Case Study of a Multi-Stage Filtration System for Remote and Northern Communities in Canada

A thesis submitted to

the Faculty of Graduate and Postdoctoral Affairs

in Partial Fulfillment of the requirements for the degree

Master of Applied Science in Environmental Engineering

By

Celia Charron

(B.A.Sc.)

Department of Civil and Environmental Engineering

Carleton University

Ottawa-Carleton Institute of Civil and Environmental Engineering

December 2015

Abstract

A pilot multi-stage filtration system for water treatment was operated at two sites with phosphorous nutrient limited (C:N:P of 546:24:1 w/w) and nutrient rich (C:N:P of 6.3:1.6:1 w/w) source waters. The system had two parallel treatment trains: Train 1 consisted of pre-ozonation, roughing filtration and slow sand filtration (SSF); and Train 2 consisted of pre-ozonation, roughing filtration and biofiltration (BF). Nutrient limited conditions exhibited lower DOC removals than nutrient rich conditions when ozone was present (9.3% vs 26% DOC removal, respectively). However, there was no difference in removal when no ozone was present (5.6% vs 6.4% DOC removal, respectively). A similar trend was seen with UVA_{254} removals (20% vs 45% with ozone, and 12% vs 13% without ozone, respectively). At the nutrient limited site, there was no overall difference between removals in the SSF and BF under conditions with and without ozone ($p>0.05$). At the nutrient rich site applied ozone resulted in a difference in removal between the SSF and BF trains ($p<0.05$), while there was no difference between the trains when no ozone was present ($p>0.05$). These findings highlight the importance of source water characterization and pilot testing when attempting to utilize the benefits of biofiltration for water treatment.

Acknowledgements

Firstly, I would like to thank my supervisor, Dr. Onita Basu, for her constant support and guidance throughout this work.

I would like to thank Bob LeCraw from MS Filter for allowing us to work with his technology, and providing a wealth of guidance throughout the experiment phase. I would also like to thank Shawn Cleary for his advice on how to properly run the trailer

I would like to thank Dr. Marie Tudoret Chow for her constant support and availability in providing laboratory assistance. I would like to thank Andy Campbell and Erin Gorman from Brittannia, and Isabelle Deslauriers and Patrick Laporte from Agnico-Eagle for their support with this project.

I would also like to thank everyone who has helped with the project in the past. Special thanks to Dejan Chortanoski and the entire Basu research group. I'd also like to thank Sylvia Corthorn who volunteered to edit this work.

I want to thank my family. I could not have done any of this without the constant support and encouragement of my father, sister and husband.

Finally, I would like to dedicate this thesis to my mother.

Table of Contents

List of Tables

List of Figures

List of Acronyms

AELM	Agnico-Eagle Laronde Mine
AOC	Assimilable Organic Carbon
AOP	Advanced Oxidation Processes
ATP	Adenosine Tri Phosphate
BAC	Biologically Activated Carbon
BDOC	Biodegradable Organic Carbon
BF	Biofiltration
BBWTP	Britannia Beach Water Treatment Plant
C:N:P	Carbon:Nitrogen:Phosphorous
COD	Chemical Oxygen Demand
DBP	Disinfection By-product
DOC	Dissolved Organic Carbon
EBCT	Empty Bed Contact Time
ES	Effective Size
GAC	Granular Activated Carbon
MSF	Multi-stage Filtration
NOM	Natural Organic Matter
SSF	Slow Sand Filtration
SUVA	Specific Ultraviolet Absorbance
THMFP	Trihalomethane Formation Potential
TOC	Total Organic Carbon
UC	Uniformity Coefficient
UVA	Ultraviolet Absorbance

1 Introduction

1.1 Background & Problem Statement

Providing clean drinking water for small and remote Canadian communities has been a problem in the past. Although clean drinking water is a human right, practical restrictions such as space, costs and complexity can pose limitations for smaller systems. These restrictions cause increases in the occurrences of waterborne diseases and boil water advisories, and lead to small drinking water systems that experience a number of problems, such as insufficient financial resources, costly technologies, lack of skilled operators, lack of local expertise and lack of space for full-scale plants (O'Connor, 2002). Several events have occurred that have led to an emphasis being placed on safe drinking water in small communities in Canada. One of the most well-known events is the tragedy of Walkerton in 2000. A number of people became ill and seven individuals died due to an operator error within their water treatment system. Following this event, the 2002 provincial inquiry led by Justice O'Connor identified 93 recommendations. Several of the recommended actions highlighted issues faced by remote communities, including First Nations communities (O'Connor, 2002). This inquiry identified three essential requirements for the provision of safe drinking water: performance, operation and maintenance, and cost-efficiency (O'Connor, 2002). These three requirements can be met through multi-stage filtration, the main technology used throughout this thesis.

Filtration is a widely used process for removing particles from drinking water. There are several types of filtration in use, including slow sand filtration (SSF), rapid sand filtration,

biofiltration (BF) and membrane filtration. Common pre-treatment steps ahead of these units include sedimentation, coagulation/flocculation and roughing filtration (Crittenden et. al., 2012). This work focuses exclusively on slow sand filtration and biofiltration, with roughing filtration and pre-ozonation as pre-treatment methods.

Slow sand filtration is a proven and reliable drinking water treatment technology, which has been in use for over 200 years. The technology is easy to understand and use. It is therefore ideal for smaller communities in which the reduction of operator training is a benefit. The combined physical and biological filtration will remove turbidity, organic compounds, inorganic compounds and microbial contaminants.

Biofiltration is a newer technology, with biological filtration as the main component for removal. Nutrients present in raw water: a) will form a biomass that attaches to the media bed; and b) develop a biofilm throughout the bed (Hozalski & Bouwer, 2001). The extent to which biomass forms depends on a number of factors including water quality, type of granular activated carbon (GAC), hydraulic conditions, temperature and backwash (Hozalski & Bouwer, 2001).

In addition to these treatments, the use of pre-ozonation ahead of filtration units has shown promise, specifically in small systems. Pre-ozonation ahead of biofiltration degrades natural organic matter (NOM) by changing its structure; making it easier for the media bed to remove the organics and eliminate taste, odour and turbidity (Yapsakli & Cecen, 2010).

1.2 Research Objectives

The primary objective of this research was to determine whether biofiltration could operate at the same performance level as slow sand filtration by comparing the two trains. Slow sand filters have larger area footprints, and operation costs in northern communities are therefore higher due to higher heating costs. The results of this research will help to determine what types of drinking water treatment are appropriate for communities with small populations in northern Canada.

This study employed technology supplied by MS Filter Inc. Currently, MS Filter (MSF) has 28 filtration plants in operation in Canada and the US, which use slow sand filtration as the final filtration step (MSF Systems Inc., 2015). Five phases of operation were investigated, at different conditions (nutrient availability and pre-ozonation dosages). The specific research objectives were as follows:

- Monitor the operation of two separate filtration systems in terms of overall removals and determine whether biofiltration is as suitable as slow sand filtration for use in remote communities;
- Using a range of ozone dosages, assess the impact of pre-ozonation ahead of filtration; and
- Assess the impact of different Carbon:Nitrogen:Phosphorous (C:N:P) ratios of source waters in terms of removal of parameters and filter biomass growth.

Phases 1 and 2 of the research study saw the trailer operating at a nutrient-limited source water site. Phases 3, 4 and 5 of the research study saw the trailer operating at a

nutrient-rich source water. Different ozone dosages were used for pre-ozonation in each phase. Water samples were collected and tested for several common drinking water parameters, including turbidity, colour, dissolved organic carbon (DOC), UVA_{254}, and specific ultraviolet absorbance (SUVA). In addition, media samples were taken from the filter beds to determine the level of biomass present throughout the duration of the study.

1.3 Organization of Thesis Document

This thesis is presented as five chapters. The literature review presented in Chapter 2 is a review of published literature related to this study. The review includes an overview of drinking water filtration systems, advanced oxidation processes, multi-stage filtration and membrane filtration. In Chapter 3 the materials and methods that were used during the research phase of this study, including a description of the pilot trailer, are discussed. Chapter 4 is presented in journal format, showing the results from this study in the form of a journal article. Finally, Chapter 5 sets out the conclusions and recommendations for future work.

2 Literature Review

The literature review is divided into three sections. :

1. Two common drinking water treatment technologies (slow sand filtration and biofiltration) are reviewed. The problems encountered when these treatments are used in remote and northern communities are considered.
2. Multi-stage filtration is discussed, as well as its associated advantages and disadvantages.
3. Additional work regarding the use of multi-stage filtration in northern communities is addressed, including its use as a pre-treatment ahead of membrane systems in an effort to decrease fouling.

2.1 Clean Drinking Water and Necessary Treatment

The need for clean drinking water is an ongoing global issue. In Canada, the need for clean drinking water is prevalent in remote and northern communities, where cost and temperature are an issue. 92% of all water systems in Canada serve a population of less than 5,000 people (Cleary et. al., 2008). Water systems serving small communities are often not technologically advanced or robust and may be more susceptible to breakthrough of contaminants as a result. In addition to the removal of taste and odour and the demand for aesthetically pleasing drinking water, the industry is facing demands to improve the removal of cysts and oocysts such as *Giardia* and *Cryptosporidium* because of their related health issues (Jasim et. al., 2008).

After the events of Walkerton in 2000, an inquiry conducted by Justice O'Connor identified several challenges that small drinking water systems face. These challenges

include insufficient financial resources to meet the regulatory standards, costly technologies used in high-tech treatment, lack of skilled operators, lack of local expertise, and lack of space for full-scale plants (O'Connor, 2002). The inquiry also identified three pre-requisites for the provision of safe drinking water: performance, operation and maintenance, and cost-efficiency (O'Connor, 2002). These pre-requisites must be considered when developing a drinking water treatment filtration system in small communities.

Two types of filtration processes that can be used in remote communities are slow sand filtration and rapid filtration (specifically biofiltration). Research has been done on the capability of both types of treatments in smaller communities. There are numerous working plants now in operation throughout Canada.

Slow sand filtration has been in operation for several decades, with the first documented case occurring in the 19th century. Some advantages of slow sand filtration are its low maintenance and operational costs (compared to full scale rapid filtration plants), simplicity, low energy consumption and proven good performance (Cleary et. al., 2008). Biofiltration is a type of rapid filtration, with filtration rates up to 100 times higher than those in slow sand filters. Research into biofilters and their operating parameters is still ongoing, and there are many uncertainties in terms of their operation in smaller communities.

2.1.1 Slow Sand Filtration

Slow sand filtration has been in use as a drinking water treatment technology for almost 200 years. Slow sand filters contain a bed of sand particles that have an effective size (E.S.) of 0.35 mm or less, and operate at filtration rates between 0.05 to 0.5 m/hr (Crittenden et. al., 2012). Due to the low hydraulic loading rate of the slow sand filters, a biological layer called the *schmutzdecke* develops on top of the bed. The *schmutzdecke* is a thin, slimy layer that contains various microorganisms (i.e. plankton and bacteria) and captures organic matter passing through. This layer provides additional physical filtration by straining some of the smaller particles from the influent water (Crittenden et. al., 2012). Once the water passes through the *schmutzdecke* layer, physical and physico-chemical removal processes such as adsorption are responsible for removing the remaining particles. Slow sand filtration has been shown to meet regulatory requirements for *cryptosporidium* and *giarda* removal, as well as for the removal of various bacteria and viruses (LeCraw & Jobb, 2011).

A slow sand filter plant is typically configured as follows: the water first passes through the *schmutzdecke* followed by the remaining sand bed; then through a support layer of graded gravel that is one to two feet deep (Crittenden et. al., 2012). The effluent, or treated water, is collected in an underdrain system and travels to the distribution system. Head loss builds slowly during the filter run due to the lower filtration rate. The maximum water level in the filter is dictated by the available head. When the maximum head loss is reached, the filter is usually drained and the top 1 to 2 cm of the media are

scraped off, cleaned and stored onsite. Slow sand filters do not need to be backwashed, but a backwash can be performed if necessary after several instances of cleaning.

The advantages of slow sand filtration include simplicity, low maintenance and operation costs, low energy consumption, low amounts of produced waste, and no transportation of cleaning chemicals (Cleary et. al., 2008). There are, however, limitations to slow sand filters, including sensitivity to increased turbidity levels, a smaller layer of biological activity responsible for removal (*schmutzdecke* depth is typically 0.5 to 2 cm) and low removal of organics and disinfection by-product (DBP) precursors (Cleary et. al., 2008). When using slow sand filters, operators typically do not use pre-treatment. Therefore this technology should not be used in tandem with poor-quality surface waters. It has been recommended that slow sand filters be used to treat waters that have turbidity less than 10 NTU and colour less than 15 true colour units (TCU) (Crittenden et. al., 2012). Finally, although slow sand filters do have low operation costs, the space required for full scale plants is higher compared to that required for full scale biofiltration plants (due to the lower filtration rates). Therefore, in remote communities the heating cost would be higher for a full scale slow sand filtration plant.

Although the technology is not efficient for larger full-scale plants, slow sand filters have been proven very successful in remote and northern communities. MS Filter Inc. (MSF) provided the pilot trailer which is the source of the data for this thesis, and reports 28 working slow sand filter plants in small communities across Ontario (MSF Systems Inc., 2015).

2.1.2 Biofiltration

Biofiltration is a filtration technology which has been the subject of extensive research in the past two decades. It is a form of rapid filtration. This method has been recognised as an effective technology for removing NOM and micropollutants, lowering turbidity, removing inorganic contaminants and eliminating DBPs that can be formed when chlorine is added ahead of filtration. This method also improves water quality by decreasing the potential for bacterial regrowth in distribution systems (Crittenden et. al., 2012).

In a biofiltration process, the columns are filled with granular activated carbon (GAC), which turns into biologically activated carbon (BAC) during colonisation of microorganisms that establish active biofilms (Gibert et. al., 2013). The GAC is well-suited for the removal of organic contaminants because of its large specific surface area and developed porous structure (Gibert et. al., 2013). Biomass accumulates on the filter media through deposition and growth as water that contains NOM and biological organic matter (BOM) flows through the filter (Hozalski & Bouwer, 2001). Biofilm development in the columns is very important for a well-established biofiltration procedure. The extent of biofilm formation depends on factors such as water quality, type of GAC, hydraulic conditions, temperature and backwashing procedure.

In a full-scale plant, biofiltration columns are typically preceded by coagulation/flocculation and sedimentation and followed by disinfection. Advanced

oxidation processes (AOPs) are also a common pre-treatment, and are investigated throughout this thesis. They are discussed further in Section 2.2.

Columns are filled with a supporting gravel or sand layer that is placed on top of an underdrain. The GAC is placed on top of the gravel layer. The media inside a biofilter should be selected to meet both physical and biological treatment objectives (Schulz, 2014). In a well operating system, the media type will affect biomass accumulation; the amount of attached biomass will decrease as the filter depth is increased (Gibert et. al., 2013). However, a study by Velten et. al. (2011) found that once a steady-state operation of their biofilters was reached, the concentration of biomass in the filter bed was actually higher a third of the way down the bed than at the top. The authors of the study hypothesized that this result was due to the use of ozonation, which may have affected the concentration in the top layer, and to the temperature at which the study was conducted (Velten et. al., 2011).

2.1.2.1 Hydraulic Loading Rate

In biofilters, the hydraulic loading rate (HLR) is an important parameter with respect to contaminant removal and biomass accumulation. HLR is the rate at which water passes through the depth of the filter bed. Empty bed contact time (EBCT) is another important parameter. EBCT is the time that it takes for a volume of water (defined as the volume of the filter bed) to pass through the entire filter bed. The calculation to determine HLR is set out below (Crittenden et. al., 2012).

$$HLR = \frac{Q}{SA} \qquad \text{(Eq. 2.1)}$$

HLR: Hydraulic loading rate (m/hr)

Q: Flow through filter (m^3/hr)

SA: Surface area of filter bed (m^2)

The calculation to determine EBCT is as follows (Crittenden et. al., 2012).

$$EBCT = \frac{V}{Q} \quad \text{(Eq. 2.2)}$$

EBCT: Empty bed contact time (min)

V: Volume of media bed (m^3)

A study by Zhang et. al. (2010) investigated the impact of six different HLRs on the removal efficiency of conventional pollutants and semivolatile organic compounds. With an increase in the HLR, the removal of the organic compounds decreased. The authors concluded that HLR had the greatest impact on biofilter biomass, substrate removal performance, and microbial community structure (Zhang et. al., 2010).

A study by Ko et. al. (2007) examined organic removal efficiency as a function of HLR and EBCT. Three different loading rates were investigated (24 m/hr, 12 m/hr, and 6 m/hr) with corresponding EBCTs. At higher HLR levels, a higher effluent turbidity was seen, but the particle removal efficiency was also higher. It was also observed that biomass concentration decreased with a decrease in HLR to less than 12 m/hr. The most efficient removal of organics was seen at the lowest HLR level, which corresponded to the highest EBCT (Ko et. al., 2007).

Table 2.1 shows hydraulic loading rates and corresponding EBCTs from a selection of studies. Observations on contaminant removals are also included in Table 2.1. In conclusion, it can be seen that it is ideal to run biofilters at a lower HLR and a corresponding higher EBCT to achieve better removal of organic contaminants.

Table 2.1: Hydraulic loading rates and corresponding empty bed contact time found in literature.

Experimental Set-Up	Hydraulic Loading Rate	Empty Bed Contact Time	Observations	Literature
Three bio-filters	5 m/hr	Each filter different: 5 min, 10 min, 15 min	- Different EBCTs used to classify fouling rate of various foulants on UF membrane	Chen et. al., 2014
Four filters filled with anthracite	7.5 m/hr	5.6 min	- Simple respirometric method was developed for measurement of biomass activity	Ufer & Huck, 2001
Two bio-filters	8 m/hr	N/A	- Gradual removal of DOC with contact time	Liao et. al., 2013
- Pilot with five filtration columns filled with GAC - Full scale with two filtration columns	5 m/hr	Pilot: 3.6 min Full scale: 20-30 min	- Incomplete nitrification in pilot columns due to short EBCT - Removal in full-scale plants occurred within first 5 minutes of EBCT	Andersson et. al., 2001
Pilot filter filled with GAC	Average at 4.8 m/hr	19 min	- DNA yields exhibited a vertical stratification where attached biomass decreased as GAC filter depth increased, disappeared with time - DOC removal via adsorption and biodegradation	Gibert et. al., 2013

2.1.2.2 Impact of Water Temperature

Water temperature has been shown to affect general filtration performance, with colder temperatures influencing the rate of reactions and the settling velocity of water (Jasim et. al., 2008). Therefore, water temperature can be considered a restricting factor when assessing the best biofiltration treatment methods for a particular source water.

A study by Fonseca et. al. (2001) examined the effect of water temperature and determined that temperature did not affect levels of biomass on top of filter beds, but did affect the fraction of DOC removal (Fonseca et. al., 2001). Viable biomass is used as a parameter for modelling that predicts organics removal. The results from this study showed that biological activity was a better parameter to use when determining the impact of temperature on removal of DOC.

Another study by Chen & Wang (2012) found that treated water quality improved with increasing ozone and catalyst dosages, but was independent of water temperature at temperatures between 10 ^{o}C and 30 ^{o}C (Chen & Wang, 2012). The use of coagulation and flocculation aides, such as polymers, or of oxidation processes, such as ozonation, has been suggested for Canada and parts of the United States that are subject to temperatures below 5 ^{o}C (Jasim et. al, 2008). Depending on the source water, temperature should be considered a factor when developing an appropriate treatment.

2.1.2.3 Biofiltration Cleaning Strategies

Although the biomass in a biofilter contributes to organics and particulate removal, the filters can also become clogged due to the accumulation of these particles. In order to alleviate this clogging, and for optimum filter performance, a backwash sequence is

necessary. The purpose of a filter backwash is to fluidize the bed at a rate sufficient to remove any deposited material (Slavik et. al., 2013). The backwash procedure generally accepted for use in a full-scale plant consists of air scour, sub-fluidization water wash and full fluidization water wash.

Optimum particle removal usually occurs when the conditions favour collapse-pulsing (Amirtharajah et. al., 1990). Collapse-pulsing is the phenomenon of air cavities forming and collapsing throughout the depth of a media bed at certain air and fluidization water flow rates. A study by Slavik et. al (2013) investigated three different backwash procedures in a deep bed filtration system, with the end result being that the productivity of filtration (in terms of filter run time) was the same for each of the three backwash procedures. This study demonstrated that the backwash velocity is more important with respect to designing a process for biofilters. A suitable backwash velocity is one that provides a proper fluidization of the bed - allowing contaminants and particles to be removed. The calculations for determining backwash flow rate in a filter are set out below (Crittenden et. al., 2012).

The first calculation provides the depth of the expanded bed and the porosity of the expanded bed, which are required in further equations.

$$\frac{L_E}{L_F} = \frac{1-\varepsilon_F}{1-\varepsilon_E} \qquad \text{(Eq. 2.3)}$$

L_E = depth of expanded bed, m

L_F = depth of bed at rest, m

ε_E = porosity of expanded bed, dimensionless

ε_F = porosity of bed at rest, dimensionless

The second calculation provides the backwash calculation factor, which is needed in further equations.

$$\beta = \frac{g p_w (p_p - p_w) d^3 \varepsilon_E^3}{\mu^2} \quad \text{(Eq. 2.4)}$$

β = backwash calculation factor, dimensionless

g = acceleration due to gravity, 9.81 m/s^2

p_w = density of water, 1000 kg/m^3

p_p = density of media particle, kg/m^3

d = diameter of media, m

μ = viscosity of water, $1.139*10^{-3}$ kg/m*s

The third calculation provides the Reynold's number of the particles.

$$Re = \frac{-k_v(1-\varepsilon) + \sqrt{k_v^2(1-\varepsilon)^2 + 4k_l\beta}}{2k_l} \quad \text{(Eq. 2.5)}$$

Re = Reynolds number, dimensionless

k_v = head loss coefficient due to viscous forces, dimensionless

k_I = head loss coefficient due to inertial forces, dimensionless

The final calculation provides the backwash velocity for the filters.

$$v = \frac{\mu Re}{p_w d} \qquad \text{(Eq. 2.6)}$$

v = backwash velocity, m/s

After a backwash sequence there is a small spike in turbidity. The filter then recovers to its previous operational values. This phenomenon is called filter ripening. It is understood that more than 90% of particles that pass through a well-operated filter do so during the ripening period (Amburgey & Amirtharajah, 2005). It has also been suggested that backwashing leads to a decrease in biomass, but that the biomass can return to its previous conditions before the next backwash cycle (Gibert et. al., 2013).

Table 2.2 shows backwash procedures found in literature, including backwash velocity and bed expansion.

Table 2.2: Typical backwash procedures for conventional and biofiltration columns found in literature.

Experimental Set-up	Backwash process	Backwash Rate	Bed Expansion	Literature
Four biofilters filled with anthracite	- Air scour with collapse pulsing conditions (A) - Air scour and water for 2.5mins (B) - Water alone for 4mins (C)	- 12 m/hr (A) - 60 m/hr (B) - 54 m/hr (C)	- Bed expansion of 40% (A) - Bed expansion of 50-60% (C)	Ufer & Huck, 2001
Three pilot filters filled with anthracite and sand	- Extended terminal subfluidization wash (ETSW) was investigated - Pilot-scale filters backwashed at 24 hour intervals	- 3 min combined air and water wash - 5 min full fluidization water only wash followed by further water to push one filter-volume through - last step at various fluidization and sub-fluidization washes for 4 mins	- Bed expansion of 20%	Amburgey & Amirtharajah, 2005
Two full-scale filters containing different regenerated GACs	- Air scour - Sand filtered water - Filters backwashed on rotating schedule so plant can continue to operate	- 0.5 m^3/s for 5 mins - 0.3 m^3/s for 14 mins	N/A	Gibert et. al,, 2013
Two filter columns, one with sand-filled single media and other with sand and anthracite-filled dual media	- Procedure 1: water flush at 20 m/hr for 10 mins, then fluidisation at 68 m/hr for 5 to 6 mins - Procedure 2: water flush at 10 m/hr for 3 mins, fluidisation at 68 m/hr for 40 to 60 secs, additional flushing at 47 m/hr for 5 mins	- 68 m/hr for 5 mins - 68 m/hr for 40-60 secs, additional flushing at 47 m/hr	N/A	Slavik et. al., 2013

	- Procedure 3: water flush at 10 m/hr for 3 mins, fluidisation at 68 m/hr for 40 to 60 secs, additional flushing of 1, 1.5 and 2 filter bed volumes at 23 m/hr	- 68 m/hr for 40-60 secs, additional flushing at 23 m/hr		
Parallel downflow biofilters operated with GAC and crushed expanded clay (two different grain sizes)	- Backwashed with collected filtrate when GAC filter reached headloss of 70 cm - Headloss development in coarse EC filter was slow, and filter was only backwashed once through course of study at 23 months	- GAC filter: 37 m/hr - Fine EC filter: 26 m/hr - Coarse EC filter: 128 m/hr	-35% -35%	Persson et. al., 2006

2.2 Multi-Stage Filtration

Multi-stage filtration is defined as a filtration process that uses pre-oxidation and roughing filtration ahead of standard filtration methods (LeCraw & Jobb, 2011). It is an ideal technology for locations with poor source water quality. Pre-oxidation technologies are processes with a high oxidation potential, including ozonation and combinations of ozone/hydrogen peroxide and ozone/UV irradiation. These technologies have the potential to remove from drinking water contaminants such as inorganic species, organic species and toxic micropollutants (Camel & Bermond, 1998). A roughing filter is a standard filtration column filled with coarse media that provides robustness to the system by decreasing turbidity and solids loading, increasing filter run time and increasing hydraulic retention time (Cleary et. al., 2008). Both treatment methods are explained below.

