Evaluation of On-Farm Potato Processing Water Usage and Treatment

by

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ABSTRACT

EVALUATION OF ON-FARM POTATO PROCESSING WATER USAGE AND TREATMENT

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This research monitored water used by a potato storage facility for washing potatoes, and its subsequent treatment in an adjacent treatment system over two years. Water conservation strategies initiated in the second year (Y2) decreased water consumption by 40%, compared to the first year (Y1). The raw water had concentrations of 1121 and 558 mg L\(^{-1}\) BOD\(_5\), 5234 and 3600 mg L\(^{-1}\) TSS, 222 and 136 mg L\(^{-1}\) TN, 23.4 and 42.6 mg L\(^{-1}\) TP in year Y1 and Y2, respectively. The system had high mass removals of all parameters, ranging from 71% to 99% in both years. Laboratory experiments studied sand filtration and coagulation for supplementary TSS removal. Unsaturated filters with lowest dosage were the best performing filter and polymer Ultrion 8185 was the best performing coagulant. Further research is necessary to aid the development and implementation of simple and inexpensive standardized treatment systems for on-farm wash-water treatment.
To my family.
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List of abbreviations

Alum       Aluminum sulfate
APHA       American public health association
BCIG       5-bromo-4-chloro-3-indolyl-β-D-glucuronide
BOD₅       5-Day biochemical oxygen demand
BOD₇       7-Day biochemical oxygen demand
CH₄        Methane
CO₂        Carbon dioxide
COD        Chemical oxygen demand
DO         Dissolved oxygen
*E.coli*   *Escherichia coli*
ED         Effective dosage
EPA        Environmental protection agency
FeCl₃       Ferric Chloride
FM_flume   Flow meter to flume
FM_pump    Flow meter to pump
FM_tank    Flow meter to tank
HRT        Hydraulic retention time
N           Nitrogen
NOM        Natural organic matter
NH₃        Ammonia
NO₃        Nitrate
OCC        Optimum coagulant concentration
OPO₄       Ortho-Phosphate
ORP        Oxidation-reduction potential
P           Phosphorus
PWQG       Provincial water quality guidelines
C1         Sedimentation pond 1
C2         Sedimentation pond 2
TB1        Tipping bucket 1
TB2        Tipping bucket 2
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<tr>
<td>TDS</td>
<td>Total dissolved solids</td>
</tr>
<tr>
<td>TKN</td>
<td>Total Kheldal Nitrogen</td>
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<tr>
<td>TN</td>
<td>Total nitrogen</td>
</tr>
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List of units

mg  Milligram
kg  Kilogram
L   Liters
mL  Milliliter
m³  Cube Meter
cwt Hundred Weight
lb  US Pound
gal US Gallon
d  Day
mon Month
t  Ton
week Week
°C  Degrees Celsius
w  Watt
%  Percent
m²  Square Meter
ppm Part Per Million
mV Millivolts
g  Gram
m  Meter
$  Canadian Dollar
CFU Colony Forming Unit
min Minute
µg  Microgram
s  Second
nm Nanometers
µm Micrometers
Chapter 1 Introduction

1.1 Introduction
Management of on-farm vegetable wastewater is of growing concern to food growers, due to the increased pressures from processing industries (e.g., Pepsi Co., Frito-Lay, Unilever, Walmart) to wash and process product prior to delivery (Lehto et al., 2009). As a result on-farm washing of vegetables is increasing, however, water management strategies and removal regulations remain unclear. Thus, information on water-use, and mitigation opportunities are of interest to growers, and farm organizations, to aid in finding a convenient and inexpensive method for conserving water, while disposing of the wastewater created, and keeping within governmental regulations for surface discharge (Lehto et al., 2009; Ontario ministry of Environment, 2011).

Large volumes of wash-water used for on-farm transport and washing, are used prior to shipment. A lack of information and guidelines from governmental organizations leave farmers with either costly or poor management of the waste wash-water. Wastewater management strategies that are non-discharge are the least expensive for farmers. These include ground seepage or land spreading. Seepage can lead to infiltration of well water, while land spreading can cause runoff that pollutes surface water. Additionally, land spreading is only allowed during warmer months of the year, while processing may occur at any time of the year, or even predominantly in the winter. Discharge based management is more costly, because the water needs to be treated and the system approved by the provincial government (Ontario Ministry of Environment, 2011).

The question of wash-water management becomes more complex when put in perspective of water conservation and water usage calculation. Traditionally, water in agriculture is thought of in terms of irrigation, where water is calculated as the water input, because output cannot be easily calculated. In industrial settings, which on-farm processing is increasing becoming more similar to, water usage is calculated as the difference between water input and water output. Therefore, when water is managed through non-discharge techniques, the total water "usage"(water consumed for washing and processing) is larger, according to the agricultural formula, even though the actual water used may be much less. However, when the water is discharged it can be returned back to surface water bodies, which decreases the total water usage and in turn conserves water. There is currently no incentive for farmers to treat and discharge
water, which is more costly and labour intensive. Research is needed to aid governmental bodies in creating simple, reliable, and inexpensive treatment system designs to promote discharge management over non-discharge management of wash-water (Lehto et al., 2009).

There have been several studies analyzing crop wash-water and factory processing-water, including dried vine-fruit (Industrial; Australia; Gwatkin and Lane 1986), winery wastewater (Industrial; United States; Shepherd et al. 2001), romaine lettuce (On-farm; United States; Luo 2007), carrot (On-farm; Australia; Hamilton et al. 2008), potato processing (Industrial, United States, Hung 2006), and various root vegetables (On-farm; Finland, Lehto et al. 2009). The results of these studies show a high variability in wash-water characteristics. Differences in washing techniques, chemical additives, soil types, and other factors will influence wash-water. Therefore, it is important to have crop-specific and location-specific data since treatment design relies on accurate characterization of the wash-water attributes and production throughout the year. Lehto et al. (2009) examined on-farm processing water usage as well as its subsequent treatment. They studied three farms and found water consumption for washing and peeling of root vegetables to be 75, 35, and 25 m$^3$ wk$^{-1}$. None of the treatment systems were able to create effluent that would pass the Ontario provincial water quality guidelines.

More research on processing water is necessary to understand the variability among farms and their wash-water, as well as, the treatment necessary to meet governmental guidelines. This project attempts to fill in the knowledge gap by looking into inexpensive, low-maintenance, and sustainable options for potato crop wash-water treatment. Using the available resources that agricultural settings provide (such as large area, excavators, and other equipment), this research proposes a land-based treatment system. Series of subsurface ponds, each with a different environment, can create separate treatment processes, including sedimentation/primary treatment, aeration/nitrification/secondary treatment, anoxic/denitrification/secondary treatment, and treatment wetlands/adsorption/tertiary treatment (Tchobanoglous and Schroeder, 1987; Kadlec and Wallace, 2009).

Lehto et al.(2009) and Hung et al. (2006), who examined industrial potato processing water, found high concentrations of total suspended solids (TSS), organic matter measured through biochemical oxygen demand (BOD), and nutrients, such as N and P. In a treatment system each contaminant can be specifically targeted for treatment with a unique process and treatment cell
type. Sedimentation ponds are specialized ponds allowing slow water movement to let solids settle (Kadlec and Wallace 2009). ADAI (1995) found 95% reduction in TSS from a settling pond, composing 15% of the total treatment area. Sedimentation will remove also organic matter, and nutrients associated with solids, while the remaining can be removed through aerated and wetland cells. Aeration of water allows ammonia to be converted to nitrate by nitrifying bacteria, while in anoxic ponds nitrate is converted to nitrite and then to N₂ gas by denitrifying bacteria. Wetlands ponds inhabited with micro- and macro-organisms can breakdown and remove contaminants through adsorption or volatilization. Wetlands are often used best as tertiary treatment to remove residual contaminants.

The main contaminant expected is TSS, due to soil from the surface of the tubers (Hung et al., 2006). TSS is not always easily removed using sedimentation alone, depending on the size (Tchobanoglous and Schroeder, 1987). Clay and fine organic matter can be too small for sedimentations where the attracting forces between particles are larger than the downward gravitational pull. These particles are called colloidal. In this case, other, supplementary treatments are necessary to remove TSS. In order to enhance the treatment of wash-water, coagulation and filtration are explored.

Along with flocculation, coagulation allows for fast settling of solids and therefore a decrease in TSS and associated organic matter and nutrients. Coagulants work by destabilization of colloidal particles, such that particles attract each other more and grow in size, making them more suitable for sedimentation and filtration (Tchobanoglous and Schroeder, 1987). There are varying types of coagulants, each with a specific range of functionality based on pH. A study on pre-treatment of beverage industry wastewater by Amunda et al. (2006) found a decrease in chemical oxygen demand (COD), TSS, and total phosphorus (TP) by 78, 74, and 75 %, respectively. In winery wastewater experiments, coagulants were found to decrease TSS up to 95.4 %, turbidity 92.6 % and COD 29.7 % (Braz et al. 2010). However, no literature from on-farm coagulation of potato processing wastewater has been found during literature review.

Sand filtration has been used extensively in industrial wastewater treatment. It removes solids from wastewater through several processes including: mechanical, chance contact straining, impaction, interception, adsorption, flocculation, and sedimentation (Tchobanoglous and Schroeder, 1987). A study on turkey processing wastewater found sand filtration reduced total
organic carbon (TOC) by 94 % and BOD by 98% (Kang et al. 2006). Wilson et al. (2011) used lateral flow sand filters to treat sewage waste and found the decrease ranged from 82.4 to 96.9 % for TSS, and 96.7 to 98.4 % for BOD. Lastly, Liu et al. 2003 found a COD decrease of up to 53 % when sand filtration inoculated with lipolytic bacteria were used to treat a solution of 0.1 % milk fat and 0.04 % detergent. Currently there is no research on potato wastewater filtration available, however, it could be useful as an inexpensive supplementary treatment for TSS removal.

There is a need for research that explores on-farm wash-water requirements and its subsequent treatment using inexpensive, effective, and simple system designs to promote water conservation through discharging treated effluent. The use of land-based treatment systems and supplementary treatment with coagulants or filters can meet these criteria while discharging effluent that meets the governmental discharge guidelines and protects surface waters.

1.2 Hypotheses
A land-based treatment system can decrease the levels of contaminants in wash-water from a potato storage facility to meet Ontario discharge limits, while being low cost and easy to maintain.

The use of supplementary treatment systems, filtration and coagulation can help decrease wash-water TSS concentrations through removal of colloidal particles.

1.3 Research Objectives
- Monitor water-use of a potato storage facility for 12 months and calculate the ratio of water used for potato washing and how it changed following implementation of water conservation strategies. Water monitoring will include flow meter data from the facility and conservation metrics will include volume reductions (%) in order to quantify water usage by a potato storage facility and explore water conservation techniques.
- Quantify the efficiency of two treatment system designs at an Ontario potato farm, based on reduction of water contaminants (BOD$_5$, TSS, phosphorus, nitrogen, and E. coli) by determining initial and final water flow and composition throughout the wetland. Efficiency metrics will include concentration reductions (%), mass reductions (%)
- Determine efficiency of supplementary treatments, coagulation and filtration, to decrease water contaminants (BOD$_5$, TSS, phosphorus, nitrogen, and E. coli) as pre- or post-treatments. Efficiency
metrics will include concentration reductions (%), mass reductions (%), and cost-effectiveness calculations ($/removal).

1.4 **Significance of Work**
This research will explore the effectiveness of two treatment system designs and other supplementary systems for on-farm treatment of wash-water. A comprehensive study of one potato storage facility will give a detailed view of year-round total water usage, with a specific focus on water used for shipping. Lastly, this research will provide scientifically tested treatment options for potato growers and other farmers to use as a basis for treating wash-water and other on-farm water treatment, as well as guide for governmental agencies in creating standardized treatment designs that can further aid farmers in managing their wash-water.
Chapter 2 Water Usage and Balance of a Potato Storage Facility and Treatment System

ABSTRACT

Vast quantities of water are used every year for irrigation; however, little information is available on other agricultural uses. This study looks at the water used by an on-farm potato storage facility for transporting (fluming) and washing potatoes, and the flow through its adjacent treatment system over two years. A comparison between two years found that simple conservation strategies decreased water consumption by 40%, including installing flow restrictors, increasing dry scrubbing, and storage workers' water conservation consciousness. The amount of water used for processing potatoes was found to be 16.6 L cwt$^{-1}$ potatoes in Y1 and 8.68 L cwt$^{-1}$ in Y2. However, due to unclear methods for calculating water usage, if calculated as an industrial facility, Y2 of the study used 20% more water. Industrial systems count their water usage as the difference between the storage water intake and discharge. These usage values were 1829 m$^3$ in Y1 and 2285 m$^3$ in Y2, due to water loss in Y2 from the treatment system due to ground seepage. A zero-discharge system eliminates the need for a treatment system and dealing with governmental guidelines that saves farmers a lot of work and money. However, this also increases the water usage which is bad in terms of water conservation. More research is necessary to understand the water usage for processing crops on farms, to eliminate, or minimize the dichotomy of water conservation and cost and inconvenience to farmers.

Key words: Potato wash-water, wastewater treatment, water usage, conservation

2.1 Introduction

Water is essential in agriculture for varying uses, from irrigating fields to cleaning machinery. Although Canada has a large quantity of natural water, the annual renewable supply of fresh water from 1971 to 2004 has dropped by 8.5% (Statistics Canada, 2010). Statistics Canada (2001) estimated 4.8 billion cubic meters (4.6% total water used in all of Canada) of water were used in agriculture for irrigation, livestock, chemicals sprays and cleaning in 2001. Of this, 92% was used for irrigation. Under the Ontario water resources act, only farms that irrigate crops and rear livestock are required to meter their water usage (De Loe et al., 2001). Because of this, very
little focus is put on studying other water uses in agriculture. A Statistics Canada (2010)
publication on agricultural water use made no mention of other water uses aside from irrigation. Another Statistics Canada (2009) report, on human activity and environment, noted that 12% of farmers that used irrigation also used water for processing and packaging, with no mention of farmers who do not irrigate. In addition, the food industry accounts for 3% of total annual water used in Canada, but this value also does not include on farm processing and packaging (Statistics Canada 2010). Since most farmers use water sources that are not connected to a main water system, there is little information about the actual water usage (De Loe et al., 2001). Therefore, there is a knowledge gap in regards to data that accounts for on-farm water use for transporting/fluming, washing, processing and packaging. Water usage values are important for conservation purposes, and mapping out patterns of supply and use for future water availability, especially since on-farm processing of crops has seen an increased demand by food processing industries (De Loe et al., 2001).

Food processing water use is measured in a different manner than traditional agricultural water use. Agricultural irrigation requirements, or animal water consumptions, are measured only as the water input into the system, because the output cannot be easily measured (Statistics Canada, 2010). Legally, irrigation is not considered to have discharge, because field run-off is considered a non-point source discharge (Ontario Ministry of Environment, 2011). In industries, however, water use is calculated as the difference between the water taken in and the water discharged out (Statistics Canada, 2010). Once the water has been used, however, it is classified as wastewater and must be treated before discharging into the environment. Therefore, treatment should automatically be considered a part of the system. The water in the treatment system undergoes changes with both artificial (chemicals, dilutions, etc.) and natural inputs into and out of the system (precipitation, snowmelt, evaporation, soil leaching, etc.). Therefore, the final output from the industrial process is the output following the treatment (Statistics Canada, 2010).

One food crop that requires on-farm washing is the potato. Total water requirements for potato growing are smaller than for other crops, accounting for <1% total of agricultural water use. (For comparison, wheat accounts for 43% of total water use (Statistics Canada, 2010). However, this does not account for storing, processing and washing of the potatoes (Statistics Canada, 2010). Potato storage requires temperature and humidity control to maintain quality. The water used for creating humidity is lost to air and therefore can be measured only as an input. Processing
requires water for transport and washing – this is called wash-water. Transporting and washing potatoes happens within the storage facility while they are being moved from storage bins to shipping trailers using a flume and scrubbers. The flume is a channel system with water that transports the potatoes and the scrubbers are a set of moving finger rollers that propel the potatoes forward to remove dirt and debris from the potatoes. Unlike other agricultural waters, wash-water can be measured as an input and output, since this water is discharged into a treatment system at the end of each day (Statistics Canada, 2010).

This study measured the wash-water usage from an on-farm potato storage facility, for washing and transporting potatoes over the course of two storage seasons. The water flow through the treatment system, and the final discharge volumes were measured to find the overall water usage as a difference of input and output for potato processing. Additionally, the total water usage over the course of one year from the entire storage facility was determined based on the total input.

2.2 Materials and Methods

2.2.1 Storage Facility Description
The storage facility of SunRise Produce potato farm in Alliston, ON, housed approximately 300,000 cwt (3.0 x 10^7 lb; 1.4 x10^7 kg) potatoes each year. Several varieties of potatoes are grown, traditionally and organic, for varying purposes, although the majority of potatoes are shipped to potato chip factories. Beginning September, following harvest, potatoes are scrubbed of dirt and piled into storage bins, which are temperature and humidity controlled. Potato shipments may occur directly from the field; however, most potatoes are stored up to 10 months. When it was time for shipment, the potatoes were dry-cleaned with dirt-eliminating finger rollers to remove solids and rotten potatoes before entering water. Additional finger-rollers were added increasing the effective area to remove more solids in Y2 of the study. Following this they entered a flume system, used to wash and transport potatoes from the bins to the trailer. Further washing was done with more scrubbers and workers manually sorted potatoes on a rotating belt to remove rotten or damaged ones. Before being loaded into the trailer, the potatoes were sprayed with potable water. Wash-water, therefore, is the water used for transport and washing of potatoes prior to being shipped. The wash-water was pumped out of the storage facility directly into the primary sedimentation cell of the treatment system after each day of shipping. This creates an irregular loading schedule based on the demand of processing factories, with one
large volume loaded at the end of the shipping day. Because of the variability in number of potatoes shipped and amount of rot and soil, the wash-water will be of variable quantity and quality. All water used by the storage facility is local well water.

