplied by each concentration were ratios of the volume at the sampling depth ($v_1 \dots v_i$) over the total volume in the sampled basin (v_t).

$$C_{vw} = \sum c_1 \frac{v_1}{v_t} + c_2 \frac{v_2}{v_t} + c_3 \frac{v_3}{v_t} \dots \quad c_i \frac{v_i}{v_t} \qquad [3.5]$$

Table 4 shown below displays volumes calculated for depth ranges using panimetry. A planimeter and bathymetric map (Government of Nova Scotia, 2015b) were used to calculate areas at each contour level, which were then multiplied by depth ranges to calculate volumes.

Table 3 Volumes of water at different sectional depths of the lake.

F Stn 1 – Inlet Basin		**F Stn 2 – Outlet Basin**	
Depth (m)	**Volume** (x 10^4 m^3)	**Depth** (m)	**Volume** (x 10^4 m^3)
0		0	
	58.268		28.284
1		1	
	45.277		18.002
2		2	
	35.875		10.986
3		3	
	27.091		4.753
4		4	
	12.205		2.710
5		6	
	2.712		1.607
6		8	
	2.233		1.129
8		10	
	1.453		0.402
10		11	
	0.952		0.022
12		11.2	
	0.022		
12.2			

A two-sample, independent t-test, assuming un-equal variances was performed on lake TP measurements from this study (averages for each sampling date of surface basin samples and outlet samples) and a TP data set from the HRM Lakes Water Quality Monitoring Program from 2007 – 2011 (Stantec, 2012) Both data sets passed a normality test at the 0.05 level of significance using Mini-tab 16 ® (Minitab Inc., State College, PA, U.S.A.) normality test after an outlier of 30 µg/L was removed from the HRM dataset.

3.3.6 Quality Assurance and Quality Control

To ensure that all laboratory measurements were representative and did not deviate due to confounding factors, triplicates on all laboratory samples taken on the same day were analyzed for turbidity, TOC and TN once over the course of the year. The same was done for sampling, by obtaining three different samples from each location on one day, to obtain confidence intervals for sampling consistency using turbidity, TOC, TP, DP, SRP. Triplicate means and 95 % confidence intervals for parameter concentrations, as well as for triplicate samples are shown in Appendix A. Due to logistics of limited time and available equipment, TSS was analyzed in singular samples. TP was analyzed in duplicate. Sampling triplicates and method triplicates for TSS were not measured due to logistical constraints of collecting and transporting multiple 1 L volumes of water sample for each site. Sampling triplicates and method triplicates were also not measured for lake water quality parameters, again due to logistical constraints of the limited time and space for multiple sample bottles while lake sampling.

CHAPTER 4 RESULTS AND DISCUSSION

4.1 Introduction

The following section provides the delineated subwatersheds for stream sampling sites and the export coefficient measurements as well as the land use breakdown of each subwatershed. A subsequent discussion comparing the measured P export coefficients to previously used literature values is included.

The following mass balance calculation section provides an overview of input and output values to the lake, calculated mass of P lost to sedimentation, and a breakdown of the proportions of all inputs. A summary with the results from an analysis used to select appropriate P septic system retention coefficients is also included. Section 4.4 Stream Water Quality presents the water quality parameters measured in the streams and inlet and outlet over the course of the year in charts with corresponding hydrographs and precipitation events, providing a discussion of observed seasonal trends. Lastly, Section 4.5 In-lake conditions and trophic state provides summaries of measured trophic indices, dissolved oxygen (DO) and temperature profiles, and resulting trophic state of the lake, compared with historically measured TP values for Fletchers Lake.

4.2 P Export Coefficients

The subwatersheds delineated for all stream sampling sites (FD, FND, FTB, FBB) are shown in Figure 4.1. Phosphorus export coefficients calculated for each of the delineated subwatersheds are summarized in Table 5, and land use type percentages for each subwatershed are also shown.

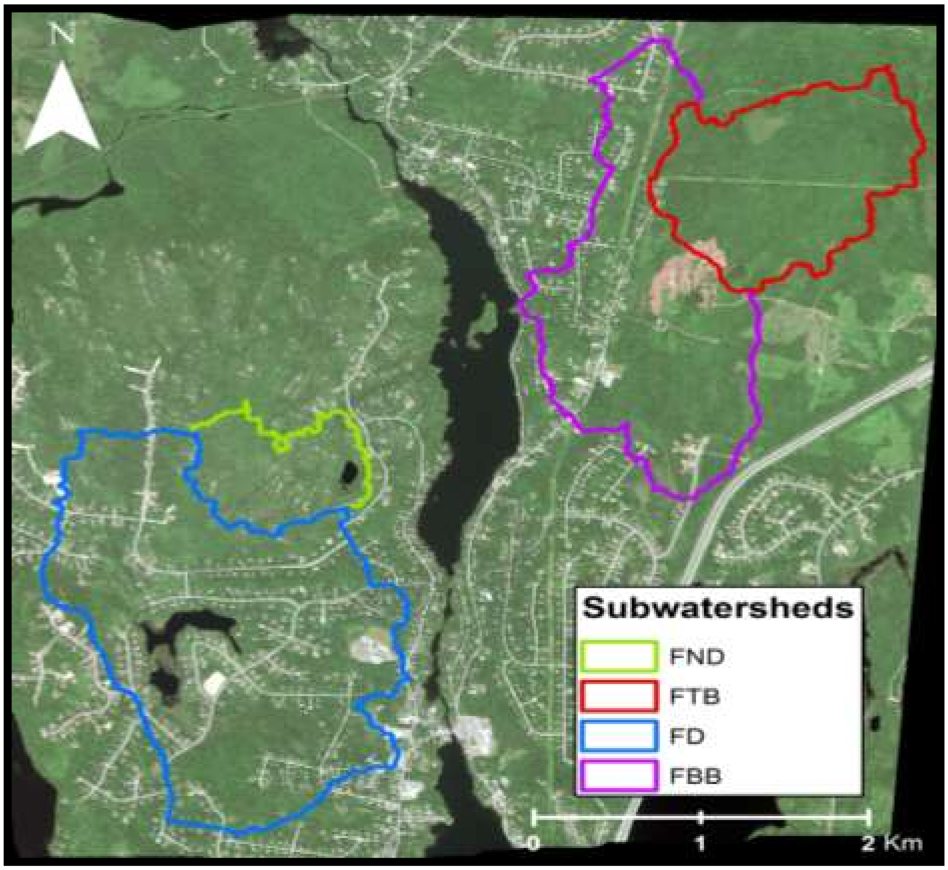


Figure 4.1 Subwatersheds labelled by corresponding sampling location labels. 2015 satellite image (0.6 m, MDA Corporation).

Table 4 Phosphorus export coefficients calculated for subwatersheds and land class breakdown in percentages for each subwatershed. The forest land class is further classified into softwood, mixedwood and hardwood. Dominant land classes are shown with *; where forest was the dominant land class, all tree types are shown with *.

Subwatersheds	FND	FD	FTB	FBB
P export coefficients ($mg/m^2/yr$)	**22.2**	**9.1**	**4.1**	**4.9**
Land class	**Land class percentages**			
Cleared	16	4	9	12
Low-density residential	3	58*	2	37
Institutional	0	3	0	1
Wetland	7	2	0	2
Inland water	2	3	0	0
Softwood	63*	17	30*	27*
Mixedwood	9	11	10*	15*
Hardwood	0	2	50*	5*

The highest measured P export originated from FND at 22.2 $mg/m^2/yr$, followed by FD (9.1 $mg/m^2/yr$), FBB (4.9 $mg/m^2/yr$) and finally FTB (4.1 $mg/m^2/yr$). The dominant land class in FND was forested land (72 %), particularly softwood, which covered 63 % of the total area. 16 % of the land was classified as cleared, which mainly consisted of exposed bedrock, and 7 % was classified as wetland. Low-density residential land was the main land class in FD followed by 30 % forested land consisting of all three tree types. FTB was classified as 90 % forested, made up of predominantly hardwood, which covered 50 % of the total area. FBB was 47 % forested, with a large low-density residential component that covered 37 % of the total area.

The measured P export coefficients are compared with a widely cited P export coefficient review study, Reckhow (1980), and with literature values used in consulting reports in Table 6 below. Each measured export coefficient is shown with the dominant land type of its subwatershed, and the percentage of the second-most defining feature, or second-most dominant land type was also noted: FD- residential, FND – forested with 7 % wetland, FBB- forested with 37 % residential, FTB- forested. The low-density residential export coefficient measured at FD is compared with residential land use P export coefficients. The forested P export coefficients measured at FND, FTB and FBB are compared with forested land-use P export coefficients.

The following two coefficient sources in Table 6 comprise of consulting reports, which utilized the P prediction model on various lakes within HRM: Scott and Hart (2004) analyzed 21 water bodies flowing into Paper Mill Lake, and Jacques-Whitford (2009) analyzed Fletchers Lake and other lakes within the Shubenacadie Grand Lake watershed. Both consulting reports used 52 mg/m^2/yr for residential land use measured by Waller (1977) in a high-density urbanized watershed on the Halifax peninsula, 6.9 mg/m^2/yr for forested land (Scott et al., 2000), and 8.3 mg/m^2/yr for forested land with a >15 % cleared component (Scott et al., 2000).

The measured 9.1 mg/m^2/yr export coefficient for residential land is outside of the range summarized in Reckhow (1980) and much lower than the 52 mg/m^2/yr measured by Waller (1977). This difference in P export is likely due to the difference in densities of development between the measured subwatershed in this study and the watershed measured by Waller (1977). However, this difference identifies the variability in export between different levels of development and supports the argument against using one P export value for all levels of residential development.

The measured forest export coefficients of 4.1 and 4.9 mg/m^2/yr fall within the range reported in Reckhow (1980), and are slightly lower than the values used in the consulting reports and measured by Scott et al. (2000). The forest export coefficient of 22.2 mg/m^2/yr also falls within the range in Reckhow (1980), and is more than double that of the forest P export coefficients in the consulting reports. This is hypothesized as a result of P loading contributions from both the dominant forest species, and the wetland component. The forest is predominantly softwood, which can export higher levels of P due to increased soil acidity from coniferous tree needles, which increases P leaching from soils (Tamm & Hallbäcken, 1988; SanClements et al., 2010). Wetlands can significantly contribute to P export, particularly if the wetland contains peat, as Scott et al. (2000) has previously observed and reported, and as Paterson et al., (2006) found with a regression between peat percent cover and mean annual TP export.

It should be noted that the total precipitation for the sampled year (July 2015 – July 2016) was 1569.6 mm, which is 173.4 mm higher than the average annual precipitation for the 1981-2010 period recorded as 1396.2 mm at the Halifax Stanfield International Airport weather station (Environment Canada, 2011). As such, it is possible that the sampled year may have exhibited slightly higher P export than normal due to the higher precipitation. However, the measured P export coefficients were at the low end of the ranges reported in the literature (Reckhow, 1980), and they were also lower than previously measured P export coefficients in NS (Waller, 1977; Scott et al., 2000), aside from the 22.2 mg/m^2/yr measured from the forested subwatershed with a wetland component. Thus, it is assumed that the P export measured in this study is reasonable and representative of average annual export, despite the higher than average precipitation for the sampling year.

Table 5 Comparison of measured phosphorus export coefficients (mg/m^2/y P) to literature values.

Subwatersheds by land use	P Export (mg/m^2/yr)	Reckhow (1980)	Scott & Hart (2004)	Jacques-Whitford (2009)
Residential	9.1	19-270	52	
Forested, 7% wetland	22.2	1-830	6.9	
Forested	4.1			
Forested, 37% residential	4.9		8.3*	

* 8.3 for forest + >15% cleared (Scott et al., 2000)

4.3 Mass Balance

A mass balance summary of measured and estimated inputs (tributaries, inlet, atmospheric deposition, WWTF, land and septic systems from rest of watershed) and their percentages of total input, outputs (outlet), and resulting difference (sedimentation) is shown in Table 7. A visual diagram of inputs and outputs from the system, as summarized by quantified results is shown in Figure 4.2. The masses in the mass balance table are calculated as totals over the sampling period, from July 2015 – July 2016 (measured P export from July 2015 – August 2015 was multiplied by the factor of 365/404).

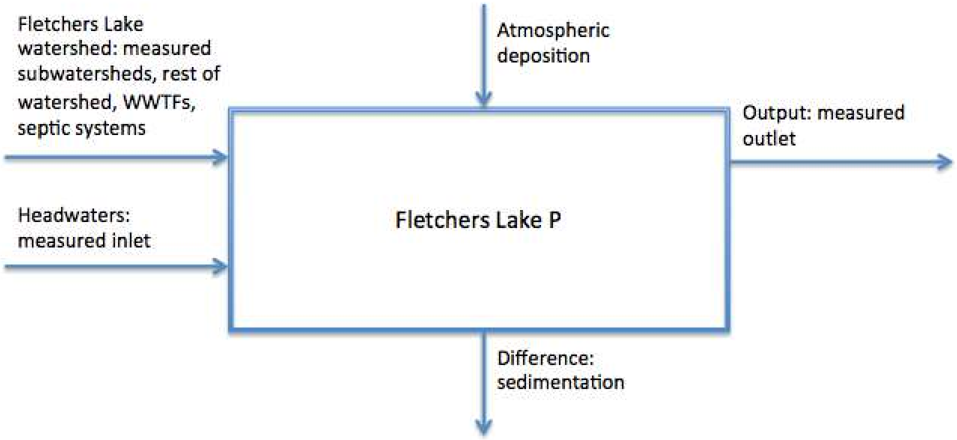


Figure 4.2 A diagram of P inputs and outputs to the Fletchers Lake system quantified by the mass balance analysis results shown in Table 7.

Table 6 Summary of the mass balance calculation listing masses of all components.

Input sources (kg)		% of total	Sources Total (kg)	Output (kg)	Difference (kg)
Measured subwatersheds	66.1	6	1087.0	695.7	391.3
Measured Inlet	631.1	58			
Atmospheric Deposition	18.1	2			
WWTFs	69.3	6			
Rest of watershed	64.0	6			
Septic systems	238.5	22			

The largest source of P to the lake system are the headwaters entering through the inlet at 631.1 kg (58 % of total input), and the second largest source are the septic systems on the land outside of measured subwatersheds, which contributed 238.5 kg (22 % of total input). The measured tributaries, WWTFs and runoff from the land outside the measured subwatersheds all contributed comparable amounts at 66.1 kg, 69.3 kg and 64.0 kg, respectively. The smallest source of P to the lake system was atmospheric deposition at 18.1 kg.

The R_{sp} represents the two main P removal mechanisms in the soil for septic system subsurface sewage plumes: precipitation and sorption (Sinclair et al., 2014). With age of a septic tank, the soil treatment of septic discharge decreases as the soil P sorption sites become saturated with the subsurface discharge plume, decreasing the soil P retention coefficient (Sinclair et al., 2014). The R_{sp} of 0.5 originating from Hart et al., (1978) has previously been used in P prediction models in HRM (Scott and Hart 2004; Jacques-Whitford, 2009). The value of R_{sp} = 0.8 from Sinclair et al., (2014) was found for septic systems prior to 30 years post-instalment. It was found that the selection of a soil P retention coefficient (R_{sp}) highly impacted the mass balance results. For instance, the mass of total inputs to the lake system decreased by 24.4 %, with an increase in the soil retention coefficient from 0.5 (Hart et al., 1978) to 0.8 (Sinclair et al., 2014). The mass difference between the inputs and output, or mass lost to sediments, was also decreased by 46.4 % with the same increase in soil retention. The two R_{sp}s were therefore examined in order to select the most representative coefficient.

Differences (between inputs and outputs) resulting from both R_{sp}s were compared to the theoretical sedimentation calculated with the Kirchner and Dillon (1975) formula shown in Section 3.4.2. With a 0.5 R_{sp} applied to septic tank loading, 596.2 kg of P entering the lake from septic tanks was calculated. This resulted in a total of 1444.7 kg of P entering the lake system from all input sources, and a difference of 749.0 kg (between inputs and output from the lake outlet) as the P settled to the lakebed. This settled amount measured at 51.8 % P retention (R_p) in the lake. The R_{sp} of 80 % gave a mass of 238.5 kg loaded from septic tanks, and a total input of 1087.0 kg, giving a new difference (between inputs and output) of 391.3 kg. The new difference reflected 36.0 % P retention within the lake. The theoretical in-lake P retention (R_p) value calculated using the Kirchner and Dillon (1975) formula (Section 3.4.2) was 0.11, or 11 % retention. An apparent settling velocity (v) of 12.4 was chosen for an oxic hypolimnion (Section 4.5), and an areal hydraulic load *(q_s)* of 97.8 m (Equation 3.4 in Section 3.4.2) was calculated using measured hydraulic loads from tributaries and the inlet (9.97 x 10^7 m^3), precipitation (1.57 m), evaporation (0.552 m; Jacques-Whitford, 2009) and runoff from the rest of the watershed which was estimated using measured runoff coefficients for developed and non-developed land, 0.45 and 0.43, respectively. It is important to note that the Kirchner and Dillon (1975) sedimentation model was calibrated on oligotrophic lakes in central Ontario, located on the unique, igneous geology of the Pre-Cambrian shield (Dillon and Rigler, 1975). The differing region and geology creates an uncertainty as to how appropriate this formula is for lakes in other regions such as NS. Furthermore, the model has its own specific limitations, as it has less predictive ability for anoxic hypolimnia than for oxic hypolimnia, and does not consider lakes with short or periodic anoxic hypolimnia (Peterson et al., 2006).

As an additional comparison to select the most representative R_{sp}, the septic system loading from dwellings within the measured FBB subwatershed was calculated using both R_{sp}s and compared to the measured export at FBB (12.5 kg with a P export coefficient of 4.9 mg P/m^2/y). The calculated septic system load was 176.6 kg for a R_{sp} of 0.5 and 70.7 kg for a R_{sp} of 0.8. The R_{sp} of 0.8 was chosen for use in the mass balance as the resulting in-lake P retention (R_p) of 37.3 % was closest to the calculated theoretical R_p of 11 %, and the resulting septic load of 70.7 kg was closest to the measured 12.5 kg at FBB.

The P export coefficients used to calculate P loading from the rest of the watershed land, outside measured subwatersheds, are shown in Table 8 below. The P export coefficients calculated in this study for forested land, 4.1 mg P/m^2/y, and for low-density residential land, 9.1 mg P/m^2/y P, were used. Coefficients used in previous P modelling studies in HRM (Jacques-Whitford, 2009; Scott and Hart, 2004) were applied to the other land use types.