2.2.1 Ozonation

Ozone has been a widely used component in drinking water treatment throughout the past twenty years, due mainly to its high oxidation potential (Camel & Bermond, 1998). Ozone is unstable in water and will undergo reactions with other components to provide specific oxidation properties. The oxidation of inorganic or organic compounds in drinking water occurs by ozone, OH radicals, or a combination of the two (von Gunten, 2003).

Ozone can be introduced at several different points during the treatment process: pre-oxidation, intermediate oxidation and final disinfection (Camel & Bermond, 1998). Pre-

oxidation will eliminate mineral compounds, colour, turbidity, taste and odour, and partly degrade NOM (Camel, 1998). Intermediate oxidation will degrade toxic micropollutants, eliminate trihalomethane (THM) precursors, and increase biodegradability (Camel & Bermond, 1998). Final disinfection will eliminate all remaining microorganisms and disinfection by-products (Camel & Bermond, 1998).

Depending on the application of ozone that is desired, the two species (ozone and OH radicals) are of different levels of importance (von Gunten, 2003). Disinfection occurs through reactions with ozone; whereas oxidation processes occur with either ozone or OH radicals or a combination of the two (von Gunten, 2003). There are over 500 rate constants measured for ozone reactions, and an even larger number measured for OH radical reactions (von Gunten, 2003).

Several organic compounds can react with ozone and form unwanted by-products (von Gunten, 2003). One such example is bromide. Source waters in which bromide is present can form hypobromous acid; leading to the formation of bromate, a dangerous by-product (von Gunten, 2003). Strategies for feasibly reducing bromate formation include adding ammonia, lowering the pH, adding iron or using UV irradiation (von Gunten, 2003).

2.2.1.1 Pre-Ozonation in Combination with Biofilters

The use of ozone with biofiltration is popular and achieves three main objectives: 1) meeting federal drinking water regulations for primary disinfection, DBPs and turbidity; 2) removing algal derived taste and odour causing compounds; and 3) removing biodegradable organic carbon (BDOC) to produce biologically stable finished water

(Schulz, 2014). Several synergistic benefits can be seen between ozonation and biofiltration in terms of reaching a high quality effluent (Schulz, 2014).

An example of this synergistic effect is the removal of BDOC. Ozone will oxidize organics to generate BDOC, and biofiltration removes the biodegradable fraction so as to produce biologically stable drinking water that does not further react in the distribution system (Schulz, 2014). A study by Kim et. al. (1997) looked at three different processes: ozonation-BAC, conventional BAC, and chlorination-GAC adsorption. The processes were evaluated based on DOC, adsorbable DOC (ADOC) and BDOC. It was found that all processes removed DOC by adsorption; however, BDOC increased by 20% after ozonation in the ozonation-BAC process and was therefore removed more effectively by the bacteria on the activated carbon (Kim et. al., 1997). It was concluded that ozonation improved the biodegradability of organic substances, and this resulted in an overall increased DOC removal (Kim et. al., 1997).

Another study by Pei et. al. (2007) found that: 1) pre-ozonation enhanced the reduction of NOM in a conventional filtration process; and 2) total removals of UVA_{254}, chemical oxygen demand (COD_{Mn}) and TOC were improved (Pei, 2007). The removal for COD_{Mn} increased from 27% to 50% with an increase in ozone doses from 0 to 1.5 mg O_3/L, and removal for UVA_{254} increased from 39% to 86% with an increasing ozone dose (Pei et. al., 2007).

Hozalski et. al. (1999) studied the effect of ozone applications on TOC removal, using a range of doses from 1.3 mg O_3/mg TOC to 7.3 mg O_3/mg TOC. It was found that ozone

dosages in the range of 1-2 mg O_3/mg TOC were optimal for enhancing biodegradation of higher molecular weight NOM sources; there was little benefit in ozonation of NOM with lower molecular weights (Hozalski et. al., 1999).

There are optimal ozone dosages for each source water. Several studies emphasize the need to perform bench-scale tests to determine the optimum dosage for a full-scale plant. Table 2.3 highlights different ozone dosages from a number of studies considered in this literature review.

Table 2.3: Ozonation doses found in literature.

Experimental Set-up	Ozone Dosage	Observations	Literature
- Three samples representing wide spectrum of NOM compositions - Semi-batch experiments	- Doses of 0.85-1.5 mg O_3/mg TOC -Doses of 2.6-3 mg O_3/mg TOC	- An average reduction of 54% in UVA_{260}, 6.4% in DOC and 8% in trihalomethane formation potential (THMFP) - An average reduction of 72% in UVA_{260}, 16% in DOC and 43% in THMFP	Galapate et. al., 2001
- Synthetic raw water for bench-scale test and raw water for pilot-scale test - Pilot plant had two parallel trains: 1. coagulation, dissolved air flotation, sand-filter, GAC and disinfection 2. coagulation, DAF, sand-filter, mid-ozone, BGAC, and disinfection	- Ozone dose between 1.1-4.4 mg/L	- UVA_{254} removal increased significantly with an increase in O_3 dose, especially at low doses - SUVA became stable as O_3 dose was raised to 2 mg/L - Percentage of intermediate MW DOM increased with 1.1 mg/L O_3, and range of low MW DOM increased at 4.4 mg/L O_3	Yan et. al., 2010
- Bench scale ozone reactor followed by filter with 213cm of anthracite	- Experimental water was ozonated with 0.6 mg O_3/mg DOC - Practical range of ozone doses was used 0.2-1.5 mg O_3/mg DOC in order to find the fraction of initial DOC transformed into filter-removable BOM	- Highest removal between 1 and 1.5 mg O_3/mg DOC	Carlson & Gary, 1996

- Case study at water utility delivering water to approximately 3,000 residents (at 1350 m^3/day) - Two identical pilot units constructed in PVC: 1. slow sand filtration 2. ozone followed by slow sand filtration	- Mean ozone dosage at 3.5 mg/L	- Detected two species of THMs and three species of HAAs in the distribution system	Guay et. al., 2005
- Treatment processes consisted of coagulation, flocculation and flotation, sand filtration, ozonation, activated carbon filtration and disinfection	- Ozone dose in full-scale treatment varied between 0.4 to 1.5 gd/m^3 (winter was 0.4-0.7 gd/m^3 and summer was 0.9-1.5 gd/m^3) - Pilot water had ozone concentration between 0.3-1.2 mg O_3/mg TOC	- Amount of assimilable organic carbon increased with increasing ozone dose, with 0.4 mg/L removed with highest ozone dose of 4 mg/L - UVA_{254} absorbance decreased as ozone dose increased	Matilainen et. al., 2006
- Treatment process consisted of coagulation, flocculation-decantation, sand filtration, ozonation, removal of residual ozone, GAC filtration, and final chlorination - Wanted to reduce ozone doses to reduce formation of bromated levels; introduced additional UV disinfection step	- Ozone doses of 0.5 and 1 mg/L were used for all experiments - Ozone doses in plant with sand filtered water are between 1.5 and 2.5 mg/L	- At low ozone dose of 0.5 mg/L, bromate formation kept below 0.4 µg/L	Meunier et. al., 2006

2.2.2 Advanced Oxidation Processes

An advanced oxidation process (AOP) is defined as a process that combines ozonation with another form of oxidation, in order to speed up the decomposition of organics. The formation of highly reactive OH radicals will occur when combining two oxidation processes, and will oxidize any compounds that are ozone-resistant, such as pesticides, certain aromatic compounds and chlorinated solvents (von Gunten, 2003). Ozone-based AOPs are the most common method of AOP. They have shorter reaction time than other methods, leading to higher ozone dosages without excess ozone concentrations in the treatment (von Gunten, 2003).

2.2.2.1 Hydrogen Peroxide Addition

A combination of hydrogen peroxide and ozone is the most commonly applied AOP in drinking water treatment (von Gunten, 2003). The addition of hydrogen peroxide to an existing ozone system will decrease the optimum ozone dose required, therefore decreasing costs and potential contamination (Camel & Bermond, 1998). The biggest advantage of the O_3/peroxide process is the acceleration of the ozone transformation process that happens during ozonation (von Gunten, 2003). The mechanism for the O_3/peroxide process occurs through the formation of hydroxyl radicals, as seen below (taken from Zhou & Smith, 2001):

$$H_2O_2 + H_2O \rightarrow HO_2^- + H_3O^+ \qquad \text{(Eq. 2.7)}$$

$$HO_2^- + O_3 \rightarrow \bullet OH + O_2^- + O_2 \qquad \text{(Eq. 2.8)}$$

The HO_2^- base reacts rapidly with molecular ozone. Even at low concentrations, the base ion can initiate ozone decomposition and facilitate the formation of hydroxyl radicals (Zhou & Smith, 2001). The base rate increases with increasing pH (more H_2O_2 becomes dissociated into HO_2^- ions). Following the Walkerton event in 2000, Rahman et. al. (2010) conducted a study on the impact of ozonation and AOPs on Lake Huron water. In the experiment, ozone was injected upstream of the treatment process. Hydrogen peroxide was injected either upstream or downstream of the ozone addition. The ozone dose was varied between 2 and 2.3 mg/L. The peroxide dose remained constant at 0.2 mg/L (Rahman et. al., 2010). The results from this part of the study showed that AOPs achieved higher particle removal and improved turbidity compared to conventional treatment processes (Rahman et. al., 2010).

The use of peroxide with ozone has also been shown to reduce bromate formation, a problem commonly seen in ozonation of drinking water. Wang et. al. (2014) investigated the possibility of controlling the formation of bromate and reducing odour by adding peroxide ahead of an O_3/BAC process. The source water required ozone doses as high as 4 mg/L in order to remove a characteristic septic odour. This dosage lead to a high production of carcinogenic bromate (Wang et. al., 2014). The authors found that with a peroxide/ozone ratio of 0.5: a) the concentration of bromate decreased significantly at ozone doses between 2 to 3 mg/L; and b) this ratio provided an enhanced removal of septic odour (Wang et. al., 2014).

Another study by Sanchez-Polo et. al. (2006) compared the efficiency of an ozone/GAC system with two common AOPs (ozone/OH^- and ozone/peroxide). The catalytic activity of GAC has the ability to transform ozone into OH radicals by affecting ozone concentration, activated carbon dose, and chemical and textural properties of the carbon (Sanchez-Polo et. al., 2006). The study found that both processes (applied to the same source water) had similar efficiency with regard to OH radical yield and rate of formation (Sanchez-Polo et. al., 2006).

Conclusions to be drawn from these studies are: a) different source waters react differently to oxidation treatments; and b) it is necessary to perform pilot studies to determine the rate of OH radical formation given by different AOPs.

Irabelli et. al. (2008) investigated the impact of O_3/peroxide treatment with respect to THM formation. THMs are of concern in drinking water treatment because they can form when chlorine has been used as a disinfectant (Irabelli et. al., 2008). The study found that the addition of peroxide before or after ozonation increased THM concentrations, and that increasing the peroxide/ozone ratio also increased THM concentration (Irabelli et. al., 2008).

2.2.2.2 UV Irradiation and Other AOPs

An alternative to peroxide addition is the use of ozone with UV irradiation. The first step of the process is a photolysis of ozone to oxygen and O(^{1}D and ^{3}P) atoms (von Gunten, 2003). The O(^{1}D) atom is very reactive. It reacts with H_2O to form hydrogen peroxide. The O(^{3}P) atom reacts with either O_3 or with organic solutes to form oxygen. The oxidation capacity is somewhat lost with the transformation from ozone to hydrogen

peroxide. However, an additional effect is the reaction of organic compounds with the $O(^3P)$ atom (von Gunten, 2003).

The original use of the O_3/UV process was for the destruction of wastewaters containing cyanide. O_3/UV is now used for treatment of groundwater containing trichloroethylene and perchloroethylene (Zhou & Smith, 2001). This process is now considered to be less economical when compared with the O_3/H_2O_2 process.

Other ozone-based AOPs currently in use include the combinations of ozone with activated carbon or with Mn(II) or Mn(IV). One other AOP is the combination of H_2O_2 with UV irradiation; a process which eliminates the need for ozone. These alternative methods have been applied in research for oxidation-based drinking water treatment. However, the application of hydrogen peroxide with ozone remains the most widely used.

2.2.3 Roughing Filtration

Roughing filtration is an integral part of multi-stage filtration systems for drinking water. Roughing filters can decrease turbidity and solids loading, increase the filter run length, increase the hydraulic retention time and increase the robustness of a system (Cleary et. al., 2008). A typical roughing filter is composed of coarse gravel media, and can be designed as upflow, downflow or horizontal flow. Roughing filter design is simplistic, and operation and maintenance are relatively easy. Use of roughing filters in cold water conditions can improve the performance of slow sand filters, by providing increased retention time that compensates for downstream loss of efficiency (Gottinger et. al.,

2011). Most roughing filters (depending on design) can handle turbidity ranging from 50 to 200 NTU and turbidity spikes that reach 1,000 NTU (Gottinger et. al., 2011).

A study by Ochieng et. al. (2004) investigated multistage filtration with a combination of slow sand filters and horizontal roughing filters as an alternative to conventional drinking water treatment. Horizontal roughing filters were chosen due to their success in tropical climates (Ochieng et. al., 2004). The study found that the combination of a slow sand filter and a roughing filter, with proper design specifications, performed better than conventional systems under similar raw water and environmental conditions (Ochieng et. al., 2004).

Another study by Pacini et. al. (2005) investigated the removal of iron and manganese from groundwater using biological systems preceded by up flow roughing filter technology. Iron and manganese removal efficiencies between 85% and 95% were observed throughout the study. The higher solids retention capacity of the roughing filter allowed for removal by both biotic and abiotic mechanisms (Pacini et. al., 2005). It was concluded that up flow roughing filtration allowed for high metal removal efficiency, with iron and manganese removal in one step without having to control pH or dissolved oxygen (Pacini et. al., 2005).

2.2.4 Multi-stage Filtration as Pre-Treatment for Membrane Fouling

Multi-stage filtration has received attention in recent years due to its potential use as a pre-treatment system ahead of a membrane. Membrane fouling is a barrier during water treatment, particularly with groundwaters that have high amounts of NOM.

Despite the advantages of membrane technology over conventional treatments, biofouling continues to be an operational problem for membrane facilities. This problem is prevalent in areas where operating conditions promote the growth of microorganisms (Hu et. al., 2005). Foulants of nano-filtration membranes include protein- and polysaccharide-like substances, colloidal macromolecules, and high molecular weight macromolecules (Halle et. al., 2009). Fouling on microfiltration and ultrafiltration membranes is predominantly caused by small, organic colloids (Halle et. al., 2009). There have been many studies directed towards lowering the rate of occurrence of membrane fouling in drinking water. One method has been to use biofiltration or multi-stage filtration ahead of membrane filtration. In several northern communities, this method is ideal because it removes the high organic content seen in groundwater ahead of the membrane system, while minimizing the need for costly chemicals (Chan et. al., 2014).

A study done by Halle et. al. (2009) investigated rapid biological filtration without coagulant addition as a pre-treatment to reduce membrane fouling. The water passed through a pilot scale roughing filter, followed by two parallel anthracite/sand biofilters and an ultrafiltration membrane (Halle et. al., 2009). Biopolymer removal was used as an indicator, and it was found that the biofilter with a longer contact time led to higher reductions in both hydraulically reversible and irreversible fouling (Halle et. al., 2009).

Another study done by Hu et. al. (2005) investigated the performance of a lab-scale reverse osmosis system set up at a local water reclamation plant. The biofiltration

system was found to remove assimilable organic carbon (AOC) at an efficiency of 40-49% and DOC at an efficiency of 35-45% with an EBCT of 30 minutes (Hu et. al., 2005). It was found that without a biofiltration pre-treatment system, biofouling would occur after 72 hours. However, with the pre-treatment in place the operational length of the RO membrane increased by five times, to more than 300 hours between fouling periods (Hu et. al., 2005).

A study done by Chan et. al. (2014) examined a biofiltration pre-treatment set-up at an existing plant in Saddle Lake Cree Nation. The treatment plant included an ultrafiltration membrane system, which was unable to reduce the raw water DOC of 20 mg/L (Chan et. al., 2014). The study ended with a recommendation for a full scale plant. The steps recommended were:

- To condition water by oxygenation;
- Particle filtration consisting of two filters;
- Biological filtration consisting of two filters;
- Four trains of RO treatment units;
- Calcium-magnesium contactor to stabilize treated water; and
- Disinfection (Chan et. al., 2014).

2.2.5 Parameters Affected by Multi-Stage Filtration

Depending on the country of origin and the regulating body, there are several drinking water standards to which adherence is required. Parameters monitored in Canada include bacterial waterborne pathogens, turbidity, metals, inorganic compounds,

organic compounds, hardness, pH, odour, taste, colour, and by-product formation potential (Health Canada, 2014). Below are concise descriptions of the most common parameters considered for safe drinking water in remote communities. The majority of these parameters were investigated in this thesis.

2.2.5.1 Turbidity

Turbidity is one of the most common parameters tested in drinking water systems. It is an indicator of the effectiveness of the treatment in removing potential microbial pathogens. It is a measure of the scattering and absorbing effect suspended particles have on light. In Canada, the relevant guidelines specify a turbidity of less than or equal to 0.3 NTU for conventional filters; less than or equal to 1.0 NTU for slow sand filters; and less than or equal to 0.1 NTU for membrane filtration (Health Canada, 2014). These results must be achieved in at least 95% of measurements either per filter cycle or per month.

2.2.5.2 Colour

The appearance of colour in drinking water is due to the presence of coloured organic substances that occur from decay of natural vegetation and the presence of one or more of metals, highly coloured industrial wastes, human-made organic substances and coloured suspended solids. In Canada, the relevant guidelines specify a true colour of less than 15 TCU (Health Canada, 2014).

2.2.5.3 Organic Matter

The removal of NOM is widely used in research as an indicator of the performance of the treatment process studied. NOM is a heterogeneous mixture of organic compounds and their degradation products. These compounds and products can cause harmful

contaminants to be present during drinking water production (Metsamuuronen et. al., 2014). The concentration of NOM in drinking water has increased since 1990, due to climate change, changes in soil acidifications, severe drought seasons and intense rain events (Metsamuuronen et. al., 2014). The high variability in the composition of NOM makes it difficult to remove NOM completely from drinking waters. In addition to colour, NOM can also be identified through the presence (or absence) of taste and odour. The relevant Canadian guidelines identify several organic parameters that can cause harm to human health or cause unpleasant aesthetic conditions in drinking water.

2.2.5.4 Inorganic Matter

Inorganic matter in surface water consists of naturally occurring and anthropogenic metals, such as iron, manganese, mercury and nitrogen-based compounds. The Canadian guidelines identify several parameters that cause harm to human health or unpleasant aesthetic conditions in drinking water. One important parameter is mercury, which has a maximum acceptable concentration of 0.001 mg/L. Mercury in excess of that amount has been proven to cause neurological symptoms (Health Canada, 2014).

2.2.5.5 Disinfection By-Products

Chlorine is a common disinfectant used in drinking water treatment for the inactivation of pathogens. Chlorine is added to stabilize water in the distribution system. However, it can react with NOM present in the surface water to form common DBPs, such as THMs and haloacetic acids (HAAs). It should be noted that the risks from THMs or other DBPs in drinking water are less than risks posed from consuming water that has not been disinfected at all (Irabelli et. al., 2008). Therefore, the presence of DBPs is expected in

drinking water treatment; however it is important to employ proper controls to minimize the risk. The utilization of a biofilter or slow sand filter to help remove organics that can contribute to the DBP formation potential is one example. In Canada, the maximum contaminant limit for THMs is 0.1mg/L in drinking water (Health Canada, 2014).

2.2.5.6 Microbiological Parameters

In drinking water treatment, the highest priority parameters are those that deal with microbiological contaminants. Harmful microorganisms in poorly treated drinking water pose a great threat to any population. These microorganisms cause gastrointestinal problems in humans. Two indicators are *E.coli* and total coliforms. In Canada, no *E.coli* should be detectable per 100mL of drinking water (Health Canada, 2014). The presence of *E.coli* indicates that: 1) a faecal contamination has occurred; and 2) microorganisms that can cause gastronomic illness may be present in the drinking water (Health Canada, 2014). Total coliforms indicate potential health effects from pathogenic microorganisms. Detection of coliforms in excess of 10% in samples collected during a sampling period should lead to investigation of the causes of the excess levels (Health Canada, 2014).

3 Materials and Methods

3.1 Experimental Set-Up

3.1.1 Pilot Trailer

This study involved the testing of two separate drinking water treatment trains in a pilot-scale multistage filtration system. The first and second phases of research were set up to treat surface water from the Ottawa River, in Ottawa, Ontario, and the third, fourth and fifth phases of research were set up to treat surface water from Lac Chassignole in the Abitibi-Témiscamingue region of Quebec. The pilot-scale trailer was provided by MS Filter Systems Inc., a company based in Newmarket, Ontario. An image of the pilot trailer is shown in Figure 3.1.

Figure 3.1: Exterior view of pilot trailer.

The pilot trailer consists of two treatment trains: a slow sand filtration train (Train 1), and a biofiltration train (Train 2). A description of the set-up for both trains is presented in Sections 3.1.2 and 3.1.3.

3.1.2 Description of Train 1

Train 1 consists of a multi-stage filtration system with a slow sand filter providing the main removal. The design has evolved from many trials that were performed by MS Filter Systems Inc. A snapshot of the train is provided in Figure 3.2.

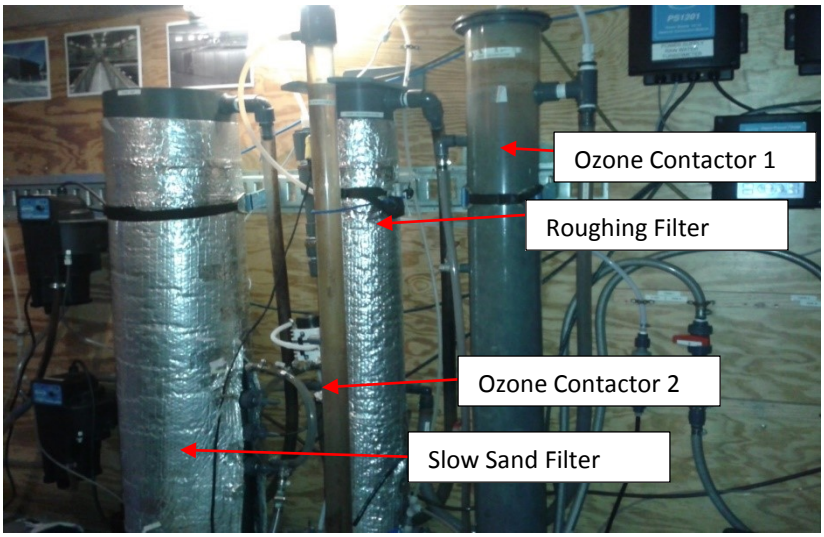


Figure 3.2: Experimental Set-Up for Train 1.

Raw water is fed through an inlet pipe and split between Train 1 and Train 2. In Train 1, water first passes through two ozone contactors at a filtration rate of 3 – 5 L/min. The purpose of the ozone contactors is to provide suitable time for the ozone to break down the organics present in the water. After the contactors, water is pushed through a peristaltic pump, and then passed into an upflow roughing filter at a lower filtration rate of 0.3 – 0.5 L/min. The remaining flow is sent directly to waste. After the roughing filter, the water flows into a standard slow sand filter. A HACH online turbidimeter was present at the effluent of the filter to measure turbidity. Table 3.1 provides design parameters for Train 1.

Table 3.1: Design parameters for Train 1.

Step	Design Parameter	Value
Ozone contactor	Diameter	150 mm
	Design Flow Rate	3 - 5 L/min
	Volume	24.7 L
	Contact Time	4.94 - 8.23 mins
Roughing filter	Diameter	150 mm
	Design Flow Rate	0.3 - 0.5 L/min
	Media Height	600 mm
Slow sand filter	Diameter	315 mm
	Design Flow Rate	0.3 - 0.5 L/min
	Volume	114 L
	Media Height	650 mm

3.1.3 Description of Train 2

Train 2 consists of a multi-stage filtration system with a biological filter providing the main removal. The design has evolved from many trials that were performed by MS Filter Systems Inc. A snapshot of the train is provided in Figure 3.3.

Figure 3.3: Experimental Set-Up for Train 2.

In Train 2, water first passes through an ozone contactor (as in Train 1) at a flow rate of 3 – 5 L/min. After the contactor, water is passed through an upflow roughing filter, and the flow is then split into two identical biological filters. A HACH online turbidimeter was present at the effluent of the filter to measure turbidity. Table 3.2 provides design parameters for Train 2.

Table 3.2: Design parameters for Train 2.