Three flow meters (Rain Bird ©, Azusa, CA, USA) were installed in August 2012 in the storage facility to monitor water usage. The flow meters were the same model fitted to 5/8” piping with resolution to 0.1 USG. Each meter measured a different water source with a different intended usage during processing. Flow meter 1 (FM_pump) measured water used to lubricate the water recirculation pump, flow meter 2 (FM_flume) measured water used to fill up the flume, and flow meter 3 (FM_tank) measured water used to fill up a large water tank used to wash the flume following a processing and flume water discharge. In Y2 FM_pump was equipped with an adjustable flow valve to restrict the water flow. Since the pump lubrication needed minimal water flow that was hard to control by the storage workers, the flow restrictor decreased the allowable area through which the water could flow out to a pre-set, minimal and no extra water was wasted. The other way water usage was modified was through the tank for flume washing. This was modified indirectly through the lowering of the solids in the water. Fewer solids meant fewer washing events were necessary and less water during each event. This water usage was controlled by the storage workers and therefore, it was their consciousness efforts for using minimal water on every given washing event that allowed for water reduction.

An additional flow meter (Stenner pump comp. ©, Jacksonville, FL, USA) was installed in February 2013 to measure the water usage by the entire storage facility. This flow meter was attached to a main pipe from carrying well water for the storage facility. The flow meter measured flow rate to a resolution of 0.1 USG, manufacturer specified accuracy of ±1.5%, and maximum flow rate of 160 gpm.

Data was recorded manually by the farm workers. Values for washing/processing were recorded following every shipping day, while values from the whole-storage flow meter were recorded approximately once a month to have total monthly usage. Each meter showed a cumulative flow (gal), therefore values were recorded in the same way. Flow meter data record sheets were under the responsibility of the storage secretary. The sheets were kept near the meters for ease of access by the farm workers. Once a sheet was full the secretary filed them for safe keeping.
Recorded sheets were scanned and emailed to researchers. Values for each shipping event were calculated using this equation:

\[
Water\ used = (\text{Volume}_{\text{Day } ii} - \text{Volume}_{\text{Day } i}) \times 3.78
\]  

[2.1]

where water used is the water used for the given shipping event (L); Volume\ _{\text{Day } ii} is the cumulative volume of water (gal) as recorded on the day in question; Volume\ _{\text{Day } i} is the cumulative volume of water (gal) as recorded on the day prior to the day in question; 3.78 is the conversion factor between gallons and liters.

2.2.2 Treatment Wetland Description

A wastewater treatment system was constructed August 2008, on SunRise Produce Potato Farm in Alliston, ON. The surface-flow treatment system had two components: two sedimentation basins (C1 and C2) and two wetlands (C3 and C4). The wetland was designed with practicality in mind, taking into account the local landscape, farm-used excavator, and least amount of labor. The size of the system was determined using the first-order plug flow design equation to meet 5–day biochemical oxygen demand (BOD\textsubscript{5}) and total suspended solids (TSS) treatment (Kadlec and Wallace, 2009):

\[
\ln \left( \frac{C_{\text{out}} - C^*}{C_{\text{in}} - C^*} \right) = -\frac{k_a}{q}
\]  

[2.2]

where \(C_{\text{out}}\) is the outlet concentration (mg L\textsuperscript{-1}), \(C_{\text{in}}\) is the inlet concentration (mg L\textsuperscript{-1}), \(C^*\) is the background concentration (mg L\textsuperscript{-1}), \(q\) is the hydraulic loading rate (m month\textsuperscript{-1}), and \(k_a\) is the first-order rate constant (m month\textsuperscript{-1}). The background concentrations used were 2 mg L\textsuperscript{-1} BOD\textsubscript{5} and 2 mg L\textsuperscript{-1} TSS, which are typical concentrations observed in the area (Jameison et al, 2007). However, \(k_a\) values and wash-water quality and quantity were estimated from literature values for other wastewater types because the characteristics of potato wash-water were unknown and no literature was available. The initial design estimated BOD\textsubscript{5} and TSS of 100-150 mg L\textsuperscript{-1} and degradation constants were estimated from literature to be 9 for TSS and 10 for BOD\textsubscript{5} (Table 2.1). This resulted in a design with overall area of 1356 m\textsuperscript{2}, volume capacity of 2457 m\textsuperscript{3}, and hydraulic retention time (HRT) of 82 d.

The system remained this way during Y1 of the study (September 2012 – July 2013). Prior to Y2 of the study (September 2013-July 2014), the system was expanded to meet the unexpectedly
high BOD<sub>5</sub> and TSS values found in the first year. The changes were based on BOD<sub>5</sub> of 1200 mg L<sup>-1</sup> and TSS of 5400 mg L<sup>-1</sup>. Wetland specifications for predicted, Y1 and Y2 are in Table 2.1.

Table 2.1 Comparison of Predicted, Y1, and Y2 effluent characteristics used for wetland design.

<table>
<thead>
<tr>
<th></th>
<th>BOD&lt;sub&gt;5&lt;/sub&gt; (mg L&lt;sup&gt;-1&lt;/sup&gt;)</th>
<th>TSS (mg L&lt;sup&gt;-1&lt;/sup&gt;)</th>
<th>Area (m&lt;sup&gt;2&lt;/sup&gt;)</th>
<th>Volume (m&lt;sup&gt;3&lt;/sup&gt;)</th>
<th>Average Inflow/month (m&lt;sup&gt;3&lt;/sup&gt;)</th>
<th>HRT (months)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Predicted</td>
<td>100-150</td>
<td>100-150</td>
<td>1356</td>
<td>2457</td>
<td>n/a</td>
<td>n/a</td>
</tr>
<tr>
<td>Y1</td>
<td>1200</td>
<td>5400</td>
<td>1119</td>
<td>740</td>
<td>404</td>
<td>2</td>
</tr>
<tr>
<td>Y2</td>
<td>956</td>
<td>3626</td>
<td>1321</td>
<td>1650</td>
<td>226</td>
<td>7</td>
</tr>
</tbody>
</table>

Water movement through the wetland was monitored by continuous measurement of two calibrated tipping buckets set up at the outflow of the third cell into the fourth and out of the entire treatment system. Hourly counts were recorded by a Campbell Scientific CR21X datalogger (Campbell Scientific, Edmonton, AB, Canada). This datalogger was replaced by a CR3000 (Campbell Scientific, Edmonton, AB, Canada) May 2013 (9 months after start of monitoring). Data was downloaded on a regular basis through a manual USB connection or wirelessly using two radios (RF401, Campbell Scientific, Edmonton, AB, Canada). One radio was attached to the CR3000, while the other was attached to a laptop located inside the storage facility. Through wireless internet access supplied by the farm, the laptop uploaded the data for easy retrieval from anywhere.

Tipping bucket tips was converted into volume and summarized into monthly totals:

$$Total \; Volume_{month} = \sum (Count_{hour} \times 1.5L)_{month}$$  \hspace{1cm} [2.3]

where Total Volume<sub>month</sub> is the total volume passing through the tipping bucket in that given month; Count<sub>hour</sub> is the recorded number of tips; 1.5 L is the average volume of water for each tip.
2.2.3 Statistical Analysis

Regression analysis was done using Microsoft Excel Professional Plus 2010 (Microsoft, Redmond, WA, USA). Lines of best fits were selected from several options (linear, linear-log, and polynomial) and selected based on $r^2$, p-value, and residuals.

Paired t-test were performed on total monthly water flows through each flow meter using SigmaPlot 12.0 (SigmaPlot, San Jose, CA, USA) to find statistical difference between Y1 and Y2 data. One-way ANOVA analysis was performed on monthly total water usage from FM_pump, FM_tank, and FM_flume for Y1 and Y2 using Sigma Plot 12.0, to test difference between water usages from Y1 to Y2.

2.3 Results and Discussion

2.3.1 Storage

2.3.1.1 Whole Storage Water Usage

For slightly more than one year, the entire storage facility used 5533 m$^3$ water (Feb 23, 2013 – Mar 3, 2014; 373 d). The average consumption per day was 14.7 m$^3$ and the total water used for shipping potatoes from this time period was 2423 m$^3$, which was 44% of the water used for the entire storage facility. The other 56% of the water was consumed by bathroom and kitchen facilities, equipment and storage cleaning, crop spraying, and humidity control for potato storage (listed in no particular order). The highest rate of use for the entire storage was seen in July at an average 27 m$^3$ d$^{-1}$, while the lowest was during one week in November at 1.7 m$^3$ d$^{-1}$.

2.3.1.2 Shipping Water Usage

The average monthly water usage was 404 ± 222 $^1$ m$^3$ in Y1 and 234 ± 148 m$^3$ in Y2 (Water use in Y1 was highest in January and July (Figure 2.1). Similarly, in Y2, the highest water usage was July and December. This suggests that January/December and July are seasonally busier times for shipping.

The average monthly potato shipment was 30,000±24,000 cwt month$^{-1}$ in Y1 and 25,000±19,000 cwt month$^{-1}$ in Y2. The large deviations reflect the varying demand for potatoes by processing

$^1$standard deviation
factories that buy potatoes from the farm (e.g., Frito-Lay Inc.). The number of trailer loads shipped was anywhere between 1 and 10 per day, and 0 to 112 per month. The total water usage for shipping alone in Y1 (October 2012 – July 2013) was 4044 m$^3$ and Y2 (October 2013 – July 2014) was 2345 m$^3$. There was a 42% reduction in total water usage from Y1 to Y2 ($p = 0.045$), while the total monthly potato shipment was not found to be statistically different ($p = 0.602$). This suggests that the decrease in wash-water in Y2 was not due to a difference in potatoes shipped but due to the conservation techniques.

Water use in Y1 was highest in January and July (Figure 2.1). Similarly, in Y2, the highest water usage was July and December. This suggests that January/December and July are seasonally busier times for shipping.

The ratio of FM_pump:FM_flume:FM_tank was different in Y1 and Y2. In Y1, the ratio was 3:3:4 and in Y2 it changed to 3:5:2 (Figure 2.2). One-way ANOVA analysis found that in Y1 the three flow meter volumes were not statistically different from each other, while in Y2 FM_pump and FM_tank were statistically the same, but FM_flume was different ($p = 0.513$ vs $p = 0.006$).

Water used for lubricating pump bearings (FM_pump) should depend on the amount of time spent shipping and the amount of potatoes shipped. Water used from FM_pump was 1289 m$^3$ (4.22 L cwt$^{-1}$) in Y1 and decreased by 53% in Y2 (611 m$^3$; 2.36 L cwt$^{-1}$; $p = 0.025$). After installation of the flow restrictor in January 2014, we saw a 69% reduction in water use from 4.84 L cwt$^{-1}$ in Y1 to 1.52 L cwt$^{-1}$ in Y2 ($p = 0.011$; comparing February – July; Figure 2.1). Due to this decrease, the proportion of total water used for shipping needed for lubricating the pump was reduced from 32% (Y1) to 26% (Y2). Regression analysis showed a strong correlation between FM_pump and total water used for shipping ($r^2 = 0.75$ for both years).

Water used for transporting the potatoes (FM_flume) should be dependent upon the amount of potatoes shipped, as well as the number of shipping days. Water used from FM_flume was 1274 m$^3$ (4.18 L cwt$^{-1}$) in Y1 and only a 2% reduction in Y2 (1250 m$^3$; 4.83 L cwt$^{-1}$; $p = 0.939$; Figure 2.2). Monthly average of water used was 5.47 L cwt$^{-1}$ in Y1 and 6.31 L cwt$^{-1}$ in Y2, however, this was not found to be statistically different ($p = 0.617$). Due to the decrease of flow through the other flow meters, the proportion of total water use for shipping needed to fill the flume increased from 32% (Y1) to 53% (Y2). Regression analysis showed a strong correlation between FM_flume and total water used for shipping ($r^2 = 0.80$ (Y1) and $r^2 = 0.82$ (Y2)).
Water used for filling up the reservoir tank (FM_tank), which in turn is used to clean the flume following shipping, should be dependent upon the number of shipping days (since cleaning the flume would occur following each shipping event), the amount of cleaning required (dependent on the amount of sediment in the water), and the cleaning process used. Water used from FM_tank was 1481 m³ (4.85 L cwt⁻¹) in Y1 and decreased by 67% in Y2 (483 m³; 1.87 L cwt⁻¹; p = 0.023; Figure 2.2) in Y2. The number of shipping d month⁻¹ were not found to be significantly different in Y1 (12.3 d month⁻¹) and Y2 (12.6 d month⁻¹; p = 0.904), however the amount of water used by FM_tank per shipping day was 67% lower in Y2 (3970 m³) than Y1 (12110m³; p = 0.002). Therefore, the storage workers were able to reduce the water through FM_tank in Y2 by decreasing flume washing events and decreasing water used for each individual washing event. Due to this decrease, the proportion of total water used for washing the flume was reduced from 37% (Y1) to 21% (Y2). Regression analysis showed a very strong correlation between FM_tank and total water used for shipping in Y1 and a modest correlation in Y2 (r² = 0.92 and r² = 0.39, respectively). This could be due to the decrease in flume cleaning events. Since the flume cleaning became less regular, it did not correlate as closely.
Table 2.2. Total water usage from the shipping season (10 months) and average monthly water usage from Y1 and Y2.

<table>
<thead>
<tr>
<th></th>
<th>Y1</th>
<th></th>
<th>Y2</th>
<th></th>
<th>p-value</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Total</td>
<td>Average</td>
<td>Total</td>
<td>Average</td>
<td></td>
</tr>
<tr>
<td>Potatoes (cwt)</td>
<td>305150</td>
<td>30515</td>
<td>258700</td>
<td>25870</td>
<td>0.602</td>
</tr>
<tr>
<td>Flow meters</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>FM-tot (m³)</td>
<td>4044</td>
<td>404</td>
<td>2344</td>
<td>234</td>
<td>0.045</td>
</tr>
<tr>
<td>FM_pump (m³)</td>
<td>1289</td>
<td>129</td>
<td>611</td>
<td>61</td>
<td>0.025</td>
</tr>
<tr>
<td>FM_flume (m³)</td>
<td>1274</td>
<td>127</td>
<td>1250</td>
<td>125</td>
<td>0.939</td>
</tr>
<tr>
<td>FM_tank (m³)</td>
<td>1481</td>
<td>148</td>
<td>483</td>
<td>48</td>
<td>0.023</td>
</tr>
<tr>
<td>FM_pump (L/cwt)</td>
<td>5.7</td>
<td>4.5</td>
<td></td>
<td></td>
<td>0.084</td>
</tr>
<tr>
<td>FM_flume (L/cwt)</td>
<td>5.5</td>
<td>6.3</td>
<td></td>
<td></td>
<td>0.617</td>
</tr>
<tr>
<td>FM_tank (L/cwt)</td>
<td>6.1</td>
<td>3.3</td>
<td></td>
<td></td>
<td>0.064</td>
</tr>
<tr>
<td>FM_tot (L/cwt)</td>
<td>17.3</td>
<td>14.4</td>
<td></td>
<td></td>
<td>0.131</td>
</tr>
<tr>
<td>Tipping buckets</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>TB1 (m³)</td>
<td>2215</td>
<td>222</td>
<td>1780</td>
<td>178</td>
<td>0.341</td>
</tr>
<tr>
<td>TB2 (m³)</td>
<td>473</td>
<td>47</td>
<td>53</td>
<td>5</td>
<td>0.055</td>
</tr>
<tr>
<td>Climate</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Rainfall (mm)</td>
<td>577</td>
<td>57.7</td>
<td>554</td>
<td>55.4</td>
<td>0.758</td>
</tr>
<tr>
<td>Temperature (°C)</td>
<td>n/a</td>
<td>5.6</td>
<td>n/a</td>
<td>3.3</td>
<td>0.028</td>
</tr>
<tr>
<td>Solar Radiation (W m²)</td>
<td>n/a</td>
<td>123</td>
<td>n/a</td>
<td>117</td>
<td>0.145</td>
</tr>
</tbody>
</table>
Figure 2.1. Water used for shipping (m$^3$), quantity of potatoes shipped (cwt), and wetland flow through the first tipping bucket (m$^3$), monthly totals from Y1 (2012-2013) and Y2 (2013-2014) of study.
Figure 2.2 Total water usage from each flow meter and their totals from Y1 (2012-2013) and Y2 (2013-2014) of the study.
2.3.1.3 Literature Comparison

Few studies that have looked at on-farm crop processing wastewaters. Our study farm was 10 × more efficient than the farms studied by Lehto et al. (2009) and more than 3 × efficient than industrial factory potato processing (Hung, Y. 1983). Y1 had a monthly average of 0.017 m³cwt⁻¹ and Y2 0.014 m³cwt⁻¹. Lehto et al. (2009) looked at on-farm washing and peeling of various root vegetables on three different farms in Finland (farms A, B, and C). Farm A used 108 m³ mon⁻¹ (25 m³ week⁻¹ for roughly 10,000 kg of carrots) and 0.11 m³ cwt⁻¹ (2.5 m³ t⁻¹) for washing carrots. Farm B used 152 m³ mon⁻¹ (35 m³ week⁻¹ for 12,000 kg of vegetables) and 0.13 m³ cwt⁻¹ (2.9 m³ t⁻¹) for dry and wet peeling various vegetables. Lastly, farm C used 108 m³ mon⁻¹ (25 m³ week⁻¹ for 8,000 kg potatoes) and 0.14 m³ cwt⁻¹ (3.1 m³ t⁻¹) for dry peeling potatoes. Lehto et al. (2009) do not give a more detailed account of how this water was used aside from peeling and washing. Industrial factory potato processing used even more water with 0.25 – 0.4 m³ cwt⁻¹ for fluming and washing potatoes (Guttormsen and Carlson, 1969). Lastly, research on a French fry processing factory from more than 20 years ago used on average 0.038 m³cwt⁻¹, or 17 L to process 1 kg of potatoes, although modern factories may be more efficient (Hung, 1983). More studies are needed to get a better representation of on-farm wash-water usage.

2.3.2 Treatment System Outflow

The total volume of water passing through TB1 in Y1 was 2215 m³ and 1780 m³ in Y2. Monthly total flow through TB1 was not found to be statistically different from Y1 to Y2 (p = 0.341; Water use in Y1 was highest in January and July (Figure 2.1). Similarly, in Y2, the highest water usage was July and December. This suggests that January/December and July are seasonally busier times for shipping.