Although the value of 0.8 was chosen for the R_{sp} as it best represented measured P loading at FBB, and produced the closest value to the theoretical P settling in the lake, it should be noted that soil P retention decreases with age of septic systems as was shown in Sinclair et al., (2014). The 80 % soil retention dropped to about 50 % after approximately 30 years in septic system P loading models calibrated by Sinclair et al. (2014). Dillon and Molot (1996) have reported that septic plumes can take decades to reach waterways via migration due to slow subsurface movement as low as < 1 m/yr, and that the soil P saturation may still not be at capacity once this has occurred. Thus, although the measured P loading from the FBB subwatershed was only 12.5 kg compared to an estimated 70.7 kg P loading from septic system discharge, it is possible the subsurface septic plumes have not yet reached the waterways, and that soils are not yet fully saturated with P, therefore the septic discharge is not reflected in the measured P mass at that location. Additionally, there exists much uncertainty around the 300 m boundary around waterways used to identify contributing septic systems, as this distance is not based on any scientific findings (Brylinsky, 2004; Scott et al., 2003). As a result of the uncertainties around septic system P loading, the septic system contribution seems to be the largest un-known in the mass balance model. It is likely this P component will increase with the age of septic systems, as soils become saturated with P, and as septic plumes migrate toward waterways (Sinclair et al., 2014; Paterson et al., 2006).

With regard to lake management and development planning, 58 % of total P loading was measured from the headwaters, which indicates that P loading management within the Fletchers Lake watershed would not be as effective as management around the headwaters. The headwaters are likely a large contributor of P to Fletchers Lake due to the increased density of land development along the headwater lakes toward Lake Charles. The watersheds of Lake Thomas and Lake William contain suburban developed land including residential, commercial and institutional development (Jacques-Whitford, 2009). Lake Charles is surrounded by urban residential and commercial development (Jacques-Whitford, 2009). Further significant P contributors to the headwaters could include older WWTFs such as the Frame Subdivision WWTF (built in the early 1970s) discharging into Lake William, and older on-site septic systems (Halifax Water, 2017; Jacques-Whitford, 2009). Efforts to control P loading within the Fletchers Lake watershed would be most appropriate for septic system loading, as this was the largest potential estimated P source within the watershed, and will likely increase as septic systems age, and as septic plumes migrate toward waterways (Sinclair et al., 2014; Paterson et al., 2006).

Table 7 P export coefficients for different land use types used to calculate P export from the rest of the Fletchers Lake watershed outside of the measured subwatersheds. Their sources are listed.

Land Use Type	P Export Coefficient	Source
Forested	4.1	This study
Cleared	8.3	Scott et al., 2000
Commercial	40	Waller and Hart 1985
Institutional	42	Waller and Hart 1985
Low-density residential	9.1	This study
Inland Water	17.3	Underwood 1984

4.4 Stream Water Quality

The following section summarizes stream sonde data (Table 9), flows with water quality and precipitation (Figures 4.2 – 4.7), and flows with precipitation in Figure 4.8.

The means and ranges of all sonde parameters measured at each sampling station are shown in Table 9 below. The smallest streams, FTB and FND, both showed the lowest mean temperatures, 9.05 °C and 9.07 °C respectively. FND had the lowest mean DO at 8.27 mg/L and FTB had the highest mean DO at 14.64 mg/L. The highest and lowest conductivity was observed at FI (176 µS/cm) and FTB (26 µS/cm), respectively. The mean pH was near neutral at all locations (> pH 6), except at FTB (pH 5.51) where it was moderately acidic and at FND (pH 4.33) where it was highly acidic (Clair et al., 2007).

Table 8 Stream sonde measurements: July, 2015 – August, 2016

	Temperature °C	**DO (mg/L)**	**Cond. (µS/cm)**	**pH**
FI				
Mean (Range)	13.38 (1.28 - 25.33)	11.59 (8.31-16.35)	176 (152 - 212)	6.55 (5.39– 7.67)
FO				
Mean (Range)	13.99 (1.70 - 26.45)	11.63 (8.02-17.16)	168 (122-199)	6.97 (6.30 –7.95)
FBB				
Mean (Range)	10.39 (0.04-24.20)	12.52 (8.35 - 17.20)	78 (51 - 160)	6.62 (5.43 – 7.90)
FTB				
Mean (Range)	9.50 (0.64-18.68)	14.64 (6.61 – 16.52)	26 (21 - 41)	5.51 (4.13 – 7.25)
FD				
Mean (Range)	11.40 (0.82-22.94)	11.96 (8.33 – 16.68)	134 (100 - 402)	6.53 (5.41 – 7.75)
FND				
Mean (Range)	9.07 (0.59-19.35)	8.27 (3.14 – 15.56)	32 (28 - 43)	4.33 (3.57 – 6.30)

Figures 4.2 – 4.7 summarize all water quality parameters along with precipitation and average daily discharge for each sampling site over the course of the sampling timeframe. TSS and TN often measured below their detection limits (DL) of 1 mg/L (TSS) and 0.120 mg/L (TN), in which case the DL was used as the data point for that sample. SRP was also often below its DL (1 µg/L) and the same method was applied. Hydrographs paired with precipitation over the course of the sampling year are plotted for each site in Figure 4.8 to show that the regressed flows from pressure loggers coincided with precipitation in terms of their peak alignments.

The lake outlet, FO (Figure 4.2), and the lake inlet, FI (Figure 4.3), show largest volumes of discharge and least responsive peaks to precipitation events. Flows at all sites were highest in the late fall, through the winter, until spring; from approximately October until May. FI and FO show more subtle peak reactions to precipitation, likely due to their larger size and ability to accommodate for larger discharge. FTB (Figure 4.7), FBB (Figure 4.6), FD (Figure 4.5), FND (Figure 4.4), show more reactive peaks on smaller magnitudes, 0 – 0.4 m^3/s, compared with inlet and outlet magnitudes of 0 – 6 m^3/s, which show more subtle peak reactions to precipitation events. Flows are seen to increase throughout the winter and spring, between December and May, at all locations.

At all sites, turbidity was the most reactive parameter to discharge. Turbidity exhibited higher levels in the winter and spring, from January to April, at FO and FD, but this seasonal trend was not apparent at other locations where turbidity showed more correlation with precipitation. The smaller streams, FND and FTB, had higher levels of TP and DOC during drier, low flow periods in the spring and summer (May-September). TOC during the summer at FND ranged from 14 – 19 mg/L, and 3 – 9 mg/L at FTB, compared with 9 – 14 mg/L at FND and 2 – 4 mg/L at FTB in the winter. Spring and summer TP at FND ranged from 12 – 50 μg/L, and at FTB ranged from 4 – 40 μg/L, compared with fall and winter ranges: 9 - 11 μg/L at FND and 1 - 6 μg/L at FTB. The larger sites, FO and FI, exhibited slightly higher TOC levels in the fall and winter, between 3 - 4 mg/L for both sites compared with summer TOC levels at 4 - 5 mg/L for both sites, but they did not show strong seasonal trends for TP. TP and TOC response to precipitation was most pronounced at FD and FBB, with peaks reaching 20 – 38 μg/L compared with baseline export around 10-15 μg/L for FD and around 5 – 10 μg/L for FBB. Due to TN and TSS often measuring at or below their detection limits, variations or trends in their measurements were little-to-none, except for occasional higher TSS values on precipitation days.

Three heavy precipitation events were captured by the water quality sampling. The first was in 2015 on December 3 (19.6 mm), and the following two were in 2016, on March 2 (36.2 mm) and June 8 (37.4 mm). The water quality peaks corresponding to the December (2015) rainfall event are most prominent at FBB for TSS, TOC, TP and turbidity. Sites FO, FI, FD and FTB also showed turbidity peaks on that day. The March (2015) rainfall event corresponds to water quality peaks at FI for TSS, TP and turbidity and at FD for TP and turbidity. The June (2016) rainfall event corresponds to peaks at all locations for TOC, TSS and TP, and turbidity (only DP was measured for FD that day).

Out of all the sampling sites, FND had the highest concentrations of TOC and TP which is hypothesized as a result of P loading contributions from the dominant forest species (coniferous), and the wetland component, both of which can exhibit higher amounts of nutrient export (Tamm & Hallbäcken, 1988; Scott et al., 2000; SanClements et al., 2010; Paterson et al., 2006).

Seasonal nutrient loading trends are to be expected, as shown by the stream water quality graphs where the majority of nutrient export occurred during seasonal precipitation events (Winter and Duthie, 2000; Santos et al., 2015; Esbroeck et al., 2016). Increases in levels of nutrient loading during low flow conditions in the summer at smaller streams, and during high flow conditions in the inlet and outlet, emphasize the importance of collecting nutrient data throughout all seasons when sampling at different sizes of discharge. The occurrence of higher nutrient measurements during rain falls and high flow at the majority of the sampling sites stresses the importance of capturing these events when measuring nutrient loading. A single heavy precipitation event could account for as much as 56 % of average annual P loading as has been reported by Esbroeck et al., (2016). Therefore, sampling designs for quantifying nutrient loads would be most accurate with focused sampling efforts during rainfall and snowmelt events in addition to regular sampling for baseline conditions.

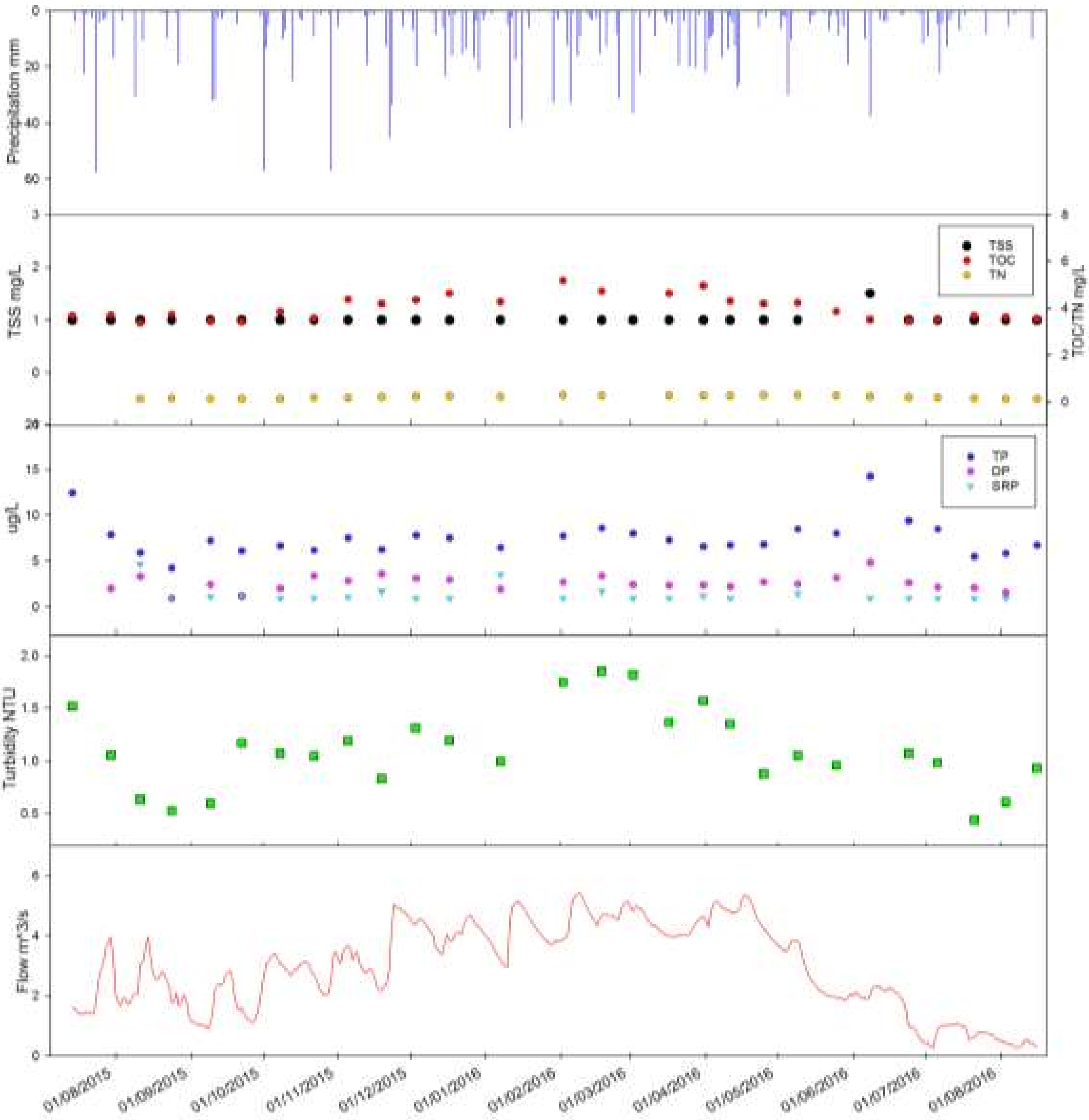


Figure 4.3 FO stream water quality parameters (TSS, TN, TOC, TP, DP, SRP, and turbidity) stacked with precipitation (m) on top, and average daily discharge (m^3/s) on the bottom.

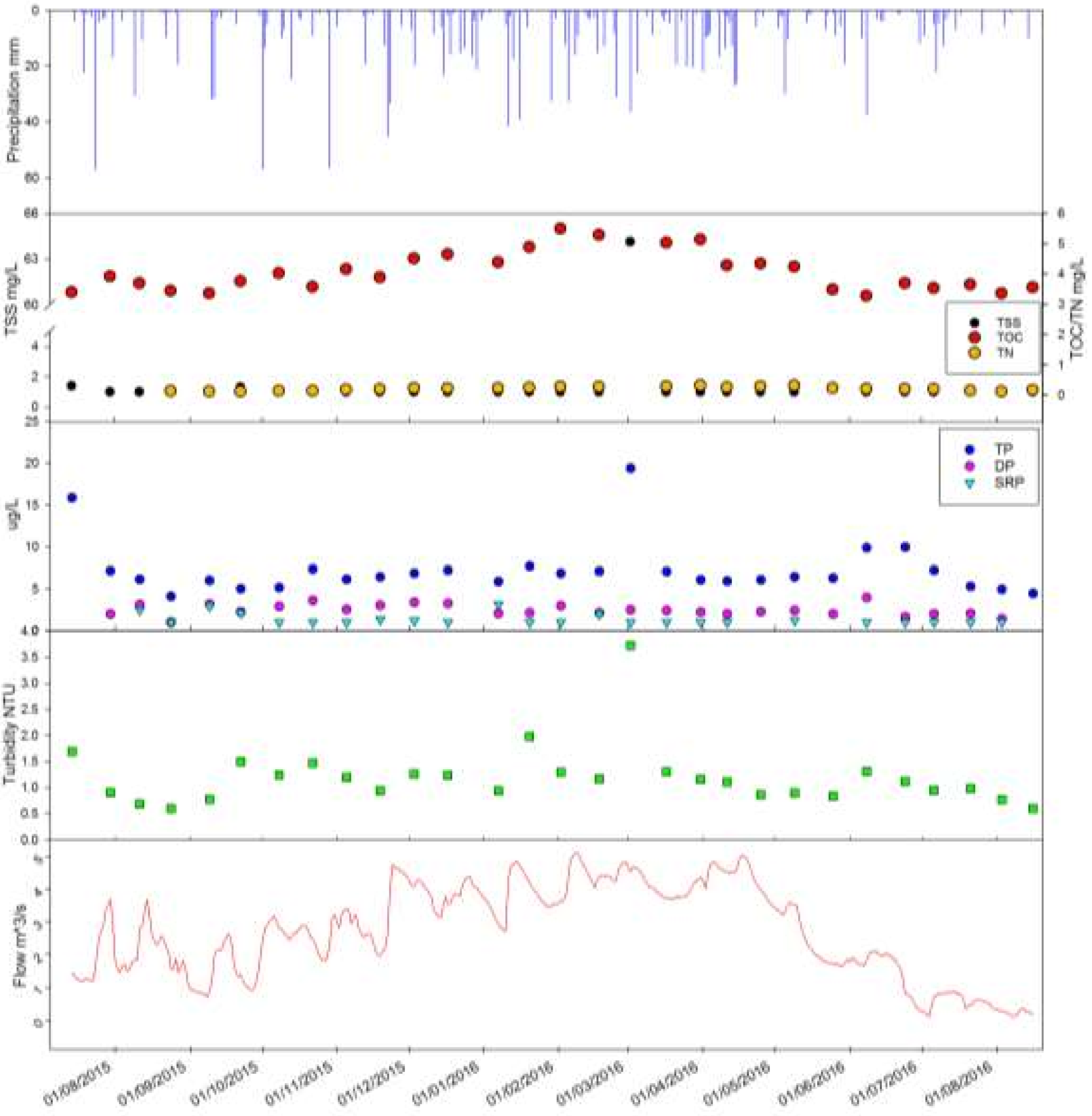


Figure 4.4 FI stream water quality parameters (TSS, TN, TOC, TP, DP, SRP, and turbidity) stacked with precipitation (m) on top, and average daily discharge (m^3/s) on the bottom.