Step	Design Parameter	Value
Ozone contactor	Diameter	150 mm
	Design Flow Rate	3 - 5 L/min
	Volume	24.7 L
	Contact Time	4.94 - 8.23 mins
Roughing filter	Diameter	150 mm (Fall 2013) 305 mm (Summer 2014 onwards)
	Design Flow Rate	3 - 5 L/min
Biological activated carbon filter	Diameter	315 mm
	Design Flow Rate	3 - 5 L/min
	Volume	121 L
	Contact Time	Approximately 11 min
	Media Height	700 mm

3.1.4 Ozonation System

Ozone is generated onsite using an O-series ozone generator from Pacific Ozone. The ozone travels through Teflon-coated tubing and mixes with the raw water through the use of venturi meters, after which it travels through the ozone contactor. The ozone contactors are filled with PVC saddles to assist in the contact of the ozone gas and raw water. Excess ozone that is present as on off-gas is redirected towards a titanium oxide ozone destructor followed by an air pump, in order to prevent ozone breakthrough into the columns.

Ozone dosing is calculated based on the voltage of the unit and the flow rate of gas through the unit. Figure 3.4 shows the performance curve for ozone output for this generator, provided by Pacific Ozone. The calculations can be seen in Appendix F.

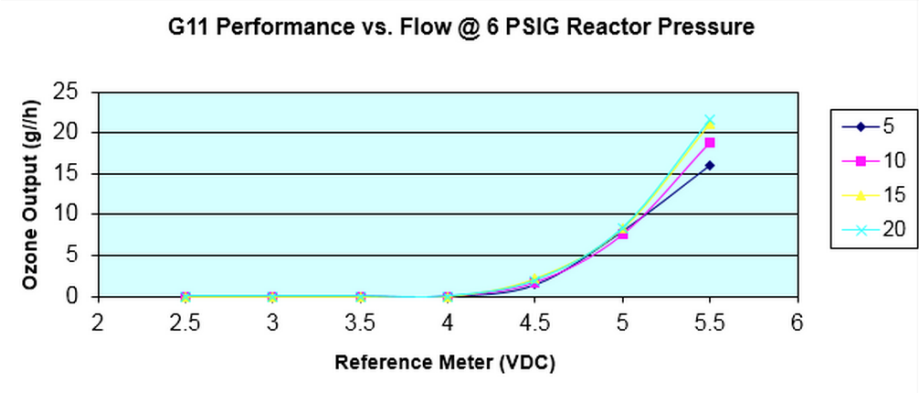


Figure 3.4: Ozone reference graph for calculation dosage.

The use of the Teflon-coated tubing that the ozone was sent through to reach the venturi meters was a constant source of issues throughout the operation of the trailer. The ozone was too strong for the tubing, and would attack weaker spots within the lines, causing leaks in the pilot system. In addition, it was difficult to locate the source of a leak. This in turn led to periods of time where ozone was not being dosed into the filters at the proper rate, affecting the performance of both trains.

3.2 Site Locations

The research took place at two separate sites. Phase 1 and 2 occurred at the Ottawa River, in Ottawa, Ontario. The pilot was set up at the Britannia Beach Water Treatment Plant (BBWTP), and received raw feed water from the plant. Phases 3, 4 and 5 occurred at Lac Chassignole, in the Abitibi-Témiscamingue region of Quebec. The pilot was set up at the Agnico-Eagle Laronde Mine (AELM), and received raw water directly from the lake.

3.2.1 Ottawa River

The pilot trailer was located outdoors at the Britannia Beach Water Treatment Plant. The overall raw water characteristics for the Ottawa River site are provided in Table 3.3.

Table 3.3: Raw water characteristics from Ottawa River (provided by BBWTP).

Raw Water Quality Overview (2013)			
		Average	**Range**
Physical	Temperature (^{o}C)	9.9	0 - 26.1
	Turbidity (NTU)	3.4	1.2 - 10.5
	Colour (TCU)	42	32 - 61
Chemical	Dissolved Organic Carbon (mg/L)	7.2	6.7 - 8.1
	Total Alkalinity (mg/L $CaCO_3$)	28	15 - 44
	pH	7.35	6.9 - 7.66
	UVA_{254} (abs/cm)	0.28	0.23 - 0.38
	Total Hardness (mg/L $CaCO_3$)	31.8	22.7 - 40.9
Microbiological	Total Coliform (cfu/100mL)	821	8 - >2420
	E. coli (cfu/100mL)	25	0 – 614
C:N:P Ratio	Carbon (as DOC, mg/L)	7.10	N/A
	Nitrogen (as TKN, mg/L)	0.31	N/A
	Phosphorous (as TP, mg/L)	0.013	N/A

The average concentration of carbon (as DOC), nitrogen (as TKN) and phosphorous (as TP) was used to calculate the C:N:P ratio of the water. Based on the data in Table 3.3, the ratio was calculated to be 546:24:1 (w/w). Two testing phases occurred at this site, one during the fall of 2013 (Phase 1), and the second during the summer of 2014 (Phase 2). Due to the fact that the pilot was located outdoors, it was not in operation during the winter of 2014 because of freezing problems that had occurred in previous studies.

3.2.2 Lac Chassignole

Agnico-Eagle expressed interest in using the pilot to determine its ability in providing safe drinking water for approximately 1,000 employees at the Laronde Mine, located in

the Abitibi- Témiscamingue region of Quebec. The pilot was located outdoors at the Laronde Mine for the first two and a half months of testing (August 2014 - October 14). It was then moved indoors for the remainder of the study to avoid any issues with freezing, until January 2015. The overall raw water characteristics for the Lac Chassignole site are provided in Table 3.4.

Table 3.4: Raw water characteristics for Lac Chassignole (provided by AELM).

Raw Water Quality Overview (2014)			
		Average	**Range**
Physical	Temperature (^{o}C)	17.26	13 - 20
	Turbidity (NTU)	6	4.6 - 7.6
	Colour (TCU)	72	36 - 117
Chemical	Dissolved Organic Carbon (mg/L)	12	6-17
	Alkalinity (mg/L $CaCO_3$)	17	N/A
	pH	6.6	6.31 - 7.18
	UVA_{254} (abs/cm)	0.555	0.33 - 1.5
	Total Hardness ($CaCO_3$)	N/A	N/A
Microbiological	Total Coliform (cfu/100mL)	8	2 - 16
	E. coli (cfu/100mL)	10	N/A
C:N:P Ratio	Carbon (as DOC, mg/L)	11.6	N/A
	Nitrogen (as N, mg/L)	3	N/A
	Phosphorous (as TP, mg/L)	1.83	N/A

The average concentration of carbon (as DOC), nitrogen (as TKN) and phosphorous (as TP) was used to calculate the C:N:P ratio of the water. Based on the data in Table 3.4, the ratio was calculated to be 6.3:1.64:1 (w/w). Three testing phases occurred at this site.

3.3 Operation of Trailer

Proper operation of the trailer was a high priority during this research. Daily checks such as measuring temperature and water height inside the columns, routine inspections and

maintenance were necessary to ensure the trailer remained in working conditions for a long period of time. A weekly backwash procedure on Train 2 was necessary to avoid any clogging of the filter material. In addition, the slow sand filter was backwashed a total of four times throughout the experimentation period. These are expanded further on in this section.

3.3.1 Daily Operations

Daily measurements were taken in order to have a good indication of the level of operation of the trailer. This included noting the temperature, daily turbidity readings, the pressure of influent water, the flow rates of both trains, the ozone output, and the head loss inside the slow sand filter column and both biofilter columns (measured as the difference between the column height and the top of the water in the column).

3.3.2 Backwash

As explained in the literature review, it is necessary to backwash the biofilter columns at a frequent rate to prevent clogging of the media, which may lead to inefficient removals. During phase 1, the backwash procedure used tap water supplied from the BBWTP (which had a low chlorine residual of 0.06 mg/L Cl_2 (free chlorine)) which may have affected the media bed. During phase 2, the backwash procedure was rearranged so that no chlorine would be present in the water during a cleaning. Instead, the incoming raw water was used to clean the columns. This procedure was continued during phases 3, 4 and 5.

The BF backwash procedure that was used was roughly the same throughout the entire experimentation period. The columns were drained to media height before backwashing. The backwash flow rates and duration are presented in Table 3.5. The reason for the different backwash rates is due to the maximum raw water flow that was available from each site. The maximum raw water flow was a limitation during the backwash procedure, as a higher rate was theoretically required to provide appropriate cleaning of the biofilters (approximately 50 L/min with a bed expansion of 30%).

Table 3.5: Biofilter backwash procedure for each site.

Sample Site	Backwash Rate	Backwash Time
Site 1	Low - 18 L/min High - 24 L/min Low - 18 L/min	7 minutes 2 minutes 7 minutes
Site 2	Low - 15 L/min High - 30 L/min Low - 15 L/min	5 minutes 15 minutes 2 minutes

Slow sand filters do not need to be cleaned as often as biofilters and can in fact last anywhere from 6 months to a year before the head loss becomes too much. However, it was necessary to backwash the slow sand filter a total of three times at Site 2 due to head loss development (once during operation at Site 1, and three times during operation at Site 2). The procedure also used incoming raw water, but the backwash occurred at a much lower rate than that for the BFs (maximum of 15 L/min) and lasted until the outgoing water appeared relatively clean (approximately 15 to 20 minutes).

3.4 Analytical Methods

3.4.1 Turbidity

Turbidity was measured using a HACH PS1201 Low Range Turbidimeter. The turbidity meter constantly monitored the influent and effluent for both Train 1 and Train 2 inside the trailer. The range of measurement was between 0 and 100 NTU, and a minimum flow rate of 0.25 L/min was required. A measurement was collected every 15 minutes for the raw water influent and the effluent from each train, and compiled electronically for data analysis.

3.4.2 Colour

Colour measurements were done in accordance with Standard Methods 2120C Spectrophotometric Method (APHA, 2012). Both apparent and true colour were measured, where true colour represents the filtered sample. Samples were filtered through a 0.45 µm filter paper from Millipore prior to analysis.

3.4.3 Dissolved Organic Carbon (DOC) & Total Organic Carbon (TOC)

DOC and TOC concentrations were measured in accordance with Standard Methods 5310C (APHA, 2012). A Shimadzu TOC Analyzer was used to analyze the concentrations.

The difference between DOC and TOC is that DOC samples have been filtered, therefore removing any undissolved solids. Samples were filtered through a 0.45 µm filter paper from Millipore prior to analysis.

3.4.4 Ultraviolet Absorbance (UVA) & Specific UVA (SUVA)

Ultraviolet absorbance (UVA) was measured at 254 nm in accordance with Standard Methods 5910B (APHA, 2012). A Cary 100 Bio UV-Visible Spectrophotometer with a 1 cm pathlength quartz cuvette was used. Samples were filtered through a 0.45 µm filter paper from Millipore prior to analysis.

SUVA was calculated based on the normalization of UVA_{254} by DOC. The calculation is seen below.

$$SUVA = \frac{UVA_{254}}{DOC} * 100$$

Where,

SUVA = the specific UV absorbance (L/mg*m)

UVA_{254} = absorbance at 254 nm (cm^{-1})

DOC = dissolved organic carbon content (mg/L)

3.4.5 Microbiological Parameters

Total coliforms, E.coli and enterococci were monitored throughout experimentation during phases 1 and 2. These measurements were not done at Site 2 due to the low influent values in Lac Chassignole.

3.4.5.1 Total Coliforms and E. coli

Total coliforms and E.coli were analysed using the Standard Methods 9223B Quanty-Tray 2000 Method (APHA, 2012). IDEXX Colisure packets were used to perform this

method. 100 mL samples were treated with Colisure, an enzyme substrate, and allowed to incubate for 24 hours at 35 ^{o}C. Once the trays were incubated, the number of cells that changed to a magenta colour were counted, and based on a conversion chart this number corresponds to the equivalent amount of total coliforms present in the water. The tray was then placed under a UV light source, and the number of fluorescing cells were counted. Based on a conversion chart, this number corresponds to the amount of E.coli present in the water in terms of CFU per 100 mL.

3.4.5.2 Enterrococci

Enterrococci was analyzed using the Standard Methods 9230D Fluorogenic Substrate Enterococcus Test (APHA, 2012). IDEXX Enterolert packets were used to perform this method. The sample presentation is similar to that of Colisure, with the exception being incubation being controlled at 41 ^{o}C for 24 hours. After the incubation, the number of fluorescing cells is directly counted, and based on a conversion chart this number corresponds to the number of enterococci present in the water in terms of CFU per 100mL.

3.4.6 Biomass Analysis (ATP)

The viable biomass was analyzed in terms of adenosine triphosphate (ATP). Media samples were collected from the top 5 cm each column and allowed to dry for several hours. ATP was analyzed using the LuminUltra Measured Deposit Procedure. The procedure is a direct and interference-free indicator of total living biomass. It involves introducing the sample containing ATP to a solution with the enzyme Luciferase. This in

turn will produce light, which is detected by a luminometer and provided as a reading of ATP in pg ATP/g sample.

3.4.7 Trihalomethanes (THMs)

Trihalomethanes were analyzed at Site 2 only using the HACH THM Plus Method 10132. The method provides an indication of THMs present as chloroform ($CHCl_3$).

3.4.8 Temperature

A standard handheld thermometer was used to measure the raw water temperature daily. The water was collected in a beaker, and allowed to run through the line for approximately 30 seconds to ensure that the temperature was representative of the actual raw water.

3.4.9 pH

Throughout the duration of the experiments, pH was measured using an Accumet pH Meter 50. Measurements were taken periodically to ensure that the raw water pH was in the range of 6-7.

3.4.10 Ozone

Ozone residual was tested regularly in each train. Residual was analyzed using an Ozone Vacu-Vials Kit (K-7423). Samples were taken at different depths of the ozone contactors, at the outlet of the roughing filter, and at the outlet of the columns.

3.4.11 Iron (Fe) and Manganese (Mn)

Iron and manganese samples were taken during phases 3, 4 and 5 of the research and analyzed externally by Agnico-Eagle employees.

3.4.12 Nitrogen

Nitrogen samples were taken during phases 3, 4 and 5 to verify that there would be enough concentration to promote growth of biomass. Total nitrogen in the raw water was analyzed using the HACH Persulfate Digestion Method (Method 10072).

3.4.13 Phosphorous

Phosphorous samples were taken during phases 3, 4 and 5 to verify that there would be enough concentration to promote growth of biomass. Total phosphorus in the raw water was analyzed using the HACH USEPA PhosVer 3 with Acid Persulfate Digestion Method (Method 8190).

3.4.14 Chemical Oxygen Demand

Filtered and unfiltered chemical oxygen demand were tested during phases 3, 4 and 5. For the filtered samples, the 0.45µm Millipore filter paper was used. COD was analyzed using the HACH Chemical Oxygen Demand Method 8000.

3.5 Quality Control and Assurance

3.5.1 Glassware

Due to the low concentrations of all of the parameters, the glassware was cleaned prior to use to minimize any potential contamination. Vials used for DOC, TOC and UV

analysis were put in an acid bath for 24 hours, and then baked in a Vulcan A-550 oven at 400 ^{o}C for one hour.

3.5.2 Sample Collection and Storage

Samples were collected from each sampling point and stored until further analysis. During sample collection, water was passed through to the waste stream for approximately 30 seconds to one minute in order to ensure a representative sample from each column. Samples were then collected in 1 L bottles, which were meticulously rinsed beforehand to avoid contamination. Samples were stored in the fridge until analysis, and if samples were stored for more than 4 hours nitric acid was added until the pH of the sample reached 2. All samples were analyzed within a week.

3.6 Experimental Design

3.6.1 Sampling Regime Ottawa River

Table 3.6 shows the sampling regime that was used at the Ottawa River site.

Table 3.6: Sampling frequency and point for Ottawa River site.

Parameter	Frequency	Sampling Point
DOC	Twice a week	RW, RF1, SSF, BAC1, BAC2
TOC	Twice a week	
UVA	Twice a week	
True Colour	Once a week	
E.coli	Once a week	
Enterrococci	Once a week	
Total Coliform	Once a week	
Turbidity	Once a week	RW, Train 1, Train 2
pH	Once a week	
ATP	Once a week	SSF, BAC1, BAC2
Temperature	Once a day	RW
Backwash	Twice a week	BAC1 & BAC2

*RW = raw water, RF1 = roughing filter on Train 1

3.6.2 Sampling Regime Lac Chassignole

Table 3.7 shows the sampling regime that was used at the Lac Chassignole site.

Table 3.7: Sampling frequency and point for Lac Chassignole location.

Parameter	Frequency	Sampling Point
DOC*	Twice a week	RW, SSF, BAC1, BAC2
TOC*	Twice a week	
UVA	Twice a week	
COD	Twice a week	
True Colour	Twice a week	
THM	Once a week	RW, Train 1, Train 2
Nitrogen	Once every two weeks	RW
Phosphorus	Once every two weeks	
Turbidity	Once a week	RW, Train 1, Train 2
pH	Once a week	
ATP	Once a week	SSF, BAC1, BAC2
Temperature	Once a day	RW
Backwash	Once a week	BAC1 & BAC2

* Due to instrument availability, DOC and TOC tests were not carried out until November 2014

3.7 Statistical and Data Analysis

Statistical analysis was performed in the form of a student's t-test at a 95% confidence interval. The statistical test was used to compare the actual difference between two means in relation to the variation in the data. The t-test was performed using Microsoft Excel software. Average, maximum, minimum and standard deviations of sample ranges were determined. Error bars in the figures represent standard deviation.

4 Case Study of a Multi-Stage Filtration Pilot for Remote and Northern Communities in Canada

Chapter 4 will be presented as a journal article. Therefore, this section is presented in the format of a journal paper, starting with an abstract and introduction. A shortened version of the materials and methods section seen in Chapter 2 will be presented here, followed by a discussion of results relevant to this thesis. This journal chapter will be submitted to the Water Quality Research Journal of Canada published by co-authors Celia Charron (student), Dr. Onita Basu (supervisor) and Shawn Cleary (co-investigator).

Abstract

A pilot scale multi-stage filtration system with two treatment trains (slow sand filtration (SSF) and biofiltration (BF)) was used to study the impacts of source water quality and ozone on system performance. The pilot trailer was operated at two separate sites: one with a phosphorous nutrient limited source water (C:N:P of 546:24:1 w/w) and one with a nutrient rich source water (C:N:P of 6.3:1.6:1 w/w). Each train consisted of pre-ozonation and roughing filtration ahead of the final filtration step. A range of ozone doses (0 mg O_3/mg DOC, 1.55 mg O_3/mg DOC, 1.9 mg O_3/mg DOC, 3.45 mg O_3/mg DOC) were investigated. Nutrient limited conditions exhibited lower DOC removals than nutrient rich conditions when ozone was applied (8.3% SSF and 10.3% BF vs 26.1% for both BF and SSF, respectively). However, there was no difference in removal between the two sites when no ozone was present (5.0% SSF and 6.3% BF vs 7.1% SSF and 5.7% BF, respectively). Similar trends were seen with UVA_{254} removal. These findings highlight

the importance of source water characterization and pilot testing when attempting to utilize the benefits of biofiltration and slow sand filtration for water treatment.

4.1 Introduction

The global need for clean drinking water continues to increase due to environmental contamination from industrial and agricultural sources, pharmaceutical and personal care products, as well as continuing global population pressures. Common water contaminants include: 1) chemical contaminants such as heavy metals, natural and synthetic organic compounds; and 2) microbiological contaminants including fecal coliforms, E.coli, enterococci and protozoa. Several events have occurred in the past that have put an emphasis on safe drinking water in small communities in Canada. Perhaps the most well-known event is the tragedy of Walkerton in May of 2000. Following this, a provincial inquiry led to 93 recommendations for upgrading drinking water treatment. Several of the recommended actions highlighted issues faced by First Nations communities (O'Connor, 2002). This inquiry identified three essential requirements for the provision of safe drinking water: performance, operation and maintenance, and cost-efficiency (O'Connor, 2002). For small systems, it is essential to find a balance between all three of these requirements.

A promising method of providing safe drinking water for these communities is multi-stage filtration, which uses pre-ozonation and roughing filtration ahead of filtration columns; such as slow sand filters and biofilters. Slow sand filters (SSF) have fine particles of sand as their media and low filtration rates (0.05 – 0.5 m/hr) whereas biofilters (BF) use GAC, anthracite, sand or a combination of all three and have higher

filtration rates (up to 100 times greater than SSF). Attached biomass is present in the media beds of both filters. In an SSF, a biological layer called the *schmutzdecke* provides the majority of biological removal, and is typically located within the first 5 cm of the filter bed. In a BF, the biomass is present within the entire depth of the media bed (Hozalski & Bouwer, 2001). The combination of roughing filtration, pre-ozonation and slow sand filtration is a viable and sustainable treatment system for remote communities. Previous research on this system has demonstrated effective short term treatment in seasonal cold water conditions (temperatures ranged from 2 to 20 ^{o}C), and demonstrated low operation and maintenance requirements (Cleary et. al., 2008). However, the use of biofiltration in place of slow sand filtration can be advantageous for small and remote communities due to the lower area footprint, and therefore lower expected heating costs (compared with SSF).

Roughing filters and pre-ozonation treatment have shown promising results in communities where balance between performance and operation and maintenance is difficult to achieve. Roughing filters are placed ahead of conventional type filtration systems, and have been shown to reduce solids loading ahead of filters due to their design (Gottinger et. al., 2011; Cleary et. al., 2008; Pacini et, al., 2005). In addition, roughing filters increase the run length and hydraulic retention time of drinking water filters (Cleary et. al., 2008).

Pre-ozonation ahead of filtration units has been shown to meet federal regulations for primary disinfection, turbidity, taste and odour compounds and biodegradable organic

carbon (BDOC), and has been shown to produce biologically stable drinking water (Schulz, 2014). These improvements are more pronounced in cold temperatures, an advantage for ozonation in northern communities in Canada. Pre-ozonation partly degrades natural organic matter (NOM) into readily biodegradable organic constituents. The advantage of having this ahead of filtration units is that the biodegradation of organics within the surface and depth of the filter media will improve due to the increase in the biodegradability of the NOM. This in turn decreases the passage of organic matter into the distribution system. Having organic matter in the distribution system can lead to an increase in disinfection by-products (DBPs) which cause health problems (Yapsakli & Cecen, 2010). Pre-ozonation causes substantial structural changes of the organic matter, leading to a decrease in colour, UVA_{254}, TOC, DOC and the high molecular weight fraction of NOM (Yapsakli & Cecen, 2010). In addition, ozone can oxidize taste compounds, odour compounds, iron and manganese, and can reduce the formation of hazardous chlorination by-products including THMs and HAAs (Yapsakli & Cecen, 2010; Chen & Wang, 2012).

The primary objective of this research was to perform a side-by-side comparison of slow sand filtration versus biofiltration under variable nutrient conditions and ozone dosages. In this study two parallel treatment trains were assessed, and their performance was monitored from an organics removal perspective which included DOC, UVA_{254}, and true colour removal. The research is intended to improve current understanding of the potential impacts of the ozone doses and source water nutrient availability on biological filter performance.

4.2 Methods and Materials

4.2.1 Pilot Plant Description and Setup

A pilot trailer consisting of two treatment trains was used throughout this research. The layout of each train can be seen in Figure 4.1. In Train 1, raw water passed through an ozone contactor at a variable flow rate of 3 to 5 L/min, followed by a peristaltic pump. Immediately after the pump, water was sent to a roughing filter followed by a slow sand filter at a variable flow rate of 0.3 to 0.5 L/min. The remaining flow was sent directly to waste. The lower flow rate of slow sand filters is ideal for removing particles of all size, but can be a problem in terms of heating cost due to the larger area footprint of the filters. The slow sand filter had a diameter of 0.315 m with a total bed depth of 0.65 m. The uniformity coefficient (UC) and effective diameter of the sand were 1.7 and 0.35 mm respectively. In Train 2, raw water passed through an ozone contactor at a variable flow rate of 3 to 5 L/min, followed by a roughing filter and two identical biologically activated carbon (BAC) filters in parallel. Each BAC filter had a diameter of 0.315 m, and an approximate media height of 0.7 m. The effective diameter of the GAC was 0.7 mm. The EBCT for each BAC filter was 11 minutes. This EBCT is on the lower end of ranges typically seen in literature, which can be between 5 minutes and 30 minutes (Ko et. al., 2007). The BF columns were backwashed once to twice a week, depending on turbidity values and head loss (i.e. if turbidity values were consistently around 3NTU, or if the head loss increased drastically throughout weekly operation). The slow sand filter column was backwashed a total of 4 times throughout this study.

Ozone was generated onsite using an O-series Pacific Ozone generator. The ozone mixed with raw water before entry into the ozone contactor through the use of venturi meters. The contactors were filled with PVC saddles, and excess ozone was redirected towards a titanium oxide ozone destructor followed by an air pump to help exhaust the line.