The total water flow through TB1 in Y1 was 55% of the total water input into the treatment system (the total of all three flow meters). In Y2, the total water through TB1 was 75% of the total water input into the treatment system. In both years, TB1 flow was statistically different than the inflow water (p<.001). Linear regression of monthly total volume from TB1 and inflow total found strong correlation in Y1 (r = 0.72; r² = 0.51). In Y2 there was no correlation found (r=0). The lack of a correlation in Y2 is most likely due to the increased size of the wetland. Because there was no water flow through TB1 until December, the pattern of flow was distorted by the pooling in the first three cells of the treatment system.
TB2 was located at the outflow of the treatment system for both years. This means that in Y2 the tipping bucket was moved to account for the two new cells created. The total volume of water passing through TB2 was 473 m$^3$ in Y1 and 53 m$^3$ in Y2. Monthly flow through TB2 was not statistically different from Y1 and Y2 ($p = 0.055$). Water use in Y1 was highest in January and July (Figure 2.1). Similarly, in Y2, the highest water usage was July and December. This suggests that January/December and July are seasonally busier times for shipping.

The outflow from TB2 does not correspond to the flow through TB1 and inflow measured by the flow meters. Therefore, we suggest that there were measurement errors from malfunctioning of TB2 in Y1. The total water through TB2 was 12% of the total water input into the treatment system for Y1. In Y2, the total water through TB2 was only 2% of the total water input into the treatment system. In both years, TB2 flow was statistically different than the inflow water ($p<0.001$). The dramatic decrease in outflow in Y2 was due to water leaching into the soil in the newly built cells. Flow into the cells, and pooling from rain and snow melt, was observed, however, each time the water would slowly disappear over the course of a few days. The leaching was so large that only in July were tips registered, when a large amount of shipping was done over consecutive days. Linear regression of monthly total volume from TB1 and inflow total was found to have no correlation in both Y1 and Y2 ($r=0$). This lack of correlation in Y1 is further evidence of the poor quality of the TB2 data in Y1. Considering there was only one treatment cell between TB1 and TB2 in Y1, there should be little difference between them. Due to the poor confidence in TB2 in Y1, the data from TB1 is used as the estimated outflow values. Since there was only C4 between the two tipping buckets we assume the flows should be very similar.

2.3.3 Water Balance

The total consumption for washing/processing of potatoes in Y2 was decreased by 42% through cooperation of researchers with farmers. Periodic updates were shared with the farm management and discussions resulted in revised protocols for water management and a combined effort to decrease water consumption. Specifically reducing water for lubricating pump, reducing water for cleaning the flume, and decreasing the amount of solids on the potatoes which in turn additionally decrease the amount of water needed to clean flume. Water consumption was decreased by 53% for pump lubrication and 67% for flume washing. None of these changes
would have been possible without the cooperation of farm staff. As many of these changes were not implemented until January of the second shipping season we expect that larger reductions are possible in future years. In the end, continued practice of water reduction strategies and consciousness on the part of the storage workers will make the biggest difference. The ability to greatly reduce water in the storage in Y2 highlights value of agricultural research conducted with ongoing dialogue between researchers and farmers.

Although water consumption was decreased in Y2, the question remains as how to calculate and report water usage. In agriculture, water usage is usually thought of in terms of irrigation—the most water intensive activity. In irrigation, field run-off is not considered a point-discharge of “used” water, therefore, water outflow is considered to be zero. On-farm processing facilities, however, are quickly becoming more like industrial institutions. In industrial operations the water usage is calculated as the difference between the water intake and the (treated) water outflow. Using the first measure, Y2 used 42% less water than Y1. Using industrial calculations, however, Y2 used 20% more water due to the smaller water discharge. Y1 ‘industrial’ net water use, with outflow values from TB1, was 1829 m$^3$ ($= 4044$ m$^3$ - 2215 m$^3$). The remainder of the water in the system did not outflow the treatment system was spread onto fields in the summer. As this water was not discharged into surface water, it is here considered water used for on-farm purposes. In Y2 the outflow from the system was measured using TB2 and the total water usage was 2285 m$^3$ ($= 2345$ m$^3$ - 59 m$^3$).

The water balance for Y1 and Y2 both show that there was considerable water loss from the treatment system, measured as the difference between the inflow rate and the outflow rate (not counting the pumping out of water onto fields at the end of the season)—greater than we would expect from evaporation. Therefore, water loss could be due to ground seepage or leakage from the treatment cells. In Y1 and Y2 leakage from the cells was observed, especially during times when the outflow pipes were clogged with debris. The pressure from the water would become too large and spillage over the side would occur. It is unclear where most of the water loss occurred in Y1. Y2 however, the majority of the water loss occurred from ground seepage in the two new wetland cells. Storage workers observed rain accumulation and water flow into the cells, there were no areas of leakage observed, and the water was gone in several days. Ground infiltration is believed to be the reason why there was more water loss in Y2 – decreased discharge due to loss to the environment.
When it comes to dealing with waste wash-water for a farmer, there are two considerations. Allowing the water to seep into the ground or spreading the water onto fields, means that the water used for shipping is all dissipated to groundwater or water vapor and there is no outflow to surface water. For the farm, if there is no water discharge, then there are no provincial discharge limits to adhere to and no additional costs associated with treatment. Therefore, a no discharge system is less hassle and no cost. As water conservation is concerned, however, this is a poor way of managing water. It would be preferable to discharge or re-use treated wastewater and keep the total amount of water usage lower. This decreases the net amount of water taken from natural water bodies, including groundwater. We can assume that most farmers will chose cheaper options for dealing with wastewater. There is limited knowledge on what practices farmers currently use for water disposal, therefore, an assessment of potato farmers and other crop farmers is necessary, in order to gain a greater understanding of the processing water volumes and treatment techniques employed. Furthermore, there is need for more research to aid governmental bodies in creating standards that highlight a clear definition of water usage and can advise farmers on the appropriate ways to deal with wastewater that can work hand-in-hand with conservation techniques. Additionally, there is need for standardized, straightforward, and inexpensive wastewater treatments to protect surface water.

2.4 Conclusion

The whole storage facility used 5500 m$^3$ of water in Y1 (February 2013-February 2014), of which 44% was used for shipping potatoes. The amount of water used for processing potatoes was found to be on average 17.3 L cwt$^{-1}$ potatoes in Y1 and 14.4 L cwt$^{-1}$ in Y2. In Y2 water usage was statistically lower in FM_pump and FM_tank, although the water to FM_flume remained the same. Water to the pump was decrease by installing a flow restrictor. Flume washing (FM_tank) was decreased by reducing flume washing events and indirectly by reducing solids entering the wash-water. The question remains how to measure water usage from an on-farm processing facility. Although the sum the total water used for shipping (FM_tot) was 4044 m$^3$ in Y1 and 2345 m$^3$ in Y2, when the water usage was calculated as an industrial system, the difference between the intake and the outflow, the usage was 1829 m$^3$ in Y1 and 2285 m$^3$ in Y2. Due to increased leaching into the soil in the second year, there was less discharge and therefore 20% higher net water usage by the farm as would be calculated for an industrial system.
Although more water was used in terms of water conservation, for the farm this has benefits. When there is no surface discharge, there is no need for treatment systems and no need to deal with governmental discharge guidelines. There appears to be a dichotomy between water conservation and the least expensive techniques of protecting surface waters from on-farm wastewaters. Therefore, there is need for more research to aid governmental bodies in creating standards that highlight a clear definition of water usage and can advise farmers on the appropriate ways to deal with wastewater that can work hand-in-hand with conservation techniques and create standardized, simple, and inexpensive wastewater treatments to protect surface- and ground-water in terms of both quantity and quality.
Chapter 3 Performance of a Land-Based On-Farm Treatment System to Treat Potato Wash-Water

ABSTRACT

There is a growing demand for farmers to process their own crops on-farm, creating large quantities of wastewater. Governmental discharge regulations and the lack of other guidance for dealing with disposing of this wastewater leave farmers with expensive or poor management. This study looks at the composition and treatment of waste wash-water from an Ontario potato farm. The raw wash-water was found to have concentrations of 1121 and 558 mg L$^{-1}$ BOD$_5$, 5234 and 3600 mg L$^{-1}$ TSS, 222 and 136 mg L$^{-1}$ TN, 23.4 and 42.6 mg L$^{-1}$ TP, and 7.36 and 6.55 log CFU 100 ml$^{-1}$ E.coli in Y1 and Y2, respectively. The treatment system had mass removals of 95 and 99 % of BOD$_5$, 98 and 99 % TSS, 77 and 99 % TN, 77 and 99 % TP, and 71 and 99% log E.coli in Y1 and Y2, respectively. These removals, however, were not enough to reduce effluent concentrations to below governmental guidelines on an annual average. Further research is necessary to further characterise wash-water from other Ontario potato farms and aid the government to create simple, affordable, and standardized treatment systems for on-farm wash-water treatment.

Key words: wastewater treatment, potato wash-water, discharge guidelines

3.1 Introduction

For fruit and vegetable farmers, the management of agricultural crop wash-water is a growing concern, since processors and retailers (e.g., Pepsi Co., Frito-Lay, Unilever, and Walmart) are increasingly requiring farmers to wash and pre-process crops. As a result, on-farm processing of fruits and vegetables is common, while regulations for treating/disposal of the resultant large quantity of waste wash-water remain unclear.

Wash-water is defined as the water used for any post-harvest on-farm processing of fruit and vegetable crops. In the case of potatoes, wash-water is used for fluming and washing, although peeling, cutting, and packaging are also common uses (Hung et al., 2006). Often fluming water is recirculated throughout the processing day. Consequently, a large sum of waste wash-water remains for disposal. Anecdotally, this is discharged into an outdoor earthen basin for holding,
and further disposal may occur through allowing seepage into the ground, spreading onto fields, or discharge to surface water. There is a problem with each of those options. Seepage into the ground can be dangerous, since it may contaminate ground water used for drinking. Spreading wastewater onto a field may cause runoff into surface water causing contamination, if done improperly. Additionally, field application is allowed only during warm seasons whereas crop processing may occur at any time of year, potentially more often in winter months. Lastly, discharging any kind of wastewater into surface water requires an Environmental Compliance Approval and the use of a recognized treatment system, both of which are a major expense for farmers. Environmental Compliance Approvals acknowledge compliance with a set of discharge criteria created on a case-by-case environmental assessment of the receiving body of water (Ontario Government, 1994). Currently, discharging farm wash-water is considered sewage waste discharge under the Ontario Water Resources Act. This act has been designed with industrial wastewaters in mind. It is unknown how wash-water compares to industrial wastewater.

There is a lack of research on crop wash-water contaminant concentrations. Only three studies have monitored crop processing wastewater and each studied a different crop, as well as analysed them for different contaminants. Romaine lettuce was studied by Luo (2007) in regards to the effect of reusing wash-water on microbial growth on lettuce leaves. They found the chemical oxygen demand (COD) was 1721 mg L\(^{-1}\), 5-day biochemical oxygen demand (BOD\(_5\)) was 526 mg L\(^{-1}\), and total dissolved solids (TDS) was 719 mg L\(^{-1}\). Carrot wash-water was found to have a BOD\(_5\) of 29.6 mg L\(^{-1}\), turbidity of 195NTU nitrite-N of 0.15 mg L\(^{-1}\), nitrate-N of 1.15 mg L\(^{-1}\), and soluble reactive phosphorus of 32.7 mg L\(^{-1}\) (Hamilton et al. 2008). Lastly, Lehto et al. (2009) measured processing water from three different farms and their subsequent treatment systems in Finland processing potatoes and other root vegetable. Potato processing water contained BOD\(_7\) of 820-3700 mg L\(^{-1}\), total solids (TS) of 150-2600 mg L\(^{-1}\), total nitrogen (TN) of 28-320 mg L\(^{-1}\), and total phosphorus (TP) of 4.0-34 mg L\(^{-1}\). There has not been any published research on potato wash-water and its treatment in Canada.

In industrial potato processing (e.g. French fries, canning) the contaminants present in the wastewater are dependent upon the characteristics of potatoes being processed and the extent of potato processing (i.e., cutting, slicing, peeling) (Hung et al., 2006; Gottormsen and Carlson, 1969). The United States Department of Agriculture states that potatoes contain 2% protein, 17%
carbohydrates, 79% water, as well as 0.6 mg g\(^{-1}\) of phosphorus (US Dept. Agriculture, 2010). The nitrogen in potatoes is 16 g N per 100 g protein (conversion factor of 6.25; VanGelder, 1981). A medium sized potato (213 g) would contain 4.30 g protein, 690 mg N, and 121 mg P (US Dept. Agriculture, 2010). Therefore, wash-water is likely to contain large concentrations of BOD\(_5\), N, and P. Soil being washed off the surface of the potatoes will also contribute high levels of total suspended solids (TSS). Contaminant concentrations will depend on the variety of the potato, smoothness of the skin, uniformity of size and shape, amount of rot, type of soil the potatoes were grown in, and even the moisture of the soil during harvest (Hung et al., 2006).

Factory processing waters have been found to contain organic matter in dissolved or finely dispersed form contributing 2-6 lb of BOD\(_5\) per ton of potatoes, up to 10,000 mg L\(^{-1}\) COD, and up to 9,700 mg L\(^{-1}\) TSS (Guttormsen and Carlson, 1969; Hung et al., 2006).

Most wastewaters require specifically designed treatment systems based on the contaminants present, volume of water, and other considerations such as location, resources, and funds. Currently there is no research available on potato wash-water treatment; however, wastewater treatment in other agricultural settings is common (e.g., slaughterhouse, dairy milkhouse, cattle, piggery). Treatment of each contaminant (TSS, BOD\(_5\), P, and N) requires different techniques. TSS can be solids of varying size and origin, for example, sand, clay, or suspended organic matter. The speed of sedimentation depends on physical characteristics and the strength of the attracting/repelling forces acting upon them. Sedimentation ponds are specialized treatment cells which passively remove solids by slowing and extending horizontal water movement to allow gravitational forces to settle the solids (Kadlec and Wallace, 2009). BOD\(_5\) represents the amount of dissolved oxygen needed for micro- and macroorganisms in the water to break down organic matter in 5 d. Therefore, BOD\(_5\) is an indirect measure of the amount of organic matter in the wastewater. Organic matter can be removed through chemical addition (oxidation or coagulation), however, biological treatment is usually lower in cost and labour. Biological treatment involves creating an environment suitable for microorganisms to breakdown the organic matter into its constituents for adsorption and volatilization. This suitable environment can be achieved by adjusting pH and injecting activated sludge or creating treatment wetlands that have many surfaces for the bacteria to grow on, as well as plants to uptake additional carbon and nutrients during their growth. BOD\(_5\) has been shown to be reduced by 68 - 99% in cold climate treatment wetlands (Pries et al., 1996; Newman et al., 1999; Schaafsma et al., 1999;
Knight et al., 2000; Smith et al., 2005). Phosphorus is removed through movement into soil and plant uptake. Phosphorus has been found to have 35-96% removal in treatment wetlands (Majer Newman et al., 1999; Schaafsma et al., 1999; Poach et al., 2003; Gottschall et al., 2007). Nitrogen has a complicated process of removal through nitrification, denitrification, and volatilization or adsorption. Since most of the N in wash-water is likely to be in the form of ammonia, or organic nitrogen, the nitrogen will have to go through the whole process before volatilization. Nitrification requires an oxygenated environment, while denitrification requires an anoxic one. This process of removal can be achieved through repeated aerated and non-aerated treatment cells. Following this, denitrifying bacteria will use nitrate and carbon to make nitrogen and carbon dioxide gases that volatilize. Treatment wetlands have resulted in nitrogen removal rates of 32-98% (Hammer, Pullin et al. 1993; Majer Newman, Clausen et al. 1999; Schaafsma, H. Baldwin et al. 1999; Poach, Hunt et al. 2003; Gottschall, Boutin et al. 2007).

Currently there is little published research characterising the production or treatment of potato wash-water in Canada, therefore this research examined the contaminants in raw potato wash-water over two shipping seasons in an Ontario on-farm potato storage facility. This research also assessed contaminant removal through a low-cost, low-manpower wash-water treatment system in two configurations. The aim of this study was to inform the design of affordable treatment systems to aid farmers in consistently treating wash-water to meet discharge guidelines.

### 3.2 Materials and Methods

#### 3.2.1 Site Description

The storage facility of SunRise Produce Potato Farm in Alliston, ON, can store over 300,000 cwt potatoes each year. Following harvest, in late summer, potatoes are scrubbed of dirt and piled into storage bins which are temperature and humidity-controlled. Potatoes are then gradually taken out of storage and shipped by trailer-load to potato processing factories, which use potatoes throughout the year. Potato shipment may occur directly from the field, however, most potatoes are stored (for up to 10 months). When shipping from storage bins, potatoes are dry scrubbed with finger rollers once again to remove solids and rotten pieces before entering a flume system (a narrow channel in-laid in the flooring to move water and potatoes) that is used to wash and transport potatoes from the storage bins to the shipping trailer. More solids are then removed using a wet scrubber and finally, workers manually sort the potatoes to remove those
with defects. Before being loaded into the trailer, the potatoes are spray-washed with potable water. All water that has been used is collected and is referred to as wash-water. After each day of shipping the wash-water is pumped out of the storage facility into the treatment system. Therefore, the treatment system is loaded with one large volume once a day. Due to the variability in the amount of potatoes shipped, occurrence of rot and soil presence on any given shipping day the wash-water may be of variable quantity and quality.

Based on the results found in Y1 of the study (September 2012 - July 2013), measures were taken to decrease the solids and other contaminants entering wash-water and water usage during shipping. The finger rollers were expanded to 4 x its length with additional rollers to remove more solids. Cull piles which leaked liquid rot into the flume were moved to an outside storage. Lastly, overall water usage was decreased by 42% (see Chapter 2).

A surface-flow, land-based treatment system was constructed near the storage facility on the farm property to collect wash-water discharge. The system was originally built (2008) with two components: two sedimentation basins (C1 and C2) and two wetlands (C3 and C4) planted with locally growing cattails (Typha latifolia; Figure 3.1). The system was designed with practicality in mind, taking into account the local landscape, farm-used excavator, and minimal labour requirements for upkeep. The size of the system was determined by assuming plug-flow kinetics and using the first-order design equation to meet BOD₅ and TSS treatment (Kadlec and Wallace, 2009). Due to the lack of potato or other crop wash-water research, literature on other wastewater types was used to estimate values for BOD₅ and TSS. The initial design was based on an estimated BOD₅ and TSS of 100-150 mg L⁻¹. This resulted in a design with overall area of 1356 m², volume capacity of 2457 m³, and nominal hydraulic retention time (HRT) of 82 d (Table 3.1).