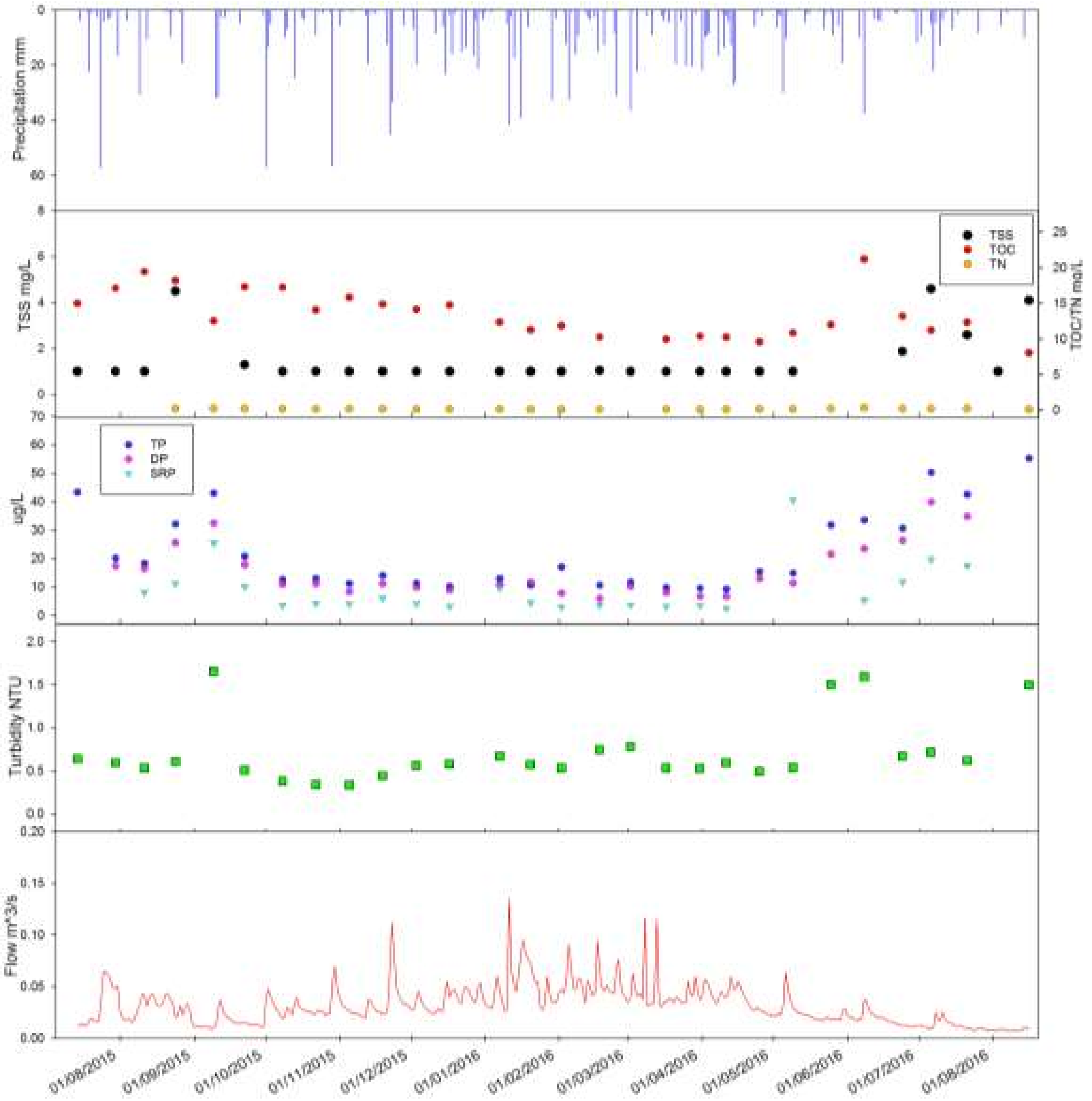


Figure 4.5 FND stream water quality parameters (TSS, TN, TOC, TP, DP, SRP, and turbidity) stacked with precipitation (m) on top, and average daily discharge (m^3/s) on the bottom.

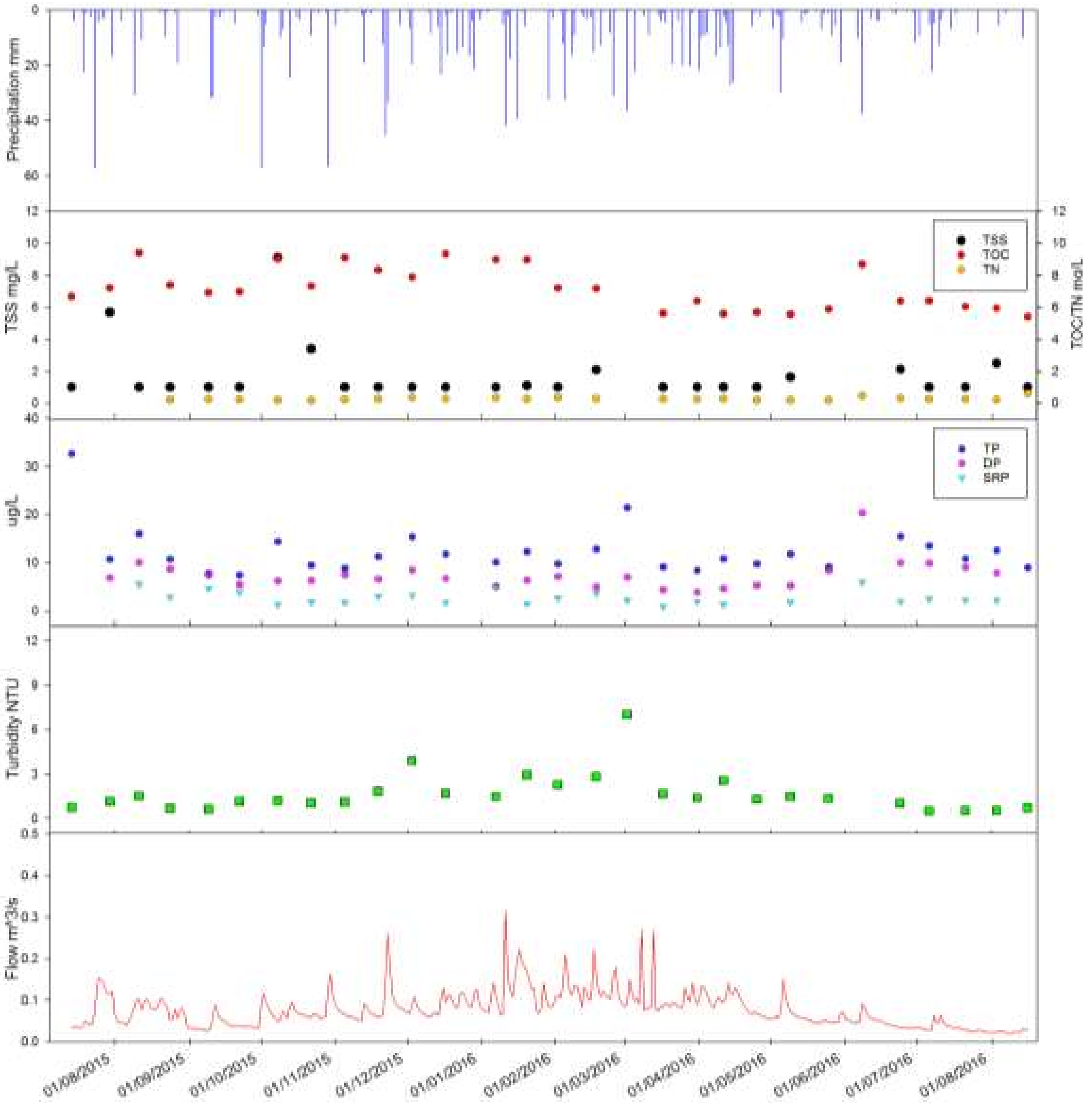


Figure 4.6 FD stream water quality parameters (TSS, TN, TOC, TP, DP, SRP, and turbidity) stacked with precipitation (m) on top, and average daily discharge (m^3/s) on the bottom.

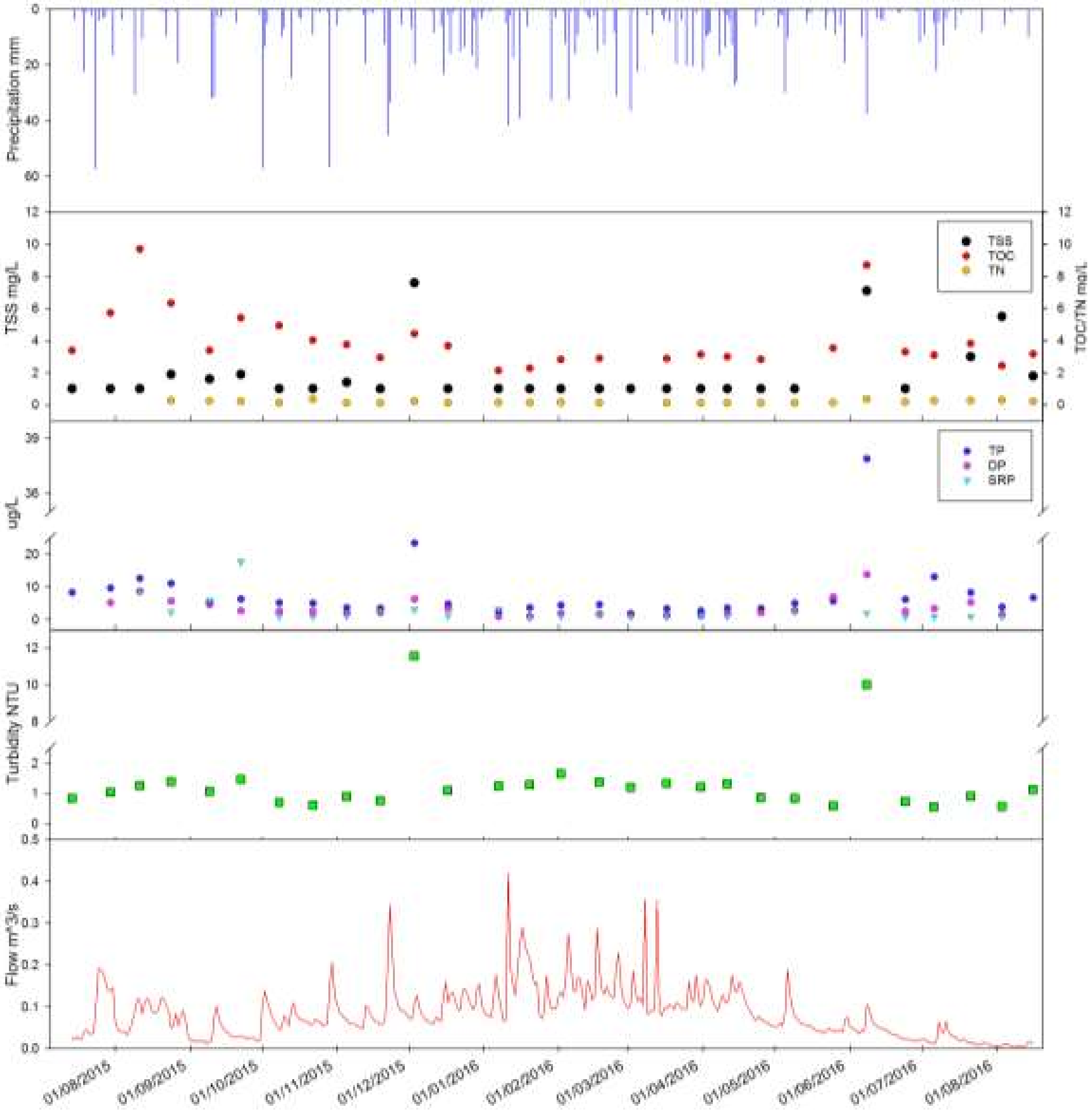


Figure 4.7 FBB stream water quality parameters (TSS, TN, TOC, TP, DP, SRP, and turbidity) stacked with precipitation (m) on top, and average daily discharge (m^3/s) on the bottom.

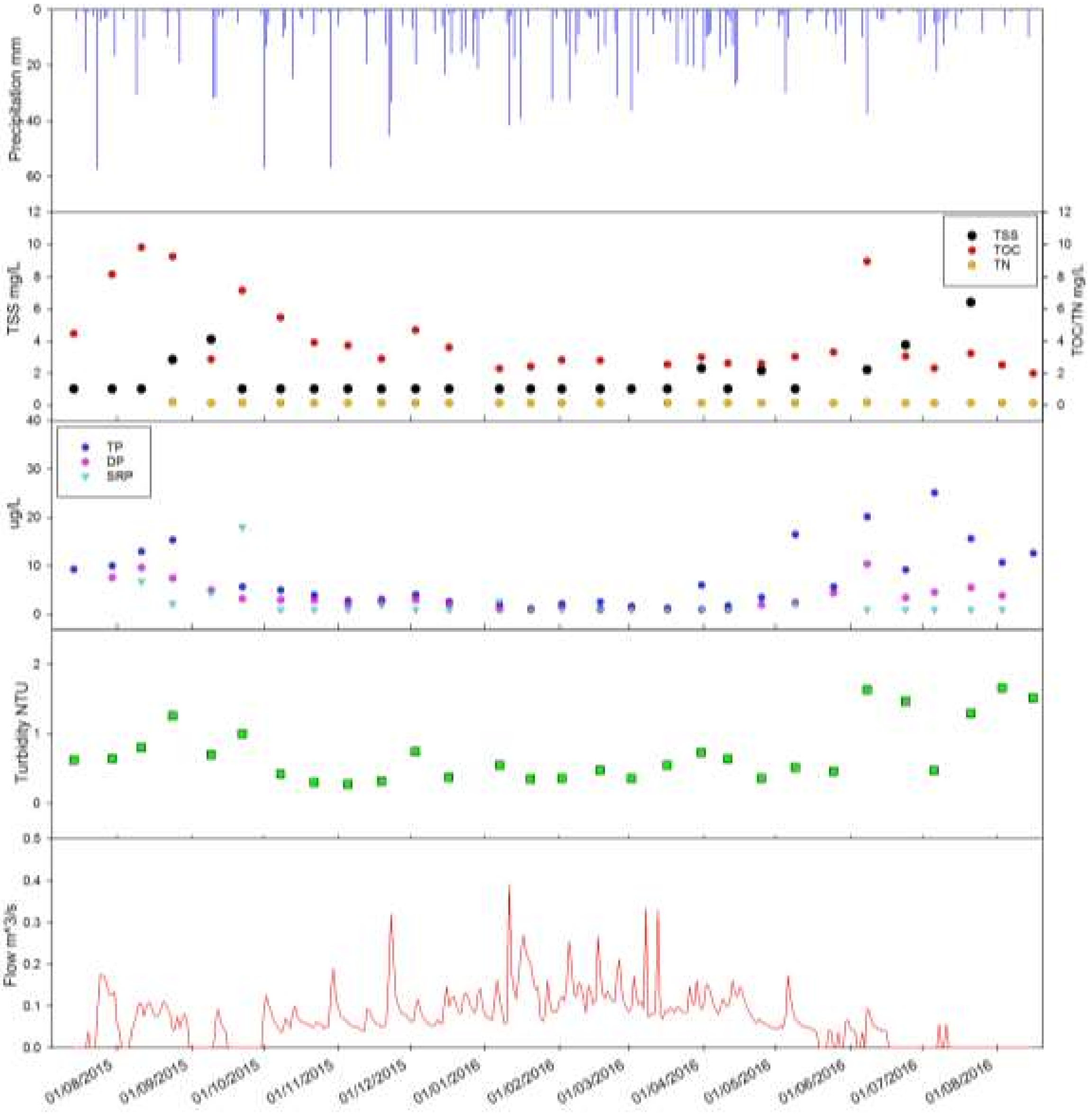


Figure 4.8 FTB stream water quality parameters (TSS, TN, TOC, TP, DP, SRP, and turbidity) stacked with precipitation (m) on top, and average daily discharge (m^3/s) on the bottom.

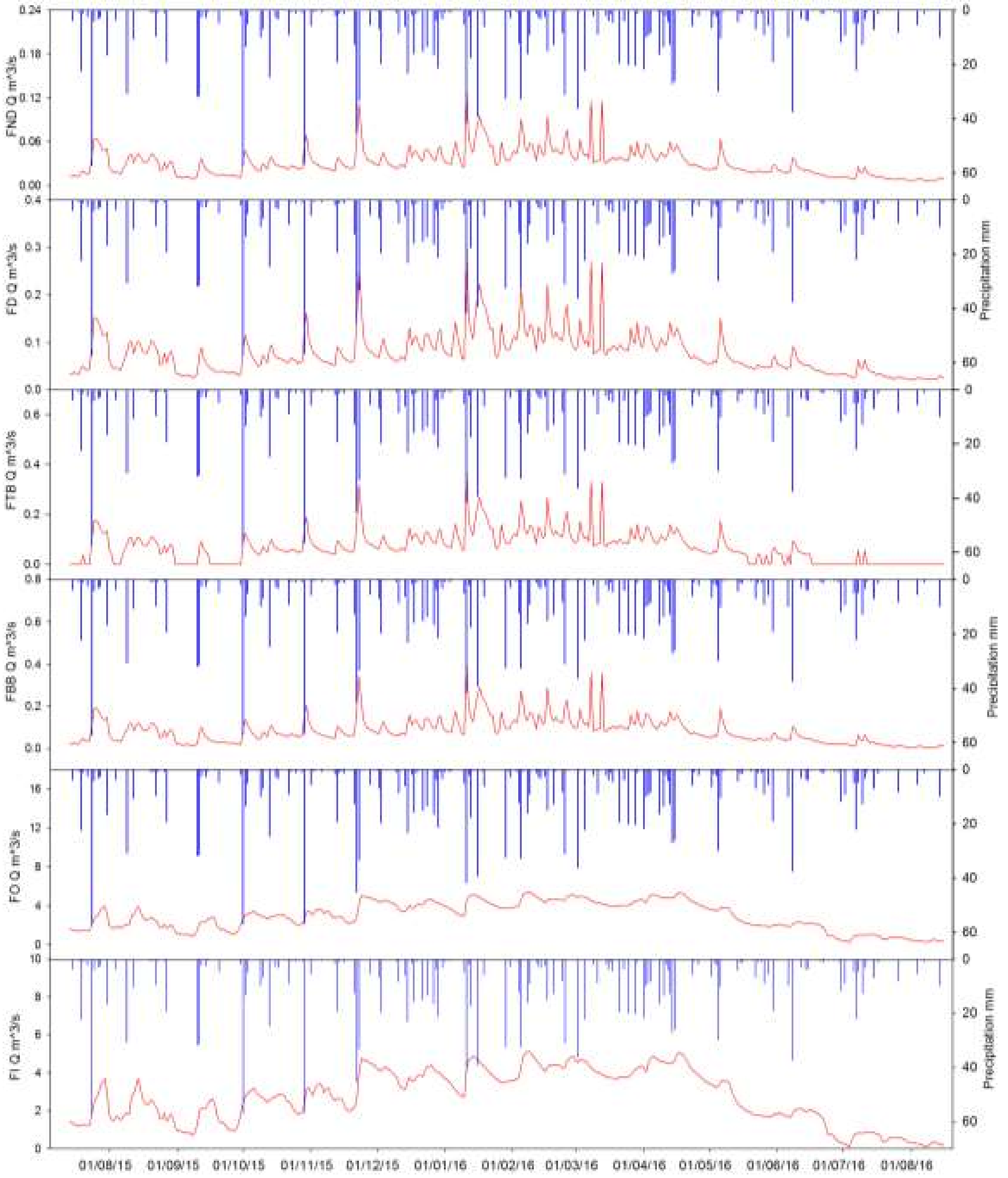


Figure 4.9 Compiled hydrographs and precipitation charts for sampling sites.