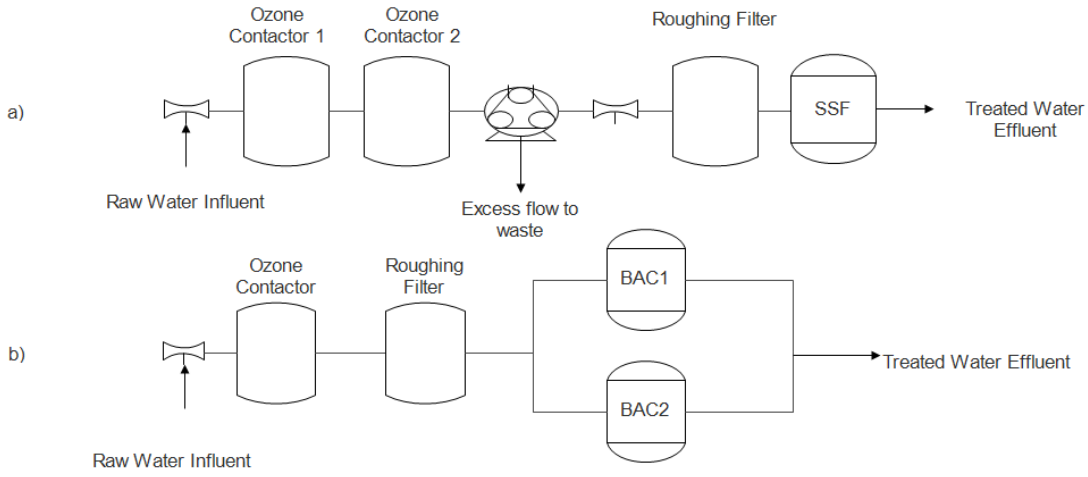


Figure 4.1: Experimental Set-Up. a) Train 1 flow diagram b) Train 2 flow diagram.

4.2.2 Collection and Storage

The pilot trailer was operated continuously over five study phases. Phase 1 occurred from September 2013 to December 2013 (88 days), phase 2 occurred from May 2014 to July 2014 (28 days), and phases 3, 4 and 5 (total of 119 days) occurred from September 2014 to January 2015. Routine measurements were taken daily, including turbidity, temperature, head loss (measured as a difference between top of column and height of water) and average incoming flow rate. Water samples were collected periodically from four different sample points: raw water entry into the trailer, after the slow sand filter, after BAC1 and after BAC2. These samples were analyzed for TOC/DOC, true colour, COD

and UVA_{254}. Samples were either analyzed immediately after collection, or refrigerated and analyzed within a week. In addition, biomass samples were taken from the top 5 to 10 cm of each filter bed and analyzed for ATP.

4.2.3 Analytical Methods

TOC and DOC concentrations were measured in accordance with Standard Methods 5310C. A Shimadzu TOC Analyzer was used to analyze the concentrations. In order to prepare DOC samples, the collected water samples were filtered through a 0.45 µm Millipore filter paper with vacuum filtration prior to analysis.

Ultraviolet absorbance (UVA) was measured at 254 nm in accordance with Standard Methods 5910B. A Cary 100 Bio UV-Visible Spectrophotometer with a 1 cm pathlength quartz cuvette was used. Samples were filtered through a 0.45 µm filter paper from Millipore prior to analysis.

Specific UV absorbance (SUVA) was calculated based on the UVA_{254} and the DOC. The calculation is seen below.

$$SUVA = \frac{UVA_{254}}{DOC} * 100 \qquad \text{(Eq. 4.1)}$$

Where,

SUVA = the specific UV absorbance (L/mg*m)

UVA_{254} = absorbance at 254nm (cm^{-1})

DOC = dissolved organic carbon content (mg/L)

Colour measurements were done in accordance with Standard Methods 2120C Spectrophotometric Method. Both apparent and true colour were measured, where true colour represents the filtered sample. Samples were filtered through a 0.45 µm filter paper from Millipore prior to analysis.

The viable biomass was analyzed in terms of adenosine triphosphate (ATP). Media samples were collected from the top 5 cm each column and allowed to dry for several hours. ATP was analyzed using a LuminUltra Measured Deposit Procedure. The procedure is a direct and interference-free indicator of total living biomass, in which a sample containing ATP is introduced to a solution containing the enzyme Luciferase to produce light detected by the luminometer.

Turbidity data was collected every 15 minutes throughout operation at each phase through an online HACH PS1201 Low Range Turbidimeter.

4.2.4 Site Locations

The research took place at two sites. Site 1 (nutrient limited site) was located at the Ottawa River, in Ottawa, Ontario. The pilot trailer was set up at the Britannia Beach Water Treatment Plant (BBWTP), and received raw feed water from that plant. Site 2 (nutrient rich site) was located at Lac Chassignole, in the Abitibi-Témiscamingue region of Quebec. The pilot was set up at the Agnico Eagle Laronde Mine, and received raw water directly from the lake. Tables 4.1 and 4.2 show the raw water quality for each site. The research was conducted in phases with various ozone dosages and nutrient conditions (determined by site location). The research phases are described in Table 4.3.

Table 4.1: Raw water characteristics from Ottawa River (Provided by BBWTP).

		Average	Range
Physical	Temperature (^{o}C)	9.9	0 - 26.1
	Turbidity (NTU)	3.4	1.2 - 10.5
	Colour (TCU)	42	32 - 61
Chemical	Dissolved Organic Carbon (mg/L)	7.2	6.7 - 8.1
	Total Alkalinity (mg/L $CaCO_3$)	28	15 - 44
	pH	7.35	6.9 - 7.66
	UVA_{254} (abs/cm)	0.28	0.23 - 0.38
	Total Hardness (mg/L $CaCO_3$)	31.8	22.7 - 40.9
Microbiological	Total Coliform (cfu/100mL)	821	8 - >2420
	E. coli (cfu/100mL)	25	0 – 614

Table 4.2: Raw water characteristics from Lac Chassignole (Provided by AELM).

		Average	Range
Physical	Temperature (^{o}C)	17.26	13 - 20
	Turbidity (NTU)	6	4.6 - 7.6
	Colour (TCU)	72	36 - 117
Chemical	Dissolved Organic Carbon (mg/L)	12	6 - 17
	Alkalinity (mg/L $CaCO_3$)	17	N/A
	pH	6.6	6.31 - 7.18
	UVA_{254} (abs/cm)	0.555	0.33 - 1.5
	Total Hardness ($CaCO_3$)	N/A	N/A
Microbiological	Total Coliform (cfu/100mL)	8	2 - 16
	E. coli (cfu/100mL)	10	N/A

Table 4.3: Research phases and operating conditions.

Phases	Operating Conditions	
	Ozone Dose (mg O_3/mg DOC)	C:N:P Ratio (w/w)
1	1.55	546:24:1
2	0	546:24:1
3	1.9	6.3:1.6:1
4	3.45	6.3:1.6:1
5	0	6.3:1.6:1

4.3 Results

The results are presented first as an individual discussion on the respective research sites followed by a comparison of the two sites.

4.3.1 Site 1 – Brittannia Water Treatment Plant

The source water for Site 1 came from the Ottawa River, with a C:N:P ratio of 546:24:1 (w/w). Although there is some variation in the expected C:N:P ratio of heterotrophic bacteria, the most commonly accepted ratio is 100:10:1 (mol/mol) or equivalently 39:4.5:1 (w/w) (Lechevallier et. al., 1991). Therefore the source water from Site 1 is considered phosphorous-limited, which may have reduced the biological activity throughout the media beds of both the slow sand filter and the biofilters in the pilot train. Phase 1 ended on December 13, 2013 and phase 2 began on May 28, 2014 (as seen in Figure 4.2). In between these dates, the trailer was not in operation.

4.3.1.1 DOC Results

Natural organic matter in source waters is problematic as it is a precursor to DBPs. Due to the complexity and variation of NOM in different source waters, a number of analyses such as DOC, UVA_{254}, SUVA and true colour can provide insight into its characteristics (Chowdhury et. al., 2008).

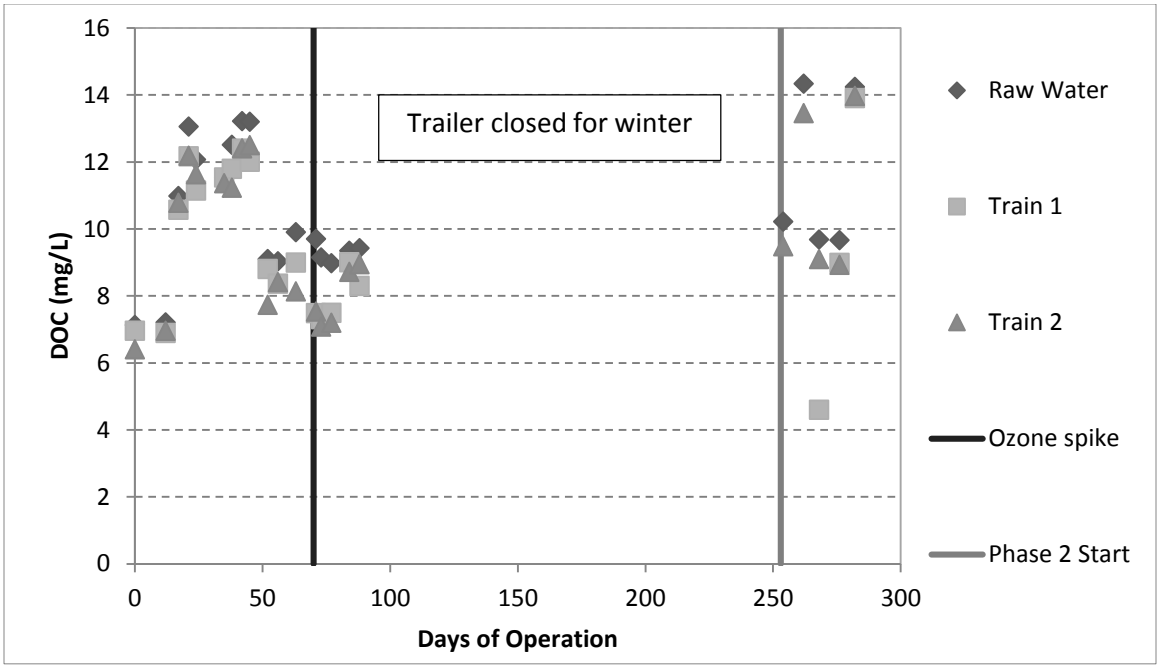


Figure 4.2: DOC values for phase 1 (nutrient limited and 1.55 mg O_3/mg DOC) and phase 2 (nutrient limited with no ozone).

The average temperature for phase 1 was 14 ^oC and the trailer was operated from September to December (although it is noted that the pilot trailer was allowed to run for two months prior for biological acclimation purposes). The first major temperature drop occurred on day 35, from 20 ^oC to 10 ^oC, and the temperature continued to drop afterwards. It was expected that DOC removal values would reach a steady-state over the course of the experiment (Simpson, 2008), but this did not occur in either phase. However, pilot issues were encountered throughout operation at Site 1 that may have prevented steady-state removal.

The first vertical line in Figure 4.2 represents when there was an ozone spike, due to an operational error, into the system on day 72 (increase to 3 mg O_3/mg DOC). The spike lasted for 5 days, and residuals of 1.8 mg O_3/L and 2.17 mg O_3/L were observed in the contactors in Trains 1 and 2, respectively. After this event, the ozone dosing rate was

returned to 1.55 mg O_3/mg DOC. Figures 4.3 and 4.4 show the correlation between biomass concentration (measured as ATP) and DOC removals (plotted on secondary and primary axis, respectively) in both trains.

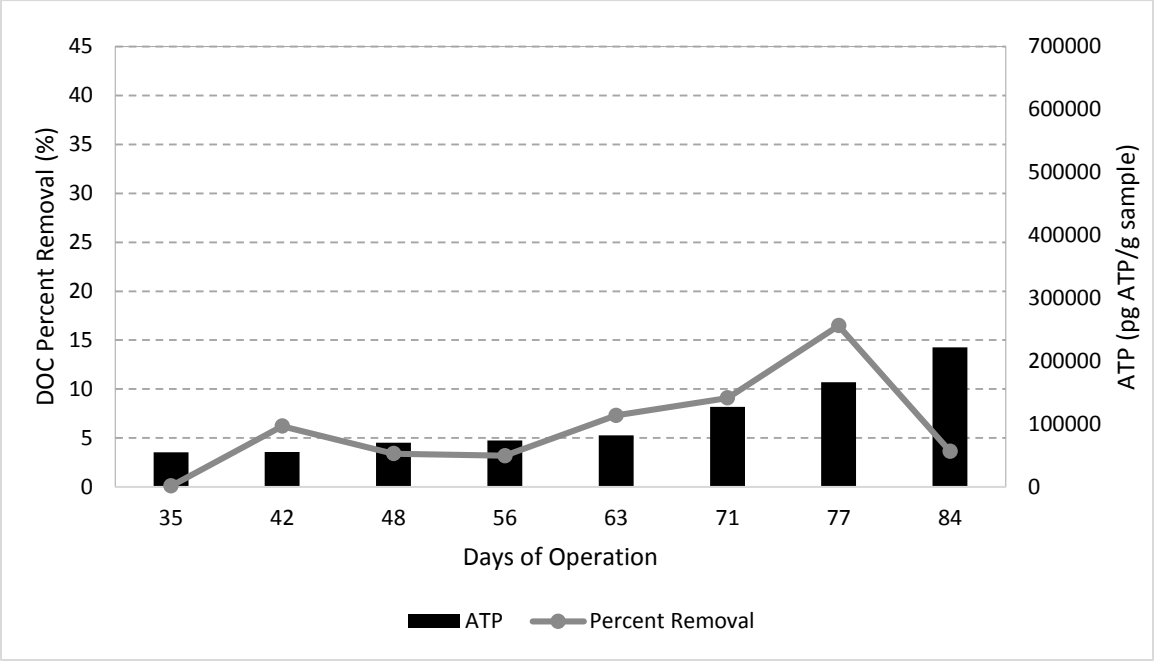


Figure 4.3: ATP concentration and DOC removal in Train 1 during phase 1 (nutrient limited with 1.55 mg O_3/mg DOC).

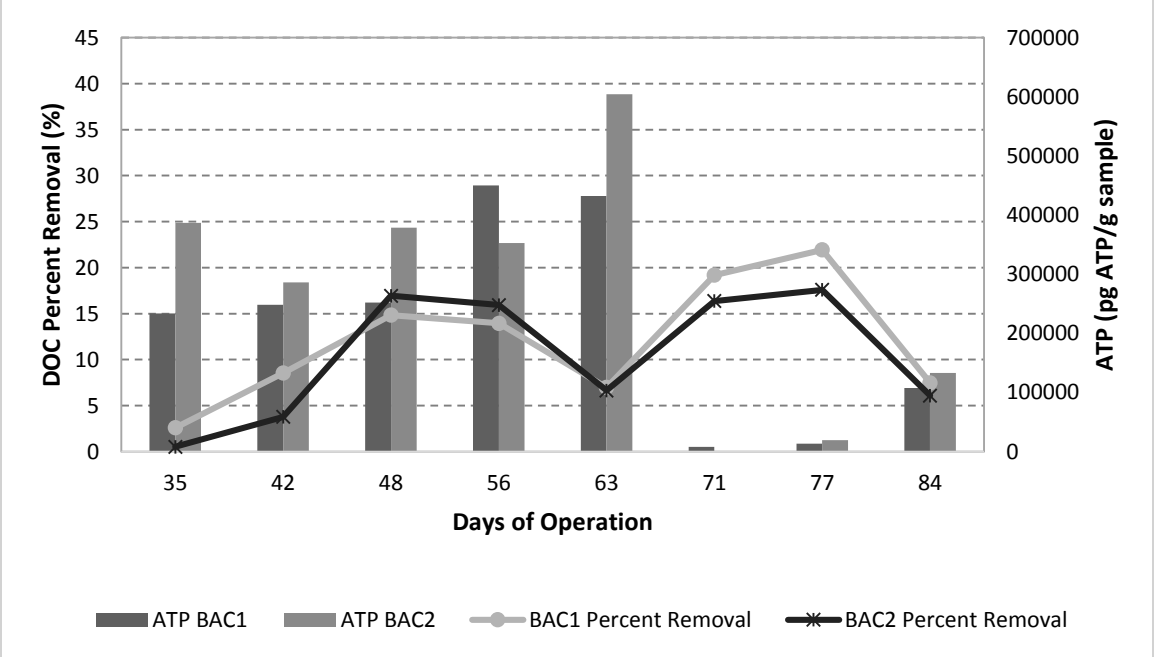


Figure 4.4: ATP concentration and DOC removal in Train 2 during phase 1 (nutrient limited with 1.55 mg O_3/mg DOC).

Between day 63 and 77, a significant drop in biomass concentration occurred in BAC1 and BAC2. The original drop from day 63 to day 71 was due to an unexpected plant shutdown that occurred on day 66, during which the filters ran dry. In addition, the

biofilters were both backwashed on day 70, which would have further affected the recovery of the biomass present in the filters. Although it was expected that the biomass concentration would recover after the backwash, an ozone spike on day 72 occurred immediately afterwards, and is responsible for the low ATP value on day 77. Interestingly, from day 63 to 77 there was an increase in DOC percent removal, which was unexpected after seeing the decrease in biomass concentration. ATP samples were collected from the top of the column; however, bacteria populate the entire media bed and thus it is likely that only the top portion of the bed was impacted by the excess ozone (Velten et. al., 2011). Therefore, although exposure to higher ozone dosages decreased biomass activity at the top of the filter bed, the bacteria within the depth of the filter likely remained unaffected. Consequently, better DOC removal occurred from days 72 to 77 due to the higher presence of readily biodegradable organics. This was also seen by Velten et. al. (2011), where the highest biomass concentrations were established 45cm from the top of the biofilters. The presence of residual ozone in the influent caused a decrease in biomass at the top of the filter, but the biomass was recovered further down the bed depth (Velten et. al., 2011).

From Figure 4.3, it appears that neither the plant shutdown nor the ozone spike affected biomass concentration in the slow sand filter. In fact, a gradual increase of ATP occurred throughout the duration of phase 1. One explanation is that the lower ozone residual (1.8 mg O_3/L SSF compared to 2.17 mg O_3/L BF) was not enough to affect the activity of the *schmutzdecke*. In addition, the lower filtration rate (0.3-0.5 L/min) and

corresponding longer contact time in the roughing filter in Train 1 caused a higher degradation of ozone compared to Train 2 (3 – 5 L/min).

After the spike, the dosing rate was returned to 1.55 mg O_3/mg DOC. The overall DOC removal for phase 1 was 8.27 ± 7% for Train 1 and 10.3 ± 8% for Train 2. These removals are lower than those found in literature for both biofiltration and slow sand filtration processes. For instance, Yan et. al. (2010) reported a 38% DOC reduction after an O_3/BAC process under an ozone dose of 1.5 mg O_3/L (0.4 mg O_3/mg DOC). In addition, Matilainen et. al. (2010) reported a 13% TOC reduction after an O_3/BAC process under a dose of 0.3 to 1.2 mg O_3/mg TOC. Furthermore, slow sand filtration has been shown to provide removals of 60 to 75% of organic matter, including DOC (Gottinger et. al., 2011). The lower values here may be due to the nutrient limited conditions encountered at Site 1.

The phase 2 DOC trend is seen in the second half of Figure 4.2. Phase 2 ran under unozonated conditions due to mechanical faults with the ozone generator. The average water temperature of phase 2 was 20 ^oC, versus 14 ^oC in phase 1. The overall DOC removal for phase 2 was 5.00 ± 4% for Train 1 and 6.34 ± 4% for Train 2. These removals are lower than those seen in Phase 1, despite being at higher temperatures; providing further evidence that ozone was beneficial in terms of DOC removal.

The nutrient-limited conditions of the source water were not ideal for achieving desired DOC removal. A C:N:P ratio of 546:24:1 (w/w) is phosphorous limited, which can be detrimental to microbiological growth on the filters. A previous study by Miettinen et.

al. (1997) found strong correlations between phosphorous and AOC availability, and determined that the presence of phosphorus in drinking water is what drives microbiological growth.

4.3.1.2 UVA_{254} and SUVA Results

In addition to DOC measurements, UVA_{254} and SUVA were investigated as characteristics of NOM. UVA_{254} represents the portion of NOM that is aromatic, and has been shown to cleave upon substantial ozone dosage (Pei et. al., 2007). Table 4.4 represents the average influent and effluent UVA_{254} values seen during phases 1 and 2.

Table 4.4: Average influent and effluent UVA_{254} values during phase 1 & 2 (nutrient limited at 1.55 mg O_3/mg DOC and without ozone, respectively).

	Raw Water Average ± SD (cm^{-1})	Train 1 Effluent Average ± SD (cm^{-1})	Train 2 Effluent Average ± SD (cm^{-1})
Phase 1	0.309 ± 0.041	0.257 ± 0.056	0.236 ± 0.055
Phase 2	0.324 ± 0.015	0.285 ± 0.012	0.275 ± 0.008

The overall UVA_{254} removal for phase 1 was 16.6 ± 16% for Train 1 and 23.5 ± 16% for Train 2. The overall UVA_{254} removal for phase 2 was 9.14 ± 6% for Train 1 and 13.7 ± 2% for Train 2. The higher removals following pre-ozonation (phase 1) were expected. A study by Pei et. al. found an increase in total UVA_{254} removal from 39% to 86% at an optimum ozone dosage of 0.85 mg O_3/L (0.45 mg O_3/mg TOC) (Pei et. al., 2007). A second study found an increasing UVA_{254} removal with increasing ozone dosages, which reached a maximum of 51% at a dosage of 4.4 mg O_3/L (1.2 mg O_3/mg DOC) (Yan et. al., 2010). In addition, Gottinger et. al. (2011) reported removals of 3 to 35% from slow sand filters. The lower removals seen in the present study are a result of the nutrient limited conditions (i.e. lack of phosphorous) found at Site 1.

SUVA is defined as the normalization of UVA_{254} by DOC concentration. A high SUVA can be indicative of a greater complexity of NOM in raw water, resulting from an increase in aromaticity and other unsaturated bonds which cause NOM to be less biodegradable (Pei et. al., 2007). Table 4.5 represents the average influent and effluent SUVA values seen during phases 1 and 2.

Table 4.5: Average influent and effluent SUVA values during phase 1 & 2 (nutrient limited at 1.55 mg O_3/mg DOC and without ozone, respectively).

	Influent Average ± SD (L/mg*m)	Train 1 Effluent Average ± SD (L/mg*m)	Train 2 Effluent Average ± SD (L/mg*m)
Phase 1	3.35 ± 1.1	2.96 ± 0.91	2.77 ± 0.94
Phase 2	3.14 ± 0.70	2.73 ± 0.90	2.86 ± 0.71

The average SUVA values for each phase at Site 1 were all below 4 L/mg*m, suggesting that the particles were low molecular weight (Yapsakli & Cecen, 2010). The lack of removal in each train corresponds with the low removal values seen for DOC and UVA_{254}, providing further proof of the negative effects of the nutrient limited water.

4.3.1.3 True Colour Results

The presence of NOM in drinking water can cause aesthetic problems including colour, taste and odour (Matilainen et. al., 2010). True colour represents an approximate measurement of all aromatic and aliphatic compounds present in source waters. Figure 4.5 shows the true colour trend during phases 1 and 2.

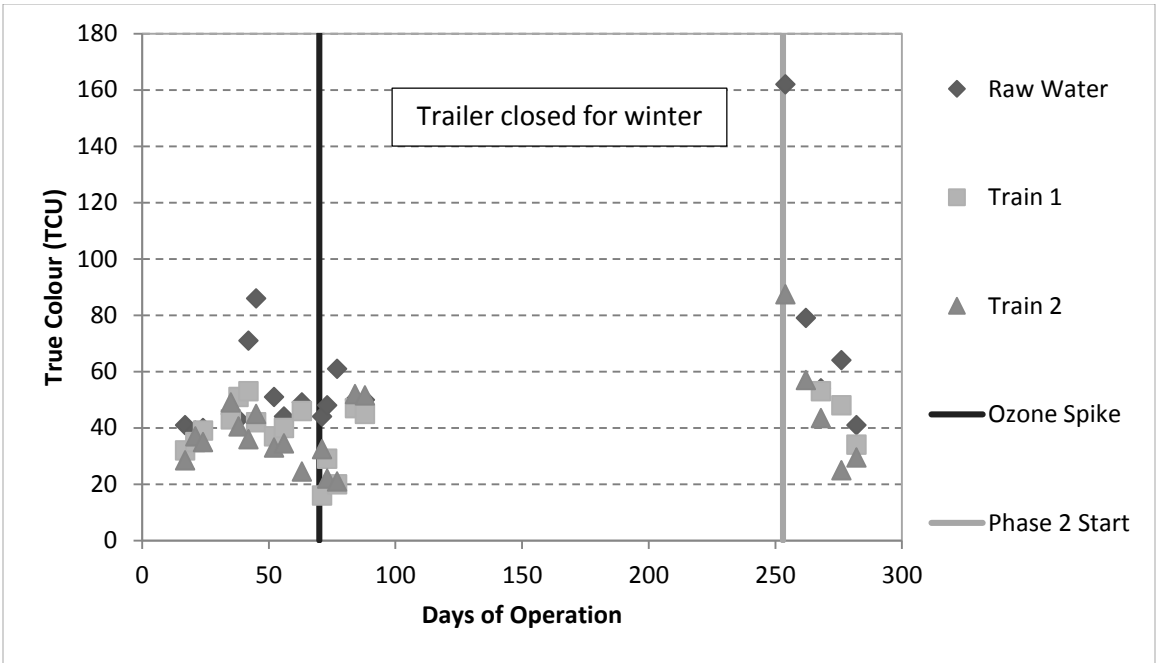


Figure 4.5: True colour values for phases 1 & 2 (nutrient limited at 1.55 mg O_3/mg DOC and without ozone, respectively).