In Y1, 2012-2013, the treatment system had an actual area of 1119 m² and volume of 740 m³ (Table 3.2). Following shipment, the wash-water was discharged into the first sedimentation cell (C1). Horizontal flow through the wetland was passive, with PVC piping connecting the cells. In order to distribute flow as much as possible, the pipes were placed on the opposite facing corner of adjacent cell. The second cell (C2) was equipped with a surface, fountain aerator (Appendix Image 8.10). The last two cells (C3 and C4) were designed as treatment wetlands. Cattails grew
only sparsely on the edges of C3, most likely due to higher water levels; however, the C4 had a rich growth of cattails throughout the cell.

In Y2, 2013-2014, the treatment system was enlarged to 1321 m$^2$ and volume of 1650 m$^3$ following annual cleaning done by the storage workers (Table 3.2). Annual cleaning involved pumping out water from non-planted cells, in the time following potato shipment and prior to harvest. The water and sediment removed from the cells were spread onto fields (Appendix 8.3.1). Two additional wetland cells were built at the end of the system (C5, C6) in Y2 as well. These cells each had an area of 100 m$^2$ and depths of 0.5 m. Cattails were transplanted, although their first year had shown little growth. These cells had a lot of ground seepage throughout Y2 making the outflow from Y2 almost non-existent.

Table 3.1 Comparison of Estimated, year 1 and year 2 effluent characteristics used for wetland design (October to July).

<table>
<thead>
<tr>
<th></th>
<th>BOD$_5$ (mg L$^{-1}$)</th>
<th>TSS (mg L$^{-1}$)</th>
<th>Area (m$^2$)</th>
<th>Volume (m$^3$)</th>
<th>Average Inflow (m$^3$ month$^{-1}$)</th>
<th>HRT (months)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Estimated</td>
<td>100-150</td>
<td>100-150</td>
<td>1356</td>
<td>2457</td>
<td>n/a</td>
<td>n/a</td>
</tr>
<tr>
<td>Y1</td>
<td>1200</td>
<td>5400</td>
<td>1119</td>
<td>740</td>
<td>404</td>
<td>1.7</td>
</tr>
<tr>
<td>Y2</td>
<td>956</td>
<td>3626</td>
<td>1321</td>
<td>1650</td>
<td>226</td>
<td>7.3</td>
</tr>
</tbody>
</table>

Table 3.2 Characteristics of each treatment cell in year 1 and year 2.

<table>
<thead>
<tr>
<th>Feature</th>
<th>C1 Y1</th>
<th>C1 Y2</th>
<th>C2 Y1</th>
<th>C2 Y2</th>
<th>C3 Y1</th>
<th>C3 Y2</th>
<th>C4 Y1</th>
<th>C4 Y2</th>
<th>C5 Y2</th>
<th>C6 Y2</th>
</tr>
</thead>
<tbody>
<tr>
<td>Depth (m)</td>
<td>0.49</td>
<td>1.40</td>
<td>0.87</td>
<td>1.31</td>
<td>0.44</td>
<td>1.58</td>
<td>0.32</td>
<td>0.32</td>
<td>1.18</td>
<td>0.99</td>
</tr>
<tr>
<td>Area (m$^2$)</td>
<td>294</td>
<td>294</td>
<td>241</td>
<td>241</td>
<td>412</td>
<td>412</td>
<td>172</td>
<td>172</td>
<td>88.6</td>
<td>105</td>
</tr>
<tr>
<td>Volume (m$^3$)</td>
<td>174</td>
<td>411</td>
<td>245</td>
<td>316</td>
<td>263</td>
<td>651</td>
<td>56</td>
<td>55</td>
<td>113</td>
<td>112</td>
</tr>
</tbody>
</table>
3.2.2 Water and Climate Monitoring

Three flow meters (Rain Bird ©, Azusa, CA, USA) were installed in August 2012 in the storage facility to monitor water usage during potato washing. Flow meter data was recorded on record sheets kept near the meters for ease of access by the farm workers. These were stored and housed by the storage secretary worker and scanned and emailed to the researchers.

Water movement through the wetland was monitored by continuous measurement of calibrated tipping buckets set up at the outflow of C3 and the final wetland outflow (C4 in Y1; C6 in Y2). Hourly counts were recorded by a Campbell Scientific CR21X datalogger (Campbell Scientific, Edmonton, AB, Canada). This datalogger was replaced by a CR3000 May 2013 (9 months after the start of monitoring). Data was downloaded on a regular basis through either a manual USB connection or wirelessly using two Campbell Scientific RF401-series spectrum radios (Campbell Scientific, Edmonton, AB, Canada). One radio was attached to the CR3000, while the other was attached to a laptop located inside the storage facility that uploaded the data for easy retrieval remotely.

Data was converted into volume and summarized into monthly totals:

\[
Total\ Volume_{mon} = \sum (Count_h \times 1.5L)_{mon}
\]

where Total Volume \(_{mon}\) is the total volume passing through the tipping bucket in that given month; \(Count_h\) is the recorded number of tips; 1.5 L is the average calibrated volume of water for each tip.

Figure 3.1 Diagram of treatment system in year 1 and year 2 configuration, with inflow on the left and outflow on the right.
A small weather station was set up and connected to the same data logger as the tipping buckets. It comprised of a Series 525 rainfall sensor (Texas Electronics, Dallas, TX, USA), a LI-200SA Pyranometer sensor (LI-COR, Lincoln, NE, USA) for solar radiation, and a thermocouple with solar radiation shield for air temperature. An SR50AT sonic range sensor for depth and an anemometer for wind speed and direction were added in May 2013.

In September 2012, two Hobo® pendant temperature loggers (Onset®, Bourne, MA, USA) were suspended in each cell to record daily average water temperature. Due to trouble retrieving the sensors, thermocouples wired to the datalogger were submerged into C3 and C4 in May 2013.

3.2.3 Wetland Sampling

Water samples were taken on average twice a month during the shipping season. Inflow samples were taken directly out of the flume prior to wash-water being discharged into the treatment system. Throughout the wetland, water samples were always taken at the location of the outflow pipe. Lastly, outflow water was taken as it was exiting the system through the out-flowing tipping bucket. Grab samples were taken using a sampling pole at mid-water level, without disturbing any sediment. Water sampling was always done from cleanest to dirtiest wastewater.

*Escherichia coli* (*E.coli*) wastewater samples were taken by filling a clean bottle and then pouring it into a sterilized bottle containing sodium thiosulfate (Na₂SO₃O₃) to neutralize any residual chlorine that may be present in the wastewater. Soil and potato dirt were gathered with a trowel and stored in zip-locking bags. Gloves were worn during all *E.coli* sampling.

All samples were kept in a cooler with ice and shipped using next day delivery currier to the Ontario Rural Wastewater Center for analysis, where they were stored at 4±1°C until analyzed.

3.2.4 Laboratory Analysis

3.2.4.1 5-day Biochemical Oxygen Demand

BOD₅ was adapted from the Standard methods 5210 B (APHA 2005). Each sample was diluted with aerated water which was injected with nutrients and Polyseed™ capsule (InterLab®, The Woodlands, TX, USA) containing known bacterial seed. Dissolved oxygen (DO) was measured using Thermo Scientific meter ORION 5 STAR and 9108 dissolved oxygen probe (Thermo Fisher Scientific, Waltham, MA, USA) prior to incubation for 5 d at 20 ± 1 °C. A seed blank was
also prepared for incubation by filling a bottle with dilution water and seed. Following incubation, DO was measured a second time and the BOD$_5$ calculated as follows:

$$BOD_5 = \frac{(DO1 - DO5)_{sample} - (DO1 - DO5)_{seed blank}}{volume\ of\ sample} \times total\ volume\ in\ bottle$$

where BOD$_5$= biochemical oxygen demand (mg L$^{-1}$); DO1 = dissolved oxygen concentration at day 1 (mg L$^{-1}$); DO5 = dissolved oxygen concentration at day 5 (mg L$^{-1}$); total volume in bottle = 300mL; volume of sample = volume of sample in BOD$_5$ bottle (mL).

3.2.4.2 Total Suspended Solids (TSS)

The TSS methods were modified from Standard Methods for water and wastewater analysis (Standard Method 2540D, APHA, 2005). Prior to analysis, 1.5 µm Whatman glass microfiber filters (GE Healthcare Life Sciences, Baied’Urfe, Quebec, Canada) were washed using demineralised water and dried at 104±1°C for at least 4 h in an oven (Isotemp 500 series, Fisher Scientific, Ottawa, ON). Filters were cooled in a desiccant chamber for 15 min and then weighed to find the initial filter weight. Water samples were filtered using a Buchner funnel and vacuum line. The filters were dried again at 105°C for at least 4 h and cooled in a desiccant chamber for 15 min prior to weighing for the final sample weight. The concentration of the total suspended solids in the sample was found using the following equation:

$$TSS\ (mg\ L^{-1}) = \frac{(Final\ sample\ weight - Initial\ filter\ weight)}{Volume\ of\ sample} \times 1000$$

[3.3]

Final sample weight (g), Initial filter weight (g), the volume of sample (L) is the volume of the sample filtered, and 1000 converts g to mg.

3.2.4.3 Nitrogen

Total Kjeldalh Nitrogen

Total Kjeldalh Nitrogen (TKN) procedure was adapted from standard method 4500-NorgB Macro-Kjeldalh Method (APHA 2005). Samples were digested for 1 h with 7 mL of concentrated H$_2$SO$_4$. Following this, the samples were distilled and the ammonia phenate method was used to measure the ammonia concentration. The TKN was calculated as follows:
\[
TKN \left( \frac{mg}{L} \right) = \frac{NH_3\text{Concentration} \times \text{Distillate volume (L)}}{\text{Sample volume (L)}} \times \text{Dilution factor}
\]

where \( TKN \) = concentration of TKN in sample or blank (mg L\(^{-1}\)); \( NH_3 \) concentration = total \( NH_3 \) in distillate (mg L\(^{-1}\)); Distillate volume = volume of distillate collected from distillation (L); sample volume = sample volume used for digestion process (L); Dilution factor = Dilution use to dilute sample (L L\(^{-1}\)).

Ammonia

Ammonia was measured using the phenate method, adapted from standard method 4500-NH\(_3\) F (APHA, 2005). A standard curve was prepared using NH\(_4\)Cl stock solution and a spectrophotometer at 640 nm. Samples were prepared with the appropriate dilution and 1mL was dispensed into a glass vial. This was mixed with 0.4 mL of phenol solution, 0.4 mL of Sodium Nitroprusside solution, and 1 mL of oxidizing solution. Vials were mixed well and allowed to sit for 2 h in low light to allow color development. The absorbency was measured using a spectrophotometer at 640nm and the concentration was found using the standard curve equation.

Nitrate

Total nitrates method was modified from standard methods 4500-NO\(_3\) electrode method (APHA, 2005). A standard curve was prepared using nitrate stock solution and mV measured using a nitrate electrode Orion 300 BNWP and Orion 900200 (Thermo Fisher Scientific, Waltham, MA, USA). Samples were also measured with the nitrate probe and dilution was carried out when necessary. Nitrate concentration was found using the logarithmic standard curve equation, ranging from 1-100 (ppm).

Total Nitrogen

Total nitrogen was found using the following equation:

\[
TN = TKN + NO_3
\]

where \( TN \) = total nitrogen; \( TKN \) = Total Kjeldahl nitrogen; \( NO_3 \) = nitrate.
3.2.4.4 **Total Phosphorus (TP) and Ortho-Phosphate (OPO₄)**

Total phosphorus procedures were adapted from standard methods 4500-P Phosphorus (Persulfate Digestion Method) and 4500-P E. Ascorbic Acid Method (APHA, 2005).

To make the total phosphorus standard curve, at least 6 different concentrations ranging from 0.01 - 1.0 mg L⁻¹ of phosphate stock solution were prepared. The standards were mixed with phenolphthalein solution, 1mL TP acid digestion solution and 0.4 g of ammonium peroxydisulfate. This was boiled until the volume was reduced to approximately 10 mL and then allowed to cool. The volume was returned to 40 mL using distilled water and a drop of phenolphthalein solution added. To this, 10 N NaOH was added until the standard solution turned light pink, followed by 5N H₂SO₄ until the pink color was discharged. Distilled water was added to make a total of 50mL and then 10mL of the solution was transferred to a glass vial.

Ortho-phosphate standard curve was made by preparing at least 6 different concentrations of phosphate stock solution ranging from 0.1 - 1.0 ppm.

For both standards and samples, 1.6mL of combined reagent was added to each vial. After colour developed – about 10 min – the absorbance was read using the spectrophotometer at 880 nm. TP samples were prepared as the standards, while the ortho-phosphate samples were diluted as necessary. Samples had 1.6 mL of combined reagent added to each and then the absorbance was read. Concentration was found by converting absorbance to concentration using the standard curve equation.

3.2.4.5 **Escherichia coli (E.coli)**

The methods described here were adapted from Difco and BBL Manual m-FC Basal Medium (Difco, 2009) and Standard Method 9222 (APHA, 2005). Water samples were diluted using serial dilution with a phosphate buffer. A range of dilutions was chosen to assure a colony count of 20-80 on at least one of the plates. Diluted water was filtered through a 0.45 µm Whatman glass microfiber filters (GE Healthcare Life Sciences, Baied’Urfé, Quebec, Canada) and plated onto a Difco m-FC Basal Medium with BCIG (5-bromo-4-chloro-3-indolyl-β-D-glucuronide) for selecting colonies. Plates were incubated for 24 h (± 2 h) at 44.5 °C (± 0.2 °C). Colonies were counted following incubation and waste was autoclaved. The bacteria count is recorded as CFU/100mL. To calculate this from colony count, the following equation was used:
\[ \frac{CFU}{100mL} = \frac{Count \times Dilution Factor}{volume \ of \ sample} \times 100mL \]  

3.2.5 Multimeter Readings

The dissolved oxygen (DO), and oxidation-reduction potential (ORP) of each wetland cell was measured as of summer 2013. Only the very first measurement from August 2013 is considered a measurement from the first season. Shortly after this the cells were drained and the treatment system filled with new wastewater from the second shipping season. Measurements were taken using the YSI 556 multimeter (YSI556, YSI Inc., Yellow Spring, Ohio, USA) by fully submersing the probe into the water. Readings were taken after all values stabilized (~5 mins). Prior to measurements the multimeter was calibrated using methods and standard solutions from the manufacturer.

3.2.6 Calculations and Statistics

Concentration reduction was calculated by taking weekly average concentrations:

\[ % \text{Reduction} = \frac{Inflow \ concentration - Outflow \ concentration}{Inflow \ concentration} \times 100 \]  

where % Reduction = percent decrease in concentration; Inflow concentration = concentration of given contaminant in the inflow sample; Outflow concentration = concentration of given contaminant in the filter discharge sample.

Mass loading was calculated for inflow and outflow as follows:

\[ Mass \ (g) = \frac{Concentration \left( \frac{mg}{L} \right) \times Flow \ (L)}{1000 \left( \frac{mg}{g} \right)} \]  

where Mass = total mass of the contaminant in the water entering or leaving the filter during the time period of interest (g); concentration = average concentration of the contaminant during the time period of interest (mg L\(^{-1}\); either inflow or outflow); Flow = total amount of water during the time period of interest (L; either inflow or outflow).
Mass Reduction was calculated as a percentage for total weekly removal and whole study total removal:

\[ \text{% Reduction} = \frac{\text{Inflow mass (g)} - \text{Outflow mass (g)}}{\text{Inflow mass (g)}} \times 100 \]  

[3.9]

where % Reduction = percent decrease in mass; Inflow mass = mass of given contaminant in the inflow; Outflow mass = mass of given contaminant in the filter discharge.

Cumulative Mass Load and cumulative mass removals were calculated as total mass that flowed through the entire study. Monthly averages were calculated as the average of all measurements from a given month and annual averages were calculated as the average October to July (shipping season). Values that were below detection were averaged as the detection limit value and then assumed to be below that value (this is only relevant to NO\textsubscript{3} concentrations).

Difference between monthly averages in Y1 and Y2 were tested using paired t-tests using SigmaPlot 12.0 (SigmaPlot, San Jose, CA, USA). Correlations were found using regression analysis in SigmaPlot 12.0. Best fits were selected from several options (linear, linear-log, and polynomial) and selected based on $r^2$ and p-value.

### 3.3 Results

#### 3.3.1 Flow

The total water usage for shipping was 4044 m\textsuperscript{3} in Y1 (October 2012 – July 2013) and 2345 m\textsuperscript{3} in Y2 (October 2013 – July 2014) and the total potatoes shipped was 305,150 cwt for Y1 and 258,700 for Y2 (Figure 2.1). The average monthly water usage was 404 ± 222 m\textsuperscript{3} in Y1 and 234 ± 148 m\textsuperscript{3} in Y2 Water use in Y1 was highest in January and July (Figure 2.1). Similarly, in Y2, the highest water usage was July and December. This suggests that January/December and July are seasonally busier times for shipping.). The average monthly potato shipment was 30,000 ± 24,000 cwt month\textsuperscript{-1} in Y1 and 25,000 ± 19,000 cwt month\textsuperscript{-1} in Y2. This large deviation reflects the varying demand for potatoes by processing factories that purchase from the farm (e.g., Frito-Lay Inc.).

The first tipping bucket (TB1) was located at the inflow into the fourth cell (C4). TB1 had flow from the start of the shipping season in Y1 (October), while in Y2, TB1 did not register flow until December. This was due to the larger size of the treatment system in Y2 and lower inflow
volume (Table 3.1 and Table 3.2). The total volume of water passing through TB1 in Y1 was 2215 m³ and 1780 m³ in Y2 (Figure 2.1). The tipping bucket at the outflow of the treatment system (TB2 located at C4 in Y1; C6 in Y2) measured 473 m³ in Y1 and 53 m³ in Y2 (Water use in Y1 was highest in January and July (Figure 2.1). Similarly, in Y2, the highest water usage was July and December. This suggests that January/December and July are seasonally busier times for shipping.). Flow and climate is discussed in more detail in Chapter 2 and Appendix 8.1.