4.5 In-Lake Conditions and Trophic State

The following Tables (10 - 14) summarize all in-lake trophic parameters from four sampling events: August 2015, May 2016, September 2016, and November 2016. Samples were taken throughout the water column at each basin, as well as the lake outlet. Values listed were volume- weighted for each basin, and then averaged with the outlet values. Values in with a DL label and are reported as their DLs (1.0 μg/L for SRP, 0.12 mg/L for TN). Values missing are shown with '-'. Sampling was completed after spring and fall turnovers, when the water column was well-mixed, as was evident from the dissolved oxygen (DO) and temperature profiles (Figure 4.9), which showed uniformity throughout the water column, aside from a slight thermocline during the first sampling event in August, 2015. The DO and temperature profiles are summarized for each basin (F Stn 1 – inlet basin and F Stn 2 – outlet basin) by sampling event in Figure 4.9. The DO-T profiles indicate the lake is well-oxygenated, and they confirm an oxic hypolimnion (DO > 2 mg/L; Nurnberg, 1996) making it unlikely that internal sediment P loading is a P source to the lake system due to changing redox conditions resulting from an anoxic sediment-water interface (Mortimer, 1942; Hupfer and Lewandowski, 2008). However, it is still possible to have anoxic sediments that release P into an oxic water column (Hupfer and Lewandowski, 2008), thus, P release from an oxic sediment-water interface can still occur and cannot be ruled out on this condition alone. Additional factors that can affect internal loading in oxic hypolimnia include changing redox conditions from sulphide formation (Bostrom et al., 1988), microbial decomposition and bioturbation from abundant macroinvertebrates (Kalff, 2002), temperature increases and wind mixing in shallow lakes (Søndergaard, Jensen, & Jeppesen, 2003) and macrophyte beds (Stephen et al., 1997). Internal loading on smaller temporal scales can still be occurring in Fletchers Lake as a result of these additional P release drivers, however, the mass balance calculated on a year time step shows net sedimentation at 391.3 kg. If P release is occurring, it likely does so on much smaller temporal and spatial scales, and is insignificant in relation to the net sedimentation as calculated from the mass balance.

Excluding Secchi depth, which indicated a mesotrophic state, all other trophic state parameters indicated that Fletchers Lake is currently oligotrophic (Table 14). The high trophic category interpreted by the secchi depth can be explained by the higher levels of DOC, or humic matter, limiting light transparency, as is typical for most lakes in NS (Underwood, 1986). By Nurnberg's (1999) DOC threshold (2 mg/L), Fletchers Lake classifies as a humic lake. The results shown here give an example of a lake lying along the dystrophic and oligotrophic spectrum (Webster et al., 2008; Brylinsky, 2004), for which the secchi depth, a widely used parameter for measuring lake trophic status in North America, is misrepresentative of the true lake productivity. For this reason it is suggested that colour or DOC are included with traditional trophic state indices, to improve interpretation of trophic state in lakes along the dystrophic-oligotrophic spectrum (Webster et al., 2008; Nurnberg, 1996).

The highest lake average TP measured at 6.8 μg/L and the lowest was 3.7 μg/L. This range is comparable to, but slightly lower than, historical measurements: 7 μg/L in 1978 (Hart et al., 1978), 6 μg/L in 1987 (Mudroch et al., 1987), and 7.5 μg/L in 2007 (Jacques-Whitford, 2009). The results from this dataset do not indicate that Fletchers Lake is increasing in trophic state with respect to TP. All historical TP averages are compared to the surface TP average from the HRM dataset (Stantec, 2012) and the surface TP average from this study (calculated as the average of all surface and outlet samples throughout sampling period) in Figure 4.9. The oligo-mesotrophic boundary is shown at 10 μg /L and the 95 % confidence intervals for the HRM 2007 – 2011 TP mean as well as for this study's TP mean are shown on each respective bar. The average TP from Hart et al., (1978) was measured on one summer sampling occasion (August) at the outlet basin (F Stn -2) from two samples: one surface sample and one bottom sample (Richard Scott, personal communication, April 4, 2017). The average TP from Mudroch (1987) was calculated from four separate sampling events (April, June, August and November) during the ice-free season of 1983, each of which included 3 lake depths (top, middle and bottom) sampled at the lake outlet. The Jacques-Whitford (2009) average TP was calculated from one surface sample obtained at both lakes basins, on two separate sampling events, in the spring (June) and summer (August) of 2007. The HRM dataset average TP was calculated from surface samples taken at both lake basins on sampling dates summarized in Table 15 (Stantec, 2012).

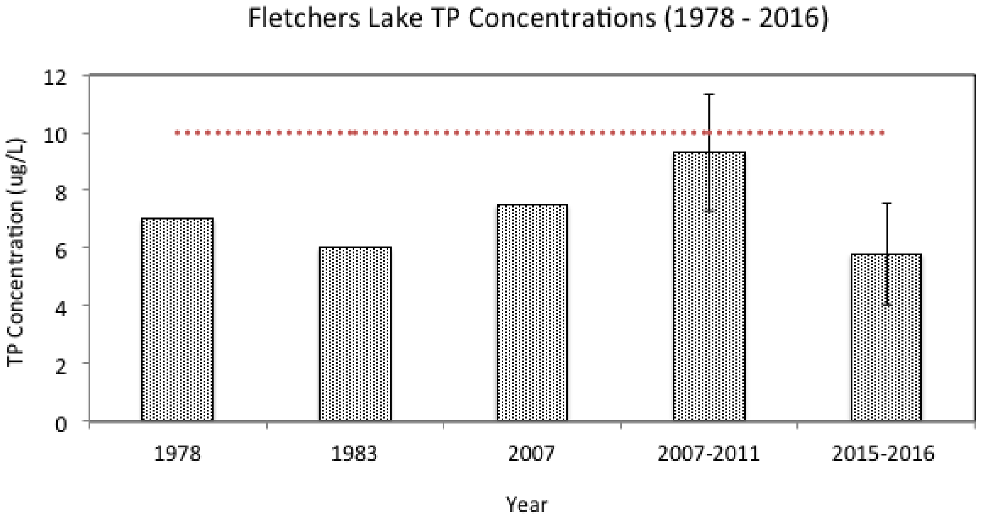


Figure 4.10 Fletchers Lake mean TP concentrations (µg /L) for 1978 to 2016. Mean for 1978 is from Hart et al., (1978), mean for 1983 is from Mudroch (1987), mean for 2007 is from Jacques-Whitford (2009), and mean from 2007-2011 is from the HRM Lakes Water Quality Monitoring Program (Stantec, 2012). The oligo-mesotrophic boundary is shown at 10 µg /L. 95 % confidence intervals are shown for the TP means from this study and from the HRM dataset.

Table 9 Lake Sampling August 2015. Basin values are volume-weighted, lake averages are an average of basin values and outlet values.

Parameters	Inlet Basin	Outlet Basin	Lake Average
TP (µg/L)	2.9	3.0	3.4
DP (µg/L)	1.0	1.2	1.3
SRP (µg/L)	1.0 ^{DL}	1.0 ^{DL}	1.0 ^{DL}
Turbidity (NTU)	0.461	0.459	0.497
TOC (mg/L)	2.15	2.34	2.69
DOC (mg/L)	1.85	2.01	2.31
TN (mg/L)	0.12 ^{DL}	0.12 ^{DL}	0.12 ^{DL}
Chlorophyll *a* (µg/L)	1.04	1.15	1.18
Color (Pt Co Units)	5	4	5
Secchi depth (m)	4.8	5.6	5.2

Table 10 Lake Sampling May 2016. Basin values are volume-weighted, lake averages are an average of basin values and outlet values.

Parameters	Inlet Basin	Outlet Basin	Lake Average
TP (µg/L)	3.0	2.7	3.9
DP (µg/L)	1.1	1.0	1.3
SRP (µg/L)	1.0 ^{DL}	1.0 ^{DL}	1.0 ^{DL}
Turbidity (NTU)	0.422	0.364	0.520
TOC (mg/L)	2.07	1.88	2.66
DOC (mg/L)	2.11	1.85	2.67
TN (mg/L)	0.14	0.14	0.20
Chlorophyll *a* (µg/L)	0.77	0.71	0.88
Color (Pt Co Units)	17	10	17
Secchi depth (m)	3.9	4.1	4.0

Table 11 Lake Sampling September 2016. Basin values are volume-weighted, lake averages are an average of basin values and outlet values.

Parameters	**Inlet Basin**	**Outlet Basin**	**Lake Average**
TP (µg/L)	3.7	3.8	4.9
DP (µg/L)	-	-	-
SRP (µg/L)	-	-	-
Turbidity (NTU)	0.828	0.836	0.855
TOC (mg/L)	1.37	1.54	1.96
DOC (mg/L)	-	-	-
TN (mg/L)	0.12 ^{DL}	0.12 ^{DL}	0.12 ^{DL}
Chlorophyll *a* (µg/L)	1.84	1.37	3.3
Color (Pt Co Units)	3	3	4
Secchi depth (m)	3.2	3.4	3.3

Table 12 Lake Sampling November 2016. Basin values are volume-weighted, lake averages are an average of basin values and outlet values.

Parameters	**Inlet Basin**	**Outlet Basin**	**Lake Average**
TP (µg/L)	1.9	2.8	3.5
DP (µg/L)	-	-	-
SRP (µg/L)	-	-	-
Turbidity (NTU)	0.310	0.405	0.514
TOC (mg/L)	1.22	1.75	2.27
DOC (mg/L)	-	-	-
TN (mg/L)	0.12 DL	0.12 DL	0.12 DL
Chlorophyll *a* (µg/L)	0.26	0.30	0.39
Color (Pt Co Units)	5	8	10
Secchi depth (m)	5.4	5.2	5.3

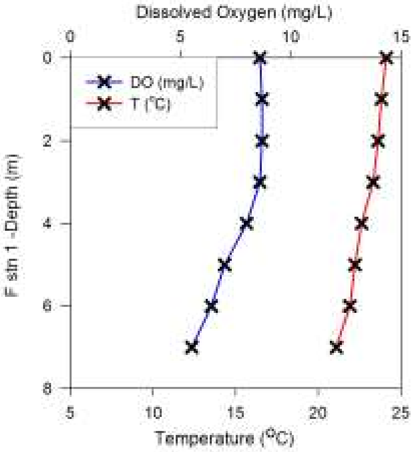
DL - reported as the detection limit (1.0 µg/L for SRP, 0.12 mg/L for TN)

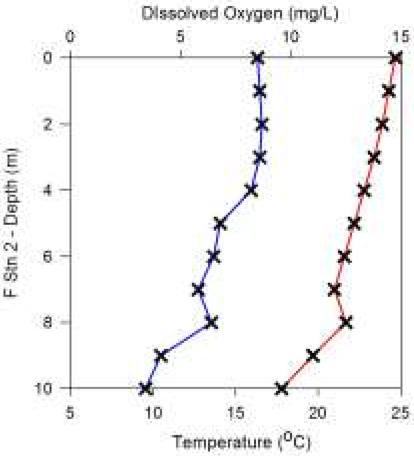
Table 13 Lake Averages Summarized

Parameters	**Aug. 2015**	**May 2016**	**Sept. 2016**	**Nov. 2016**	**Trophic State**
TP (µg/L)[a]	3.4*	3.9	4.9	3.5	Oligotrophic*
TN (mg/L)[b]	0.12 DL	0.20	0.12 DL	0.12 DL	Oligotrophic
Chlorophyll *a* (µg/L)[a]	1.18	0.88	1.32	0.39	Oligotrophic
Secchi depth (m)[a]	5.2	4.0	3.3	5.3	Mesotrophic

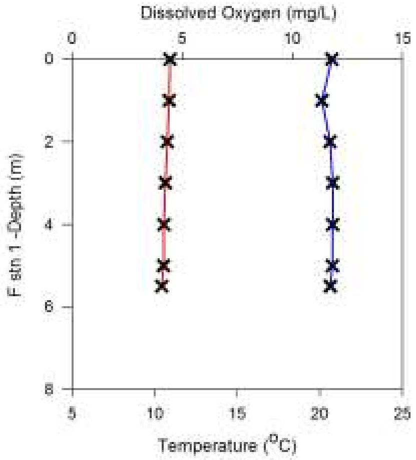
* 3.7 µg/L P (<4) = ultra-oligo; <1 µg/Lchl a = ultra-oligotrophic
a (CCME, 2004)
b (Dodds, 1998)

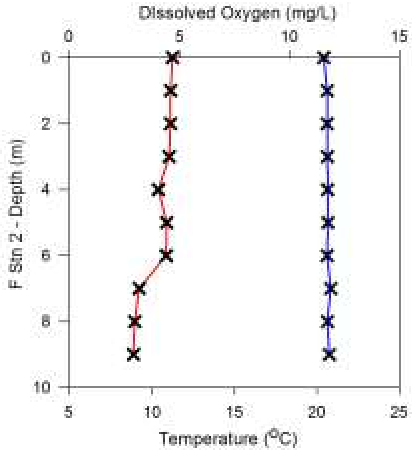
August 2015



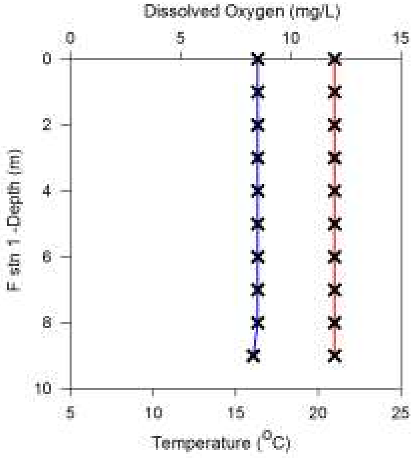


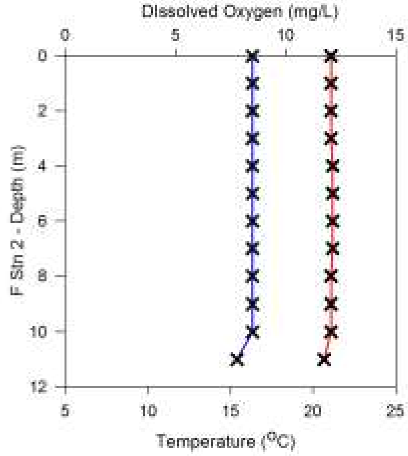
May 2016





September 2016





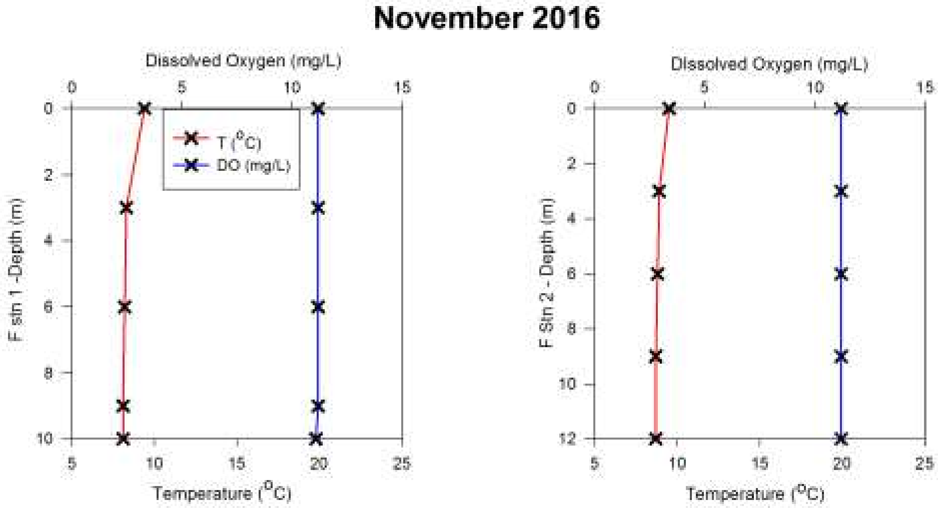


Figure 4.11 Isotherms for both Fletchers Lake basins (F Stn 1 on left, F Stn 2 on right) showing temperature (°C)(x-axis bottom) and Dissolved Oxygen (mg/L) (x-axis top) with depth (Y).

The TP dataset produced by the HRM Lakes Water Quality Monitoring Program from 2007 – 2011 (Stantec, 2012) was statistically compared to the TP data collected in this study with an independent t-test using Mini-tab 16 ® (Minitab Inc., State College, PA, U.S.A.). Both data sets passed a normality test at the 0.05 level of significance using Minitab's inbuilt normality test. The mean TP in this study was significantly lower than the HRM TP mean at the 0.05 level of significance (p-value of 0.003). The data sets are shown in Table 15 below. Each of HRM's data points comprised of three surface samples taken across the lake length, thus only surface basin samples and outlet samples were averaged for each sampling date from this data set.

Although the TP range from this study did not exceed TP values measured in earlier studies (Hart et al., 1978; Mudroch et al., 1987; Jacques-Whitford, 2009) and the statistical comparison of current TP with HRM TP from 2007-2011 showed a significant decrease in TP in Fletchers Lake, it is important to remember that TP is not a direct measure of primary productivity. The most direct measure of trophic state is algal biomass, for which Chl *a* is an appropriate indicator (Nurnberg, 1996).

As is being put forth by the EU WFD, direct trophic state indicators, such as Chl *a* should be additionally considered in the assessment of trophic state with future monitoring (Carvalho et al., 2006).

Fletchers Lake high flushing of 35 times/yr (Jacques-Whitford, 2009) can make the system less responsive to P concentration changes. Therefore, it is recommended that future monitoring include Chl *a* as the most direct measure of trophic state. Although the turnover time of approximately 11 days can make it difficult for algal communities to establish themselves, P and Chl *a* still show strong positive correlation over vast ranges of WRT and many studies report that nutrient enrichment to flowing rivers and streams can still result in increased in algal growth (Hoyer and Jones, 1983; Smith, 1999). Thus a short WRT cannot entirely prevent algal blooms from occurring, and increases in trophic state are not out of the question for Fletchers Lake.

Furthermore, due to the variability of alkalinity (Clair et al., 2007) and the abundance of humic lakes (Underwood, 1986) in NS, these additional parameters which have shown strong relationships with TP:Chl *a* (Nurnberg, 1996; Spears et al., 2013), should be generally considered for future trophic state monitoring in NS. Nurnberg (1996) has shown that classification of lakes into groups of alkalinity and color can improve the TP:Chl *a* relationship for better representation of trophic state from TP measurements. Carvalho (2006) also grouped lakes by mean depth and altitude to improve TP:Chl *a* regressions. Although such further classification can improve the accuracy of TP as a trophic state indicator, it can be difficult to obtain additional lake characteristics due to limitations of resources. If an additional parameter is to be included in trophic state analysis, aside from TP, it is most appropriate to use Chl *a,* as it is the most direct trophic state measure.