A decrease in true colour concentration occurred immediately after the ozone spike for both trains, suggesting that the higher ozone dose (3 mg O_3/mg DOC) was instrumental in degrading NOM. A true colour of ≤ 15 TCU is the established objective in Canada (Health Canada, 1995). From Figure 4.5 it is evident that this objective was not met, and that both the SSF and BAC have similar true colour effluent values, indicating that neither system was advantageous for colour reduction at this site. A higher ozone dose (similar to that of the spike) may be necessary to reach the aesthetic objective.

4.3.1.4 Turbidity Results

On-line turbidity data (Figures 4.6 and 4.7) was collected throughout the experiment from influent and effluent turbidimeters. Raw water turbidity values ranged from 0.78 NTU to 100 NTU, with an average turbidity value of 2.78 NTU. During phase 1, the SSF

effluent turbidity ranged between 2 to 4 NTU, while the BF effluent turbidity ranged from 0.1 to 10 NTU during regular turbidity events.

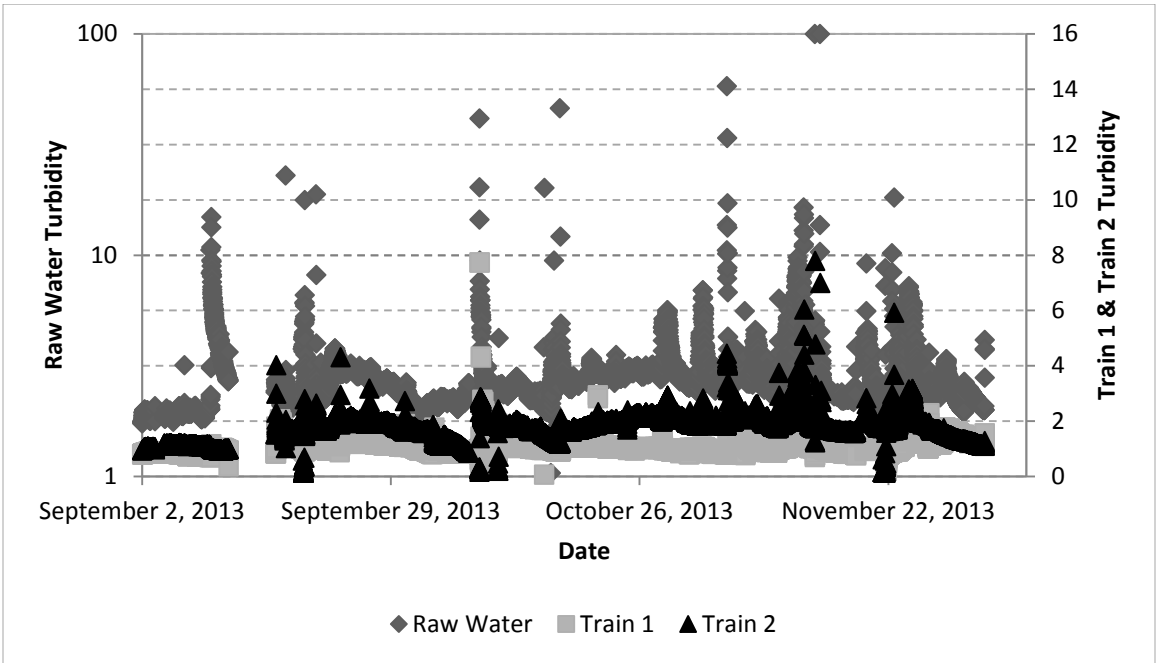


Figure 4.6: Turbidity trend for phase 1 (nutrient limited at 1.55 mg O_3/mg DOC).

During raw water turbidity breakthroughs seen in Figure 4.6, when the incoming turbidity was 100 NTU, the SSF performed better than the BF. The SSF maintained an effluent turbidity of 1 NTU, while the BF turbidity increased to over 10 NTU. This demonstrated that the SSF was more robust towards high influent turbidity. One reason for the fluctuating BF effluent values may have been the backwash procedure used. The calculated backwash rate necessary for the filters was approximately 50 L/min (30% bed expansion), but the highest backwash rate available at Site 1 was 24 L/min (3.7% bed expansion). Therefore, the BFs were not cleaned efficiently during operation at Site 1. In the future, steps should be taken to ensure that a proper bed expansion can be obtained. Otherwise, if a high backwash rate is not possible, a membrane system can be placed after the BFs to provide turbidity removal.

Raw water turbidity values during phase 2 ranged from 1.67 to 39.1 NTU, with an average turbidity value of 5.77 NTU. The SSF effluent turbidity ranged between 0.7 to 4 NTU, while the BF effluent turbidity ranged from 0.2 to 25 NTU, and the SSF was more robust to high influent turbidity.

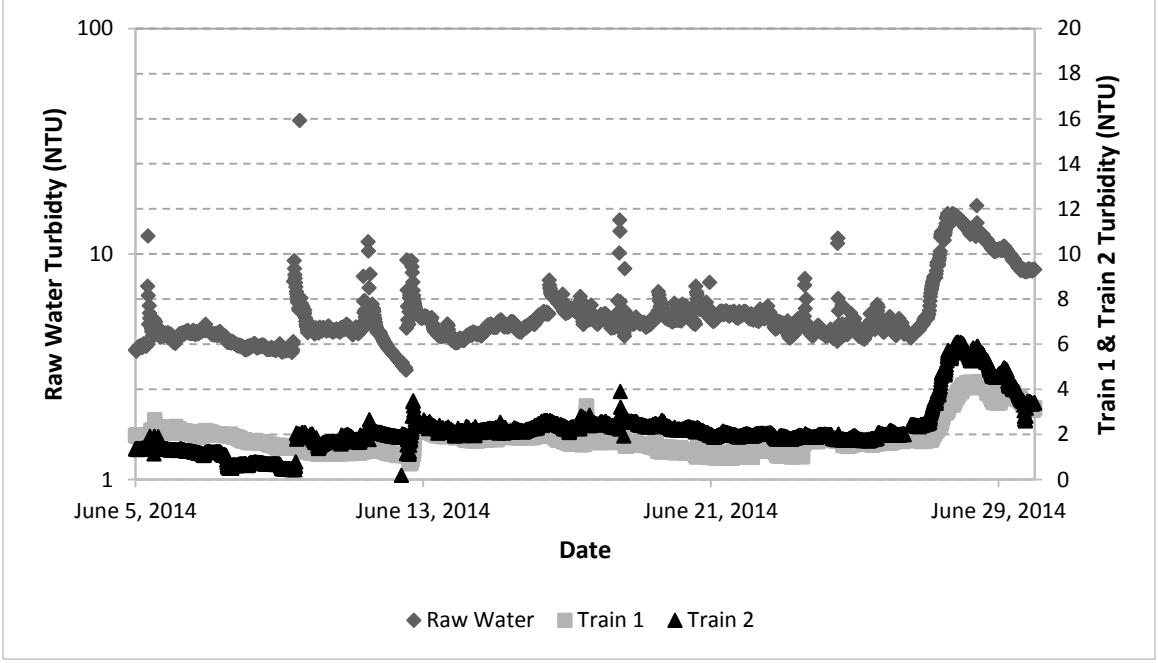


Figure 4.7: Turbidity trend for phase 2 (nutrient limited without ozone).

An operating filter must meet specific turbidity criteria depending on the location of the treatment plant. Health Canada guidelines recommend a treated water value of less than 1.0 NTU for slow sand filtration and 0.3 NTU for direct and conventional filtration (which applies to BFs) (Health Canada, 2013). Although turbidity removal occurred during phases 1 and 2, the effluent values remained over the regulation values. It should be noted that the purpose of this study was for organics removal, and reaching the turbidity criteria was not an objective.

4.3.2 Site 2 – Agnico-Eagle Laronde Mine

The second study site was the Agnico-Eagle Laronde Mine, with source water from Lac Chassignole. The C:N:P ratio for the raw water was 6.3:1.6:1 (w/w) on a mass basis. This is higher than the C:N:P ratio required for microbial cell growth (Lechevallier et. al., 1991), and is therefore considered nutrient-rich.

4.3.2.1 DOC Results

Phases 3 to 5 occurred during the months of August 2014 to January 2015, with an average raw water temperature of 16 ^{o}C. Due to the nutrient rich conditions of the source water it was expected that the Site 2 location would have higher overall removal of DOC compared to Site 1.

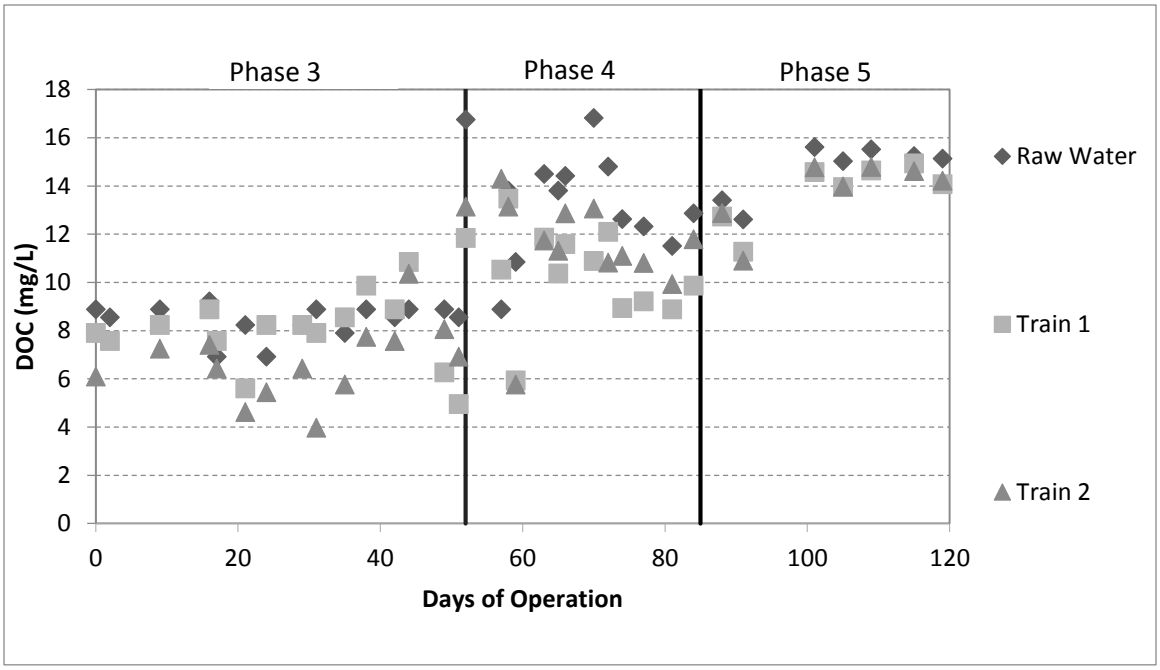


Figure 4.8: DOC values for phases 3, 4 & 5 (nutrient rich at 1.9 mg O_3/mg DOC, 3.45 mg O_3/mg DOC and with no ozone, respectively).

DOC versus time is shown in Figure 4.8 for the pilot trailer at site 2. Overall, it can be observed that the influent DOC increased over the 120 day time period of the piloting,

which would have hampered the ability of the filters to reach steady state. The overall DOC removal during phase 3 was 10.9 ± 10% for Train 1 and 26.1 ± 13% for Train 2. The overall DOC removal during phase 4 was 26.1 ± 11% for Train 1 and 17.8 ± 10% for Train 2. While during phase 5, the overall DOC removal dropped to 7.14 ± 3% for Train 1 and 5.73 ± 3% for Train 2. This was expected as phase 5 occurred during unozonated conditions and at a lower operational temperature of 11 ^{o}C, compared to 19 ^{o}C and 15 ^{o}C in the prior two phases.

The drop in removal in the BF between phases 3 and 4 was unexpected, and can be linked to a corresponding drop in biomass concentration. The ATP values for BAC1 and BAC2 were $1.2*10^6$ pg ATP/g sample and $6.8*10^5$ pg ATP/g sample respectively during phase 3. During phase 4, the ATP values dropped to $5.2*10^5$ pg ATP/g sample for BAC1 and $6.3*10^5$ pg ATP/g sample for BAC2. BAC1 and BAC2 were backwashed twice a week throughout operation due to high headloss and associated increased effluent turbidity; therefore, it is likely that the frequent backwashing impaired the biomass recovery and associated DOC removal from the BAC filters. In addition, the ozone breakthrough events that occurred during phase 4 would also have negatively affected the biomass concentration in the BFs. In contrast, the biomass concentration in the SSF increased from $3.2*10^5$ pg ATP/g sample during phase 3 to $5.2*10^5$ pg ATP/g sample during phase 4. This finding suggests that the increase in DOC removal in Train 1 (from phase 3 to 4) occurred due to the higher biomass concentration. The influence of biomass concentration on overall DOC removal were plotted (as seen in Figures 4.9 and 4.10).

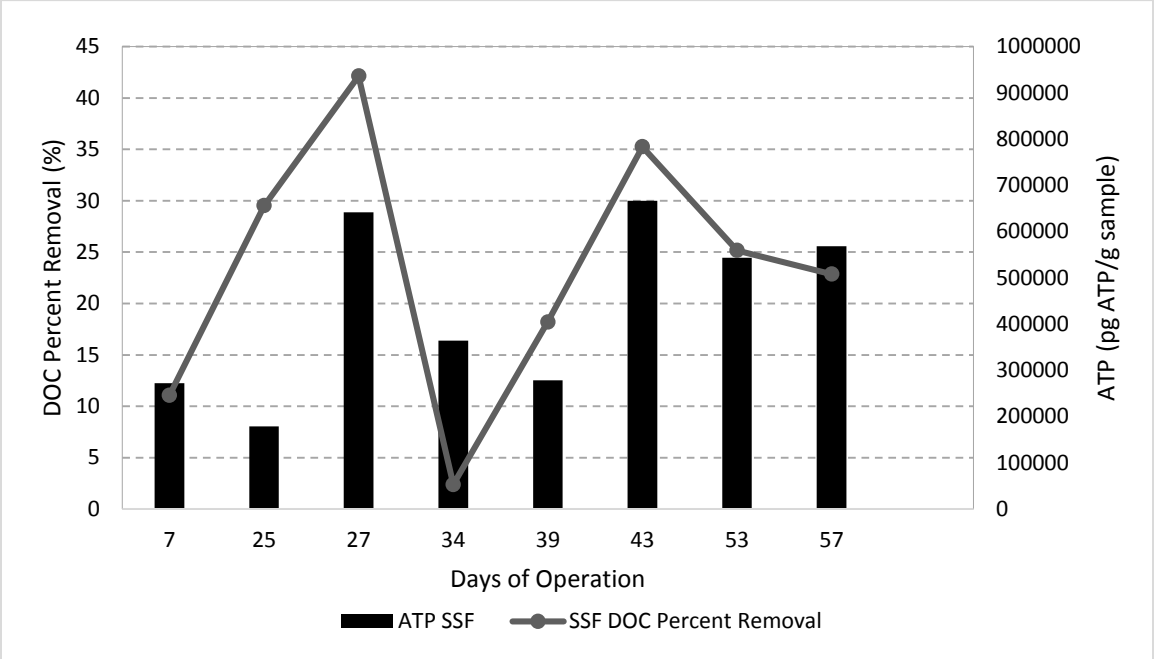


Figure 4.9: ATP concentration and DOC percent removal correlation for Train 1 (nutrient rich site).

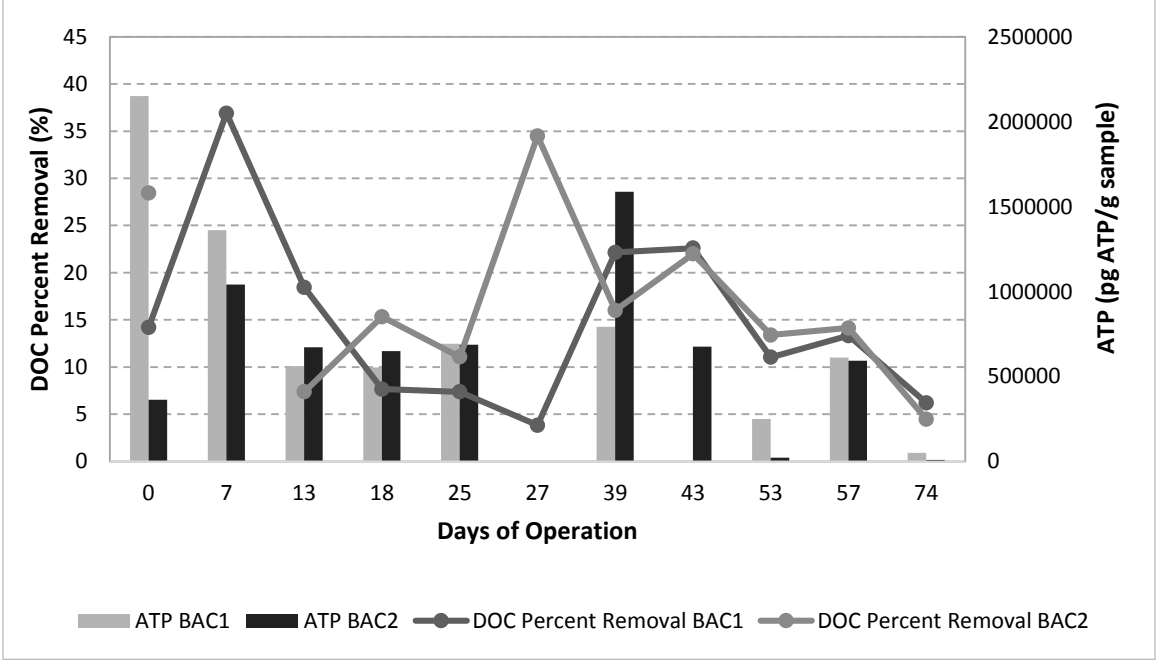


Figure 4.10: ATP concentration and DOC percent removal correlation for Train 2 (nutrient rich site).

The correlation between DOC removal and ATP values was not as apparent as it was during phases 1 and 2. During phase 3, there were operational issues with the ozone generator that prevented the use of a higher dosing rate. At the start of phase 4, the issues were resolved and a higher dosing rate was used (3.45 mg O_3/mg DOC). However, the higher rate led to ozone breakthrough events. For instance, on day 49 (five days after the increase to 3.45 mg O_3/mg DOC), the residual ozone after the contactors was 2.29 mg O_3/L in Train 1 and 0.4 mg O_3/L in Train 2. This led to a residual of 0.07 mg O_3/L and 0.06 mg O_3/L after the slow sand filter and biofilters, respectively. Therefore, ozone passed throughout the entire filter bed, affecting the biomass concentration and consequently the DOC removal.

4.3.2.2 UVA_{254} and SUVA Results

An improvement in UVA_{254} removal was expected during phases 3 and 4 compared to those at Site 1 due to the higher ozone doses applied and the nutrient rich conditions. The average influent and effluent values at Site 2 are reported in Table 4.6.

Table 4.6: Average influent and effluent values for UVA_{254} during phases 3, 4 & 5 (nutrient rich at 1.9 mg O_3/mg DOC, 3.45 mg O_3/mg DOC and with no ozone, respectively).

Phase	Influent Average ± SD (cm^{-1})	Train 1 Effluent Average ± SD (cm^{-1})	Train 2 Effluent Average ± SD (cm^{-1})
Phase 3	0.525 ± 0.34	0.463 ± 0.23	0.424 ± 0.27
Phase 4	0.506 ± 0.078	0.243 ± 0.079	0.324 ± 0.085
Phase 5	0.720 ± 0.057	0.610 ± 0.064	0.642 ± 0.069

During phase 3, the overall UVA_{254} removal was 9.57 ± 8% for Train 1 and 24.9 ± 8% for Train 2. During phase 4, the overall UVA_{254} removal was 52.7 ± 11% for Train 1 and 36.6 ± 10% for Train 2. Finally, during phase 5, the overall UVA_{254} removal was 15.3 ± 5% for Train 1 and 10.9 ± 6% for Train 2. As expected, higher removal occurred at a higher

ozone dose of 3.45 mg O_3/mg DOC. It was hypothesized that the slight increase in removal for Train 1 from phase 3 to phase 5 was due to the difference in biomass concentration in the SSF. The average ATP value for Train 1 was $3.2*10^5$ pg ATP/g sample during phase 3, and increased during the subsequent phases.

SUVA calculations were carried out using a normalization of UVA_{254} by DOC concentration for operation at Site 2. Table 4.7 represents the average influent and effluent SUVA values.

Table 4.7: Average influent and effluent values for SUVA during phases 3, 4 & 5 (nutrient rich at 1.9 mg O_3/mg DOC, 3.45 mg O_3/mg DOC and with no ozone, respectively).

Phase	Influent Average ± SD (L/mg*m)	Train 1 Effluent Average ± SD (L/mg*m)	Train 2 Effluent Average ± SD (L/mg*m)
Phase 3	6.39 ± 4.32	5.53 ± 2.61	4.73 ± 1.05
Phase 4	4.13 ± 0.99	2.64 ± 1.07	3.09 ± 0.93
Phase 5	4.94 ± 0.30	4.50 ± 0.26	4.65 ± 0.21

The average influent SUVA values at Site 2 were above 4 L/mg*m, suggesting that the water at Site 2 was more humic and had higher molecular weight molecules than the water at Site 1. A reduction in SUVA values during phase 4 was due to the effect of the higher ozonation dose on the humic source water, as previously seen in literature (Hozalski et. al., 1999).

4.3.2.3 True Colour

It was expected that the higher ozone doses seen in phases 3 and 4, compared to phase 1, would lead to a higher removal of true colour at Site 2. Although the influent values experienced an increase during phase 4, a higher removal also occurred in Trains 1 and 2 compared to phase 3 (as seen in Figure 4.11). This was expected due to the higher ozone dose providing better breakdown of NOM ahead of the filters.

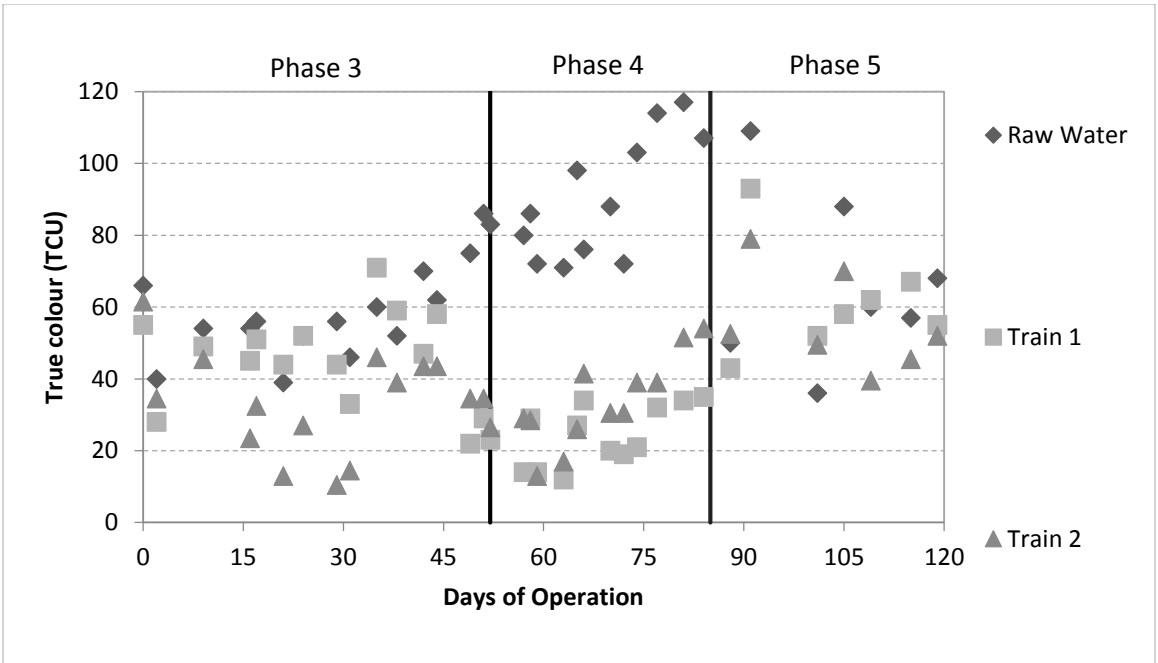


Figure 4.11: True colour values for phases 3, 4 & 5 (nutrient rich at 1.9 mg O_3/mg DOC, 3.45 mg O_3/mg DOC and with no ozone, respectively).

The objective of ≤ 15 TCU was not consistently met throughout the three phases; however, there were several days during phases 3 and 4 where the true colour values were less than 15, confirming that this dose was able to reduce the aesthetic appearance of colour.

4.3.2.4 Turbidity Results

Raw water turbidity values during phases 3 to 5 ranged from 0.2 to 70 NTU, with an average turbidity value of 6 NTU. The SSF effluent turbidity ranged between 0.2 to 17 NTU, while the BF effluent turbidity ranged from 0.1 to 60 NTU.