3.3.2 **Wetland Performance**

3.3.2.1 **Biochemical Oxygen Demand**

The BOD₅ inflow in both years was high, however, in Y2 average outflow concentrations within governmental limits were achieved (Table 3.3). In Y1 the average inflow concentration was 1243 mg L⁻¹ and average inflow mass was 616 kg month⁻¹ (Table 3.3). The total amount of BOD₅ in the inflow for the entire Y1 shipping season was 5500 kg. In Y2 the inflow was 162 mg L⁻¹ and 89 kg month⁻¹. The total amount of BOD₅ in the inflow for Y2 was 1600 kg, which is a 71% from Y1.

The average BOD₅ outflow in Y1 was 135 mg L⁻¹ and 29.8 kg month⁻¹. The total BOD₅ outflow from the system was 298 kg from the entire shipping season. This was a 95% total reduction of BOD₅ mass. Although the BOD₅ removal was high, the monthly outflow concentrations were not low enough to pass governmental guidelines. The Ontario Provincial Surface Water Quality Guidelines (PWQG) recommend a concentration <25 mg L⁻¹ (Ontario Ministry of Environment, 1994b). Therefore, an extra 78% reduction would be necessary to meet these guidelines.

The outflow in Y2 was almost non-existent due to ground seepage in the last two wells of the wetland system which were added to increase treatment in the Y2. The only outflow from the system was seen at the end of the shipping season in July, following a series of large shipping days (14340 cwt shipped in the week prior) and a large rainfall (18 mm of rain). The average BOD₅ concentration in the last treatment cell (C6) in Y2 was 25 mg L⁻¹. The total BOD₅ outflow from the system was 1.5 kg from the entire shipping season (there was only outflow in 1 month (July)). This is a 99.7% total reduction of BOD₅ mass. The average outflow concentrations of BOD₅ were low enough to pass governmental guidelines.
The average C6 concentration in Y2 was 78% less than the average outflow concentration in Y1. Within the treatment system, the largest concentration difference was between C3 and C4, where the average concentrations dropped from 443 mg L\(^{-1}\) to 35 mg L\(^{-1}\) in Y2 and from 283 mg L\(^{-1}\) to 146 mg L\(^{-1}\) in Y1 (Figure 3.2). The increased treatment in C4 could be attributed to warmer weather when water reached C4 along with organic matter more decomposed and hence more biologically available.

The BOD\(_5\) is a measure of organic carbon in the wastewater, therefore lowering BOD\(_5\) is accomplished through removal of organic carbon. Carbon can be present in the wastewater as dissolved carbon or particulate carbon and can be removed through aerobic decomposition creating carbon dioxide (CO\(_2\)) or aerobic decomposition producing CO\(_2\) and anaerobic decomposition producing methane (CH\(_4\)) (Kadlec and Wallace, 2009). Particulate carbon can be suspended in the wastewater and therefore also contributes to TSS concentrations. This form of carbon can be treated similarly to TSS, through sedimentation, and decomposition will continue within the sediment. The removal of BOD\(_5\) and TSS throughout the wetland had a strong positive correlation on a monthly basis in Y1 (\(r^2 = 0.96\)) and in Y2 (\(r^2 = 0.70\)). Since carbon decomposition and nitrification both require aerobic environments for removal we should see a correlation of BOD\(_5\) and NH\(_3\) removals through the treatment system. Additionally, denitrification requires a carbon source for NO\(_3\) to create N\(_2\) and CO\(_2\) gasses. BOD\(_5\) had a strong positive correlation with TN concentration removal through the wetland (C4) in Y1 (\(r^2 = 0.78\)) and in Y2 (\(r^2 = 0.77\)).

### 3.3.2.2 Total Suspended Solids

TSS concentrations in the raw wash-water were very high and neither Y1 nor Y2 had average concentrations within governmental discharge guidelines. The TSS is the amount of suspended organic or inorganic solids in the wastewater which are small enough to remain suspended. For this thesis TSS is defined as all particles \(\geq 1.5\) µm. In Y1 the average monthly inflow concentration was 5200 mg L\(^{-1}\) and mass was 2500 kg. The total amount of TSS in the inflow for Y1 was 24,000 kg. In Y2 the monthly average inflow concentration was 3900 mg L\(^{-1}\) and 629 kg. The total amount of TSS in the inflow for Y2 was 4,400 kg. There was an 82% reduction from Y1 to Y2 in the total TSS mass loaded into the system throughout the shipping season.
The TSS outflow in Y1 was 176 mg L\(^{-1}\) and 39 kg. The total TSS outflow from the system was 387 kg from the entire shipping season. Thus, the treatment system reduced TSS mass by 98% in Y1. Although the TSS removal was high, the outflow concentrations were not low enough to pass governmental guidelines. The Ontario PWQG recommends a concentration of <25 mg L\(^{-1}\) (Ontario Provincial Ministry, 1994b). Therefore, an extra 86% reduction was necessary to meet this guideline. The average TSS concentration in the last treatment cell (C6) in Y2 was 91.2 mg L\(^{-1}\) and the total TSS outflow from the system was 5.4 kg from the entire shipping season. This is a 99% total reduction of TSS mass in Y2; however, the C6 concentration was still above governmental guidelines.

The removals through the treatment system were similar in Y1 and Y2; however in Y2 outflow concentrations were consistently lower (Figure 3.3). The average C6 concentration in Y2 was 48% less than the average outflow concentration in Y1. This can be explained by the changed farm practices in Y2 to remove large quantities of solids off potatoes prior to washing (reflected in the 31% reduction of average inflow concentration from Y1 to Y2). There is no evidence that the treatment system was able to treat TSS more effectively in Y2 than Y1 (The TSS concentration in C6 actually increased, presumably due to water loss from ground seepage.).

Both outflow TSS concentration from Y1 and the C6 TSS concentration from Y2 were above the governmental guideline. Therefore, the sedimentation cell was not able to remove enough solids. The sedimentation cell (C1) in Y1 and Y2 was created with a centre baffle to force water to move through the cell in a slower manner, however the tracer results suggest there may have been short-circuiting, therefore sedimentation was not optimal (Appendix 8.2). Annual cleaning of C1 showed accumulation of sedimentation, as well as ample time for sedimentation to occur throughout other cells of the treatment system. The inability for solids to settle suggests that the particles are colloidal, which means that their size is small enough that the electrical charges that attract them to other particles are stronger than the force of gravity acting upon it (Tchobanoglous and Schroeder 1987; Image 8.11). Although sedimentation could be further optimised, in the last treatment cell in Y2 there was a re-suspension of solids with an increase in TSS from 60 to 90 mg L\(^{-1}\). The reason for this re-suspension is unknown, however, it suggests that this may be the limit in sedimentation the treatment system. Research into supplementary treatment options may be necessary for further lowering TSS (Chapter 4 and 5).
3.3.2.3 Nitrogen

TN is the total of TKN and NO₃. In Y1 the average inflow concentration was 222 mg L⁻¹ and average inflow mass was 117 kg month⁻¹. The total amount of TN in the inflow for the entire Y1 shipping season was 1170 kg. In Y2 the inflow averaged 142 mg L⁻¹ and 36.3 kg month⁻¹. The total amount of TN in the inflow for the entire Y2 shipping season was 453 kg, which is 61% less than Y1.

Due to the negligible concentrations of nitrate in the wastewater in Y1, the TN was mostly from TKN; hence the results were very similar. The TKN is the total organic N, ammonia (NH₃) and ammonium (NH₄⁺) in water. In Y1 the average inflow TKN concentration was 222 mg L⁻¹ and average inflow mass was 117 kg. The average NH₃ inflow concentration was 69.9 mg L⁻¹ month⁻¹ and average inflow mass was 33.4 kg month⁻¹. The total amount of NH₃ in the inflow for the entire Y1 shipping season was 334 kg. Therefore, TKN was 28% NH₃ and 72% organic N. In Y2 the inflow was 225 mg L⁻¹ month⁻¹ and 21.7 kg. The total amount of TKN in the inflow for the entire Y2 shipping season was 450 kg. In Y2 the inflow was 90.3 mg L⁻¹ month⁻¹ and 28.6 kg month⁻¹. The total amount of NH₃ in the inflow for the entire Y2 shipping season was 236 kg. TKN was 52% NH₃ and 48% organic N. There was a shift in Y2 to a higher ratio of NH₃ to organic matter. There was a 62% reduction from Y1 to Y2 in the TKN total mass and 29% reduction in the NH₃ total mass into the system during the entire shipping season. The reduction of organic N in Y2 inflow could be due to the reduction of rot and potato chunks into the flume in Y2 due to relocation of cull piles and increase in finger rollers prior to washing.

TN reduction is the overall reduction in all N constituents, which leave the system either as NH₃ or N₂ gas following nitrification and denitrification, excavated solids, or harvested plants. The average TN outflow in Y1 was 114 mg L⁻¹ and the total TN outflow from the system was 272 kg from the entire shipping season. Thus, the treatment system reduced TN mass by 77% in Y1. The average TN concentration in the last treatment cell (C6) in Y2 was 43.1 mg L⁻¹ and the total TN outflow from the system was 2.54 kg from the entire shipping season—a 99% total reduction of TN mass in Y2.

Removal of TKN is achieved through nitrification of NH₃ to NO₃, therefore the decrease in TKN and NH₃ should have a corresponding decrease in NH₃ and increase in NO₃. The average outflow concentrations in Y1 were 84.3 mg L⁻¹ TKN, 24.4 mg L⁻¹ NH₃, and <1.41 mg L⁻¹ NO₃. The total
outflow from the system was 257 kg TKN, 59.0 kg NH₃, and 3.00 kg NO₃ from the entire shipping season. This was a 79% total reduction of TKN and 82% reduction of NH₃. Although the N removals were high, the monthly concentrations were not low enough to pass governmental guidelines. The average concentration in the last treatment cell (C6) in Y2 was 23.0 mg L⁻¹ TKN, 6.65 mg L⁻¹ NH₃, and 1.41 mg L⁻¹ NO₃. The total outflow from the system was 1.36 kg TKN, 0.39 kg NH₃, and 0.08 kg NO₃ from the entire shipping season. There was a slight increase in NO₃ concentrations in the spring of Y1 and Y2, although concentrations were < 10 mg L⁻¹ (Figure 3.7). Because there was no NO₃ increase corresponding to the NH₃ removal in both years, we can conclude that all NO₃ created was denitrified to N₂ gas. The Ontario PWQG recommends an NH₃ concentration <1 mg L⁻¹, therefore an extra 85% removal is needed to pass governmental guidelines.

From Y1 to Y2 there were concentration removals of 62% TN, and 73% for TKN and NH₃. The largest difference in the treatment was seen in the last three cells—similar to BOD₅. Treatment in C4 increased in Y2, while the two new cells gave additional removal (Figure 3.3). N removal in both years was limited by nitrification in the treatment system, which is evident from the lack of increase of NO₃ and the high remaining NH₃ in the outflow. Nitrification can be limited by pH, temperature, NH₃ concentrations, and DO. The pH for nitrification must be 7.5 - 8.5. With the exception of C1 during August 2013 measurement, all cells had pH values within that range. There was no strong trend of nitrification increasing with temperature, which suggests that temperature was not limiting the reaction. DO, however, is readily depleted in the water through respiration, carbonaceous BOD₅ and nitrification (Kadlec and Wallace 2009). Additionally, slow moving bodies of water, such as those in this treatment system, are inefficient at oxygen transfer (Kadlec and Wallace 2009). Therefore, low DO concentrations are most likely limiting nitrification. This is exemplified by the increased TN removal in Y2 (after the installation of an aerator) compared to Y1 (Figure 3.3).

For nitrification to occur, a minimum of 0.25 mg L⁻¹ is necessary for nitrification. DO measurements were below the detection limit in August 2013 and December 2013 in all cells. Although the DO concentrations increased in the spring, they were not consistently high throughout the treatment system and therefore nitrification was not optimal (Figure 3.8). It is also possible to have pockets within the cells with low DO which were not measured. For example, C2 and C3 had aerators causing agitation to increase oxygen concentrations, however if they did
not reach to all sides horizontally or vertically, pockets with low DO could have been created. Therefore, more DO measurements are necessary to ensure even DO aeration.

3.3.2.4 Phosphorus

The inflow P concentrations were not very high compared to other contaminants. The outflow concentrations did not meet governmental discharge guidelines (Ontario Ministry of Environment, 1994b). Phosphorus is an important nutrient for the growth of organisms, including microbes, plants and algae, and when in excess as a wastewater pollutant can contribute to eutrophication of surface waters. In Y1 the average inflow concentration of TP was 32.4 mg L\(^{-1}\) and average inflow mass was 16.3 kg. In Y1 the average inflow concentration of OPO\(_4\) was 15.5 mg L\(^{-1}\) and total mass inflow for the entire shipping season was 162 kg TP and 68.6 kg PO\(_4\). In Y2 the average inflow TP concentration was 48.8 mg L\(^{-1}\) (34% higher than Y1) and OPO\(_4\) was 10.9 mg L\(^{-1}\) and the total inflow load was 56.8 kg TP and 20.6 kg OPO\(_4\) for the entire shipping season.

The TP outflow in Y1 averaged 13.8 mg L\(^{-1}\) while OPO\(_4\) averaged 4.93 mg L\(^{-1}\). Total outflow load from the entire Y1 shipping season was 29.7 kg TP and 11.7 kg OPO\(_4\). Thus, the treatment system reduced the mass of TP by 82% and OPO\(_4\) by 83%. In Y2 the average concentration in the last treatment cell (C6) was 1.35 mg L\(^{-1}\) TP and 0.08 kg and 1.19 mg L\(^{-1}\) OPO\(_4\). The total outflow from the system was 0.08 kg TP and 0.07 kg OPO\(_4\) from the entire Y2 shipping season. Thus the treatment system provided a 99% total reduction of TP and OPO\(_4\) by mass in Y2. The TP removals in both years were not enough to have outflow concentrations within the Ontario PWQG recommended concentration (<0.3 mg L\(^{-1}\)).

Comparing years, we see the average TP concentration in C6 in Y2 was 90% less than the average outflow concentration in Y1. In Y2 there was consistently better removal throughout the treatment system, suggesting increased removal is due to the increased retention time in the system (Figure 3.2). Although the inflow TP concentrations were higher in Y2, each treatment cell had lower average outflow concentrations. Therefore, there was a correlation of increased HRT and increased TP removal. There was a very strong correlation of TP and TSS average monthly mass removal through the wetland in Y1 \((r^2 = 0.94)\) and in Y2 \((r^2 = 0.83)\). This suggests that increasing TP removal to pass governmental guidelines should be accomplished alongside TSS; therefore, supplementary treatments may be necessary (see Chapter 4 and Chapter 5).
3.3.2.5 *Escherichia Coli*

*E.coli* is used as an indicator species of faecal contamination. Although the only known source of contamination into the wash-water was soil, the concentrations in the raw wash-water in both years were surprisingly high and the outflow was always above governmental limits. The average inflow concentration was 7.35 log CFU 100 mL⁻¹ in Y1 and 6.55 log CFU 100 mL⁻¹ in Y2. There was an average 1-log reduction from Y1 to Y2 in *E.coli* concentration in flow into the treatment system throughout the shipping season. This could be due to the reduction of solids, which could be harboring *E.coli* colonies.

The *E.coli* outflow in Y1 was 4.46 log CFU 100 mL⁻¹, which was a 3-log concentration reduction of *E.coli* after treatment. Although the *E.coli* removal was high, the outflow concentrations were not low enough to pass governmental guidelines. The Ontario PWQG recommends a concentration of <2 log CFU 100 mL⁻¹. Therefore, >2-log reduction is necessary to meet those guidelines. The average *E.coli* concentration in the last treatment cell (C6) in Y2 was 2.39 log CFU 100 mL⁻¹, which was a 4-log reduction of *E.coli* concentration. There was a 46% decrease in *E.coli* log concentrations (2-log) in the outflow of Y2 compared to Y1. In Y2 the removal was steeper C1 to C3 as well as additional treatment in C5 and C6.

3.3.2.6 *Dissolved Oxygen, Oxidation-Reduction Potential*

The wastewater was both actively and passively aerated in the treatment system. The second and third cells of the system (C2 and C3) had floating aerators, which was the largest input of oxygen into the system. Passive aeration had multiple sources including wind, flow through tipping buckets, plant roots (aerenchyma), and gravel spillways between the last three wetland cells.

The highest DO was seen in C5 with 13.8 mg L⁻¹ in April 2014 while the lowest was 0.02 mg L⁻¹ in December 2013 in C2. The overall highest average DO was C6, while the lowest was in C1. The DO in all locations was low in the first three measurements (<2 mg L⁻¹ in August, December, and January 2013, with a huge increase in April (average 10.6 mg L⁻¹) and a slow decline in the last two months (6.55 mg L⁻¹ in May and 2.39 mg L⁻¹ in July). This can be assumed to be due to seasonal changes. The increase in oxygen in April would be due to aerators turning on, ice and snow melting, water movement, and temperature change. The subsequent decrease in summer can be due to increasing biological activity using the oxygen. We see a
coinciding increase in NO$_3$ concentrations in the outflow after April (Figure 3.7). There also appears to be a pattern of increasing DO throughout the wetland, although C4 had unusually low concentrations which do not fit into this pattern. The average concentrations throughout the treatment system in Y2 were 0.97 mg L$^{-1}$ in C1, 5.0 mg L$^{-1}$ in C2, 5.17 mg L$^{-1}$ in C3, 1.36 mg L$^{-1}$ in C4, 8.06 mg L$^{-1}$ in C5 and 9.26 mg L$^{-1}$ in C6. Because C4 was a slow moving wetland overgrown with cattails, it has little oxygen input. Although the aerenchyma of cattails provide O$_2$ movement into the water through the root zone, the height and bulk of the vegetation prevented wind mixing and O$_2$ movement through this means. The NO$_3$ and TN concentrations in C4 have a coinciding average decrease, which suggests that NO$_3$ is being converted to N$_2$ gas in the oxygen – lacking environment.