Limitations to Chl *a* monitoring should be considered when including Chl *a* as a primary trophic state indicator, as this measure can vary seasonally, throughout the growing season, and annually (Carvalho et al., 2013). Therefore it has been suggested that 6 – 10 samples are collected throughout a growing season (Carvalho et al., 2013), as well as multiple samples throughout a lake's depths, and that minimum and maximum values are compared to capture the seasonal, and within-growing season variability.

Table 14 Surface sample TP data for Fletchers Lake statistically compared from this study and from the HRM Lakes Water Quality Monitoring Program

This study		**HRM Lake Monitoring**	
Date	**TP (μg/L)**	**Date**	**TP (μg/L)**
August 2015	4.4	Spring 2007	12
May 2016	5.8	Summer 2007	5
September 2016	7.1	Fall 2007	9
November 2016	5.9	Spring 2008	7
		Summer 2008	5
Mean	**5.8**	Fall 2008	12
n	4	Spring 2009	9
SD sample	**1.1**	Summer 2009	10
		Fall 2009	16
		Summer 2010	11
		Fall 2010	7
		Summer 2011	9
		Mean	**9.3**
		n	12
		SD sample	**3.2**

Lake sonde data is summarized as basin averages in Table 17 for all sampling dates. The temperature ranged from 8.1 – 24.7, DO from 3.4 – 11.88, conductivity from 159.6 – 206.9 μS/cm, and pH from 6.1 – 6.8. Alkalinity for all sampling dates is shown in Table 16 and was in the range 7 – 25 mg/L $CaCO_3$, with means 7.8, 12.2, 13.8, and 16.1 mg/L for all four sampling dates from August 2015 to November 2016. The pH was near neutral, and higher than the typical ranges seen for acidified lakes (pH 0 -6) in southwestern NS (Clair et al., 2007) but it exhibited an identical range of pH 6.1 – 6.9, as was measured in 1983 (Mudroch, 1987). The alkalinity is classified as low to moderate as per Phillips et al., (2008) classification of European lakes, using the Hem (1985) conversion factor of 0.02 to convert from mg $CaCO_3$ to meq/L. Compared with other lakes in NS, however, alkalinity in Fletchers Lake is well above the range of 0 – 2 mg/L $CaCO_3$ common for lakes in southwestern NS, giving it a comparably high acid-buffering capacity (Clair et al., 2007). Compared to a historical alkalinity range of 0.3 – 4.4 mg/L CaCO3 from 1983 (Mudroch, 1987), Fletchers Lake appears to have substantially increased its buffering capacity. The lake exhibits a softwater alkalinity range (Shannon and Brezonik, 1972), and a hardwater conductivity range and as per Nurnberg (1996) and Shannon and Brezonik (1972). Calcium concentrations ranged from 6.08 - 7.47 mg/L for all sampling dates, and fall within the hardwater range (Shannon and Brezonik, 1972). Calcium was near the center of the range reported for lakes in Atlantic Canada, and higher than values reported for acidified lakes in south-western NS (Clair et al., 2007). Summarized results for alkalinity, metals and major cations data is presented in Appendix B.

Table 15 Alkalinity (mg/L $CaCO_3$) and Calcium (mg/L) means and ranges by sampling date

Sampling dates	**Alkalinity (mg/L $CaCO_3$)** ***mean (range)***	**Calcium (mg/L)** ***mean (range)***
August 2015	14 (11 – 25)	6.88 (6.46 – 7.36)
May 2016	8 (7 – 9)	6.18 (6.08 – 6.30)
September 2016	16 (14 – 18)	7.38 (7.33 – 7.47)
November 2016	12 (11 – 14)	6.19 (6.11 – 6.37)

Table 16 Lake sonde data: means and ranges for temperature, Dissolved Oxygen (DO), conductivity and pH measured at both Fletchers Lake basins: F stn 1 and F stn 2. Measurements were taken at least every 3 meters if the water column was not stratified, and every meter if the water column was stratified.

August 2015

	Temp. (°C)	DO (mg/L)	Cond. (µS/cm)	pH
F Stn 1				
Mean (Range)	22.8 (21.1-24.1)	7.7 (5.5-8.7)	176.3 (175-179)	6.4 (6.1-6.8)
F Stn 2				
Mean (Range)	22.1(17.8-24.7)	6.9 (3.4-8.7)	180 (177-197)	6.6 (6.3-6.8)

May 2016

	Temp. (°C)	DO (mg/L)	Cond. (µS/cm)	pH
F Stn 1				
Mean (Range)	10.67 (10.42-10.92)	11.73 (11.35-11.84)	159.6 (159-160)	6.51 (6.45-6.62)
F Stn 2				
Mean (Range)	10.37 (8.87-11.24)	11.74 (11.57-11.88)	162 (160-164)	6.66 (6.58-6.72)

September 2016

	Temp. (°C)	DO (mg/L)	Cond. (µS/cm)	pH
F Stn 1				
Mean (Range)	21.0 (21.0)	8.5 (8.3-8.5)	205 (205)	6.9 (6.8-6.9)
F Stn 2				
Mean (Range)	21.1 (20.7-21.2)	8.4 (7.8-8.5)	206.9 (206-209)	6.8 (6.7-6.9)

November 2016

	Temp. (°C)	DO (mg/L)	Cond. (µS/cm)	pH
F Stn 1				
Mean (Range)	8.4 (8.1-9.4)	11.18 (11.1-11.2)	161 (161)	6.82 (6.8-6.9)
F Stn 2				
Mean (Range)	8.9 (8.7-9.5)	11.2 (11.2)	165.2 (164-166)	6.8 (6.8)

4.6 Quality Assurance and Quality Control

Method and sampling triplicate means are plotted as bar graphs with 95 % confidence intervals in Appendix A. Turbidity method triplicates showed the greatest variability compared with TN and TOC method triplicates. FO and FD showed greatest sampling variability with the turbidity method, and FND showed greatest sampling variability with the TP method. All other sampling sites showed similar sampling variability results with other methods.

CHAPTER 5 CONCLUSION

In summary, baseline P export was measured within the Fletchers Lake watershed to calculate new P export coefficients for forested and low-density residential land use. A mass balance model was built for Fletchers Lake using measured P export from tributaries, measured P inputs and outputs at the lake inlet and outlet, as well as estimated P export from the rest of the watershed, WWTFs, atmospheric deposition and septic system contribution. The mass balance model identified the two largest sources of P to the lake system: the headwaters and septic system contribution. A trophic state analysis of Fletchers Lake characterized the current baseline TP, TN, SD, and Chl *a* within the lake as well as the DOC content, for referencing with post-development conditions. The lake is currently oligotrophic (TP, TN, Chl *a*), which is consistent with historical trophic state. Secchi depth measured as mesotrophic, likely due to high humic content with DOC > 2 mg/L (Nurnberg, 1999).

The measured P export coefficient for low density residential land was much lower than the commonly used value of 52 mg/m^2/yr (Waller, 1977), and forested P export coefficients showed a variability between subwatersheds with different dominant tree species, and with wetland components. The large differences between the measured export coefficient values in this study, and those commonly used from the literature, emphasizes the importance of detailed subwatershed land use description, and the selection of appropriate export coefficients for the different land use components in a watershed. Additionally, it has been shown how greatly P export can vary between low-density and high-density residential land. The measured P export coefficients from this study can be used in future P modelling studies to improve the accuracy of estimated P export from forested and low-density residential land types with similar bedrock geology and soil characteristics, which were characterized as a mix of granite, and the Halifax Formation consisting of slate, silt, and sandstone (Keppie, 2000), and thin sandy clay loam Rockland soil (MacDOugall et al., 1963; Jacques-Whitford, 2009).

Nutrient loading data from measured streams displayed opposite seasonal trends between small streams and the inlet and outlet, as well as trends with precipitation events and high flow periods.

The mass balance results showed net sedimentation of P within the lake of 391.3 kg; however, the soil retention coefficients were found to have a large influence on these results. As a result of the uncertainties around septic system P loading, the septic system contribution seems to be the largest un-known in the mass balance model. Additionally, it is likely this P component will increase with the age of septic systems, as soils become saturated with P, and as septic plumes migrate toward waterways (Sinclair et al., 2014; Paterson et al., 2006).

The trophic analysis of Fletchers Lake exemplified a dystrophic-oligotrophic lake, where SD is misrepresentative of the true lake productivity, and stresses the importance of the inclusion of DOC as an additional trophic parameter, or for the direct measurement of primary productivity through Chl *a* for the assessment of lakes in NS. Lastly, a comparison of surface TP values measured in this study against a data set collected over 2007 – 2011 (Stantec, 2012) showed a significant decrease in surface TP from 2007-2011 to 2015-2016.

5.1 Recommendations for Future Research

It is recommended that future P modelling studies conducted in NS use density-specific residential P export coefficients such as those measured in this study. It was found that P export from different residential densities can largely vary; 9.1 mg /m^2/yr measured in low-density development in Fall River, to 52 mg /m^2/yr measured in urban Halifax (Waller 1977). As the total precipitation for the sampled year was above average for the area compared with the average from 1981 - 2010 (Environment Canada, 2011), further studies to validate the measured export coefficients would be useful. More research on residential P export in NS is also recommended, as the number of studies examining P export from this land use type is limited, and there currently is no measured export coefficient for medium-density residential land in NS. Further research is also recommended to investigate differing P export between dominant tree species, particularly between softwood and hardwood, as historical ranges of P export in Reckhow (1980), and measured P export in this study, suggest large differences between tree species P export. The sampling design of future work should span at least one year to capture seasonal nutrient loading trends, and should focus on obtaining baseline data plus storm and snowmelt events to capture the variation of loading response to different precipitation and flow conditions.

With regard to analyzing trophic state in HRM and generally in NS, additional indices are recommended for inclusion in trophic state analysis studies and with regard to lake management. Chl *a* is the most direct measure of primary productivity, and its measure alone, can help eliminate the problem of influencers (such as DOC and alkalinity) on the TP: Chl *a* relationship (Lyche-Solheim et al., 2013; Carvalho et al., 2013). This is particularly important in NS where lakes have high humic contents and variable alkalinity (Underwood 1986; Clair et al., 2007). Further studies should include thorough Chl *a* sampling in their data collection design to obtain the most accurate and direct measure of trophic state in the province's lakes.

Furthermore, as the contribution of septic system P loading to Fletchers Lake was found to be a potentially significant component of total loading, and as this source still remains a large uncertainty in the P modelling process, it should be further investigated.

In order to improve the comparison of future data sets, it must be stressed that sampling designs for all trophic indexes are thorough and consistent for each measured parameter. Additionally, the same methodologies for laboratory measurements should be applied for consistency and to facilitate easy comparisons between results and identify current and future trends. The same consistency should be applied to P export coefficient studies with regard to sampling frequency and focus on storm and snowmelt events.

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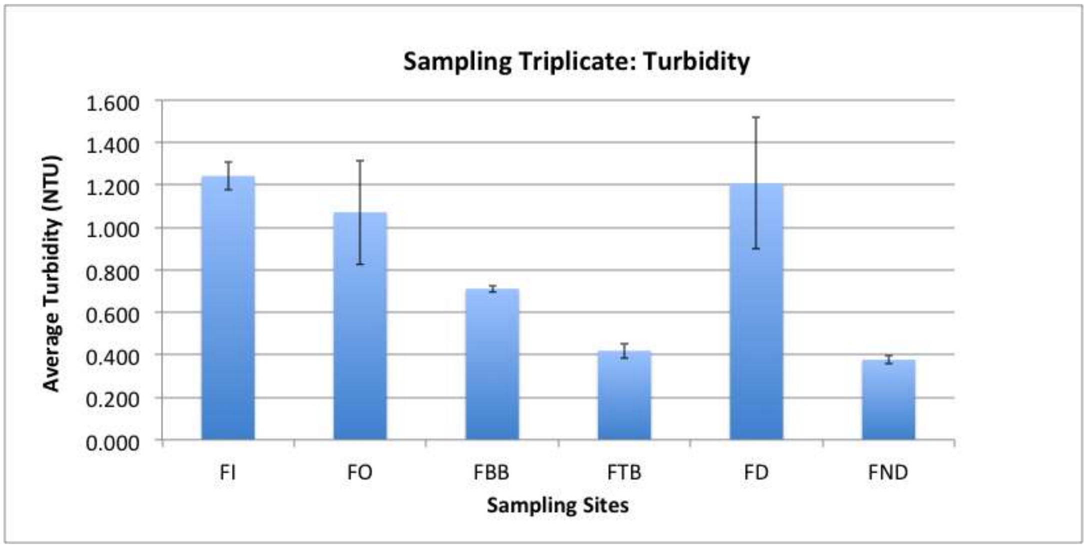
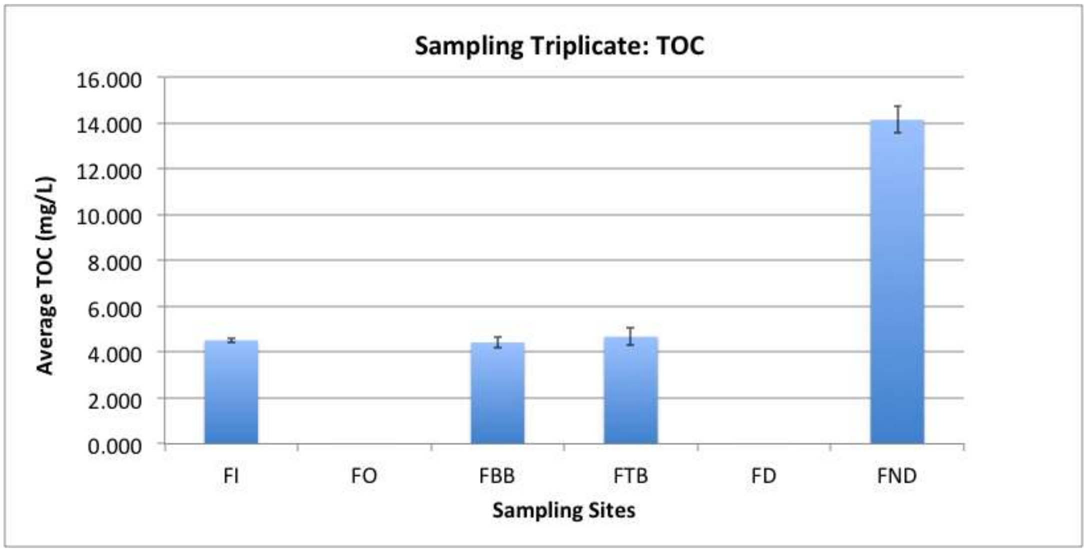
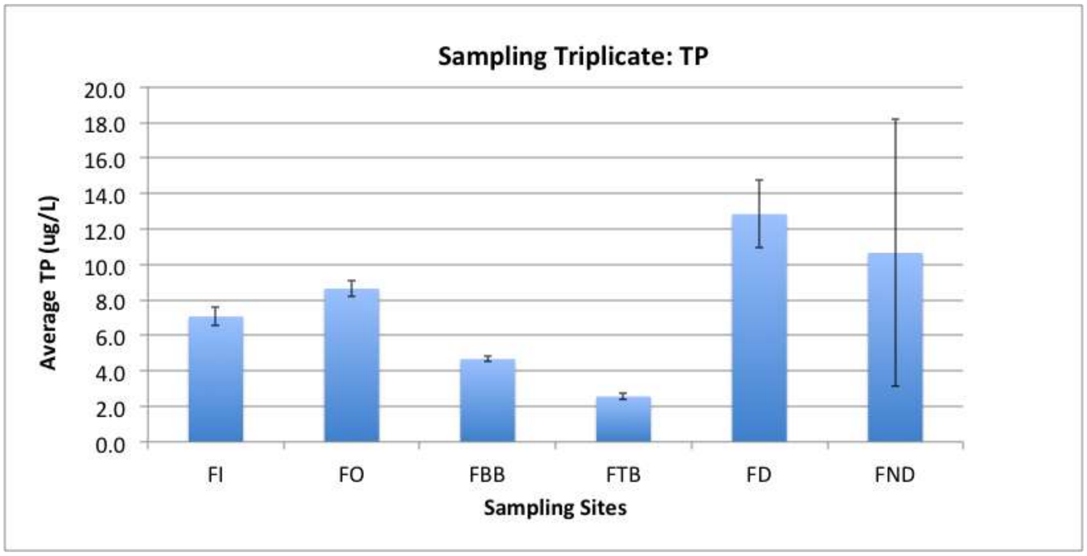
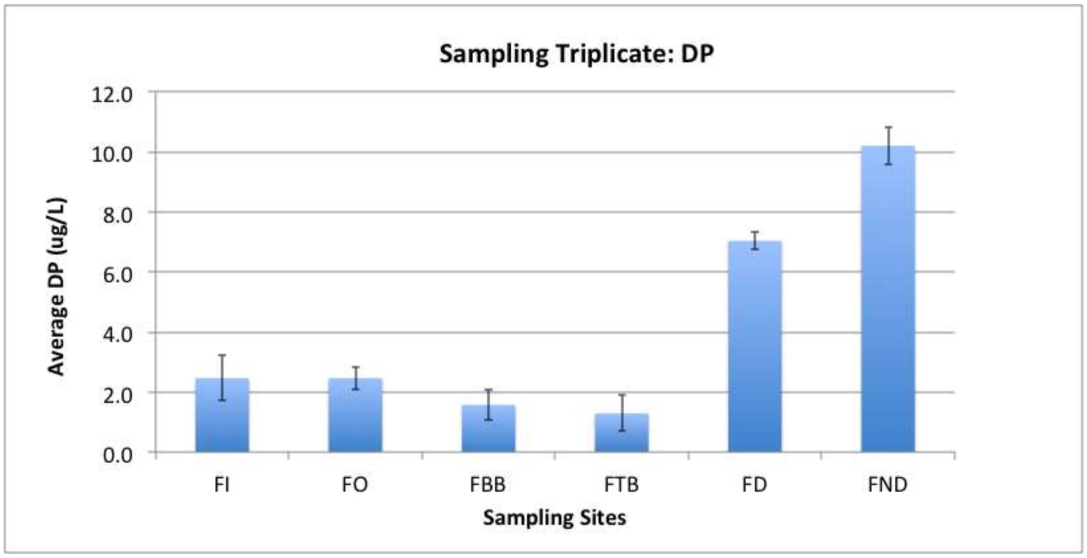
APPENDIX A

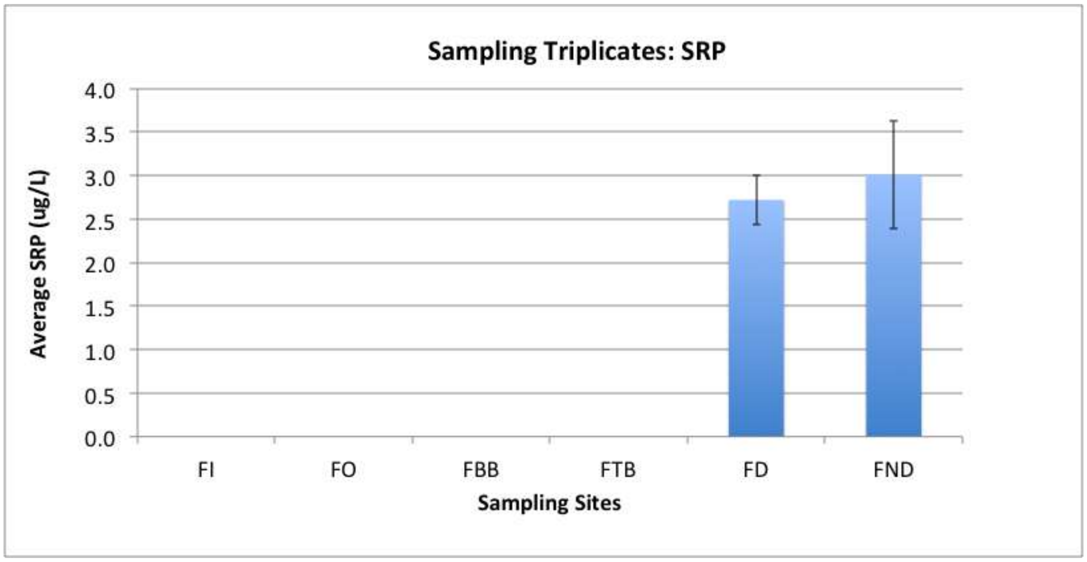
Sampling triplicate means for each water quality paramter are shown below with 95 % conifdence intervals. Missing sample site values were a result of sample loss (TOC), and values below the detection limit (SRP).