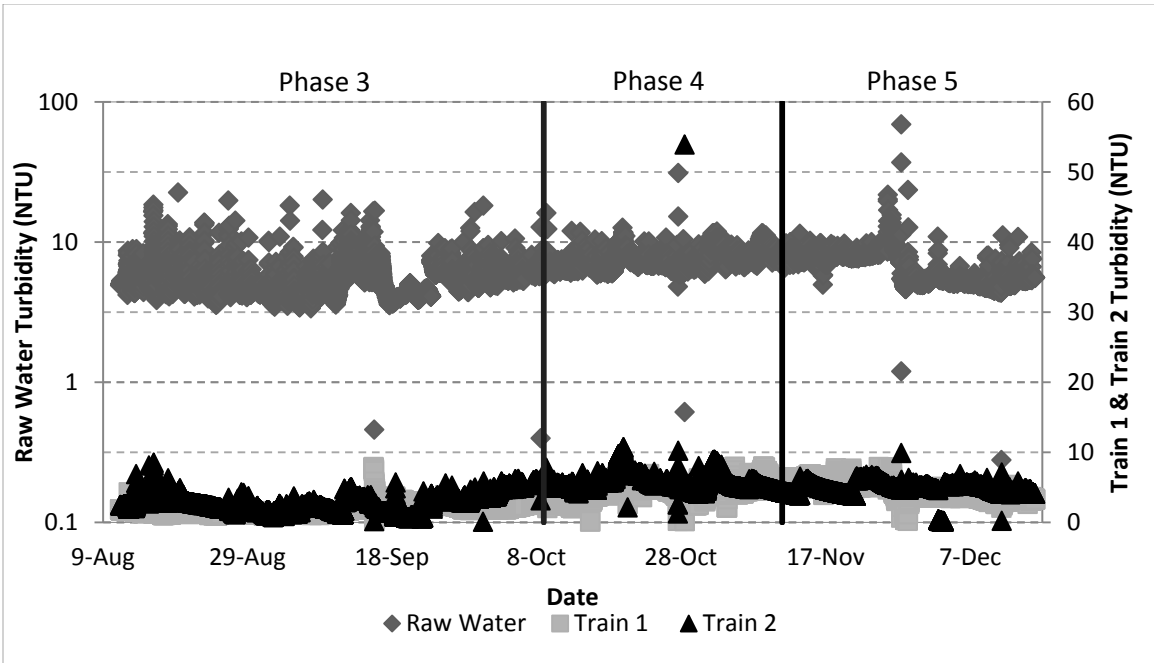


Figure 4.12: Turbidity trend for phases 3, 4 & 5 (nutrient rich at 1.9 mg O_3/mg DOC, 3.45 mg O_3/mg DOC and with no ozone, respectively).

The SSF was more robust to high turbidity events than the BF, most likely due to the inefficient cleaning of the BFs. The maximum backwash rate available at Site 2 was 30 L/min (7.1% bed expansion), which was lower than the calculated rate of 50 L/min (30% bed expansion). Therefore, the cleaning of the biofilters at Site 2 was insufficient. It is evident that the effluent turbidity did not reach the requirements from Health Canada, similar to Site 1.

4.3.3 Overall Comparison of Nutrient Conditions and Pre-Ozonation

This section compares the performance between Train 1 and Train 2 under the different nutrient and pre-ozonation conditions throughout the study.

4.3.3.1 Turbidity Removal

Effluent turbidity is an important parameter when investigating the suitability of a treatment plant. Figure 4.13 shows the average influent and effluent turbidity values at Sites 1 and 2.

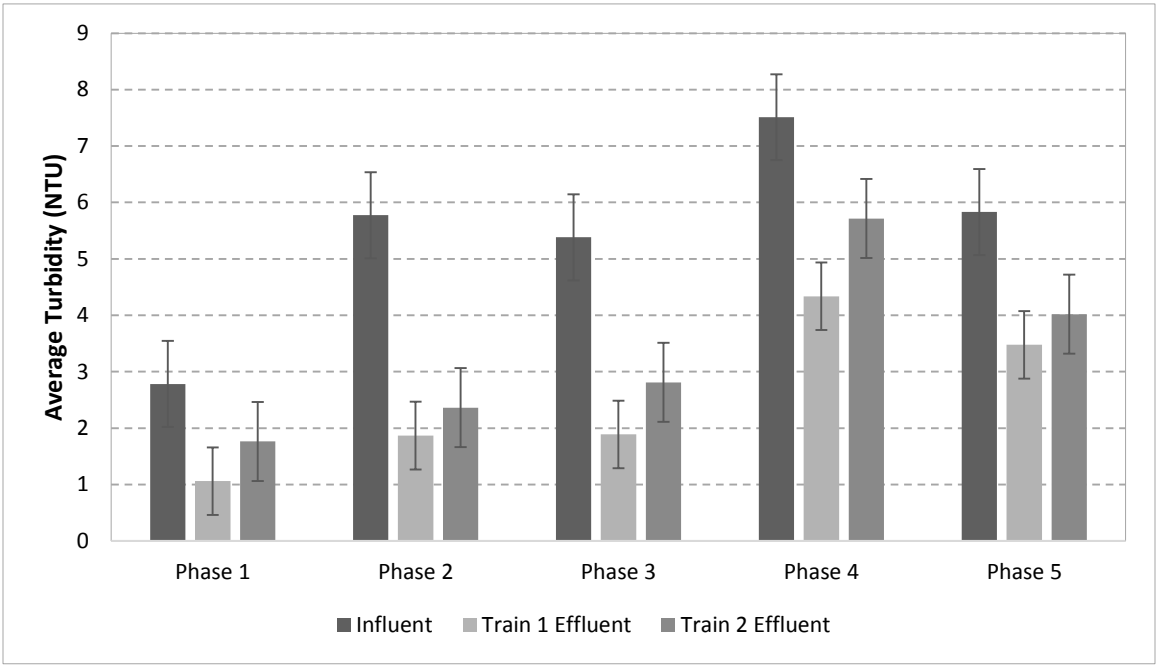


Figure 4.13: Influent and effluent turbidity values. Phase 1: Site 1, 1.55 mg O_3/mg DOC, Phase 2: Site 1, no ozone, Phase 3: Site 2, 1.9 mg O_3/mg DOC, Phase 4: Site 2, 3.45 mg O_3/mg DOC, Phase 5: Site 2, no ozone.

The turbidity removal was higher for Train 1 than Train 2 at each condition. This was expected, as slow sand filters have been shown to provide high removal of turbidity regardless of the source waters. This is due to the finer size of particles inside the SSF (effective diameter of 0.35 mm), compared with the GAC particles in the BF (effective diameter of 0.7 mm). It is evident that at each phase in Figure 4.13, the turbidity requirements of less than 1.0 NTU (SSF) and less than 0.3 NTU (BF) were not met. However, the effluent turbidity values from ozonated water were lower than effluents from unozonated water, with the exception of phase 4. This was seen in a previous study, in which ozone-based oxidation processes facilitated enhanced turbidity removal

regardless of influent turbidity (Rahman et. al., 2010). The deviation during phase 4 may have been a result of higher influent values (compared to those seen in remaining phases) and to the backwash issues and ozone leak issues.

4.3.3.2 DOC Removal

It was expected that the higher ozone dosages used throughout the experiments would provide high DOC removal, compared with periods of low ozone dose. In addition, the nutrient rich conditions at Site 2 were expected to provide higher DOC removal, compared to Site 1. Figure 4.14 shows the average removals for DOC at Sites 1 and 2.

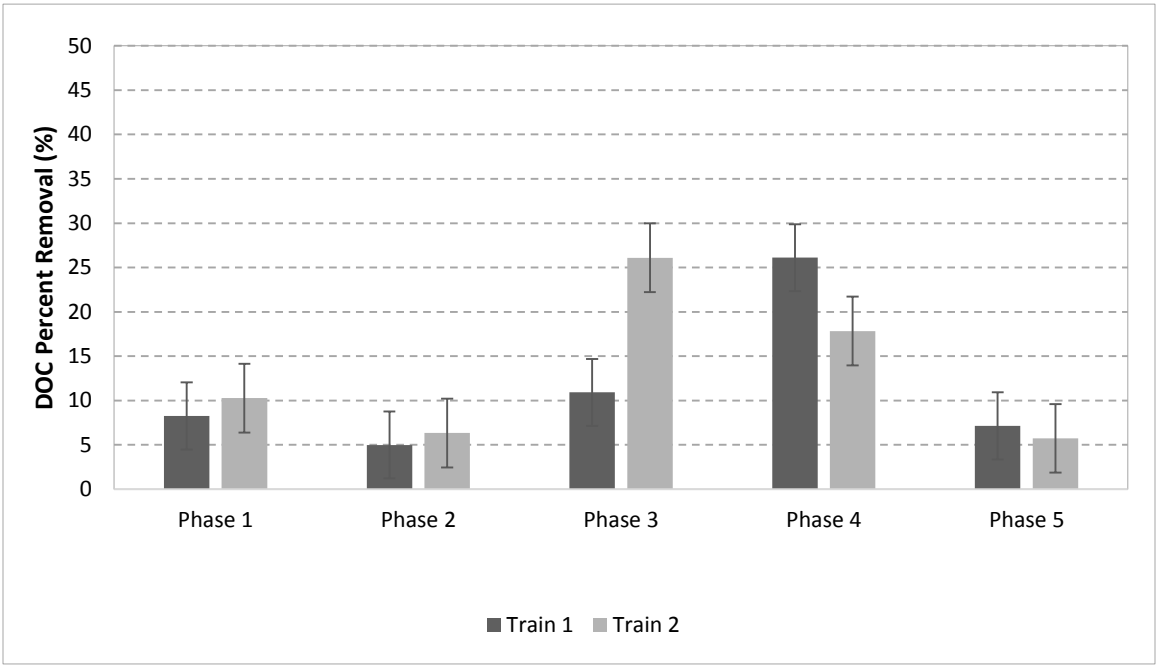


Figure 4.14: DOC removal. Phase 1: Site 1, 1.55 mg O_3/mg DOC, Phase 2: Site 1, no ozone, Phase 3: Site 2, 1.9 mg O_3/mg DOC, Phase 4: Site 2, 3.45 mg O_3/mg DOC, Phase 5: Site 2, no ozone.

The observed DOC removals were in agreement with previous literature (Yan et, al., 2010; Kim et. al., 1997; Galapate et. al., 2001). Matilainen et. al. (2010) reported a TOC removal of 13% under a dosage of 1.5 g O_3/L. Galapate et. al. (2001) observed a DOC removal of 6.4% under a dose of 1.5 mg O_3/mg DOC, and an increased removal of 16%

under a dosage of 3 mg O_3/mg DOC. In addition, a previous study at the nutrient limited site reported removals of 38% (SSF) and 31% (BF) at an ozone dose of 4 mg O_3/mg DOC, and removals of 11% (SSF) and 19% (BF) at a dose of 2 mg O_3/mg DOC (Black et. al., 2013). These are comparable to a pilot study performed by Yan et. al. (2010), where a 38% removal in DOC was seen at a maximum ozone dosage of 4.4 mg O_3/L (1.15 mg O_3/mg DOC).

An increase in DOC removal was seen during phases 3 and 4 (compared to phases 1 and 2) due to the improved nutrient conditions and higher ozone doses used. During phase 3, the removal in Train 2 was better than in Train 1. This was a result of the biomass levels being higher in the BF than the SSF. During phase 4, the DOC removal in the SSF increased as a result of the improved biodegradability provided by the higher ozone dosage, as well as an increase in biomass concentration (from $3.2*10^5$ pg ATP/g sample during phase 3 to $5.2*10^5$ pg ATP/g sample during phase 4). However, the removal in the BF experienced a decrease from 26.1% to 17.8%, as a result of the decreased biomass concentration that occurred due to issues with the ozone generator (0.06 mg/L ozone residual after BF bed).

Figure 4.15 shows the correlations between influent and effluent DOC concentration for each phase. During the unozonated phases, a strong correlation occurred, suggesting that there was no difference between the influent and effluent concentrations and therefore a low removal of DOC during these phases. The same was true at the nutrient limited site under low ozone dose conditions (1.55 mg O_3/mg DOC). However, at the

nutrient rich sites under ozonation, the correlation between influent and effluent was poor, due to the better DOC removal that occurred (average of 19% (SSF) and 22% (BF)).

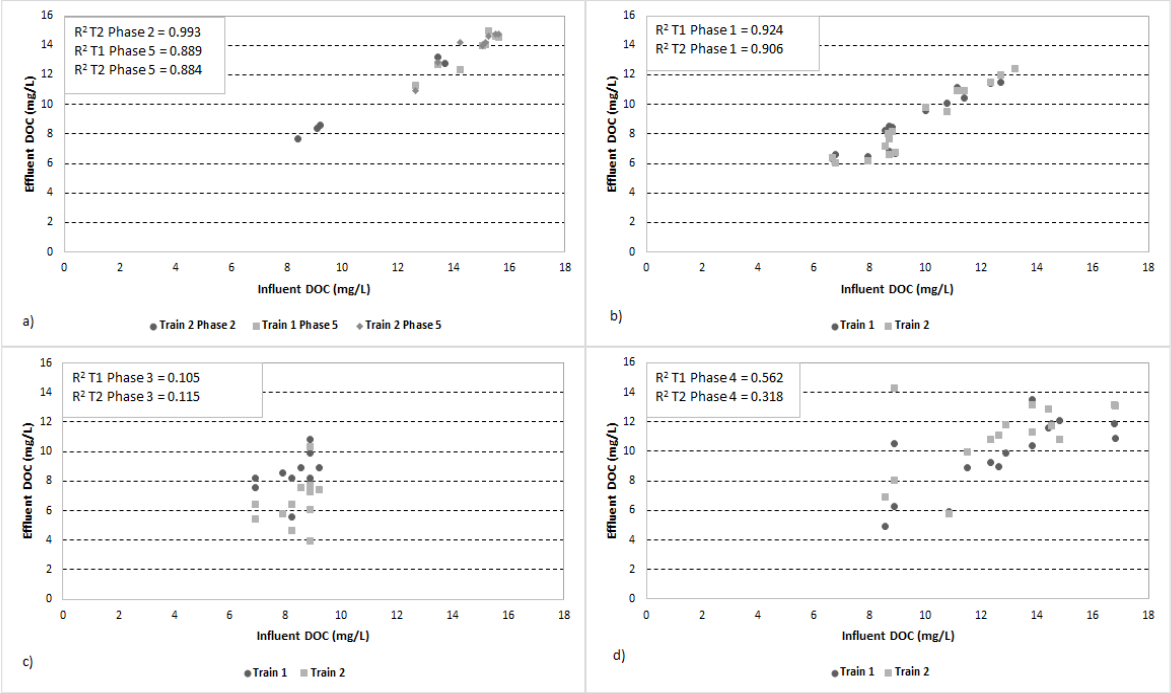


Figure 4.15: Correlation between influent and effluent DOC concentration. a) No ozone (Phases 2 and 5) b) 1.55 mg O_3/mg DOC (Phase 1) c) 1.9 mg O_3/mg DOC (Phase 3) d) 3.45 mg O_3/mg DOC (Phase 4).

4.3.3.3 UVA_{254} Removal

Higher ozone doses led to a better removal of UVA_{254} regardless of site location as is shown in Figure 4.16. The highest UVA_{254} removal (52.7% (SSF) and 36.6% (BF)) occurred during phase 4 which coincides with the highest ozone dosage (3.45 mg O_3/mg DOC), and the second highest removal (16.6% (SSF) and 23.53% (BF)) occurred during phase 1 (dose of 1.55 mg O_3/mg DOC). Previous pilot studies have shown a similar increase in UVA_{254} removal with increasing ozone dosage, as discussed in Section 4.3.1.2 (Galapate et. al., 2001; Pei et. al., 2007; Yan et. al., 2010).

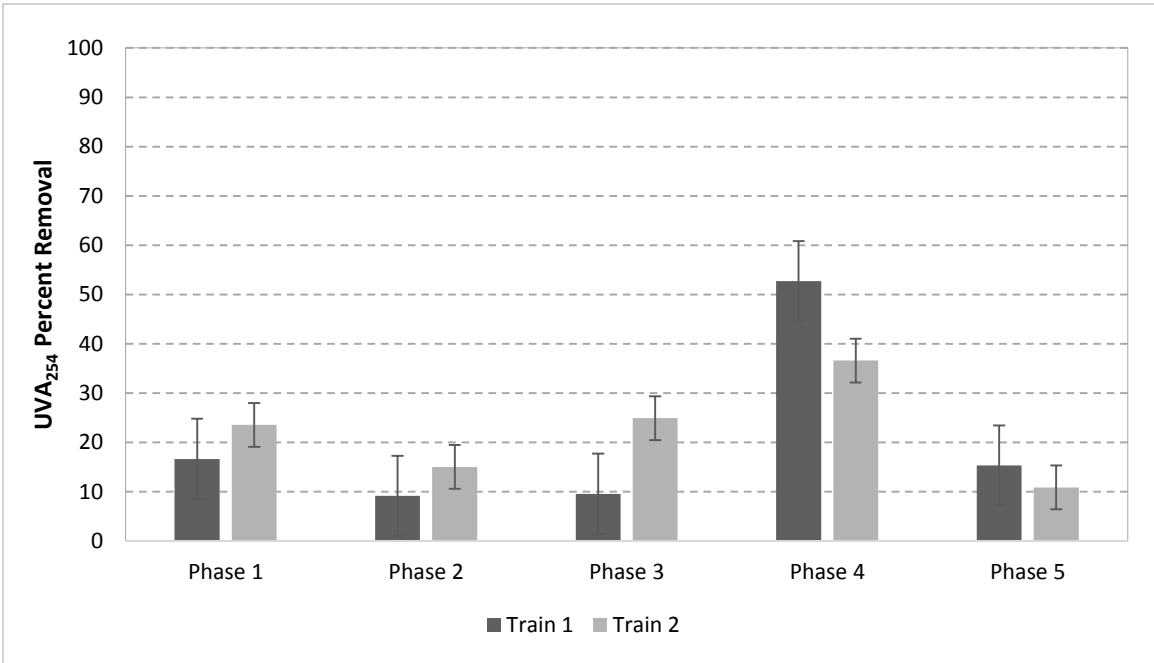


Figure 4.16: UVA254 removal. Phase 1: Site 1, 1.55 mg O3/mg DOC, Phase 2: Site 1, no ozone, Phase 3: Site 2, 1.9 mg O3/mg DOC, Phase 4: Site 2, 3.45 mg O3/mg DOC, Phase 5: Site 2, no ozone.

The highest UVA_{254} removals for both trains occurred at the nutrient rich site, at a high ozone dose of 3.45 mg O_3/mg DOC. The lowest removals for both trains occurred during unozonated conditions (phases 2 and 5). The SSF also exhibited low removal during phase 3, although the ozone dosage at this phase was slightly higher than that at phase 1. This was unexpected, as the nutrient rich conditions and higher ozone dose (1.9 mg O_3/mg DOC vs 1.55 mg O_3/mg DOC for phase 1) were expected to lead to higher removals.

4.3.3.4 True Colour Removal

For true colour, it was expected that higher removal would be seen at higher ozone doses. The highest removal was seen during phase 4 (as seen in Figure 4.17). This was expected due to the ability of ozone to react with humic acid in NOM and degrade aromatic bonds. In addition, the true colour influent during phase 4 was extremely high.

This indicated that regardless of influent true colour values, the nutrient rich conditions and high ozone dose provided true colour removal.

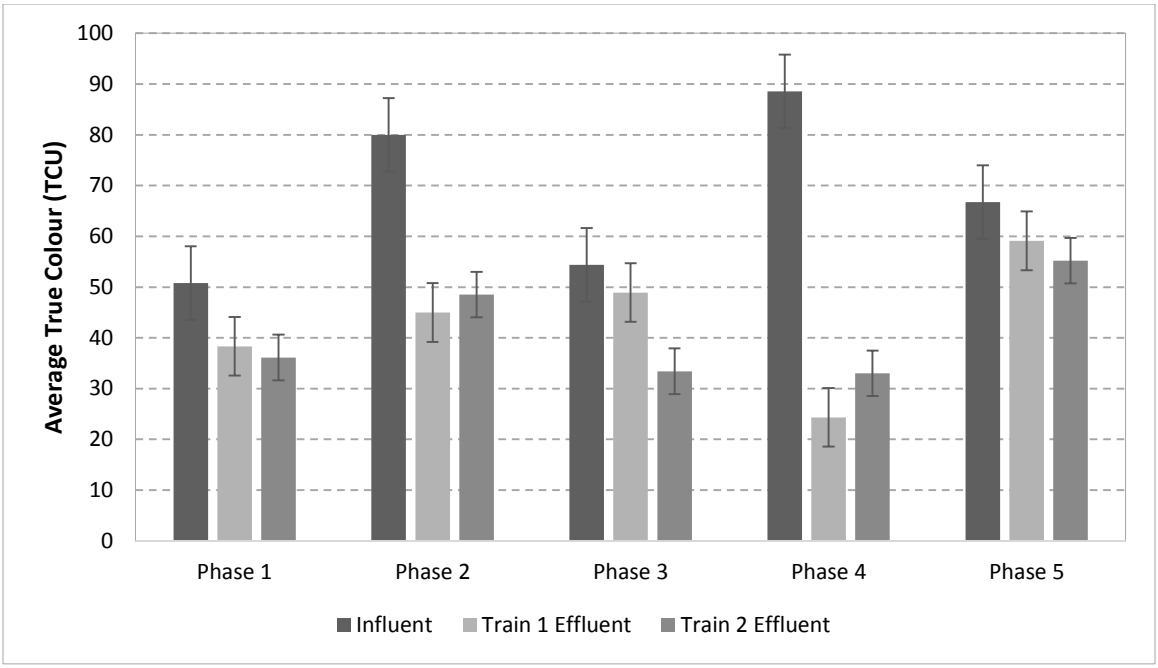


Figure 4.17: Influent and effluent true colour values. Phase 1: Site 1, 1.55 mg O3/mg DOC, Phase 2: Site 1, no ozone, Phase 3: Site 2, 1.9 mg O3/mg DOC, Phase 4: Site 2, 3.45 mg O3/mg DOC, Phase 5: Site 2, no ozone.

4.3.3.5 Statistical Analysis

A student's t-test was carried out to compare DOC and UVA_{254} removals between the SSF and the BF. One of the objectives of this research was to have the BF operating at the same performance level as the SSF. Slow sand filters are popular units in remote and northern communities, but require a larger space for plants due to the lower filtration rate. In contrast, biofilters require less space, and therefore will require less heating costs. Removals for each filter during the separate phases were analyzed using a student's t-test (two sample assuming unequal variances). The resulting p-values are listed in Table 4.8. A p-value greater than 0.05 signifies that the removals are statistically similar, and a p-value less than 0.05 signifies that the removals are statistically different.

Table 4.8: Results of statistical t-test (two-sample assuming unequal variance) comparing SSF versus BF – p-values shown.

	Site 1		Site 2		
	Phase 1	**Phase 2**	**Phase 3**	**Phase 4**	**Phase 5**
DOC	0.19	0.36	0.0076	0.023	0.27
UVA_{254}	0.11	0.062	0.0013	0.00016	0.057

The t-test analysis indicates that at Site 1, under nutrient limited conditions, there was no overall difference in results between the SSF and the BF ($p > 0.05$). However, at the nutrient rich site, when ozone was applied, the SSF and BF performed differently ($p < 0.05$), while when no ozone was applied there was no overall difference. One reason for the different performance during ozonated conditions at Site 2 could be the varying ATP values in the SSF and BF during phases 3 and 4. During phase 3, the ATP values of BAC1 and BAC2 were $1.2*10^6$ and $6.8*10^5$ pg ATP/g sample respectively, and they dropped to $5.2*10^5$ and $6.3*10^5$ pg ATP/ g sample (respectively) during phase 4. Alternatively, the ATP value for the SSF was $3.2*10^5$ pg ATP/g sample during phase 3, and increased to $5.2*10^5$ pg ATP/g sample during phase 4. This, combined with the higher ozone dose during phase 4, resulted in different removals between the SSF and the BF.

The removals for each train were compared to the pilot site location to investigate potential differences in removal related to the nutrient availability at the two sites (refer to Table 4.9).

Table 4.9: Results of statistical t-test (two-sample assuming unequal variance) comparing nutrient-limited and nutrient rich conditions.

Train	Parameter	p-value	
		Ozonated	**Unozonated**
Train 1 (Site 1 vs. Site 2)	DOC	0.00018	0.239
	UVA_{254}	0.00086	0.0578
Train 2 (Site 1 vs. Site 2)	DOC	0.00039	0.367
	UVA_{254}	0.032	0.132
*Ozone dosages of 1.55, 1.9, 3.45 mg O_3/mg DOC grouped together			

The t-test analysis indicates that during ozonated conditions, the performances at Site 1 and Site 2 were different for both trains. However, under no ozone, there was no overall difference in results between the two sites, for both trains. This is an important finding as it highlights the importance of ozone with biofiltration more so than nutrient availability in the raw source water as a key parameter in achieving successful DOC removal. During phase 1, the average DOC removals were 8.27% (SSF) and 10.3% (BF). During phase 3, the average DOC removals were 10.9% (SSF) and 26.1% (BF). Finally, during phase 4, the average DOC removals were 26.1% (SSF) and 17.8% (BF). The difference in removal is expected, as nutrient rich conditions should provide higher removal than nutrient limited conditions.

During unozonated conditions, no difference was observed between the two sites. During phase 2, the average DOC removals were 5.00% (SSF) and 6.34% (BF). The average DOC removals during phase 5 were 7.14% (SSF) and 5.73% (BF). This is unexpected, as previous studies have shown that phosphorous supplementation is beneficial in terms of increased heterotrophic plate counts, leading to increased biomass concentration and DOC removal (Nishijima et. al., 1997; Xin et. al., 2003).