The ORP is a measurement of the particle charges of ions which are an indicator of the most prominent biological reactions occurring in the wastewater (YSI environmental, 2008). Each reaction has a range of ORP values over which it is occurring. In August 2013 C2 and C3 were all within the range of biological phosphorus release and fermentation (Figure 3.8; YSI environmental, 2008). C4 was in the range of sulfide production and C1 was in the range of biological phosphorus removal (+25 to +250 mV) and carbonaceous BOD$_5$ degradation with free molecular oxygen (+50 to +250 mV). In April C3, C4, C5, and C6 all had their levels within the range of denitrification (+50 to -50 mV). This agrees with the DO drop seen in C4, which suggests that denitrification was occurring. In May only C5 and C6 had values for denitrification and in July only C3 had values for denitrification. There were no measurements in the +100 to +350 mV range, where nitrification occurs, although this may not be because nitrification was not occurring, but because other reactions were more prominent.
Table 3.3 Total inflow and outflow total mass (Kg) and average monthly concentration (mg L\(^{-1}\)) from Y1 (using TB1) and Y2 (using TB2).

<table>
<thead>
<tr>
<th></th>
<th>Y1 Inflow Kg</th>
<th>Y1 Inflow mgL(^{-1})</th>
<th>Y1 Mass red'n %</th>
<th>Y2 Inflow Kg</th>
<th>Y2 Inflow mgL(^{-1})</th>
<th>Y2 Mass red'n %</th>
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<tr>
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<td>450</td>
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<th>Y2 Log CFU/mL(^{-1})</th>
<th>Y2 Log CFU/mL(^{-1}) Conc red'n</th>
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<td>3-log</td>
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</table>
Figure 3.2 Average concentration of TSS, BOD$_5$, TP (mg L$^{-1}$) and E.coli (log CFU 100mL$^{-1}$) in each cell of the treatment system in Y1 and Y2
Figure 3.3 Average concentration (mg L$^{-1}$) of N constituents (NH$_3$, TKN, NO$_3$, and TN) in each cell of the treatment system in Y1 (bottom) and Y2 (top).
Figure 3.4: Mass of BOD, TSS, (kg) and concentration of E.coli (log CFU 100 mL$^{-1}$) in the inflow and outflow from the treatment system in Y1 and Y2.
Figure 3.5 Mass of N constituents (kg) TN, TKN, NH$_3$, and NO$_3$ in the inflow and outflow from the treatment system in Y1 and Y2.
Figure 3.6 Mass of P constituents (kg) TP and OPO₄ in the inflow and outflow from the treatment system in Y1 and Y2.
Figure 3.7 Nitrate$^2$ concentrations throughout the treatment system in winter and spring months of Y1 and Y2.

$^2$ All NO$_3$ concentrations at 1 mg L$^{-1}$ are <1 mg L$^{-1}$. 
Figure 3.8 Dissolved oxygen (DO) and oxidation-reduction potential (ORP) from August 2013 to July 2014 from each treatment cell.
3.4 Discussion

3.4.1 Comparison of Y1 and Y2

Simple changes to reduce water usage and prevent contaminants from entering wash-water in the storage facility accomplished inflow mass reductions from Y1 to Y2 of 98% BOD$_5$, 82% TSS, 76% TN, 83% TP, and 53% E.coli. Preventing contaminants from entering the wash-water was achieved by removing more solids from potatoes prior to entering the flume with finger rollers and moving cull piles, which had leaked rotten potato liquids into the flume in Y1. By reducing the volume of water used for washing (details in Chapter 2), however, the net effect on concentration values was diminished. This reduction of dilution also influenced the outflow concentrations. Therefore it is essential to look at mass loads and mass removals, not only concentrations.

With the exception of BOD$_5$, all contaminants needed further removal to meet PWQG. Most contaminants were at least partially associated with solids in wastewater, therefore removing solids early in the treatment system should be a priority to further increase treatment in the system. Although TSS outflow concentration was reduced by an extra 48% in Y2 compared to Y1, it was not enough to governmental guidelines. There is no evidence that increasing the HRT of the treatment system changed the TSS treatment in Y2. The lower TSS concentrations were most likely due to the decreased inflow concentrations. Colloidal particles that are too small to settle may need other methods of removal Filtration and coagulation were explored in this study as possible methods to improve TSS removal (see Chapter 4 and 5). Similarly, E.coli removal appeared unaffected by the changes in the treatment and reductions were mostly due to influent concentration reductions.

TP was the only contaminant that increased in the inflow in Y2. TP removal in Y2 benefited from the larger treatment system. There was consistently better removal in all cells, which also seemed unhindered by winter freezing (Figure 3.2). This suggests that the mechanism for TP removal was correlated with increased HRT.

BOD$_5$ and TN both had much better removals in C4 in Y2. This increased efficiency in Y2 could be due to increased growth of cattails (Cattails had been harvested in the fall 2013, which they had not been the year prior) or degradation of organic matter allowed for more plant uptake. TN
was removed by 63% in Y2, however it was not enough to meet governmental guidelines. N removal was limited by nitrification, therefore, more aeration is necessary in the treatment system. Increasing aeration earlier in the system, such as C2, would allow for more nitrification, and allow denitrification to occur later in the system. This could also assist BOD$_5$ removal as a carbon source is necessary for further denitrification. Although there were two aerators in the treatment system, the tracer results suggest that, in Y2, C2 and C3 were too large to be fully mixed by the aerator, thereby not fully oxygenating all the water (Appendix 8.2). NO$_3$ and BOD$_5$ concentrations suggest that the aerators were contributing to treatment, however, since most of the inflow is during winter months when the aerators were turned off due to ice, much of the water would have passed through those cells before aerators were functioning (Figure 3.7).

3.4.2 Literature Comparison

A few studies have characterised on-farm vegetable processing wastewater (Table 3.4). Luo 2007 studied lettuce wash-water and the effect of re-using wash-water on microbial growth on the surface of the lettuce leaves. As part of the research, BOD$_5$ of the wash-water was found to be 526 mg L$^{-1}$. This is similar to the average concentration of BOD$_5$ we measured in Y2 (558 mg L$^{-1}$), but much higher than Hamilton and Mebalds (2000) who found carrot wash-water had a BOD$_5$ of 29.6 mg L$^{-1}$. Lastly, Lehto et al. (2009) measured BOD$_7$, total solids (TS), TN, TP, and pH in the processing water of three different farms that peeled and washed various root crops. One farm had BOD$_7$ concentrations similar to Y1, however the other farms had 2 and 3× higher concentrations, which is expected if they are peeling and cutting the vegetables. The TS was at least 3× less than our TSS inflow in Y2, which suggests that their BOD$_7$ and TS values are not correlated in the same way as the ones in this study. On average, the TN and TP concentrations were all less than those in this study, however the highest concentrations were comparable to our lowest concentrations. The pH in both Y1 and Y2 stayed within the biologically neutral values, however all three farms in Lehto et al. (2009) had pH values of 4.5 - 4.7. Overall, the values from these studies highlight the variability between the farms. In addition, Lehto et al. (2009) documented variability among repeated measurements at individual farms. This agrees with our finding that variability in wash-water is an inherent aspect of on-farm processing waters. These studies did not have any information on the processing mechanisms and the sources of contaminants. In future research, assessing on-farm washing/processing facilities of other potato...
farms could shed light on the causes of variability and offer other strategies to minimize contaminants in wash-water.

Although there are a few studies which characterize on-farm vegetable processing water, only one looked at the subsequent treatment of these wastewaters. Lehto et al. (2009) measured on-farm processing water in Finland from three different farms: A, B, and C. Each farm had a different treatment system, influent concentration and subsequent removals. Farm A's treatment system consisted of three small sedimentation basins (10 m$^3$ each) and one larger (35 m$^3$), followed by the addition of a coagulant in a following aeration basin. The concentration reductions were 99-100% BOD$_7$, 97% TS, 94% TN, and 95% TP. Farm B's treatment system consisted of three larger sedimentation basins (20 m$^3$, 15 m$^3$, and 10 m$^3$), coagulant addition followed by a trickling filter, and a stone wool filter. The concentration reductions were 63% BOD$_7$, 67% TS, 67% TN, and 77% TP. Lastly, farm C’s treatment system consisted of a small (7 m$^3$) and a large (60 m$^3$) sedimentation pond with additions of coagulant prior to each. The concentration reductions were 67% BOD$_7$, 94% TS, 73%TN, and 95% TP. The best removal of solids was from the system on farm A. The complete treatment systems at Farms A, B, and C had low HRTs of 9, 0.8, and 12 d, respectively. The HRT in our study was much larger with 2 months in Y1 and 7 months in Y2. The removals from these farms were comparable to both Y1 and Y2, where most effluent concentrations were not low enough to pass PWQG.
Table 3.4 Summary of three on-farm vegetable washing or peeling wastewater characterization studies.

<table>
<thead>
<tr>
<th></th>
<th></th>
<th></th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td>Crop</td>
<td>washing</td>
<td>washing</td>
<td>Farm A</td>
</tr>
<tr>
<td>COD mg L(^{-1})</td>
<td>1721</td>
<td>4870</td>
<td>Farm B</td>
</tr>
<tr>
<td>BOD mg L(^{-1})</td>
<td>526</td>
<td>29.6</td>
<td>Farm C</td>
</tr>
<tr>
<td>TS mg L(^{-1})</td>
<td>1103</td>
<td>213</td>
<td>potato</td>
</tr>
<tr>
<td>TDS mg L(^{-1})</td>
<td>719</td>
<td></td>
<td></td>
</tr>
<tr>
<td>TN mg L(^{-1})</td>
<td>64</td>
<td>44</td>
<td>175</td>
</tr>
<tr>
<td>TP mg L(^{-1})</td>
<td>17</td>
<td>6.7</td>
<td>23.1</td>
</tr>
<tr>
<td>pH</td>
<td>4.5</td>
<td>4.7</td>
<td>4.7</td>
</tr>
<tr>
<td>Nitrite-N mg L(^{-1})</td>
<td>0.15</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Nitrate-N mg L(^{-1})</td>
<td>1.15</td>
<td></td>
<td></td>
</tr>
<tr>
<td>SRP-P mg L(^{-1})</td>
<td>32.7</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Turbidity NTU</td>
<td>195.2</td>
<td></td>
<td></td>
</tr>
<tr>
<td>E.coli &gt; 0/100mL</td>
<td>14/25 samples</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Coliforms &gt; 1,000/100mL</td>
<td>18/25 samples</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

3.5 Conclusion

The treatment system had mass removals of 95% and 99% of BOD\(_5\), 98% and 99% TSS, 77% and 99% TN, 77% and 99% TP, and 3-log and 4-log E.coli in Y1 and Y2, respectively. The combination of reducing inflow concentrations and increasing the treatment system size resulted in concentration reductions from Y1 to Y2 by 81% BOD\(_5\), 48% TSS, 62% TN, 73% TKN and NH3, 90% TP, and 46% E.coli. The largest contributor to these reductions were changes inside that the storage facility that decreased the amount of contaminants entering the wash-water.

There is little evidence that the treatment system in Y2 was able to remove extra TSS. Although
most of the contaminant loads were removed, the outflow concentrations were still above
governmental discharge guidelines, on average. More research is needed to find effective ways
to reduce the particularly difficult-to-remove colloidal solids in potato wash-water, which were
not removed even after several months in the treatment system. Additionally, N removal was
limited by nitrification, most likely due to lack of dissolved oxygen, therefore, more research is
needed to find the best ways to increase aeration. Other research is also necessary to characterise
wash-water compositions of other Ontario farms to better understand wash-water and the
treatment required, in order to assist governmental agencies in creating standardized
management strategies that are straightforward and affordable for farmers to use.
Chapter 4 On-farm Treatment of Potato Wash-Water: Supplementary Treatment with Sand Filtration

ABSTRACT

On-farm treatment of wastewater is needed to be low in cost and labour. This study examined sand filtration as a supplementary treatment option for removing total suspended solids (TSS), biochemical oxygen demand (BOD$_5$), and nutrients in potato wash-water. Filters were studied at a lab-scale at two different orientations, horizontal – saturated (H) and vertical – unsaturated (V), and three different dosages: 34 L m$^{-2}$ (150 mL d$^{-1}$), 68 L m$^{-2}$ (300 mL d$^{-1}$), and 136 L m$^{-2}$ (600 mL d$^{-1}$). A total of 24 filters were built within a block design, with 4 replicates. The study ran for 10 weeks dosing once every weekday (5d wk$^{-1}$). Both orientation and dosage were found to have statistical effect on treatment, with vertical filter and lower dosages having better treatment. The best treatment was found in the unsaturated filters with the lowest dosage (150V). The 150V filters removed 88% TSS, 90% BOD$_5$, and 98% total phosphorus. The V filters had more oxygenation and hence more nitrification, decreasing ammonia concentration up to 93% with a commensurate increase in nitrate concentrations. Additionally, the 150V filters did not show any signs of clogging whereas the other filters, especially the higher loaded ones did. This study found that sand filtration, in both saturated and unsaturated filters, even at the highest dosage would benefit treatment removal by removing solids, BOD$_5$ and phosphorus, although only vertical filters have additional treatment of nitrogen removal. Practical application of vertical sand filters at these dosages; however, may require too much infrastructure and cost for potato farmers, therefore other treatment possibilities to supplement nitrogen removal in horizontal filters must still be explored.

Key words: wastewater treatment, filtration, total suspended solids, potato wash-water, vegetable wash-water

4.1 Introduction

Filtration has been identified as an easy, low cost option for the removal of organic and inorganic solids from wastewater, with limited upkeep cost and convenience of handling (Gaur et al., 2010). It can treat wastewater using physical or biological means. Filters remove contaminants
through several physical processes: mechanical, chance contact straining, impaction, interception, adsorption, flocculation, and sedimentation (Tchobanoglous and Schroeder, 1987). All filters treat using physical processes, although not all are able to create the proper environment for biological treatment.

Filtration consists of four main characteristics that affect treatment efficiency: orientation, media type, dosage, and inflow wastewater (Huisman and Wood, 1974). The orientation of the filter is the angle of flow – this will change the behaviour and speed of the water flow. Media type is the largest factor controlling filtration time, though media pore size will also change the distribution of the active settling zone in the filter. Dosage and inflow wastewater will directly affect the treatment efficiency of the filter. The dosage will affect the contact time of water to filtration media, while the contaminant concentration in the inflow water will dictate the amount of debris caught in the filter.

Sand filters have been used in wastewater treatment since the early 1800’s (Huisman and Wood, 1974). Although filters can have any orientation, horizontal and vertical are the two main types. Horizontal filters are referred to as saturated filters, since the water pools in the filter and water only flows when there is lateral pressure. Vertical filters are referred to as unsaturated or trickling filters, as effluent flows by gravity through the unsaturated pore spaces. Therefore, the contact time of water with sand is longer in the horizontal filters, while the vertical filters have more air and sand contact time. This creates very different environments within the filters.

Saturated filters have been used in many situations, including fast filtering gravel beds and filtering planted wetland cells. Kalibbala et al. (2012) studied the removal of natural organic matter (NOM) and iron in drinking water using horizontal roughing filters and found 27% removal of NOM and 39% removal of total iron. Dastanaie et al. (2007) conducted a similar study treating river water in Iran with a horizontal gravel roughing filter with a filtration rate of 1.8 m h⁻¹. They found 89.7% removal of TSS, 94% removal of coliforms and 15.6% removal of iron. Headley et al. (2001) looked at horizontal flow wetlands planted with Phragmites australis in 10 mm gravel media with 2 and 5 d hydraulic residence times for treating nursery irrigation runoff. They found 84% removal of total nitrogen (TN) and 65% removal of total phosphorus (TP). Zurita et al. (2008) found 82% TSS and 47.7% NO₃ removal from domestic wastewater
using a continuously fed (128 L d\(^{-1}\)) horizontal subsurface flow wetland with tezontle gravel – a volcanic, extrusive rock.

Unsaturated filters have also found good contaminant removal in various studies. A study on turkey processing wastewater found vertical filtration with layers of coarse sand, fine sand and gravel, reduced total organic carbon (TOC) by 94% and 5-day biochemical oxygen demand (BOD\(_5\)) by 98% at a dosage of 132 L m\(^{-2}\) d\(^{-1}\) (Kang et al., 2006). Another study treating municipal sewage found removals of 95% of TSS and 99% of BOD\(_5\) when sand filters were used as a tertiary treatment in three different wastewater treatment plants designed to treat 190000, 120000, and 70000 m\(^3\) d\(^{-1}\) (Hamonda et al., 2004). Lastly, Christopherson et al., (2005) looked at treatment of domestic wastewater in a recirculating sand filter and found removals of 92% BOD\(_5\), 87% TSS, 33% TP, and 43% TN.

In addition to orientation, dosage affects the treatment and the filter functioning. Larger doses will have more water accumulation in the headspace, causing a higher filtration force. This means that wastewater will filter through faster and collect solids faster (Huisman and Wood, 1974). Kang et al. (2006) evaluated three dosing rates of 66, 132, and 264 L m\(^{-2}\) d\(^{-1}\). The authors found that all doses had similar removal, although the lowest rate had the same treatment efficiency throughout the study, while the 132 and 264 L m\(^{-2}\) d\(^{-1}\) rates started to lose their efficiency as early as 10 d. In contrast, Zheng et al. (2010) studied the removal of organic foulants from secondary effluent and found that their lower rate of 0.25 m h\(^{-1}\) (250 L m\(^{-2}\) h\(^{-1}\)) had consistently better removal efficiency than their higher rate of 0.5 m h\(^{-1}\) (500 L m\(^{-2}\) h\(^{-1}\)).

Intermittent loading is used almost exclusively in modern wastewater filtration. Sand filters have also been successfully used due to their low cost, easy construction and maintenance; however, sand filtration has not been traditionally used for treating potato wash-water (Kang et al. 2006, Headley et al., 2001). Potato wash-water is wastewater from transporting and washing potatoes in an on-farm storage facility. Prior to shipment, potatoes are washed and sorted, which creates wastewater containing soil and sand from potato surface, as well as organic matter from broken or rotten potatoes. Following shipment, all the water is discharged into a holding pond in one bulk volume. Potato wash-water is high in TSS, BOD\(_5\) and nutrients such as nitrogen (N) and phosphorus (P), therefore it needs to be treated to meet
governmental discharge criteria. There are no studies available on determining the most effective methods for treating potato wash-water.