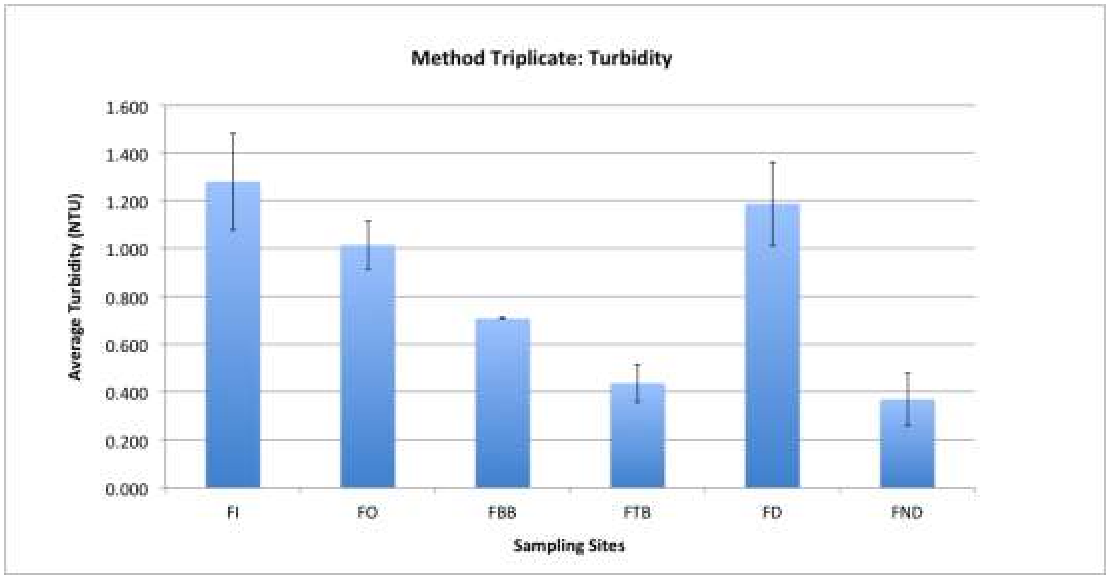


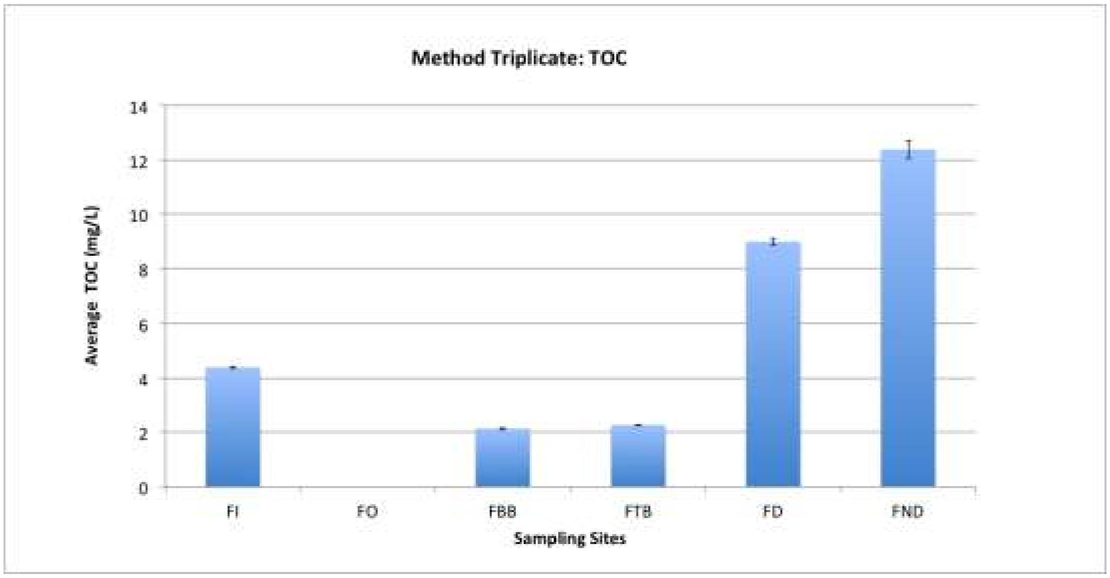

Sampling Triplicate: TP
Average TP (ug/L)
0.0
2.0
4.0
6.0
8.0
10.0
12.0
14.0
16.0
18.0
20.0
FI
FO
FBB
FTB
FD
FND
Sampling Sites

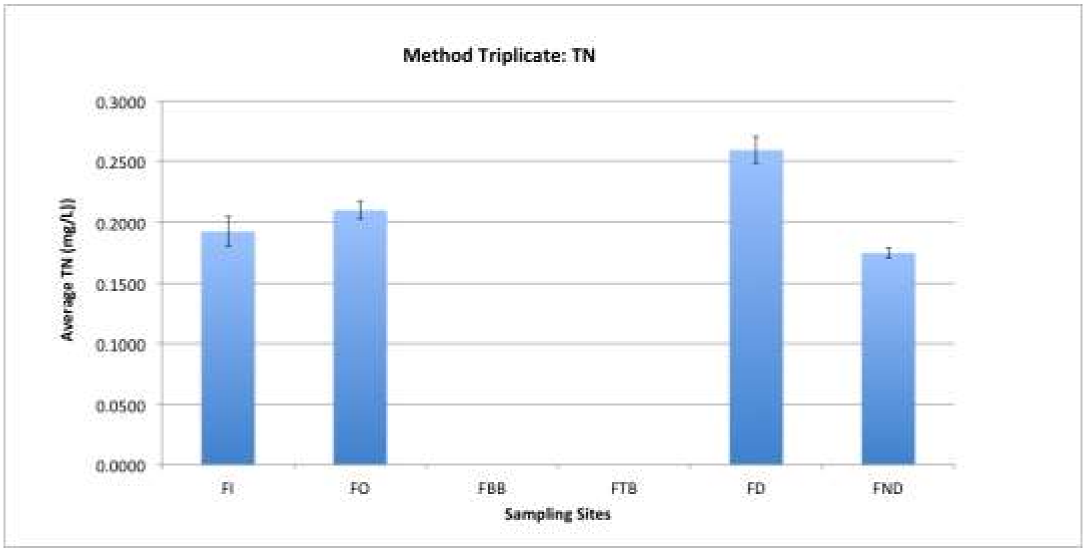

Sampling Triplicate: DP
Average DP (ug/L)
0.0
2.0
4.0
6.0
8.0
10.0
12.0
FI
FO
FBB
FTB
FD
FND
Sampling Sites


Sampling Triplicates: SRP
Average SRP (ug/L)
0.0
0.5
1.0
1.5
2.0
2.5
3.0
3.5
4.0
FI
FO
FBB
FTB
FD
FND
Sampling Sites

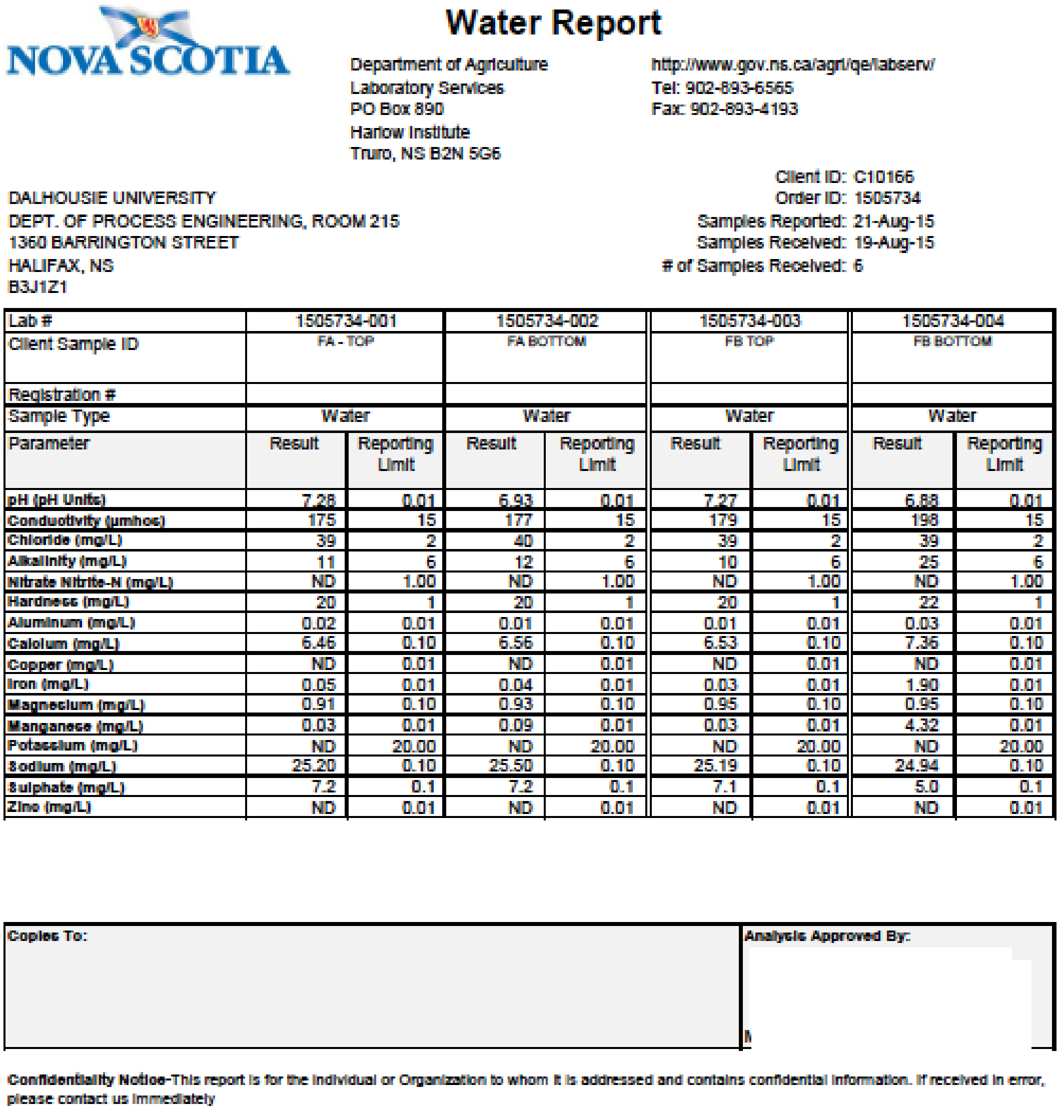
Method triplicate means for each water quality parameter shown below with 95% confidence intervals. Missing triplicate means were a result of sample loss (TOC), and values below the detection limit (TN).






Method Triplicate: TN
0.3000
0.2500
0.2000
0.1500
0.1000
0.0500
0.0000
Average TN (mg/L)
FI
FO
FBB
FTB
FD
FND
Sampling Sites

APPENDIX B



Water Report

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DALHOUSIE UNIVERSITY
DEPT. OF PROCESS ENGINEERING, ROOM 215
1360 BARRINGTON STREET
HALIFAX, NS
B3J1Z1

Client ID: C10166
Order ID: 1505734
Samples Reported: 21-Aug-15
Samples Received: 19-Aug-15
# of Samples Received: 6

Lab #	1505734-001		1505734-002		1505734-003		1505734-004	
Client Sample ID	FA - TOP		FA BOTTOM		FB TOP		FB BOTTOM	
Registration #								
Sample Type	Water		Water		Water		Water	
Parameter	Result	Reporting Limit	Result	Reporting Limit	Result	Reporting Limit	Result	Reporting Limit
pH (pH Units)	7.28	0.01	6.93	0.01	7.27	0.01	6.88	0.01
Conductivity (µmhos)	175	15	177	15	179	15	198	15
Chloride (mg/L)	39	2	40	2	39	2	39	2
Alkalinity (mg/L)	11	6	12	6	10	6	25	6
Nitrate Nitrite-N (mg/L)	ND	1.00	ND	1.00	ND	1.00	ND	1.00
Hardness (mg/L)	20	1	20	1	20	1	22	1
Aluminum (mg/L)	0.02	0.01	0.01	0.01	0.01	0.01	0.03	0.01
Calcium (mg/L)	6.46	0.10	6.56	0.10	6.53	0.10	7.36	0.10
Copper (mg/L)	ND	0.01	ND	0.01	ND	0.01	ND	0.01
Iron (mg/L)	0.05	0.01	0.04	0.01	0.03	0.01	1.90	0.01
Magnesium (mg/L)	0.91	0.10	0.93	0.10	0.95	0.10	0.95	0.10
Manganese (mg/L)	0.03	0.01	0.09	0.01	0.03	0.01	4.32	0.01
Potassium (mg/L)	ND	20.00	ND	20.00	ND	20.00	ND	20.00
Sodium (mg/L)	25.20	0.10	25.50	0.10	25.19	0.10	24.94	0.10
Sulphate (mg/L)	7.2	0.1	7.2	0.1	7.1	0.1	5.0	0.1
Zinc (mg/L)	ND	0.01	ND	0.01	ND	0.01	ND	0.01

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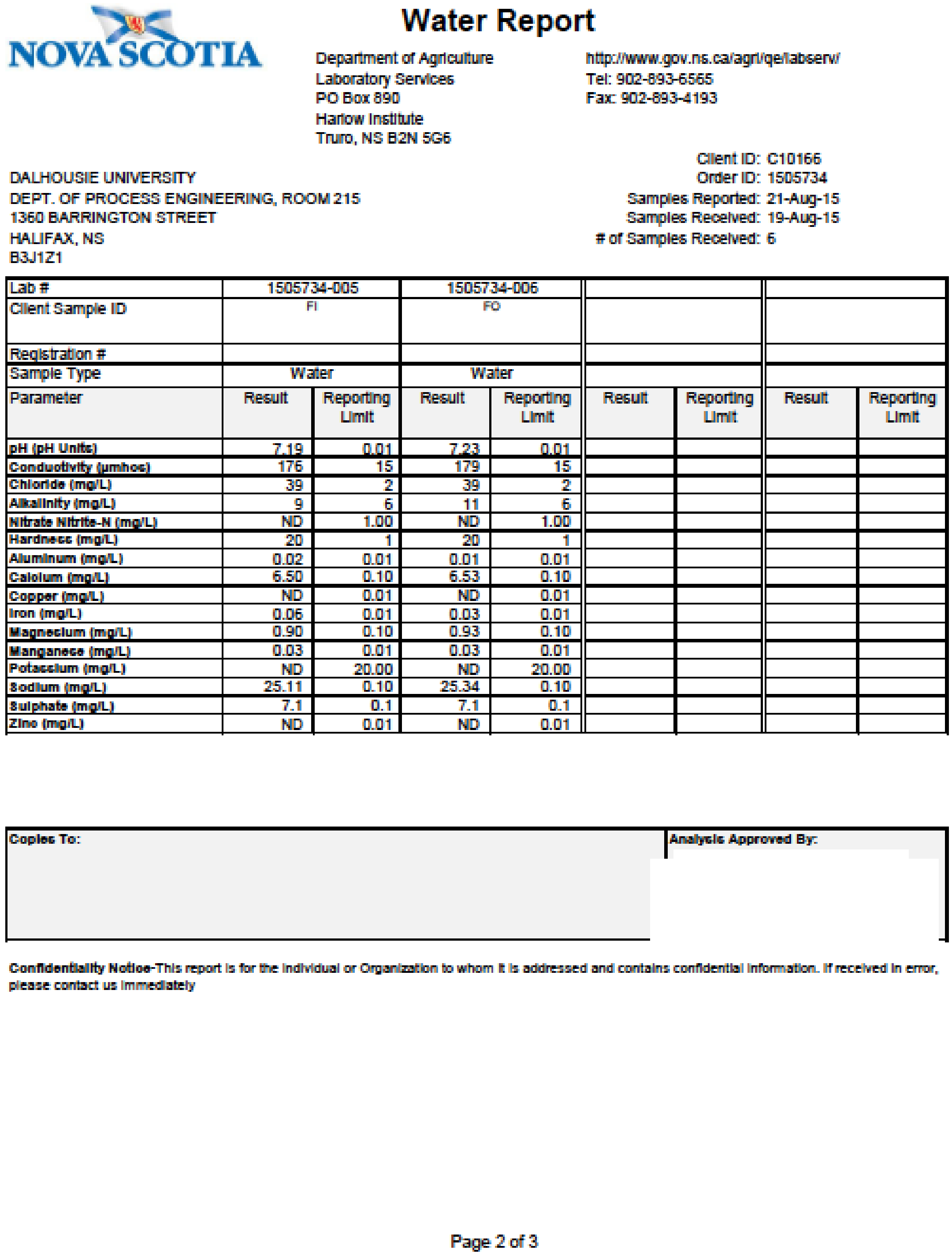

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DALHOUSIE UNIVERSITY
DEPT. OF PROCESS ENGINEERING, ROOM 215
1360 BARRINGTON STREET
HALIFAX, NS
B3J1Z1