4.4 Conclusions

A multi stage filtration pilot trailer with two trains (one with a biofilter and the other with a slow sand filter) was used to complete a case study comparing the impact of nutrient conditions and pre-ozonation on organics removal. The research highlighted the importance of both parameters as being important for obtaining effective DOC removal in a biological filtration system with the key points listed outlined below:

- Nutrient limited conditions (C:N:P of 546:24:1 w/w) resulted in low DOC removals for both a biofilter and slow sand filter operating in parallel. A low ozone dosage of 1.55 mg O_3/mg DOC improved the DOC removal (8.27% for the SSF and 10.3% for the BF) compared to unozonated conditions (5% for the SSF and 6.34% for the BF). UVA_{254} removals were also better at the low dosage (16.6% and 23.5%, respectively for both systems) compared to unozonated conditions (9.14% and 13.7%, respectively).
- Nutrient rich conditions (C:N:P of 6.3:1.6:1 w/w) resulted in higher DOC removals for both a biofilter and slow sand filter operating in parallel under low and high ozone doses, compared to the nutrient limited conditions. The highest DOC removal for the biofilter occurred during low ozone dosing conditions (26.1% at a dose of 1.9 mg O_3/mg DOC) and the highest removal for the slow sand filter occurred during high ozone dosing conditions (26.1% at a dose of 3.45 mg O_3/mg DOC). Ozone control issues resulting in ozone passing through the biofilter may have adversely impacted the results when the ozone dosage was 3.45 mg O_3/mg

DOC. The highest UVA_{254} removals occurred at the highest ozone dose (3.45 mg O_3/mg DOC) for both trains (52.7% and 36.6%, respectively).

- The better overall DOC removal at site 2 was attributed to the phosphorous availability at the nutrient rich site (1.83 mg/L vs 0.013 mg/L at nutrient limited) and to the higher ozone dose that was used.
- Turbidity removal in both trains and at both sites did not reach the target effluent turbidity of 1 NTU. The slow sand filter was more robust to changes in raw water quality turbidity changes while the biofilter effluent quality was less stable under variable raw water turbidity conditions. This was attributed to an inefficient backwash step with the biofilter.

References

1 Black, K., Basu, O., Cleary, S., & LeCraw, R. (2013). Effect of a pilot-scale multi-stage filtration system on the removal of natural organic matter (NOM). *OWWA/OMWA Joint Annual Conference & OWWEA Trade Show* (pp. 1-9). Ottawa: Ontario Water Works Association.

2 Camel, V., & Bermond, A. (1998). The use of ozone and associated oxidation processes in drinking water treatment. *Water Research*, *32(11),* 3208-3222.

3 Chen, K., & Wang, Y. (2012). Control of disinfection by-product formation using ozone-based advanced oxidation processes. *Environmental Technology, 33 (4)*, 487-495.

4 Chowdhury, F. L., Berube, P. R., & Mohseni, M. (2008). Characteristics of natural organic matter and formation of chlorinated disinfection by-products from two source waters that respond differently to ozonation. *Ozone: Science and Engineering, 30 (3)*, 321-331.

5 Cleary, S. A. (2005). *Sustainable Drinking Water Treatment for Small Communities Using Multistage Slow Sand Filtration.* (Master's thesis, University of Waterloo, Waterloo). Retrieved from https://uwspace.uwaterloo.ca/handle/10012/926.

6 Cleary, S. A., Collins, M. R., DeLoyde, J., & Lecraw, R. A. (2008). How Far North Can We Take It? Pushing the Limits of Slow Sand Filtration. *Proceedings of the British Columbia Water & Waste Association Annual Conference* (p. 67). Whistler, B.C.: British Columbia Water & Waste Association, Burnaby, B.C.

7 Galapate, R. P., Baes, A.U., & Okada, M. (2001). Transformation of dissolved organic matter during ozonation: effects on trihalomethane formation potential. *Water Research*, *35(9),* 2201-2206.

8 Gottinger, A. M., McMartin, D.W., Price, D., & Hanson, B. (2011). The effectiveness of slow sand filters to treat Canadian rural prarie water. *Canadian Journal of Civil Engineering, 38 (4)*, 455-463.

9 Health Canada. (1995). *Colour.* Government of Canada.

10 Health Canada. (2013). *Guidelines for Canadian Drinking Water Quality: Guidline Technical Document: Turbidity.* Ottawa: Health Canada.

11 Hozalski, R. M., Bouwer, E.J., & Goel, S. (1999). Removal of natural organic matter (NOM) from drinking water supplies by ozone-biofiltration. *Water Science and Technology*, *40 (9),* 157-163.

12 Kim, W. H., Nishijima, W., Shoto, E., & Okada, M. (1997). Pilot plant study on ozonation and biological activated carbon process for drinking water treatment. *Water Science and Technology*, *35 (8),* 21-28.

13 Lechevallier, M. W., Schulz, W., & Lee, R.G. (1991). Bacterial nutrients in drinking water. *Applied and Environmental Microbiology 57 (3)*, 857-862.

14 Matilainen, A., Vepsalainen, M., & Sillanpaa, M. (2010). Natural organic matter removal by coagulation during drinking water treatment: A review. *Advances in Colloid and Interface Science, 159 (2)*, 189-197.

15 Miettinen, I.T., Vartiainen, T., & Martikainen, P.J. (1997). Phosphorous and bacterial growth in drinking water. *Applied and Environmental Microbiology*, *63 (8)*, 3242-3245.

16 Miltner, R., Summers, R., & Wang, J. (1995). Biofiltration Performance: Part 2, effect of backwashing. *Journal of American Water Works Association*, *87 (12),* 64-70.

17 Moll, D. M., Summers, R.S., Fonseca, A.C., & Matheis, W. (1999). Impact of Temperature on Drinking Water Biofilter Performance and Microbial Community Structure. *Environmental Science and Technology, 33 (14)*, 2377-2382.

18 Nishijima, W., Shoto, E., & Okada, M. (1997). Improvement of biodegradation of organic substance by addition of phosphorous in biological activated carbon. *Water Science and Technology, 36 (12)*, 251-257.

19 O'Connor, T. H. (2002). *Report of the Walkerton Inquiry : The Events of May 2000 and Related Issues.* Toronto: Ontario Ministry of the Attorney General.

20 Pacini, V. A., Ingallinella, A.M., & Sanguinetti, G. (2005). Removal of iron and manganese using biological roughing up flow filtration technology. *Water Research, 39 (18)*, 4463-4475.

21 Pei, Y. , Yu, J., Guo, Z., Zhang, Y., Yang, M., Zhang, J., & Junji, H. (2007). Pilot study on pre-ozonation enhanced drinking water treatment process. *Ozone: Science and Engineering, 29 (3)*, 317-323.

22 Peng, X., Murphy, T., & Holden, N.M. (2008). Evaluation of the Effect of Temperature on the Die-Off Rate for Cryptosporidium Parvum Oocysts in Water, Soils and Feces. *Applied and Environmental Microbiology, 74 (23)*, 7101-7107.

23 Rahman, M. F., Jasim, S.Y., Yanful, E.K., Ndiongue, S., & Borikar, D. (2010). Advanced oxidation treatment of drinking water: Part II. Turbidity, Particles, and Organics Removal from Lake Huron Water. *Ozone: Science & Engineering, 32 (5)*, 295-304.

24 Simpson, D. R. (2008). Biofilm processes in biologically active carbon water purification. *Water Research, 42 (12)*, 2839-2848.

25 Velten, S., Boller, M., Koster, O., Helbing, J., Weilenmann, H-U., & Hammes, F. (2011). Development of biomass in a drinking water granular active carbon (GAC) filter. *Water Research, 45 (19)*, 6347-6354.

26 von Gunten, U. (2003). Ozonation of drinking water: Part I. Oxidation kinetics and p. *Water Research, 37 (7)*, 1443-1467.

27 Xin, Y., Xiao-jian, Z., Xiao-ling, L., Xiao-dong, Z., Xiao-dongz, Z., & Zhan-sheng, W. (2003). Phopshorous limitation in biofiltration for drinking water treatment. *Journal of Environmental Sciences, 15 (4)*, 494-499.

28 Yan, M., Wang, D., Ma, X., Ni, J., & Zhang, H. (2010). THMs precursor removal by an integrated process of ozonation and biological granular activated carbon for typical Northern China water. *Separation and Purification Technology, 72 (3)*, 263-268.

29 Yapsakli, K., & Cecen, F. (2010). Effect of type of granular activated carbon on DOC biodegradation in biological activated carbon filters. *Process Biochemistry, 45 (3)*, 355-362.

5 Conclusions and Future Work

This chapter will present a summary of the conclusions reached through this research and recommendations for future work.

5.1 Summary of Findings

Multi-stage filtration is a reliable technology that can be used in remote and northern communities in Canada through the use of pre-ozonation and roughing filtration ahead of standard filtration methods (LeCraw & Jobb, 2011). Slow sand filtration has been in use for over 200 years and has proven to be a viable treatment method for small communities. However, biofiltration require less space and therefore will require less heating cost. The use of pre-ozonation with biofiltration is a process that can assist in meeting federal drinking water regulations for primary disinfection, DBPs and turbidity and in removing BDOC, taste and odour causing compounds (Schulz, 2014). Synergistic benefits can be seen between ozonation and biofiltration in terms of reaching a higher quality effluent, through the oxidation of organics to generate BDOC. The biofilters can then remove the biodegradable fraction and produce stable drinking water that does not react further down the distribution system (Schulz, 2014).

The purpose of this research was to operate a pilot system with two parallel treatment trains (SSF and BF) and compare the performance of each train. A pilot multi-stage filtration trailer was set up at two different sites with different quality source waters. . Site 1 drew source water from the Ottawa River, and had a C:N:P ratio of 546:24:1 (w/w). Site 2 drew source water from Lac Chassignole, and had a C:N:P ratio of 6.3:1.6:1 (w/w). Turbidity, biomass levels, DOC, UVA_{254}, SUVA and true colour were all monitored

throughout the duration of each phase of the study. Ozone was dosed into the system at different levels, in addition to being turned off for a period of time at both sites. The main findings of this thesis are summarized in the following sections.

5.1.1 Site 1

Site 1 was the nutrient limited site, with a C:N:P ratio of 546:24:1 (w/w). At Site 1, the highest removals occurred during ozonated conditions at 1.55 mg O_3/mg DOC, compared to when the ozone was turned off. During ozonated conditions, the average DOC removals for the SSF and BF were 8.27% and 10.3% (respectively), and the average UVA_{254} removals for the SSF and BF were 16.6% and 23.5% (respectively). These removals were lower than what has previously been seen in literature, suggesting that the nutrient limited conditions were hindering the growth of biomass within both the BF columns and the SSF column. In addition, the ozone dosage of 1.55 mg O_3/mg DOC was lower than desired due to mechanical problems with the ozone generator. Ozone leaks into the system resulted in residuals of up to 1.8 mg O_3/L in the contactor of Train 1 and 2.17 mg O_3/L in the contactor of Train 2, which resulted in a sharp decrease in ATP values for the biofilters, while the DOC and UVA_{254} removals appeared unaffected. The true colour and turbidity effluents for each train did not reach those minimum values recommended by Health Canada. These results indicate that a higher ozone dosage may be necessary when dealing with nutrient limited source waters. A statistical analysis was carried out that indicated the DOC and UVA_{254} removals between Trains 1 and 2 were similar during both the ozonated and unozonated conditions. However, due to the

unfavourable conditions, this was not enough to determine whether biofilters can operate in place of slow sand filters in remote and northern communities.

5.1.2 Site 2

Site 2 was the nutrient rich site, with a C:N:P ratio of 6.3:1.6:1 (w/w). Here, higher removals of DOC and UVA_{254} occurred during pre-ozonated conditions compared to unozonated conditions. The highest DOC removal in the slow sand filter (26.1%) occurred at an ozone dose of 3.45 mg O_3/mg DOC. The highest DOC removal in the biofilter (26.1%) occurred at an ozone dose of 1.9 mg O_3/mg DOC. The highest UVA_{254} removals in both the SSF and the BF (52.7% and 36.6% respectively) occurred at the high dose of 3.45 mg O_3/mg DOC. These removals were similar to what has been seen in literature. In addition to the ozonated phases, a phase without ozone occurred during operation at Site 2. During the unozonated phase, the average DOC removals for the SSF and BF were 7.14% and 5.73% respectively, and the average UVA_{254} removals for the SSF and BF were 15.3% and 10.9% respectively. These findings suggest that the combination of nutrient rich conditions and pre-ozonation were necessary to observe higher organic removals in both trains of the pilot filter. A statistical analysis was carried out to compare the DOC and UVA_{254} removals seen between the SSF and the BF at Site 2. Firstly, it was observed that during the unozonated phase, the removals between the SSF and the BF were in fact similar, and were lower than expected for both trains despite being at nutrient rich conditions. However, during both ozonated phases the removals between the SSF and the BF were statistically different.

5.2 Overall Conclusions and Recommendations for Future Work

Multi-stage filtration showed an effective treatment of source water when pre-ozonation was applied and at nutrient rich conditions; however, the turbidity and aesthetic objectives outlined by Health Canada were not fully met. Nutrient limited conditions coupled with insufficient pre-ozonation were major limiting factors in terms of the overall performance of both a slow sand filter and a biofilter operated within a pilot trailer. Therefore, when choosing to use multi-stage filtration at a specific site, it is important to determine the nutrient availability of the source water. Additional nutrients can be added ahead of the filters if it is found that the source water is nutrient limited. In addition, bench scale ozone tests can be performed to determine the minimum ozone dose required, so as not to increase the amount of ozonation by-products in the treated water.

Several mechanical issues occurred during this research that compromised the operation of the pilot trailer. Firstly, ozone leaks occurred fairly regularly due to an improper design of the system that delivered the ozone into the venturi meters. Full scale design should incorporate a proper stainless steel pipe system, as opposed to Teflon coated tubing, which was deteriorated by continuous flow of ozone. Secondly, the backwash capacity available at both sites was also a limiting factor for operation of the pilot trailer. The theoretical backwash flow required was greater than the maximum flow rates available at each site, therefore compromising the cleaning of the biofilters.

One aspect of future work should be the use of multistage filtration as a pre-treatment ahead of membrane filtration systems. Membrane fouling is a disadvantage during

water treatment, particularly with source waters that have high amounts of NOM. Several studies have investigated the use of filtration ahead of a membrane, particularly in remote communities due to the high organic groundwater and the need to reduce costly chemicals. These studies have found that removal of AOC and DOC is possible at a higher efficiency, and that the operational length of membranes can be increased to more than 300 hours in between fouling periods (Halle et ,al., 2009; Hu et. al., 2005).

Throughout the operation of the trailer, the temperature of the source water maintained an average between 10 ^{o}C to 20 ^{o}C, sometimes reaching a low of 1 ^{o}C. It would be interesting to observe the reproducibility of the above results at a consistent cold water temperature of less than 5 ^{o}C. Considering that the communities in which the trailer will be operated will be located in northern parts of Canada, it is important that the same level of performance observed throughout this research is also observed at consistently colder temperatures. It may be beneficial to understand the effects of nutrient availability in cold water temperature as well.

Finally, biomass quantification may be helpful in fully understanding the intricate relationship between biomass levels and organics removal in both slow sand filters and biofilters. In addition, biomass quantification and measurements throughout the entire media bed during higher ozone doses may provide information regarding the effect of ozone on biomass levels. It was seen that during an overdose event, biomass concentration would decrease but organic removal would continue to increase. Since the biomass samples were taken from the top 5 cm of the filters, it was hypothesized

that the media further down the filter was unaffected by the ozone and continued to provide removal of the biodegradable components. Biomass quantification will give proof of this.

References

Amburgey, J. E. & Amirtharajah, A. (2005). Strategic Filter Backwashing Techniques and Resulting Particle Passage. *Journal of Environmental Engineering, 131 (4)*, 535-547.

Amirtharajah, A. M. (1990). Optimum backwash of dual media filters and granular activated carbon filter-adsorbers with air scour. *AWWA Research Foundation*, (p. 140). Denver.

Andersson, A., Laurent, P., Kihn, A., Prevost, M., & Servais, P. (2001). Impact of temperature on nitrification in biological activated carbon (BAC) filters used for drinking water treatment. *Water Research, 35 (12)*, 2923-2934.

Basu, O., & Huck, P. (2004). Removal of humic acid by biofiltration. *Water Science and Technology: Water Supply, 4 (4)*, 147-154.

Black, K., Basu, O., Cleary, S., & LeCraw, R. (2013). Effect of a pilot-scale multi-stage filtration system on the removal of natural organic matter (NOM). *OWWA/OMWA Joint Annual Conference & OWWEA Trade Show* (pp. 1-9). Ottawa: Ontario Water Works Association.

Camel, V. & Bermond, A. (1998). The use of ozone and associated oxidation processes in drinking water treatment. *Water Research*, *32(11),* 3208-3222.

Carlson, K. & Amy, G. (1996). The formation of filter-removable biodegradable organic matter during ozonation. *Ozone Science and Engineering, 19 (2)*, 179-199.

Chan, L., Receveur, O., Batal, M., David, W., Schwartz, H., Ing, A., . . . Tikhonov, C. (2014). *First Nations Food, Nutrition and Environment Study (FNFNES): Results from Ontario (2011/2012).* Ottawa: University of Ottawa.

Chan, R., Miller, S., & Suthaker, S. (2014). Using Biological Process to Reduce Chemical Pre-Treatment of RO Membranes. *Ontario Water Works Association*, (pp. 12-16).

Chen, F., Peldszus, S., Peiris, R.H., Ruhl, A.S., Mehrez, R., Jekel, M., Legge, R.L., & Huck, P.M. (2014). Pilot-scale investigation of drinking water ultrafiltration membrane fouling rates using advanced data analysis techniques. *Water Research, 48 (1)*, 508-518.

Chen, K., & Wang, Y. (2012). Control of disinfection by-product formation using ozone-based advanced oxidation processes. *Environmental Technology, 33 (4)*, 487-495.

Choi, Y. C., Raskin, L., Li, X., & Morgenroth, E. (2007). Effect of backwashing on perchlorate removal in fixed bed biofilm reactors. *Water Research, 41 (9)*, 1949-1959.

Chowdhury, F., Berube, P., & Mohseni, M. (2008). Characteristics of natural organic matter and formation of chlorinated disinfection by-products from two source waters that respond differently to ozonation. *Ozone: Science and Engineering, 30 (3)*, 321-331.

Cleary, S. A. (2005). *Sustainable Drinking Water Treatment for Small Communities Using Multistage Slow Sand Filtration.* (Master's thesis, University of Waterloo, Waterloo). Retrieved from https://uwspace.uwaterloo.ca/handle/10012/926.

Cleary, S. A., Collins, R., DeLoyde, J., & Lecraw, R. (2008). How Far North Can We Take It? Pushing the Limits of Slow Sand Filtration. *Proceedings of the British Columbia Water & Waste Association Annual Conference* (p. 67). Whistler, B.C.: British Columbia Water & Waste Association, Burnaby, B.C.

Crittenden, J. T. (2012). *MWH's Water Treatment: Principles and Design, 3rd edition.* Hoboken: John Wiley & Sons, Inc.

Environment, F.-P.-T. C.-P.-T. (2014). *Guidelines for Canadian Drinking Water Quality Summary Table.* Ottawa, ON: Health Canada.

Fonseca, A.C., Summers, R.S., & Hernandez, M.T. (2001). Comparative measurements of microbial activity in drinking water biofilters. *Water Research, 35 (16)*, 3817-3824.

Galapate, R. P., Baes, A.U., & Okada, M. (2001). Transformation of dissolved organic matter during ozonation: effects on trihalomethane formation potential. *Water Research*, *35 (9),* 2201-2206.

Gibert, O., Lefevre, B., Fernandez, M., Bernat, X., Paraira, M., & Pons, M. (2013). Fractionation and removal of dissolved organic carbon in a full-scale granular activated carbon filter used for drinking water production. *Water Research, 47 (8)*, 2821-2829.

Gibert, O., Lefevre, B., Fernandez, M., Bernat, X., Paraira, M., Calderer, M., & Martinez-Llado, X. (2013). Characterising biofilm development on granular activated carbon used for drinking water production. *Water Research, 47 (3)*, 1101-1110.

Gottinger, A. M., McMartin, D.W., Price, D., & Hanson, B. (2011). The effectiveness of slow sand filters to treat Canadian rural prarie water. *Canadian Journal of Civil Engineering, 38 (4)*, 455-463.

Guay, C., Rodriguez, M., & Serodes, J. (2005). Using ozonation and chloramination to reduce the formation of trihalomethanes and haloacetic acids in drinking water. *Desalination, 176 (1-3)*, 229-240.

Halle, C., Huck, P.M., Peldszus, S., Haberkamp, J., & Jekel, M. (2009). Assessing the performance of biological filtration as pretreatment to low pressure membranes for drinking water. *Environmental Science & Technology, 43 (10)*, 3878-3884.

Health Canada. (1995). *Colour.* Government of Canada.

Health Canada. (2013). *Guidelines for Canadian Drinking Water Quality: Guidline Technical Document: Turbidity.* Ottawa: Health Canada.

Hozalski, R. M., Bouwer, E.J., & Goel, S. (1999). Removal of natural organic matter (NOM) from drinking water supplies by ozone-biofiltration. *Water Science and Technology*, *40 (9),* 157-163.

Hu, J. Y., Song, L.F., Ong, S.L., Phua, E.T. & Ng, W.J. (2005). Biofiltration pretreatment for reverse osmosis (RO) membrane in a water reclamation system. *Chemosphere, 59 (1)*, 127-133.

Irabelli, A., Jasim, S., & Biswas, N. (2008). Pilot-scale evaluation of ozone vs. peroxone for trihalomethane formation. *Ozone: Science and Engineering, 30 (5)*, 356-366.

Jasim, S. Y., Ndiongue, S., Johnson, B., Schweitzer, L., & Borikar, D. (2008). The effect of ozone on cold water coagulation. *Ozone: Science and Engineering, 30 (1)*, 27-33.

Kim, W. H., Nishijima, W., Shoto, E., & Okada, M.(1997). Pilot plant study on ozonation and biological activated carbon process for drinking water treatment. *Water Science and Technology*, *35 (8),* 21-28.

Ko, Y., Lee, Y., & Nam, S. (2007). Evaluation of a pilot scale dual media biological activated carbon process for drinking water. *Korean Journal of Chemical Engineering, 24 (2)*, 253-260.

Lechevallier, M. W., Schulz, W., & Lee, R.G. (1991). Bacterial nutrients in drinking water. *Applied and Environmental Microbiology*, *57 (3),* 857-862.

LeCraw, R. A. & Jobb, B. (2011). Meeting Filtration and Disinfection By-Product Rules with Modified Slow Sand Filtration. *Clean and Safe Drinking Water Workshop* (p. 39). Newfoundland and Labrador: Department of Environment and Conservation.

Liao, X., Chen, C., Wang, Z., Wan, R., Chang, C., Zhang, X., & Xie, S. (2013). Changes of biomass and bacterial communities in biological activated carbon filters for drinking water treatment. *Process Biochemistry, 48 (2)*, 312-316.

Matilainen, A., Iivari, P., Sallanko, J., Heiska, E., & Tuhkanen, T. (2006). The role of ozonation and activated carbon filtration in the natural organic matter removal from drinking water. *Environmental Technology 27 (10)*, 1171-1180.

Metsamuuronen, S., Sillanpaa, M., Bhatnagar, A., & Manttari, M. (2014). Natural Organic Matter Removal from Drinking Water by Membrane Technology. *Separation & Purification Reviews, 43 (1)*, 1-61.

Meunier, L., Canonica, S., & von Gunten, U. (2006). Implications of sequential use of UV and ozone for drinking water quality. *Water Research, 40 (9)*, 1864-1876.

Miettinen, I.T., Vartiainen, T., & Martikainen, P.J. (1997). Phosphorous and bacterial growth in drinking water. *Applied and Environmental Microbiology*, *63 (8)*, 3242-3245.

Miltner, R., Summers, R., & Wang, J. (1995). Biofiltration Performance: Part 2, effect of backwashing. *Journal of American Water Works Association*, *87 (12),* 64-70.

Moll, D. M., Summers, R.S., Fonseca, A.C., & Matheis, W.(1999). Impact of Temperature on Drinking Water Biofilter Performance and Microbial Community Structure. *Environmental Science and Technology, 33 (14)*, 2377-2382.

MSF Systems Inc. (2015). Retrieved 01 29, 2015, from Case Studies: http://www.msfilter.com/case-studies.php

Nishijima, W., Shoto, E., & Okada, M. (1997). Improvement of biodegradation of organic substance by addition of phosphorous in biological activated carbon. *Water Science and Technology, 36 (12)*, 251-257.

Ochieng, G.M.M., Otieno, F.A.O., Ogada, T.P,M, Shitote, S.M., & Menzwa, D.M.. (2004). Performance of multistage filtration using different filter media against conventional water treatment systems. *Water SA, 30 (3)*, 361-368.

O'Connor, T. H. (2002). *Report of the Walkerton Inquiry : The Events of May 2000 and Related Issues.* Toronto: Ontario Ministry of the Attorney General.

Pacini, V. A., Ingallinella, A.M., Sanguinetti, G. (2005). Removal of iron and manganese using biological roughing up flow filtration technology. *Water Research, 39 (18)*, 4463-4475.