This study assesses the effectiveness of sand filtration as an on-farm treatment of wastewater from a potato storage facility with high levels of organic and inorganic solids, as well as nutrients. A comparison is made between saturated and unsaturated filters to compare their removal efficiencies and to determine which physical or biological processes are occurring in the filters. If treatment is mainly due to the physical process, we expect that the treatment efficiency between the two orientations will be the same for all contaminants. This comparative study of saturated and unsaturated filters will set recommendations regarding the best orientation and dosing rate for sand filtration of potato wash-water.

4.2 Materials and Methods

4.2.1 Sand Filter Construction

Saturated (12) and unsaturated (12) sand filters were constructed out of 10 cm (3 inch) diameter PVC piping of 2 L volume, and installed in a temperature and light controlled greenhouse at the Agriculture and Agri-food Canada, Central Experimental Farm, Ottawa, ON, Canada. The PVC tubing was filled with sand and landscaping fabric was layered at the bottom of unsaturated pipes to keep the sand in the pipe. Saturated, horizontal pipes were fitted with pipe angle connectors to keep sand completely saturated with wash-water (Figure 4.1).

Sand (loamy sand; 88% sand, 3% silt, 9% clay) was obtained at the potato farm research site. Although we could not guarantee the uniformity of the sand, it was used due to its ease of accessibility for the farmer and increase in external validity of research. Aquarium grade pea gravel was layered on top of the sand at both inflow and outflow, to keep the sand in place within the saturated filters. A layer of gravel was put only at the inflow of the vertical filters.

4.2.2 Experimental Design

The experimental design was a randomized complete block which controlled for spatial variability. There were four blocks, each with six filters, representing all combinations of the two treatments: two types of filter orientation, horizontal (saturated) and vertical (unsaturated) and three dosage levels. The dosages were: 34 L m⁻² (150 mL d⁻¹), 68 L m⁻² (300 mL d⁻¹), and 136 L
m² (600 mL d⁻¹) (the loading was performed 5 d wk⁻¹). These dosing rates were selected based on the on-farm treatment system in-flow rate, compatibility with other literature, and practicality (Table 4.1).

Table 4.1 Specifications of filter design and dosages.

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Low</th>
<th>Medium</th>
<th>High</th>
</tr>
</thead>
<tbody>
<tr>
<td>Diameter (cm)</td>
<td>7.62</td>
<td>7.62</td>
<td>7.62</td>
</tr>
<tr>
<td>Length (cm)</td>
<td>50</td>
<td>50</td>
<td>50</td>
</tr>
<tr>
<td>Surface Area (m²)</td>
<td>0.00442</td>
<td>0.00442</td>
<td>0.00442</td>
</tr>
<tr>
<td>Volume (L)</td>
<td>2.21</td>
<td>2.21</td>
<td>2.21</td>
</tr>
<tr>
<td>Dosage (mL d⁻¹)</td>
<td>150</td>
<td>300</td>
<td>600</td>
</tr>
<tr>
<td>Dosage (L m⁻² d⁻¹)</td>
<td>34</td>
<td>68</td>
<td>136</td>
</tr>
<tr>
<td>Dosage (L/m²/week)</td>
<td>170</td>
<td>339</td>
<td>679</td>
</tr>
<tr>
<td>Dosage (L/m²/whole study)</td>
<td>1493</td>
<td>2986</td>
<td>5973</td>
</tr>
</tbody>
</table>

¹For comparison with other studies: L m⁻² d⁻¹ is equivalent to mm d⁻¹

Figure 4.1 Schematic of unsaturated (left) and saturated (right) filters.
4.2.3 Wastewater

Potato wash-water was collected from a potato storage facility in Alliston, ON. Clean water from a groundwater well was used for washing and transporting potatoes (within the facility) prior to shipping. The untreated wash-water is pumped out of the facility at the end of each shipping day. The wash-water quality varies due to quantity of potatoes shipped, type of potato shipped, type of soil potatoes were grown in, and the amount of rotten potatoes.

Wastewater was collected from the on-farm treatment system, which was part of a larger study. The treatment system is composed of a sedimentation pond, two aeration ponds, and three wetland cells operating in series. Wastewater loading into the on-farm treatment system was done by the daily injection of large volumes of wash-water after a day of potato washing and shipping. Wastewater was collected from the 3rd cell of the treatment system, 1.3 m deep basin, following primary sedimentation where most solids settle. Wastewater was collected on three occasions for ease of transport and handling. During the winter, samples were obtained by boring a hole in the surface ice layer of the basin and removing wastewater from below the ice. Water was collected in 18L buckets and stored in a walk-in refrigerator unit at 4 °C.

4.2.4 Discharge Flow Pattern

Discharge flow was monitored using time lapse camera “TimelapseCam” (Windscapes®, Calera, ML, USA) set up to take pictures of collection beakers every 5 min following dosing. This was done three times: before and after the study with demineralised water and once during regular dosing in the middle of the study. The volumes observed from the pictures were manually entered into data files.

4.2.5 Loading and Sampling

Prior to the experiment, filters were doused with clean water for a week to test the flow rates, remove any fine soil particles, and allow for sand-settling to occur. During the study, filters were dosed for 10 weeks. Dosing of filters was done by mixing each bucket of wastewater with a glass stir-stick for one minute to assure homogeneity prior to being poured into dosing cups positioned on top of each filter. The dosing cups were drilled with small holes to allow for a more even distribution of wastewater during dosing. Dosing was carried out in two sets: (i) half the dosage
in each filter, and then (ii) the second half of the dosage within 30 min. This allowed for slower seepage to prevent overfilling and water tunnelling from the higher water pressure in each filter.

The influent wastewater sample consisted of a 150 mL mixed sample taken from the dosing bucket. Because of possible stratification of the wastewater in the bucket, half of the influent sample was taken before the start of dosing and the rest of the sample was taken following the end of dosing. Of this, 50 mL was saved for TSS and VSS analysis and 100 mL was added to a 10 d cumulative sample for analysis of BOD$_5$, NH$_3$-N, NO$_3$-N, TKN, TP, and OPO$_4$. Samples were refrigerated at 4°C. TSS analysis was performed within two weeks of samples being taken.

Effluent was collected with beakers stationed under each filter to collect effluent water. Effluent was gathered the next day prior to the next dosing, with the exception of Friday, which was sampled in the afternoon to avoid leaving effluent at room temperature over the weekend. Unlike inflow, each filter had its own effluent sample. Each filter had 50 mL saved in a tube for TSS and VSS analysis and 100 mL added to a 10-day composite sample for analysis of BOD$_5$, NH$_3$-N, NO$_3$-N, TKN, TP, and OPO$_4$. Samples were refrigerated at 4 °C. TSS analysis was performed within two weeks of samples being taken. Effluent measurements of pH and ORP were taken only once for each filter using a calibrated multimeter (YSI556, YSI Inc., Yellow Spring, Ohio, USA). Multimeter measurements were taken 2-3 hours after dosing to make sure enough effluent had accumulated to take accurate measurements.

Sand samples were taken following the end of the experiment. One sample from the center of each filter of one block, was taken for the analysis of TS, VS, TKN, and microscopy. Samples were stored at 4°C in zip locking bags.

The ambient temperature was recorded hourly using a HOBO thermistor (Onset ®, Bourne, MA, USA) and the surface temperature of each filter was recorded during filter dosing using a hand-held infra-red temperature sensor.

4.2.6 Analysis of Physical, Chemical, and Biological Properties

4.2.6.1 Total Suspended Solids (TSS)

Methods were modified from Standard methods for water and wastewater analysis (Standard method 2540D, APHA, 2005). Prior to analysis, 1.5 μm Whatman glass microfiber filters (GE Healthcare Life Sciences, Baied’Urfe, Quebec, Canada) were washed using demineralised water
and dried at 105 °C for at least 4 h in an oven (Isotemp 500 series, Fisher, Ottawa, ON). Filters were cooled in a desiccant chamber for 15 min and then weighed using Sartorius CPA225D scale to find the initial filter weight. Water samples were filtered using a Buchner funnel and vacuum line. The filters were dried again at 105 °C for at least 4 h and cooled in a desiccant chamber for 15 min prior to weighing for the final sample weight. The TSS concentration (mg L⁻¹) was found using the following:

\[
TSS = \frac{(FSW - IFW)}{V} \times 1000
\]  

where FSW = final sample weight (g); IFW = initial filter weight (g); V = volume (L) of the sample filtered; and 1000 converts g to mg.

### 4.2.6.2 Volatile Suspended Solids (VSS)

Volatile suspended solids (VSS) analysis was performed, according to Standard methods 2540 D (APHA, 2005), on one day of sample per week of study. Following TSS analysis filters were put into a muffle furnace at 500 °C for half an hour to burn off all the organic matter in the sample. Samples were then allowed to cool for a minute, and put into a desiccant chamber for 15 min before being weighed. The VSS weight represents the amount of solids in the sample from organic matter.

\[
VSS = \frac{(A - C) \times 1000}{V}
\]

where VSS = volatile suspended solids (mg L⁻¹); A = final TSS weight after drying (g); C = final VSS weight after igniting (g); V = volume of sample processed (L)

### 4.2.6.3 Volatile Solids (VS)

Volatile solids were measured on sand samples from the filters following the experiment, using Standard methods 2540 D (APHA, 2004). A 10 mL sample was put into pre-weighed crucibles and then weighed using a scale (Scietech Inc., Boulder, CO, USA). Samples were dried at 104°C for 4 h, cooled for 15 min in a desiccator, and then weighed. Following this initial drying, the samples were put into a muffle furnace at 500°C for 30 min. The final weight was taken after the samples cooled for one minute in air and 1 hour in a desiccator.
\[
V_S = \frac{(A - C)}{(\text{Weight of sample})}
\]  \[4.3\]

Where \(V_S\) = volatile solids (g \(g^{-1}\)); \(A\) = final TSS weight after drying (g); \(C\) = final VS weight after igniting (g); weight of sample = weight of wet sample processed (g).

4.2.6.4 **Biochemical Oxygen Demand (BOD5)**

The 5-day biochemical oxygen demand (BOD\(_5\)) was adapted from the Standard methods 5210 B (APHA 2005). Each sample was diluted with aerated water which was inoculated with nutrients and Polyseed\(^\text{TM}\) capsule (InterLab (R), The Woodlands, TX, USA) containing known bacterial seed. Dissolved oxygen (DO) was measured using Thermo Scientific meter ORION 5 STAR and 9108 dissolved oxygen probe (Thermo Fisher Scientific, Waltham, MA, USA) prior to incubation for 5 d at 20 ± 1 °C. A seeded blank was also prepared for incubation by filling a bottle with dilution water and seed. Following incubation, DO was measured a second time and the BOD\(_5\) calculated as follows:

\[
BOD_5 = \frac{(DO1 - DO5)_{\text{sample}} - (DO1 - DO5)_{\text{seed blank}}}{V} \times \text{total volume in bottle}
\]  \[4.4\]

Where \(BOD_5\) = biochemical oxygen demand (mg L\(^{-1}\)); \(DO1\) = dissolved oxygen concentration at day 1 (mg L\(^{-1}\)); \(DO5\) = dissolved oxygen concentration at day 5 (mg L\(^{-1}\)); total volume in bottle = 300 mL; \(V\) = volume of sample in BOD\(_5\) bottle (mL).

4.2.6.5 **Nitrogen**

Total Kjeldahl Nitrogen (TKN)

Total Kjeldahl Nitrogen (TKN) procedure was adapted from standard method 4500-NorgB Macro-Kjeldahl Method (APHA 2005). Samples were digested for 1 hour with 7 mL of concentrated H\(_2\)SO\(_4\) and two Kjeldahl tablets (each containing 3.5 g K\(_2\)SO\(_4\) and 0.4 g CuSO\(_4\)). The samples were then distilled and the ammonia phenate method was used to measure the ammonia concentration. The TKN was calculated as follows:

\[
TKN = \frac{\text{NH}_3\text{Concentration} \times \text{Distillate volume}}{\text{Sample volume}} \times \text{Dilution factor}
\]  \[4.5\]
Where \( TKN \) = concentration of TKN in sample or blank (mg L\(^{-1}\)); \( NH_3 \) concentration = total \( NH_3 \) in distillate (mg L\(^{-1}\)); Distillate volume = volume of distillate collected from distillation (L); sample volume = sample volume used for digestion process (L); Dilution factor = Dilution use to dilute sample (L L\(^{-1}\)).

Ammonia

Ammonia was measured using the phenate method, adapted from standard method 4500-NH\(_3\) F (APHA, 2005). A standard curve was prepared using \( NH_4Cl \) stock solution and a spectrophotometer set at 640 nm. Samples were prepared with the appropriate dilution and 1 mL was dispensed into a glass vial. This was mixed with 0.4 mL of phenol solution, 0.4 mL of Sodium Nitroprusside solution, and 1 mL of oxidizing solution. Vials were mixed well and allowed to sit for 2 h in low light to allow color development. The absorbance was measured using a spectrophotometer at 640 nm and the concentration was found using the standard curve equation.

Nitrate

Total nitrates method was modified from standard methods 4500-NO\(_3\) electrode method (APHA 2005). A standard curve was prepared using nitrate stock solution and mV measured using a nitrate electrode Orion 300 BNWP and Orion 900200. Samples were also measured with the nitrate probe and dilution was carried out when necessary. Nitrate concentration was found using the logarithmic standard curve equation, ranging from 1 – 100 ppm.

Total Nitrogen

Total nitrogen (TN) was found using the following equation:

\[
TN = TKN + NO_3
\]  

[4.6]

where \( TKN \) = Total Kjeldahl nitrogen; and \( NO_3 \) = nitrate.

4.2.6.6 Total Phosphorus (TP) and Ortho-Phosphate (OPO\(_4\))

Total phosphorus procedures were adapted from standard methods 4500-P Phosphorus (Persulfate Digestion Method) and 4500-P E. Ascorbic Acid method (APHA, 2005).
To make the total phosphorus standard curve, at least 6 different concentrations ranging from 0.01 – 1.0 mg L$^{-1}$ of phosphate stock solution were prepared. The standards were mixed with phenolphthalein solution, 1 mL TP acid digestion solution and 0.4 g of ammonium peroxysulfate. This was boiled until the volume was reduced to approximately 10 mL and then allowed to cool. The volume was brought back to 40mL and a drop of phenolphthalein solution was added. To this, 10 N NaOH was added until the standard solution turned light pink, followed by 5 N H$_2$SO$_4$ until the pink disappeared. Distilled water was added to make a total of 50 mL and then 10 mL of the solution was transferred to a glass vial.

Ortho-phosphate standard curve was prepared by making at least 6 different concentration of phosphate stock solution ranging from 0.1 – 1.0 ppm.

For both standards, 1.6 mL of combined reagent was added to each vial. After colour developed, about 10 minutes, the absorbance was read using the spectrophotometer at 880 nm. TP samples were prepared as the standards, while the ortho-phosphate samples were diluted if necessary. Samples had 1.6 mL of combined reagent added to each and then the absorbance was read. Concentration was found by converting absorbance to concentration using the standard curve equation.

4.2.6.7 pH and Conductivity

The pH and conductivity of each filter was measured on effluent from one day. Prior to taking measurements the multimeter was calibrated. Calibration methods and standard solutions used were based on manufacturer recommendations. Measurements were taken using the YSI 556 multimeter after enough water accumulated to take measurements (~3 h). The sensor was submersed into each beaker, rinsing the probe between measurements. Readings were taken after all values stopped fluctuating (~5 min).

4.2.6.8 Sand Particle Size

The sand was analysed for particle size distribution and coefficient of uniformity. A known weight of dried sand was separated into 8 different size categories, using a series 7 sieve. The sand that would not fall through the sieve, following mechanical shaking, was considered larger than the sieve pore size. The sand left in the sieve was weighed. The smallest size sieve had to be
filtered using water, rather than just mechanical shaking. It was oven dried overnight at 104 °C before weighing. The uniformity coefficient was found using the following:

\[ UC = \frac{D_{60}}{D_{10}} \]  \[4.7\]

where UC is the uniformity coefficient, \( D_{60} \) is the 60th percentile value, and \( D_{10} \) is the 10th percentile value.

4.2.7 Microscopy

Sand samples were suspended in a solution of water and Acridine Orange dye on a standard plate and cover slip (Chabot et al., In Preparation). Confocal Zeiss LSM 510 (ZEISS©, Jena Germany) microscope was used with a single wavelength for excitation. Images were taken at 20x magnification, with highlighted RNA (red) and DNA (green) in combined Z- stacks. Adobe Photoshop (Adobe, San Jose, CA, USA) was used to combine images to get a view of both RNA and DNA present in the image.

4.2.8 Statistical Analysis

Concentration reduction was calculated by taking weekly average concentrations:

\[ \% \text{ Reduction} = \frac{\text{Inflow concentration} - \text{Effluent concentration}}{\text{Inflow concentration}} \times 100\% \]  \[4.8\]

where \% Reduction = percentage decrease in concentration; Inflow concentration = concentration of given contaminant in the inflow sample; Effluent concentration = concentration of given contaminant in the filter discharge sample.

Mass loading was calculated for inflow and effluent as follows:

\[ \text{Mass} = \frac{\text{Concentration} \times \text{Flow}}{1000} \]  \[4.9\]

where Mass = total mass of the contaminant in the water entering or leaving the filter during the time period of interest (g); concentration = average concentration of the contaminant during the time period of interest (mg L\(^{-1}\); either inflow or effluent); Flow = total amount of water during the time period of interest (L; either inflow or effluent).
Mass Reduction was calculated as a percentage for total weekly removal and whole study total removal:
\[
\% \text{ Mass Reduction} = \frac{\text{Inflow mass} - \text{Outflow mass}}{\text{Inflow mass}} \times 100\%
\]  

[4.10]

where % Mass Reduction = percentage decrease in mass; Inflow mass = mass of given contaminant in the inflow (g); Effluent mass = mass of given contaminant in the filter discharge (g).

Cumulative Mass Load and cumulative mass removals were calculated as total mass that flowed through the entire study.