Client ID: C10166
Order ID: 1505734
Samples Reported: 21-Aug-15
Samples Received: 19-Aug-15
# of Samples Received: 6

Lab #	1505734-005		1505734-006					
Client Sample ID	FI		FO					
Registration #								
Sample Type	Water		Water					
Parameter	Result	Reporting Limit	Result	Reporting Limit	Result	Reporting Limit	Result	Reporting Limit
pH (pH Units)	7.19	0.01	7.23	0.01				
Conductivity (µmhos)	176	15	179	15				
Chloride (mg/L)	39	2	39	2				
Alkalinity (mg/L)	9	6	11	6				
Nitrate Nitrite-N (mg/L)	ND	1.00	ND	1.00				
Hardness (mg/L)	20	1	20	1				
Aluminum (mg/L)	0.02	0.01	0.01	0.01				
Calcium (mg/L)	6.50	0.10	6.53	0.10				
Copper (mg/L)	ND	0.01	ND	0.01				
Iron (mg/L)	0.06	0.01	0.03	0.01				
Magnesium (mg/L)	0.90	0.10	0.93	0.10				
Manganese (mg/L)	0.03	0.01	0.03	0.01				
Potassium (mg/L)	ND	20.00	ND	20.00				
Sodium (mg/L)	25.11	0.10	25.34	0.10				
Sulphate (mg/L)	7.1	0.1	7.1	0.1				
Zinc (mg/L)	ND	0.01	ND	0.01				

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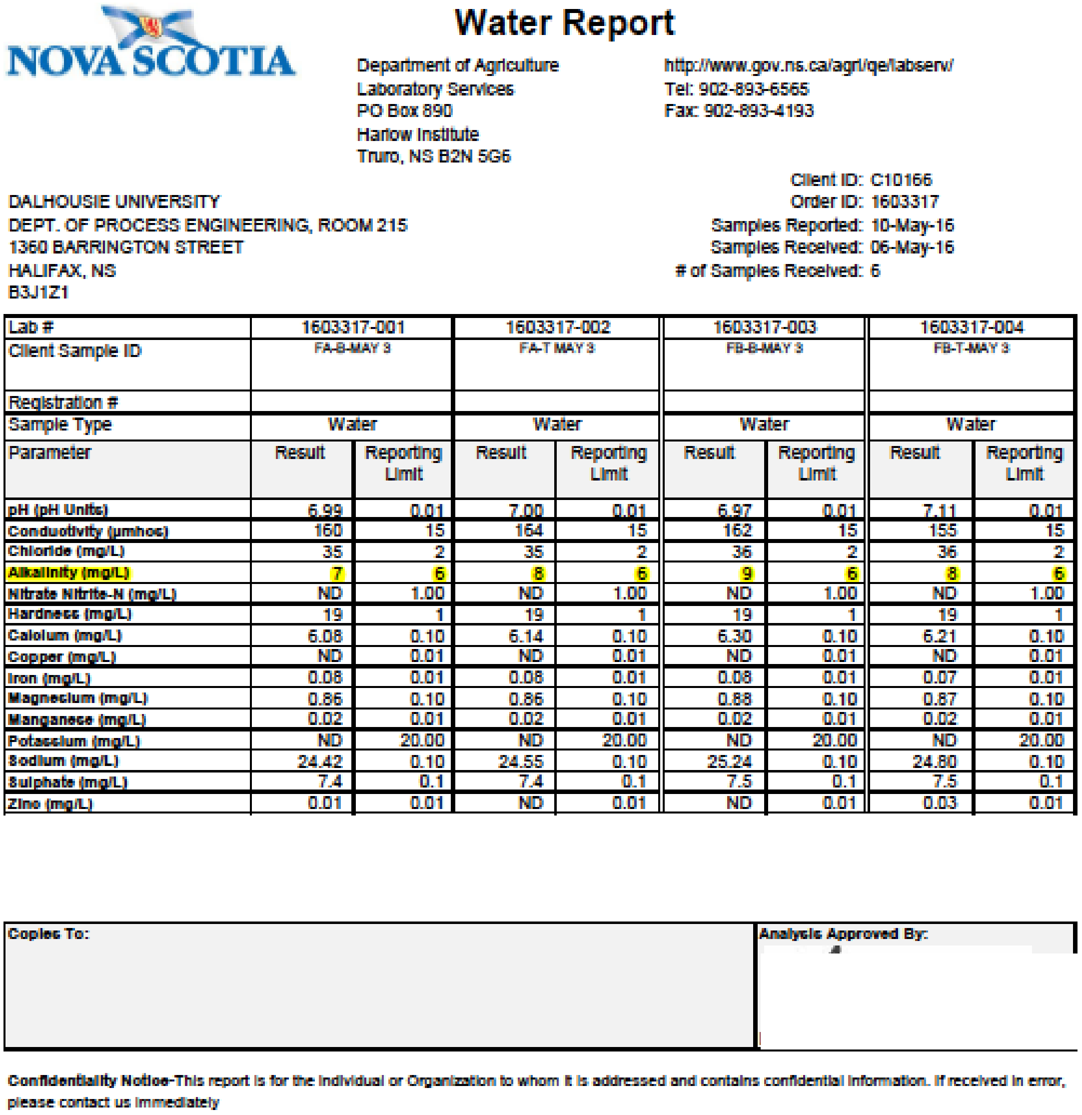

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HALIFAX, NS
B3J1Z1

Client ID: C10166
Order ID: 1603317
Samples Reported: 10-May-16
Samples Received: 06-May-16
# of Samples Received: 6

Lab #	1603317-001		1603317-002		1603317-003		1603317-004	
Client Sample ID	FA-B-MAY 3		FA-T MAY 3		FB-B-MAY 3		FB-T-MAY 3	
Registration #								
Sample Type	Water		Water		Water		Water	
Parameter	Result	Reporting Limit	Result	Reporting Limit	Result	Reporting Limit	Result	Reporting Limit
pH (pH Units)	6.99	0.01	7.00	0.01	6.97	0.01	7.11	0.01
Conductivity (µmhos)	160	15	164	15	162	15	155	15
Chloride (mg/L)	35	2	35	2	36	2	36	2
Alkalinity (mg/L)	7	6	8	6	9	6	8	6
Nitrate Nitrite-N (mg/L)	ND	1.00	ND	1.00	ND	1.00	ND	1.00
Hardness (mg/L)	19	1	19	1	19	1	19	1
Calcium (mg/L)	6.08	0.10	6.14	0.10	6.30	0.10	6.21	0.10
Copper (mg/L)	ND	0.01	ND	0.01	ND	0.01	ND	0.01
Iron (mg/L)	0.08	0.01	0.08	0.01	0.08	0.01	0.07	0.01
Magnesium (mg/L)	0.86	0.10	0.86	0.10	0.88	0.10	0.87	0.10
Manganese (mg/L)	0.02	0.01	0.02	0.01	0.02	0.01	0.02	0.01
Potassium (mg/L)	ND	20.00	ND	20.00	ND	20.00	ND	20.00
Sodium (mg/L)	24.42	0.10	24.55	0.10	25.24	0.10	24.80	0.10
Sulphate (mg/L)	7.4	0.1	7.4	0.1	7.5	0.1	7.5	0.1
Zinc (mg/L)	0.01	0.01	ND	0.01	ND	0.01	0.03	0.01

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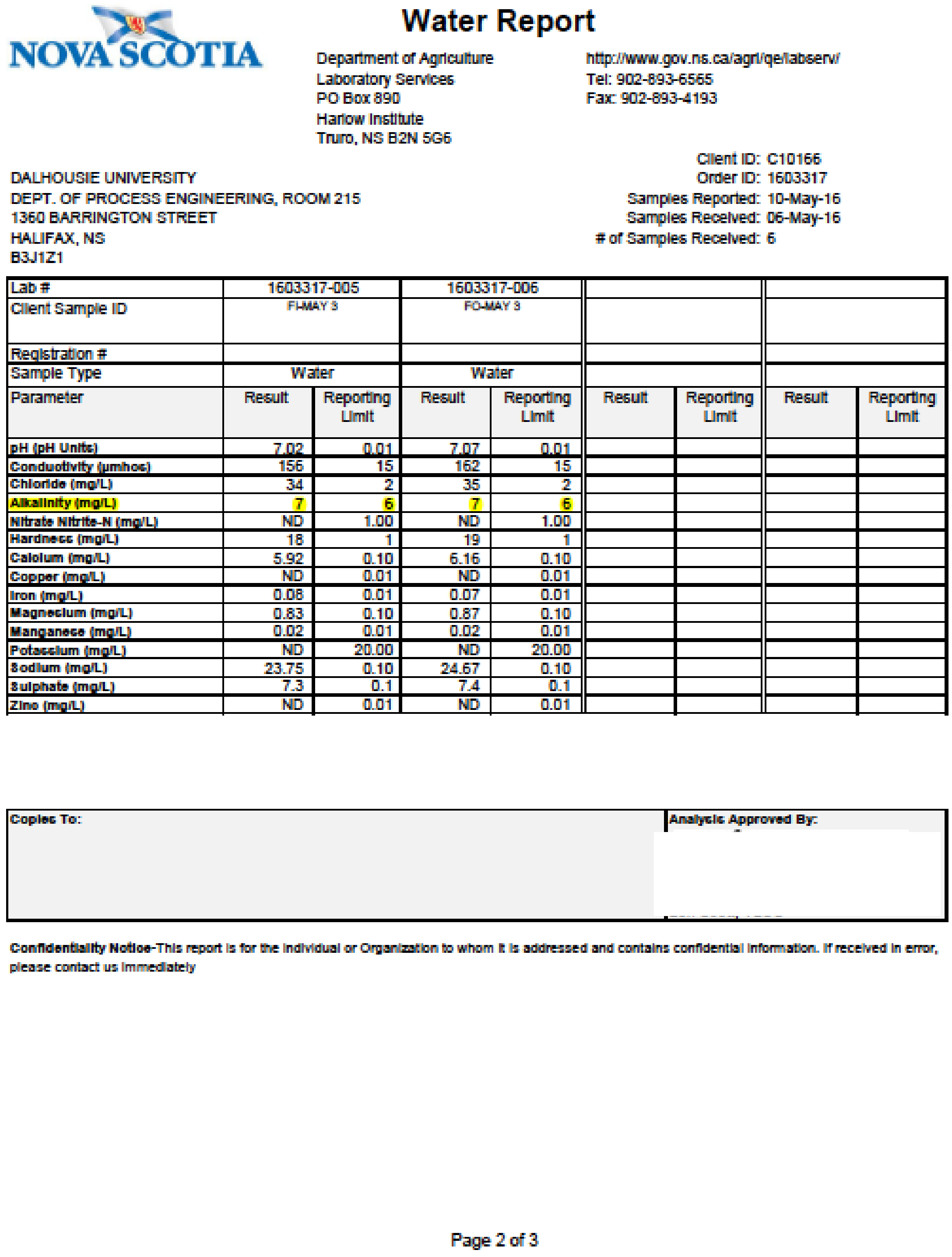

Water Report

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1360 BARRINGTON STREET
HALIFAX, NS
B3J1Z1

Client ID: C10166
Order ID: 1603317
Samples Reported: 10-May-16
Samples Received: 06-May-16
# of Samples Received: 6

Lab #	1603317-005		1603317-006					
Client Sample ID	FI-MAY 3		FO-MAY 3					
Registration #								
Sample Type	Water		Water					
Parameter	Result	Reporting Limit	Result	Reporting Limit	Result	Reporting Limit	Result	Reporting Limit
pH (pH Units)	7.02	0.01	7.07	0.01				
Conductivity (µmhos)	156	15	162	15				
Chloride (mg/L)	34	2	35	2				
Alkalinity (mg/L)	7	6	7	6				
Nitrate Nitrite-N (mg/L)	ND	1.00	ND	1.00				
Hardness (mg/L)	18	1	19	1				
Calcium (mg/L)	5.92	0.10	6.16	0.10				
Copper (mg/L)	ND	0.01	ND	0.01				
Iron (mg/L)	0.08	0.01	0.07	0.01				
Magnesium (mg/L)	0.83	0.10	0.87	0.10				
Manganese (mg/L)	0.02	0.01	0.02	0.01				
Potassium (mg/L)	ND	20.00	ND	20.00				
Sodium (mg/L)	23.75	0.10	24.67	0.10				
Sulphate (mg/L)	7.3	0.1	7.4	0.1				
Zinc (mg/L)	ND	0.01	ND	0.01				

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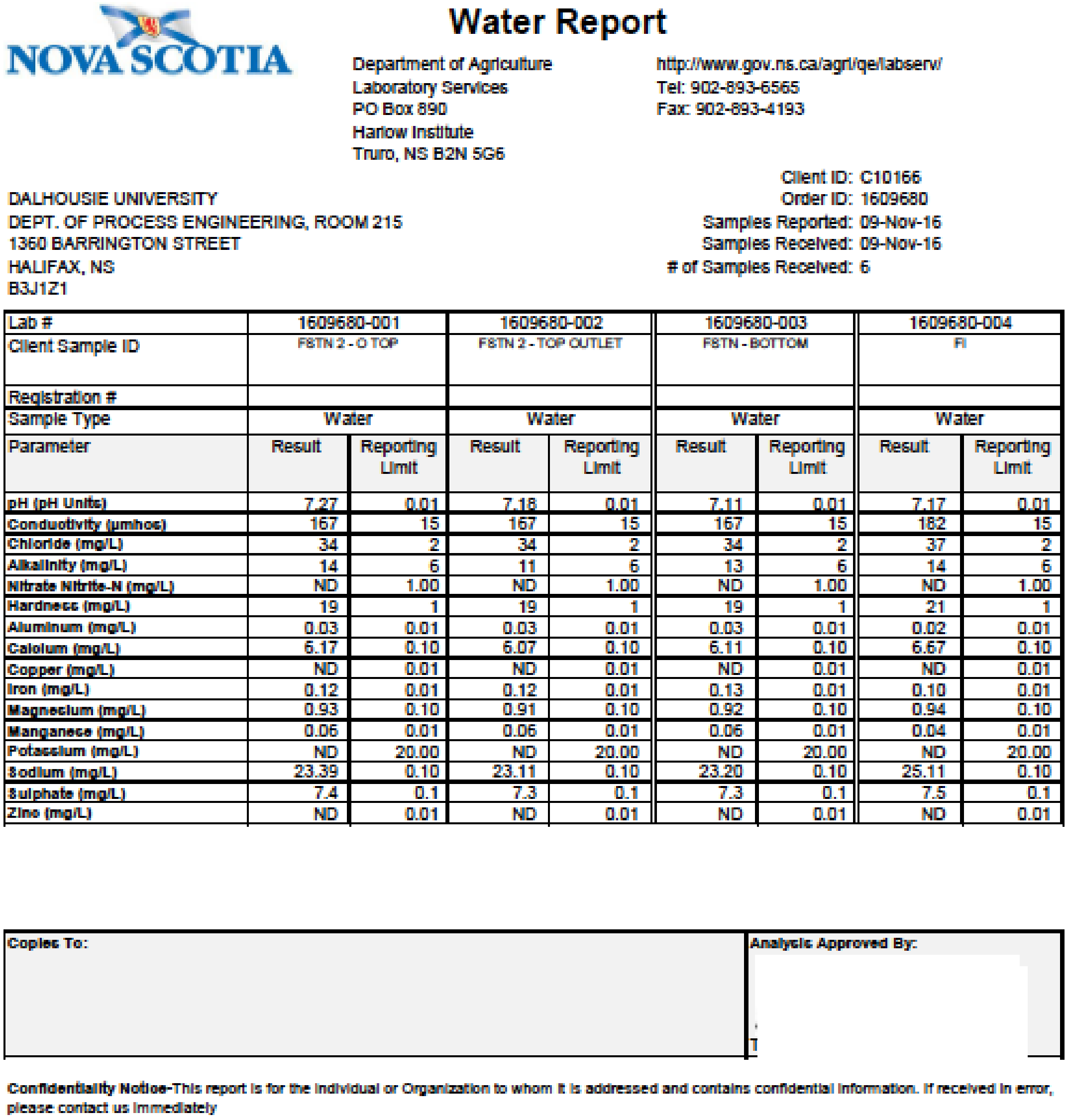

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Fax: 902-893-4193

Client ID: C10166
Order ID: 1609680
Samples Reported: 09-Nov-16
Samples Received: 09-Nov-16
# of Samples Received: 6

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1360 BARRINGTON STREET
HALIFAX, NS
B3J1Z1

Lab #	1609680-001		1609680-002		1609680-003		1609680-004	
Client Sample ID	FSTN 2 - O TOP		FSTN 2 - TOP OUTLET		FSTN - BOTTOM		FI	
Registration #								
Sample Type	Water		Water		Water		Water	
Parameter	Result	Reporting Limit	Result	Reporting Limit	Result	Reporting Limit	Result	Reporting Limit
pH (pH Units)	7.27	0.01	7.18	0.01	7.11	0.01	7.17	0.01
Conductivity (µmhos)	167	15	167	15	167	15	182	15
Chloride (mg/L)	34	2	34	2	34	2	37	2
Alkalinity (mg/L)	14	6	11	6	13	6	14	6
Nitrate Nitrite-N (mg/L)	ND	1.00	ND	1.00	ND	1.00	ND	1.00
Hardness (mg/L)	19	1	19	1	19	1	21	1
Aluminum (mg/L)	0.03	0.01	0.03	0.01	0.03	0.01	0.02	0.01
Calcium (mg/L)	6.17	0.10	6.07	0.10	6.11	0.10	6.67	0.10
Copper (mg/L)	ND	0.01	ND	0.01	ND	0.01	ND	0.01
Iron (mg/L)	0.12	0.01	0.12	0.01	0.13	0.01	0.10	0.01
Magnesium (mg/L)	0.93	0.10	0.91	0.10	0.92	0.10	0.94	0.10
Manganese (mg/L)	0.06	0.01	0.06	0.01	0.06	0.01	0.04	0.01
Potassium (mg/L)	ND	20.00	ND	20.00	ND	20.00	ND	20.00
Sodium (mg/L)	23.39	0.10	23.11	0.10	23.20	0.10	25.11	0.10
Sulphate (mg/L)	7.4	0.1	7.3	0.1	7.3	0.1	7.5	0.1
Zinc (mg/L)	ND	0.01	ND	0.01	ND	0.01	ND	0.01