Pei, Y., Yu, J., Guo, Z., Zhang, Y., Yang, M., Zhang, J., & Junji, H. (2007). Pilot study on pre-ozonation enhanced drinking water treatment process. *Ozone: Science and Engineering, 29 (3)*, 317-323.

Peng, X., Murphy, T., & Holden, N.M. (2008). Evaluation of the Effect of Temperature on the Die-Off Rate for Cryptosporidium Parvum Oocysts in Water, Soils and Feces. *Applied and Environmental Microbiology, 74 (23)*, 7101-7107.

Persson, F., Heinicke, G., Uhl, W., Hedberg, T., & Hermansson, M.(2006). Performance of direct biofiltration of surface water for reduction of biodegradable organic matter and biofilm formation potential. *Environmental Technology, 27 (9)*, 1037-1045.

Rahman, M. F., Jasim, S.Y., Yanful, E.K., Ndiongue, S., & Borikar, D. (2010). Advanced oxidation treatment of drinking water: Part II. Turbidity, Particles, and Organics Removal from Lake Huron Water. *Ozone: Science & Engineering, 32 (5)*, 295-304.

Sanchez-Polo, M., Sahil, E., Rivera-Utrilla, J., & von Gunten, U. (2006). Combination of ozone with activated carbon as an alternative to conventional advanced oxidation processes. *Ozone: Science and Engineering, 28 (4)*, 237-245.

Schulz, C. (2014). Designing Integrated Ozone-Biofiltration Treatment Systems. *Ozone: Science and Engineering, 36 (3)*, 276-286.

Shin, H.-S., & Lim, K.-H. (1999). Spectroscopic and elemental investigation of microbial decomposition of aquatic fulvic acid in biological process of drinking water treatment. *Biodegradation, 7 (4)*, 287-295.

Simpson, D. (2008). Biofilm processes in biologically active carbon water purification. *Water Research, 42 (12)*, 2839-2848.

Slavik, I., Jehmlich, A., & Uhl, W.(2013). Impact of backwashing procedures on deep bed filtration productivity in drinking water treatment. *Water Research, 47 (16)*, 6348-6357.

Ufer, D. & Huck, P.M. (2001). Measurement of biomass activity in drinking water biofilters using a respirometric method. *Water Research, 35 (6)*, 1469-1477.

Velten, S., Boller, M., Koster, O., Helbing, J., Weilenmann, H-U., & Hammes, F.(2011). Development of biomass in a drinking water granular active carbon (GAC) filter. *Water Research, 45 (19)*, 6347-6354.

von Gunten, U. (2003). Ozonation of drinking water: Part I. Oxidation kinetics and p. *Water Research, 37 (7)*, 1443-1467.

von Gunten, U. (2003). Ozonation of drinking water: Part II. Disinfection and by-product formation in presence of bromide, iodide or chlorine. *Water Research, 37 (7)*, 1469-1487.

Wang, Y., Yu, J., Zhang, D., & Yang, M. (2014). Addition of hydrogen peroxide for the simultaneous control of bromate and odor during advanced drinking water treatment using ozone. *Journal of Environmental Sciences, 26 (13)*, 550-554.

Wert, E.C., Neemann, J.J., Rexing, D.J., & Zegers, R.E. (2008). Biofiltration for removal of BOM and residual ammonia following control of bromate formation. *Water Research, 42 (1-2)*, 372-378.

Xin, Y., Xiao-jian, Z., Xiao-ling, L., Xiao-dong, Z., Xiao-dongz, Z., & Zhan-sheng, W. (2003). Phopshorous limitation in biofiltration for drinking water treatment. *Journal of Environmental Sciences, 15 (4)*, 494-499.

Yan, M., Wang, D., Ma, X., Ni, J., & Zhang, H. (2010). THMs precursor removal by an integrated process of ozonation and biological granular activated carbon for typical Northern China water. *Separation and Purification Technology, 72 (3)*, 263-268.

Yapsakli, K., & Cecen, F. (2010). Effect of type of granular activated carbon on DOC biodegradation in biological activated carbon filters. *Process Biochemistry, 45 (3)*, 355-362.

Zhang, X-X., Zhang, Z-Y., Ma, L-P., Liu, N., Wu, B., Zhang, Y., Li, A-M., Cheng, S-P. (2010). Influences of hydraulic loading rate on SVOC removal and microbial community

structure in drinking water treatment biofilters. *Journal of Hazardous Materials, 178 (1-3_*, 652-657.

Zhou, H. & Smith, D.W. (2001). Advanced technologies in water and wastewater treatment. *Canadian Journal of Civil Engineering, 28 (1)*, 49-66.

Appendix A: Supplementary Results

The purpose of this section is to present supplementary results that were not included in Chapter 4.

A.1: Microbial Results

Microbial samples were collected at Site 1, but not at Site 2 due to their absence in the source water (Lac Chassignole). E.coli and enterococci analyses were completed using the IDEXX system described in Chapter 3.

Results for total E.coli values are given in Table A.1.

Table A.1: E.coli values.

	Phase 1 (1.55 mg O_3/mg DOC)			Phase 2 (0 mg O_3/mg DOC)		
	Max	Min	Average	Max	Min	Average
Influent	98.7	9.7	38.5	13.4	2	5.56
Train 1	13.5	2	5.7	0	0	0
Train 2	35.4	1	23.3	1.5	0	0.7

During Phase 1, the average log removal was 0.83 for Train 1 and 0.22 for Train 2. During Phase 2, the average log removal was 0.9 for Train 2 (the log removal could not be calculated for Train 1). The low removals may be a result of higher influent turbidity values that were seen during operation at Site 1.

Results for total enterococci removal are seen in Table A.2.

Table A.2: Enterrococci values.

	Phase 1 (1.55 mg O_3/mg DOC)			Phase 2 (0 mg O_3/mg DOC)		
	Max	Min	Average	Max	Min	Average
Influent	67.7	7.4	34.9	42.2	2	14.8
Train 1	18.7	1	8.04	1	0	0.33
Train 2	50.9	1	28.1	6.85	1	2.79

During Phase 1, the average log removal was 0.64 for Train 1 and 0.09 for Train 2. The average log removal during Phase 2 was 1.65 for Train 1 and 0.72 for Train 2. Again, the low removals may have been due to influent turbidity levels.

A.2 THM Formation Potential

Harmful DBPs can be controlled by removing NOM before disinfection. Previous studies have shown that the use of biofiltration and combined biofiltration/ozonation processes can reduce THMs, a precursor to DBPs. THMFP was monitored at Site 2, using the HACH THM Plus Method. The average values are shown in Figure A.1.

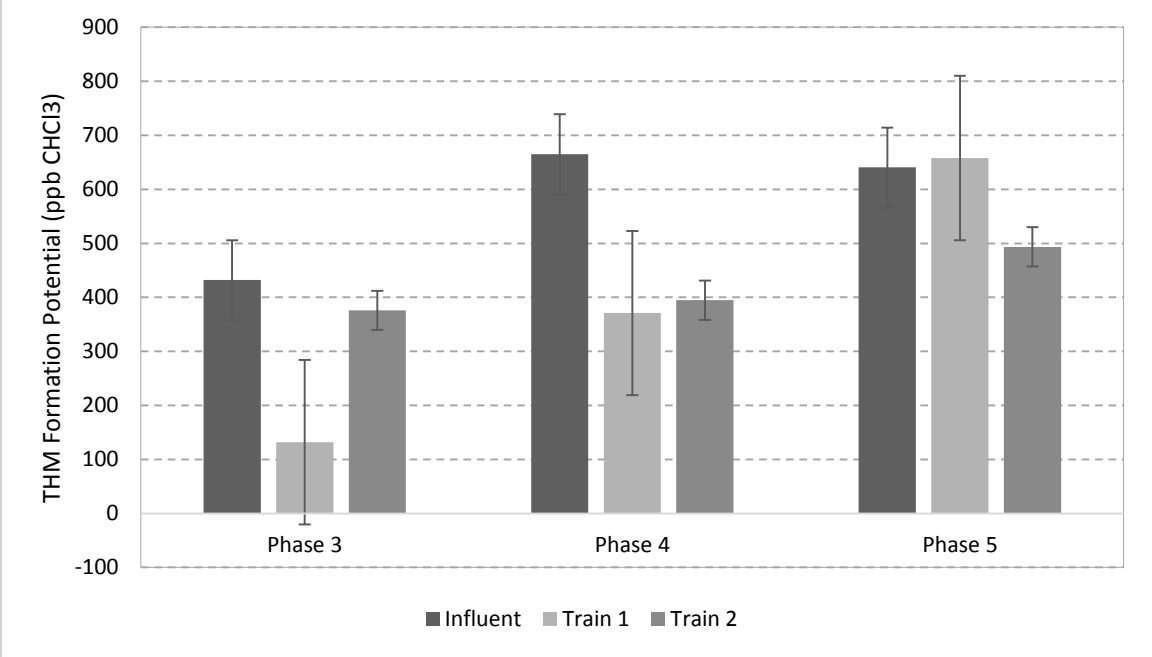


Figure A.1: THM Formation Potential.

The results from Figure A.1 are all in excess of the guideline of 100 µg/L. The average removal during phase 3 was 69.4% for Train 1 and 13% for Train 2. The average removal during phase 4 was 46.8% for Train 1 and 46% for Train 2. Finally, the average removal during phase 5 was 36.1% for Train 1 and 17.4% for Train 2. High removal at higher

ozone doses was expected based on previous literature. Chen & Wang (2012) found that biofiltration alone removed 44% of THM, and that an increasing ozone dose from 1.25 to 10 mg/L lead to a decrease of THM of 20% to 47% (Chen & Wang, 2012). In addition, it has been shown that high frequency backwash of biofilters can show an increased reducing in THM (Delatolla, et al., 2015).

A.3 Average SUVA Removal

SUVA represents a normalization of UVA_{254} by the DOC concentration. As mentioned, the SUVA values remained below 4 L/mg*m at the nutrient-limited site, and were above 4 L/mg*m at the nutrient-rich site.

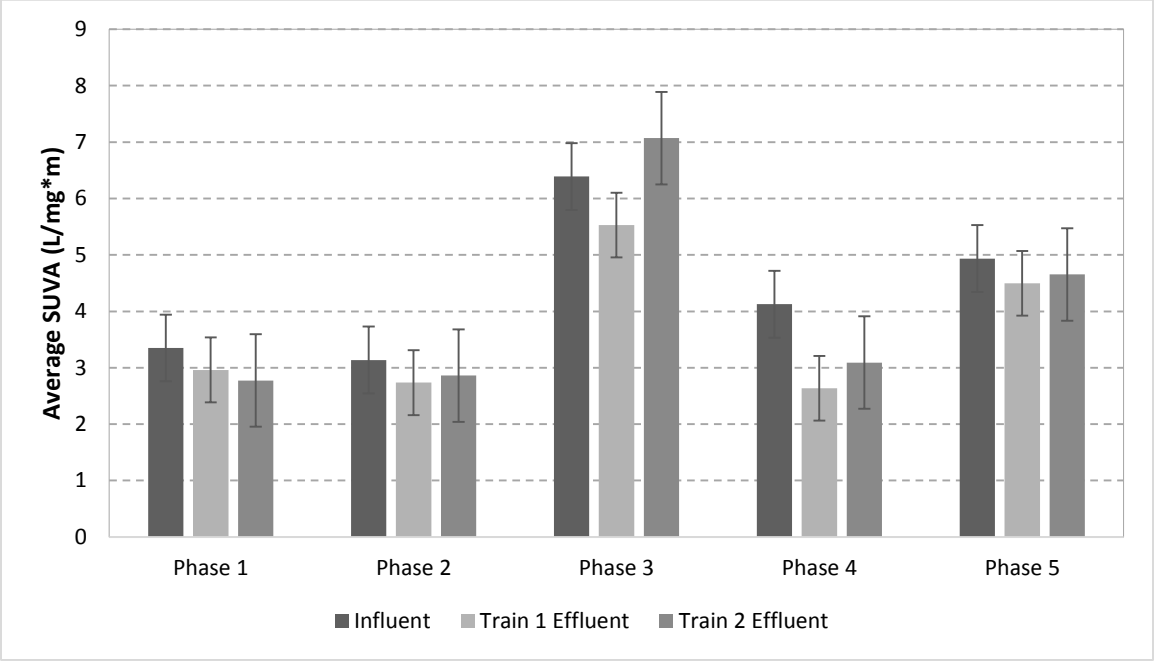


Figure A.2: Average influent and effluent SUVA values.

The correlation between DOC concentration and SUVA was plotted for each phase with the resultant R^2 values shown in Table A.3.

Table A.3: R^2 values for correlation between DOC concentration and SUVA.

	Ozonated			Unozonated	
	Phase 1	**Phase 3**	**Phase 4**	**Phase 2**	**Phase 5**
Raw Water	0.651	0.0208	0.6786	0.922	0.1526
Train 1	0.252	0.0006	0.0945	INS	0.0162
Train 2	0.243	0.1604	0.3955	0.923	0.0271

Ozonation is responsible for breaking down the aromatic portion of NOM (represented by UVA_{254}) into aliphatic compounds, which can be removed by filtration (Camel & Bermond, 1998). Phases 1, 3 and 4 all occurred at ozonated conditions; therefore, it was expected that the NOM passing through the filters was more biodegradable (von Gunten, 2003). The biodegradable portion was subsequently trapped in the filters, providing a higher DOC removal and therefore decreasing the DOC component in the SUVA calculation. The R^2 values for Train 1 and 2 effluents from the three ozonated phases were expected to be low, which was the case. During unozonated conditions (phases 2 and 5), it was expected that the DOC effluent values would be high and the correlation between DOC and SUVA would be stronger. This can be seen in Table 4.10 for phase 2, but did not occur during phase 5. This is because the SUVA values during phase 5 remained constant throughout operation, but a gradual increase in DOC was seen, leading to a lower correlation in both trains.

Appendix B – ATP Data

ATP Values Phase 1 (pg ATP/g sample)			
Days of Operation	**SSF**	**BAC1**	**BAC2**
35	54,749	233,634	387,000
42	55,542	248,377	286,281
48	70,140	252,088	378,448
56	74,022	450,018	352,578
63	82,123	432,168	604,247
71	127,183	7,834	21
77	166,481	13,379	19,404
84	221,943	107,505	133,231
ATP Values Phase 3 (pg ATP/g sample)			
Days of Operation	**SSF**	**BAC1**	**BAC2**
0	N/A	2,150,579	362,369
7	272,389	1,362,384	1,041,573
13	N/A	561,475	671,844
18	N/A	553,237	649,983
ATP Values Phase 4 (pg ATP/g sample)			
Days of Operation	**SSF**	**BAC1**	**BAC2**
25	178,460	693,741	688,014
27	641,441	N/A	N/A
34	364,304	N/A	N/A
39	278,624	792,691	1,587,891
43	666,551	N/A	675,496
53	543,189	250,597	23,167
57	568,652	611,788	593,610

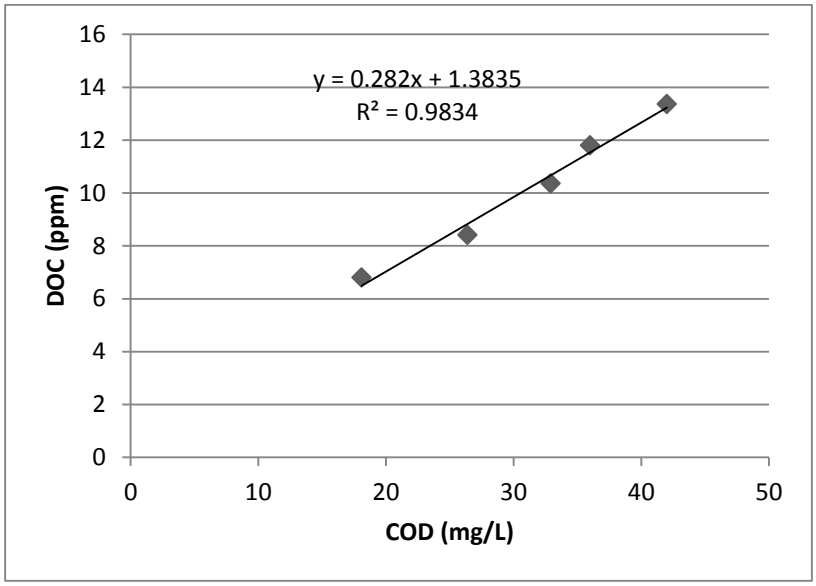
Appendix C – TOC Data

Phase 2 TOC Data				
Days of Operation	**RW**	**SSF**	**BAC1**	**BAC2**
254	8.087		7.443	7.47
262	15.9284		12.8084	12.8384
268	13.83	13.08	12.92	8.176
276	13.443	12.943	12.643	12.723
282	13.578	13.058	13.058	13.118
Phase 3 TOC Data				
Days of Operation	**RW**	**SSF**	**BAC1**	**BAC2**
0	9.5432	7.2479	8.8874	9.5432
2	8.5595	7.5758		7.9037
9	8.2316	6.5921	3.641	6.92
16	9.2153	8.5595	7.5758	7.5758
17	7.9037	7.9037	5.2805	5.9363
21	8.5595	7.5758	6.5921	5.6084
24	9.2153	7.9037	11.1827	5.2805
29	9.8711	8.2316	6.2642	4.2968
31	10.199	6.2642	4.2968	6.2642
35	8.8874	6.5921	7.2479	7.2479
38	11.1827	8.5595	8.8874	8.2316
42	9.5432	8.8874	9.5432	8.5595
44	12.1664	7.5758	6.92	8.8874
49	11.1827	5.6084	5.9363	7.2479
51	10.199	6.5921	8.5595	6.2642
52	15.7733	12.8222	14.7896	14.4617
57	15.7733	10.8548	13.8059	14.4617
58	16.757	13.1501	13.478	13.478
59	9.8711	5.2805	6.92	5.9363
63	17.0849	7.9037	15.7733	17.0849
65	11.5106	8.2316	9.5432	13.8059
66	11.5106	8.2316	9.5432	9.5432
70	12.8222	8.2316	10.199	9.5432
72	8.2316	7.5758	6.5921	8.2316
74	12.69	8.811	10.97	11.09
77	12.61	9.421	11.16	11.05
81	11.98	9.565	11.87	11.76
84	13.37	10.18	11.76	11.92
88	13.85	13.15	13.67	13.26
91	12.94	11.93	11.54	11.24
101	14.96	14.28	13.23	14.16

105	14.32	13.4	14.18	13.82
109	16.52	14.87	14.67	15.83
115	16.45	14.96	14.87	15
119	15.52	14.81	14.5	14.54
156	14.93	12.97	14.54	14.73

Appendix D – COD and DOC Calibration Curve

A calibration curve was prepared for COD and DOC concentration, due to that fact that Phase 3 occurred in a location without the capacity to carry out DOC measurements.



The equation of the line was used with COD measurements in order to find an appropriate DOC measurement.

Appendix E – C:N:P Ratio Calculations

The C:N:P ratio at Site 1 was obtained from data received by BBWTP. The C:N:P ratio at Site 2 had to be calculated with average phosphorus, nitrogen and carbon concentrations. The sample calculations are shown below.

[P] = 1.83 mg/L PO_4^{3-}

[N] = 3 mg/L N

[C] = 11.6 mg/L DOC

Therefore, C:N:P on a mass basis is based on [P].

$$C:N:P = \frac{11.6}{1.83}:\frac{3}{1.83}:\frac{1.83}{1.83}$$

$$C:N:P = 6.3:1.64:1$$

Appendix F – Ozone Dosage Calculations

Throughout the experimentation period, the ozone generator was set to two dosing positions, in addition to being turned off. The low dosage was set at 16 mg O_3/L and the high dosage was set at 44 mg O_3/L. Using the average DOC concentration in the influent stream, the ozone dosage in terms of mg DOC could be calculated. Sample calculations are seen below for Phase 3 and 4.

Phase 3 – Low Ozone Dosage

Average RW DOC Concentration = 8.38 mg DOC/L

Average Ozone Dosage = 16 mg O_3/L

$$Average\ Ozone\ Dose = \frac{Ozone\ Dosage}{RW\ DOC\ Concentration}$$

$$Average\ Ozone\ Dose = \frac{16\ mg\frac{O_3}{L}}{8.38\ mg\frac{DOC}{L}} = \frac{1.9\ mg\ O_3}{mg\ DOC}$$

Phase 4 – High Ozone Dosage

Average RW DOC Concentration = 12.76 mg DOC/L

Average Ozone Dosage = 44 mg O_3/L

$$Average\ Ozone\ Dose = \frac{44\ mg\frac{O_3}{L}}{12.76\ mg\frac{DOC}{L}} = \frac{3.45\ mg\ O_3}{mg\ DOC}$$

Appendix G – Backwash Rate Calculation

The backwash rates used throughout the experimentation period for the biofilters were found to be lower than what was calculated. The sample calculations for the theoretical backwash rate are seen below (assuming operating temperature of 15° C and bed expansion of 30%).

Step 1: Calculate depth of expanded bed

$$\frac{L_E}{L_F} = \frac{1 - \varepsilon_F}{1 - \varepsilon_E}$$

$$\varepsilon_E = 1 - \left[\frac{L_E}{L_F} * (1 - \varepsilon_F)\right]$$

$$\varepsilon_E = 1 - \left[\frac{0.845m}{0.65m} * (1 - 0.4)\right]$$

$$\varepsilon_E = 0.538$$

Step 2: Calculate the backwash calculation factor

$$\beta = \frac{g p_w (p_p - p_w) d^3 \varepsilon_E^3}{\mu^2}$$

$$\beta = \frac{\frac{9.81m}{s^2} * \frac{999.1kg}{m^3} * \left(\frac{1600kg}{m^3} - \frac{999.1kg}{m^3}\right) * (0.001378m)^3 (0.538)^3}{(\frac{0.001139kg}{m * s})^2}$$

$$\beta = 1853.57$$

Step 3: Calculate the Reynold's number

$$Re = \frac{-k_v(1-\varepsilon) + \sqrt{k_v^2(1-\varepsilon)^2 + 4k_l\beta}}{2k_l}$$

$$Re = \frac{-210(1-0.538) + \sqrt{210^2(1-0.538)^2 + 4*3.5*1853.57}}{2*3.5}$$

$$Re = 13.01$$

Step 4: Calculate the backwash velocity for the filters

$$v = \frac{\mu Re}{p_w d}$$

$$v = \frac{\frac{0.001139kg}{m*s} * 13.01}{\frac{999.1kg}{m^3} * 0.001378m}$$

$$v = \frac{0.011m}{s} = \frac{38.76m}{hr}$$

Step 5: Calculate volumetric flowrate

$$Q = v * A$$

$$Q = \frac{38.76m}{hr} * 0.07789m^2 * \left(\frac{1hr}{60mins}\right) * (\frac{1000L}{1m^3})$$

$$Q = 50.32\ L/min$$

The calculated backwash rate was therefore 50L/min, assuming a 30% bed expansion.

It was also necessary to determine what the actual bed expansion was at each of the operating sites. The sample calculation for bed expansion at Site 1 is shown below. The backwash rate at Site 1 was 24 L/min.

Step 1: Find backwash calculation factors

$$X = \frac{\mu * v}{2g(p_p - p_w) * d^2} * \left[k_v + \frac{k_l p_w v d}{\mu}\right]$$

$$X = \left[\frac{\frac{0.001139kg}{m*s} * \frac{0.0051m}{s}}{2 * 9.81\frac{m}{s^2}\left(1600\frac{kg}{m^3} - 999.1\frac{kg}{m^3}\right) * (0.001378m)^2}\right] * \left[210 + \frac{3.5 * 999.1\frac{kg}{m^3} * \frac{0.0051m}{s} * 0.001378m}{\frac{0.001139kg}{m*s}}\right]$$

$$X = 0.061$$

$$Y = \frac{k_v * \mu * v}{3g(p_p - p_w) * d^2}$$

$$Y = \frac{210 * \frac{0.001139kg}{m*s} * \frac{0.0051m}{s}}{3 * 9.81\frac{m}{s^2}\left(1600\frac{kg}{m^3} - 999.1\frac{kg}{m^3}\right) * (0.001378m)^2}$$

$$Y = 0.037$$

Step 2: Calculate porosity

$$\varepsilon_E = \sqrt[3]{X + (X^2 + Y^3)^{0.5}} + \sqrt[3]{X - (X^2 + Y^3)^{0.5}}$$

$$\varepsilon_E = \sqrt[3]{0.061 + (0.061^2 + 0.037^3)^{0.5}} + \sqrt[3]{0.061 - (0.061^2 + 0.037^3)^{0.5}}$$

$$\varepsilon_E = 0.42$$

Step 3: Calculate expanded bed depth

$$L_E = L_F * \left(\frac{1-\varepsilon_F}{1-\varepsilon_E}\right)$$

$$L_E = 0.6m * \left(\frac{1-0.4}{1-0.42}\right)$$

$$L_E = 0.62m$$

Step 4: Calculate the percent expansion of the bed

$$Expansion = \left(\frac{L_E}{L_F} - 1\right) * 100$$

$$Expansion = \left(\frac{0.62m}{0.6m} - 1\right) * 100$$

$$Expansion = 3.7\%$$

Therefore the expansion of the bed at Site 1 was 3.7%.