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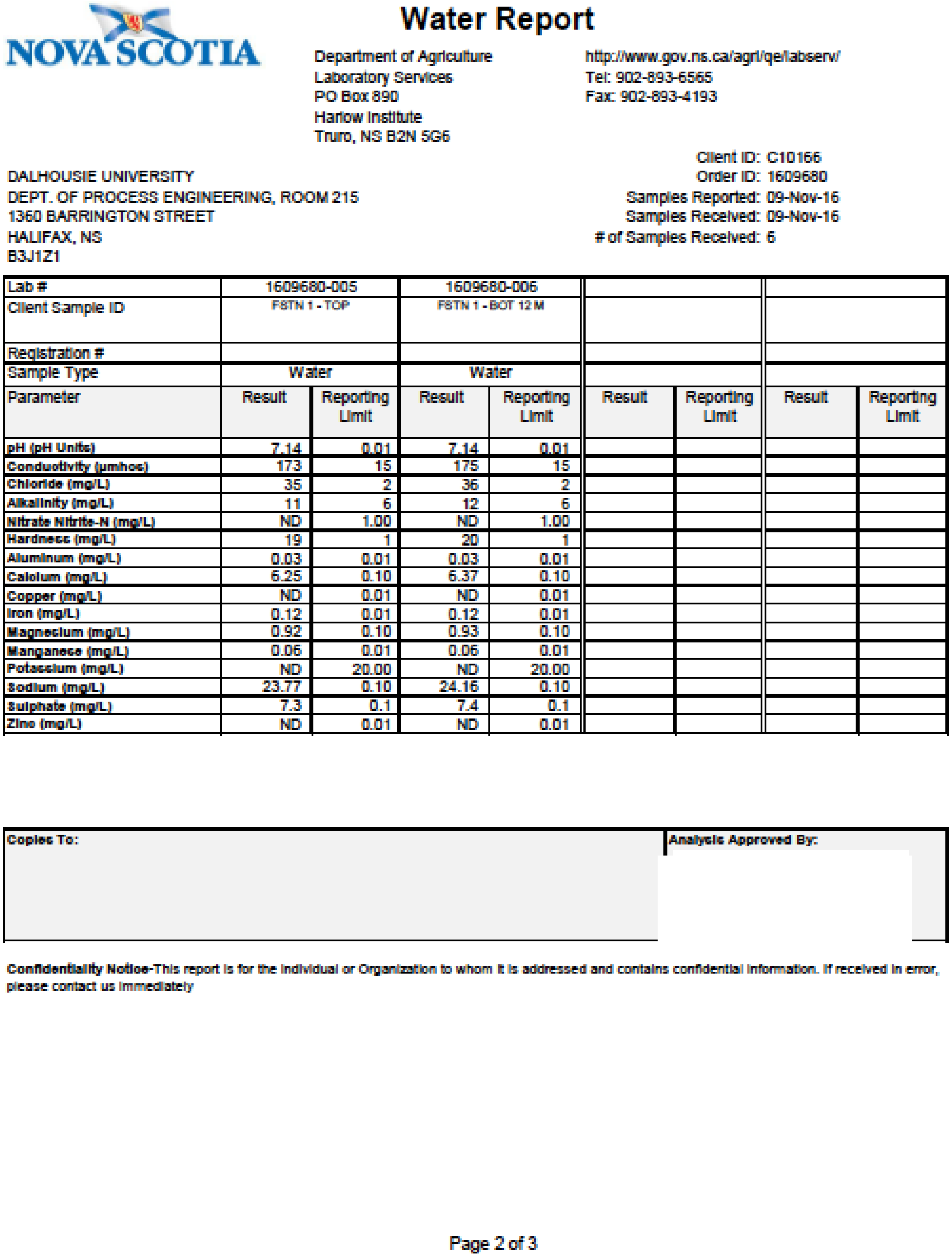

Water Report

NOVA SCOTIA

Department of Agriculture
Laboratory Services
PO Box 890
Harlow Institute
Truro, NS B2N 5G6

http://www.gov.ns.ca/agri/qe/labserv/
Tel: 902-893-6565
Fax: 902-893-4193

DALHOUSIE UNIVERSITY
DEPT. OF PROCESS ENGINEERING, ROOM 215
1360 BARRINGTON STREET
HALIFAX, NS
B3J1Z1

Client ID: C10166
Order ID: 1609680
Samples Reported: 09-Nov-16
Samples Received: 09-Nov-16
# of Samples Received: 6

Lab #	1609680-005		1609680-006					
Client Sample ID	F8TN 1 - TOP		F8TN 1 - BOT 12 M					
Registration #								
Sample Type	Water		Water					
Parameter	Result	Reporting Limit	Result	Reporting Limit	Result	Reporting Limit	Result	Reporting Limit
pH (pH Units)	7.14	0.01	7.14	0.01				
Conductivity (µmhos)	173	15	175	15				
Chloride (mg/L)	35	2	36	2				
Alkalinity (mg/L)	11	6	12	6				
Nitrate Nitrite-N (mg/L)	ND	1.00	ND	1.00				
Hardness (mg/L)	19	1	20	1				
Aluminum (mg/L)	0.03	0.01	0.03	0.01				
Calcium (mg/L)	6.25	0.10	6.37	0.10				
Copper (mg/L)	ND	0.01	ND	0.01				
Iron (mg/L)	0.12	0.01	0.12	0.01				
Magnesium (mg/L)	0.92	0.10	0.93	0.10				
Manganese (mg/L)	0.06	0.01	0.06	0.01				
Potassium (mg/L)	ND	20.00	ND	20.00				
Sodium (mg/L)	23.77	0.10	24.16	0.10				
Sulphate (mg/L)	7.3	0.1	7.4	0.1				
Zinc (mg/L)	ND	0.01	ND	0.01				

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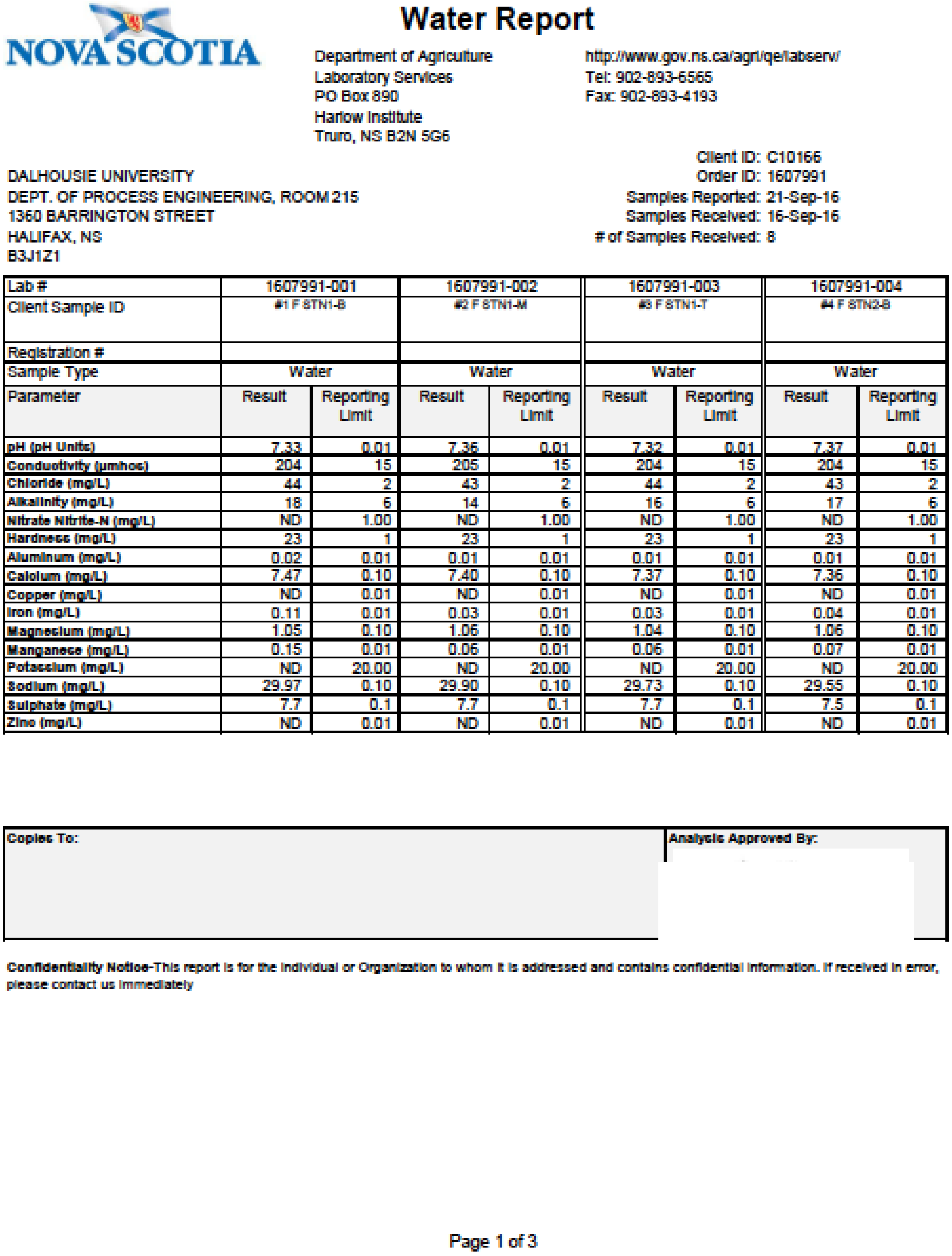

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DEPT. OF PROCESS ENGINEERING, ROOM 215
1360 BARRINGTON STREET
HALIFAX, NS
B3J1Z1

Client ID: C10166
Order ID: 1607991
Samples Reported: 21-Sep-16
Samples Received: 16-Sep-16
# of Samples Received: 8

Lab #	1607991-001		1607991-002		1607991-003		1607991-004	
Client Sample ID	#1 F STN1-B		#2 F STN1-M		#3 F STN1-T		#4 F STN2-B	
Registration #								
Sample Type	Water		Water		Water		Water	
Parameter	Result	Reporting Limit	Result	Reporting Limit	Result	Reporting Limit	Result	Reporting Limit
pH (pH Units)	7.33	0.01	7.36	0.01	7.32	0.01	7.37	0.01
Conductivity (µmhos)	204	15	205	15	204	15	204	15
Chloride (mg/L)	44	2	43	2	44	2	43	2
Alkalinity (mg/L)	18	6	14	6	16	6	17	6
Nitrate Nitrite-N (mg/L)	ND	1.00	ND	1.00	ND	1.00	ND	1.00
Hardness (mg/L)	23	1	23	1	23	1	23	1
Aluminum (mg/L)	0.02	0.01	0.01	0.01	0.01	0.01	0.01	0.01
Calcium (mg/L)	7.47	0.10	7.40	0.10	7.37	0.10	7.36	0.10
Copper (mg/L)	ND	0.01	ND	0.01	ND	0.01	ND	0.01
Iron (mg/L)	0.11	0.01	0.03	0.01	0.03	0.01	0.04	0.01
Magnesium (mg/L)	1.05	0.10	1.06	0.10	1.04	0.10	1.06	0.10
Manganese (mg/L)	0.15	0.01	0.06	0.01	0.06	0.01	0.07	0.01
Potassium (mg/L)	ND	20.00	ND	20.00	ND	20.00	ND	20.00
Sodium (mg/L)	29.97	0.10	29.90	0.10	29.73	0.10	29.55	0.10
Sulphate (mg/L)	7.7	0.1	7.7	0.1	7.7	0.1	7.5	0.1
Zinc (mg/L)	ND	0.01	ND	0.01	ND	0.01	ND	0.01

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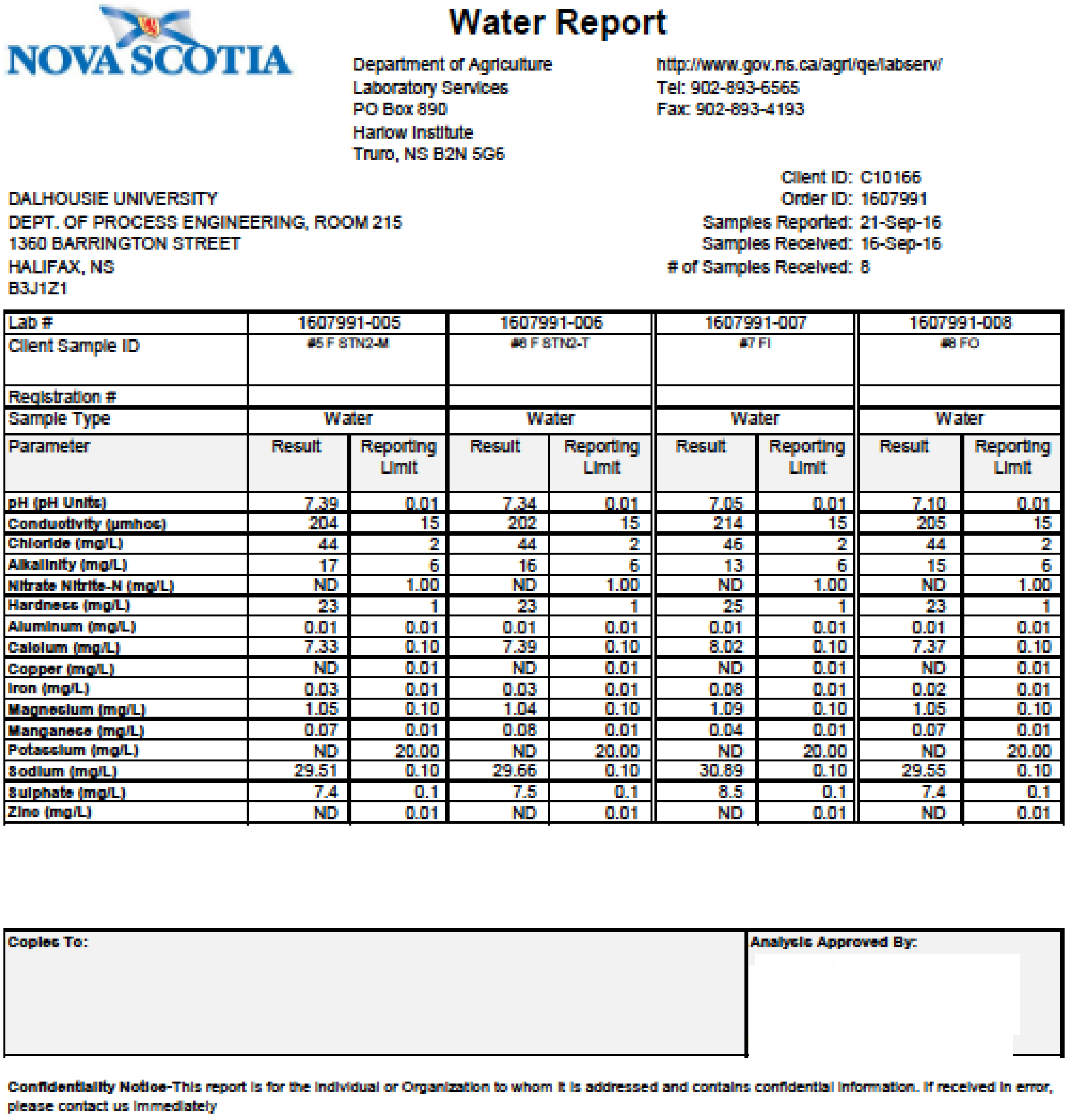

Water Report

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http://www.gov.ns.ca/agri/qe/labserv/
Tel: 902-893-6565
Fax: 902-893-4193

Client ID: C10166
Order ID: 1607991
Samples Reported: 21-Sep-16
Samples Received: 16-Sep-16
# of Samples Received: 8

DALHOUSIE UNIVERSITY
DEPT. OF PROCESS ENGINEERING, ROOM 215
1360 BARRINGTON STREET
HALIFAX, NS
B3J1Z1

Lab #	1607991-005		1607991-006		1607991-007		1607991-008	
Client Sample ID	#5 F STN2-M		#6 F STN2-T		#7 FI		#8 FO	
Registration #								
Sample Type	Water		Water		Water		Water	
Parameter	Result	Reporting Limit	Result	Reporting Limit	Result	Reporting Limit	Result	Reporting Limit
pH (pH Units)	7.39	0.01	7.34	0.01	7.05	0.01	7.10	0.01
Conductivity (µmhos)	204	15	202	15	214	15	205	15
Chloride (mg/L)	44	2	44	2	46	2	44	2
Alkalinity (mg/L)	17	6	16	6	13	6	15	6
Nitrate Nitrite-N (mg/L)	ND	1.00	ND	1.00	ND	1.00	ND	1.00
Hardness (mg/L)	23	1	23	1	25	1	23	1
Aluminum (mg/L)	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01
Calcium (mg/L)	7.33	0.10	7.39	0.10	8.02	0.10	7.37	0.10
Copper (mg/L)	ND	0.01	ND	0.01	ND	0.01	ND	0.01
Iron (mg/L)	0.03	0.01	0.03	0.01	0.08	0.01	0.02	0.01
Magnesium (mg/L)	1.05	0.10	1.04	0.10	1.09	0.10	1.05	0.10
Manganese (mg/L)	0.07	0.01	0.08	0.01	0.04	0.01	0.07	0.01
Potassium (mg/L)	ND	20.00	ND	20.00	ND	20.00	ND	20.00
Sodium (mg/L)	29.51	0.10	29.66	0.10	30.89	0.10	29.55	0.10
Sulphate (mg/L)	7.4	0.1	7.5	0.1	8.5	0.1	7.4	0.1
Zinc (mg/L)	ND	0.01	ND	0.01	ND	0.01	ND	0.01

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APPENDIX C

Lockview-MacPherson WWTF **Water Quality Data 2015-2016**

Date dd-mm-yyyy	TSS (mg/L)	Turbidity (NTU)	TP (mg/L)	DP (mg/L)	SRP (mg/L)	TOC (mg/L)	TN (mg/L)
22-10-2015	63.50					12.844	32.78
03-12-2015	6.11		0.585			7.072	31.98
07-01-2016	3.90		0.391	0.093	0.1		
18-02-2016	3.18		0.405			5.524	17.16
02-03-2016	7.33		0.660				
17-03-2016	3.14						
31-03-2016	0.94		0.117				
11-04-2016	1.81		0.1455				
25-04-2016							19.20
09-05-2016	1.80		0.212			5.349	21.47
25-05-2016						6.758	34.14
08-06-2016	1.10	0.806	0.273			7.032	17.51
24-06-2016	3.02	4.273	1.055	0.441	0.3	9.961	14.85
06-07-2016	2.67	1.130	0.555	0.284	0.2	7.603	18.19
21-07-2016		7.303				27.400	29.07
03-08-2016		2.020				7.449	25.64
16-08-2016		5.500				11.720	20.55
Average	**8.209**	**3.505**	**0.4**	**0.272**	**0.2**	**9.883**	**23.545**