The significance of treatment effects were tested using a three-way ANOVA with three factors (filter-type, dosage, and block) and two-way interactions of orientation and dosage, using SAS software (2.0.4; proc GLM) (© SAS Institute Inc., North Carolina) on total mass reduction, total effluent mass, and average temperature. ANOVA was followed by pairwise multiple comparisons of means using the Holm-Sidak method (p<0.05).

4.3 Results

4.4 Environmental Conditions

The greenhouse was temperature and light controlled. The temperature of the filters was on average 21°C, with minimum of 14.9°C and maximum of 38.6°C. Orientation was found to be a significant variable (p <0.01). Saturated filters were found to be significantly warmer than the unsaturated filters (23.0 vs 19.6°C). This was most likely because the saturated filters were set up on top of the greenhouse bench, while the unsaturated filters were positioned lower – attached to the side of the bench. There was no significant effect of dosage.

4.4.1 Sand Particle Analysis

The majority (89% (w/w)) of the sand particle size ranged between 105 – 500 µm. The coefficient of uniformity was 2.40 from a logarithmic equation of the cumulative percentage (D60 was 0.287 and D10 was 0.120). The generally acceptable coefficient of uniformity is between 1.3-1.7, which suggests that our sand was slightly less uniform than what is generally used for sand filtration (Montgomery et al., 2005). This result is not unexpected since this sand
was obtained locally by the farmer from a naturally sandy agricultural field, and it was not guaranteed to be uniform. Although screening of the sand prior to use is possible, it would add much more work for the farmers.

4.4.2 Water Flow

Clogging due to accumulation of solids and growth of biofilm occurs when a filter is nearing the end of its usability, and has to be cleaned or replaced. In this study, if the water did not completely flow through the filter within 5 h, it was considered to be clogged. There appeared to be effect of orientation and dosage, where unsaturated filters and lower dosages clogged less. Filters that never exhibited any clogging were 150 mL dosed filters, both saturated and unsaturated, and 300 mL dosed unsaturated filters. First incidents of clogging were seen at week 4 in a 600 mL dosed saturated filter. After two weeks of reduced dosage, the filter unclogged and the full dosage could again be applied. Other filters periodically clogged, although in most cases, by the next day, the wastewater had filtered through. When the inflow water TSS concentration decreased near the end of the study the clogging stopped in all but two of the 600 mL dosed filters.

The hydraulic behaviour of each filter was slightly different, influenced by both orientation and dosage (Figure 4.2). The saturated filters were found to exhibit slower discharge than the unsaturated filters, with an average time of 716 min and 621 min, respectively, to achieve 90% of expected outflow. The saturated filters can be slower due to the smaller hydraulic head and higher occurrence of clogging. The dosage was found to influence the outflow rate as well, with increasing dosages having slower outflow rates. In both saturated and unsaturated filters 600 mL dosed filters were the slowest. The average time of 90% of expected outflow to be discharged was 1155 min in 600 mL dosed, 590 min in 300 mL dosed, and 260 min in 150 mL dosed filters. There was an exception seen in the 150 mL unsaturated filter which was slower than the 300 mL dosed. This exception can be explained by under-saturation or drying of the unsaturated filters. Since the filters are dosed only once a day, the filters dry out over the course of the day before the next dosing. The fastest filter was 150 mL dosed saturated filter, where 90% of expected outflow was reached in approximately 250 min. The slowest filter was 600 mL unsaturated filter, where 90% of expected outflow was reached in ~1200 min. In the saturated filters we see that a dosage of 150 mL had the fastest outflow rate, while a 300 mL dose was more similar to the 600 mL.
mL dosage. In the unsaturated filters, 600 mL dosage is the slowest filter, while the 300 mL and 150 mL doses have a similar outflow rate, with the 300 mL dose having a higher rate through most of the time. Therefore there appears to be an interaction effect of dosage and orientation, where the 300 mL dosed filters act differently based on the orientation.
Figure 4.2 Percent outflow from mid-study readings of block 2 filters. Figure shows comparison of dosage in saturated (H) and unsaturated (V) filters.
### 4.4.3 Contaminant Removal

The TSS concentrations in the inflow water varied between 4 – 590 mg L\(^{-1}\) throughout the experiment, with an average of 264 mg L\(^{-1}\) (Table 4.2). This variability was representative of the variability found in on-farm wastewater. In the last two weeks of the study a decrease in TSS inflow was observed, coinciding with the spring snowmelt of on the treatment basin causing dilution of the wastewater. The Ontario recommended TSS discharge concentrations are <25 mg L\(^{-1}\) (MOE, 1994).

Generally, all filters were effective at removing TSS. Every treatment reduced the TSS load by >60% and TSS concentration by >79% (Table 4.3). The treatment effect showed good consistency across replicates, as indicated by low standard deviations (Table 4.4). Overall, unsaturated filters reduced TSS significantly better than saturated filters (79% vs 89%; p < 0.0001, Table 4.4). The effect of dosage was also significant (p < 0.0001). The highest removals were associated with the two lowest doses (150 mL, 89.5%; 300 mL, 87.7% TSS load removal), both of which were higher than the 600 mL dose (75.2% removal). This can also be seen in the cumulative outflow mass in Figure 4.3, where the 150 mL and 300 mL curves were quite similar, while the 600 mL curve was distinctly higher. The interaction of orientation and dosage was not statistically significant (p=0.1528). The reduction of TSS concentration had similar results to the load removal, although with about 5% lower reductions compared to TSS loads (Table 4.4). This was what we would expect, since the inflow and outflow volumes should be similar, with only a small amount of volume loss in outflow due to evaporation. None of the filters were able to have a discharge TSS concentration below the provincial limit of 25 mg L\(^{-1}\) with effluent concentrations ranging from 31 to 91 mg L\(^{-1}\).

The VSS is the fraction of suspended solids in water that are volatile, and therefore, considered to be organic matter. We expect that VSS should have a similar removal to TSS since they are both solids which are removed through similar physical mechanisms. Total VSS mass removal from the entire study ranged from 59 – 94%, with an average from all filters of 82% (Table 4.4), which was consistent with the TSS results. Orientation was found to have a significant effect (p = 0.035; Table 4.5) with saturated filters having higher removal than unsaturated filters (88.3% vs 75.7%). Dosage was also found to be significant with the best mass removal found in the 150 mL dose, followed by the 300 mL, and 600 mL being last (90.9% vs 85.9% vs 69.1%). The 150
mL dose was significantly higher than the 600 mL dose; however, the 300 mL dose was not found to be significantly different than either the 150 mL rate or the 600 mL rate. Lastly, there was no significant interaction between orientation and dosage.

The inflow BOD$_5$ was found to be on average 332 mg L$^{-1}$ throughout the experiment (Table 4.3). The Government of Ontario recommends that BOD$_5$ is <25 mg L$^{-1}$ when discharging into rivers or lakes (Ontario Ministry of Environment, 1994b). All treatments removed BOD$_5$ well, with the mean of the total BOD$_5$ load removal between 77 – 99% (Table 4.4). The unsaturated filters had a removal efficiency of 97.2%, which was statistically better than the saturated filters with removal of 83.0% (p < 0.0001, Table 4.5). Effect of dosage was also found to be significant (p = 0.0007). The best performance was observed in the 150 mL dosed filter with 96.2% removal, significantly different than the 300 mL and 600 mL doses. The 300 mL and 600 mL doses were not statistically different from each other at removal rates of 87.9% and 86.2%. The concentration had a removal between 72 – 99% (Table 4.4). There was very little difference between mass and concentration removal, with only about 5% difference, except in 150 mL dosed unsaturated filters, which have the same removal in both. Most of the unsaturated filters were able to meet the BOD$_5$ discharge criteria at 7, 15 and 27 mg L$^{-1}$, while the effluent concentrations of the saturated filters ranged from 32 to 84 mg L$^{-1}$.

Inflow TP was on average 16.8 mg L$^{-1}$, where recommended discharge limits by the provincial government are <0.3 mg L$^{-1}$ (Table 4.3; Ontario Ministry of Environment, 1994b). The TP removal was very good ranging from 60 to 99% (Table 4.4). The unsaturated filters were not significantly different than saturated filters, with TP mass removal of 88.8% and 84.4%, respectively (p = 0.0516, Table 4.5). The dosage volume effect was found to be significant (p = <.0001). The highest TP removal was 96.0% from the 300 mL dosed filters, on average. The 150 mL dosed filters had a removal of 93.0%, which was not statistically lower. The 600 mL dosed filters were statistically lower with 70.8% removal. The interaction of orientation and dosage was not statistically significant (p = 0.182). The TP concentration removal ranged between 71 – 98%, with no difference from mass removals (Table 4.4). None of the outflow TP concentrations were able to meet the provincial guideline.

Inflow OPO$_4$ was 10.0mg L$^{-1}$; Table 4.3). The OPO$_4$ mass removal was very good overall ranging from 46 to 100% (Table 4.4). The unsaturated filters were not significantly different than saturated filters, with OPO$_4$ mass removal of 83.5% and 86.9%, respectively (p = 0.4001, Table
The dosage effect was found to be significant (p = <.0001). The highest OPO₄ removal was 97.0% from the 300 mL dosed filters, on average. The 150 mL dosed filters had a removal of 94.3%, which was not statistically lower. The 600 mL dosed filters were statistically lower with 64.2% removal. The interaction of orientation and dosage was not statistically significant (p = 0.632). The OPO₄ concentration removal ranged between 43 – 99%, with no difference from mass removals (Table 4.4). OPO₄ had similar significant interactions as TP, suggesting a constant fraction of OPO₄ in TP. The % of OPO₄ in TP ranged from 64 – 100% in the inflow and 31 – 100% in the outflow. Since OPO₄ is the fraction of TP is biologically reactive, this suggests that most of the P was treatable with biological means.

Total nitrogen (TN) removal was variable, ranging from 25 – 74% average mass removal (Table 4.4). There was a significant effect of orientation and dosage (p < 0.0001; Table 4.4). The removal was higher in the unsaturated filters, rather than saturated filters (68.0% vs 37.3%) and the removal increased with dosage (44.5% vs 50.0% vs 63.7%). There was also a significant interaction between orientation and dosage, therefore, the effect of one variable will change based upon the other (p <0.0001). The TN concentration removal ranged from 9 – 74% (Table 4.4). Only at 150 mL dosage had the saturated filters higher concentration removal than the unsaturated, however, this was most likely due to remaining water in the saturated filters which was not measured. Concentration removal, found a significance in orientation, with unsaturated filters having higher removal than saturated (27.0% vs 53.3%; p <.0001). The 300 mL dosage was not found to be significantly different from the 150 mL or the 600 mL, however the 150 mL was different than the 600 mL.

Inflow TKN was 133 mg L⁻¹ (Table 4.3). Overall mass removal was good ranging from 25 –87% (Table 4.4). There was a significant orientation effect, with unsaturated filters having significantly higher removal than saturated filters (79.9% vs 36.9%; p < 0.0001; Table 4.5). There was also a significant effect of dosage (p < 0.0001). Each dosage was significantly different from the other two, with 150 mL dosage having the highest removal, followed by 300 mL, and lastly 600 mL (72.7% vs 55.6% vs 47.0%). There was also a significant interaction between orientation and dosage, therefore, the effect of one variable will change based upon the other (p <0.0001). The concentration removal ranged from 16 – 82%. It had about a 10% lower

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3The cation exchange capacity of the soil was 5.5 MEQ/100g
removal than the mass removals (Table 4.4).

Inflow NH$_3$ was found to be 93.1 mg L$^{-1}$, which was similar to TKN concentrations (Table 4.3). This suggests that there was very little organic N which contributed to the TKN concentrations. Provincial discharge guidelines limit NH$_3$ concentrations to <1 mg L$^{-1}$ (Ontario Ministry of Environment, 1994b). Overall mass removal was good ranging from 21 – 97% (Table 4.4). There was a significant orientation effect, with unsaturated filters having significantly higher NH$_3$ removal than saturated filters (87.5% vs 34.8%; p <0.0001; Table 4.5). There was also a significant effect of dosage (p <0.0001). Each dosage was significantly different from the other two, with 150 mL dosage having the highest removal, followed by 300 mL, and lastly 600 mL (79.0% vs 59.4% vs 44.9%). There was also a significant interaction between orientation and dosage, therefore, the effect of one variable will change based upon the other (p <0.0001). The concentration removal ranged from 10 –95%, with about 10% lower removal than mass (Table 4.4). None of the filters had a concentration within the discharge guidelines. Decrease in TKN and NH$_3$ corresponded to NO$_3$ increase. This suggests that unsaturated filters were undergoing nitrification; however, remaining high concentrations of NO$_3$ show that denitrification in an anoxic environment is still necessary for N removal (Figure 4.3). All three N fractions found significant treatment effects of filters orientation and dosage. This is what we would expect since the removal and increase of these are linked through nitrification and denitrification.

The NO$_3$ in the inflow water was found to be quite low, with a concentration of 4.0 mg L$^{-1}$ (Table 4.3). This gives a mass load into the filters of 0.6 mg, 1.2 mg, and 2.4 mg, on a daily basis. The provincial NO$_3$ discharge guideline is <10.0 mg L$^{-1}$ (Ontario Ministry, 1994). The NO$_3$ mass removal was on average -623% across all treatment, indicating a dramatic increase rather than decrease (Table 4.4). This increase was a sign of nitrification of NH$_3$ to NO$_3$. There were significant treatment effects of orientation, dosage, as well as their interaction (p < .0001 for all). Saturated filters had an average removal of 86.1%, while unsaturated filters have an average increase by 1333% (Table 4.5). The dosage effect showed 600 mL dosed filters had the least NO$_3$ increase followed by 300 mL dosed filters, and lastly 150 mL (-252% vs -642% vs -975%). Interactions of orientation and dosage were significant; therefore NO$_3$ quantity present will be dependent upon both dosage and filter orientation. In other words, the effect of one variable will change based upon the other. Concentration reduction of NO$_3$ ranged from -5411 – 87% (Table 4.4; Figure 4.4). Concentration removals had the same significant interactions as
NO$_3$ mass removal. All the horizontal filters had their NO$_3$ concentrations within the provincial limit. The average outflow ORP ranged from 59.1 mV to 83.7 mV (Table 4.4). The unsaturated filters had slightly higher ORP than the saturated filters (78.3 vs 70.4 mV; $p = 0.0011$; Table 4.5). There was also a significant effect of dosage ($P < 0.0001$). The 150 mL dosage had the highest ORP of 81.5 mV – not statistically different from the 300 mL dosage of 77.9 mV. The 600 mL dosage ORP was statistically lower at 63.7 mV. The interaction of orientation and dosage was not significant.

The DO in outflow spanned a large range, from 0.97 mg L$^{-1}$ to 10.32 mg L$^{-1}$ (Table 4.4). Both orientation and dosage treatment effects were significant ($p < 0.001$ for both). The unsaturated filters had an average outflow concentration of 7.3 mg L$^{-1}$, while the saturated filters were significantly lower with 2.4 mg L$^{-1}$ (Table 4.5). The outflow DO was the highest in the 300 mL dosage with 6.4 mg L$^{-1}$, which was statistically the same as the 150 mL dosage (5.7 mg L$^{-1}$) and different than the 600 mL dosage (2.5 mg L$^{-1}$). Lastly, the interaction between orientation and dosage was found to be significant ($p = 0.002$). The significant interactions were similar to the NO$_3$ results, which we would expect, since NH$_3$ nitrification to NO$_3$ is dependent upon DO levels.
Table 4.2 Total solid (TS) and volatile solid (VS) average concentration and percentage from sand of each filter of one block following the end of the study.

<table>
<thead>
<tr>
<th>Filter</th>
<th>TS (g g(^{-1}))</th>
<th>TS (%)</th>
<th>VS (g g(^{-1}))</th>
<th>VS (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Control</td>
<td>0.998</td>
<td>99.80</td>
<td>0.009</td>
<td>0.95</td>
</tr>
<tr>
<td>600 unsaturated</td>
<td>0.93</td>
<td>93.16</td>
<td>0.01</td>
<td>0.94</td>
</tr>
<tr>
<td>300 unsaturated</td>
<td>0.92</td>
<td>92.28</td>
<td>0.01</td>
<td>1.08</td>
</tr>
<tr>
<td>150 unsaturated</td>
<td>0.93</td>
<td>93.12</td>
<td>0.01</td>
<td>1.01</td>
</tr>
<tr>
<td>600 saturated</td>
<td>0.85</td>
<td>84.81</td>
<td>0.01</td>
<td>0.82</td>
</tr>
<tr>
<td>300 saturated</td>
<td>0.82</td>
<td>82.34</td>
<td>0.01</td>
<td>1.00</td>
</tr>
<tr>
<td>150 saturated</td>
<td>0.84</td>
<td>83.81</td>
<td>0.01</td>
<td>1.21</td>
</tr>
</tbody>
</table>

Table 4.3 Average contaminant concentration of the wastewater (inflow) from the entire study (10 weeks).

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Average (mg L(^{-1}))</th>
<th>STD (mg L(^{-1}))</th>
</tr>
</thead>
<tbody>
<tr>
<td>TSS</td>
<td>264</td>
<td>131</td>
</tr>
<tr>
<td>VSS</td>
<td>0.135</td>
<td>0.088</td>
</tr>
<tr>
<td>BOD(_5)</td>
<td>362</td>
<td>404</td>
</tr>
<tr>
<td>TP</td>
<td>11.3</td>
<td>8.58</td>
</tr>
<tr>
<td>OPO(_4)</td>
<td>11.2</td>
<td>6.62</td>
</tr>
<tr>
<td>TN</td>
<td>143</td>
<td>93.9</td>
</tr>
<tr>
<td>NO(_3)</td>
<td>4.75</td>
<td>6.97</td>
</tr>
<tr>
<td>NH(_3)</td>
<td>99.3</td>
<td>68.9</td>
</tr>
<tr>
<td>TKN</td>
<td>142</td>
<td>99.9</td>
</tr>
</tbody>
</table>
Table 4.4 Summary of mean removal efficiency for each treatment combination (orientation and dosage) on a mass-load basis and concentration-basis for each parameter during the entire study (10 weeks).

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Removal</th>
<th>Saturated</th>
<th></th>
<th></th>
<th>Unsaturated</th>
<th></th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>600 mL d⁻¹</td>
<td>300 mL d⁻¹</td>
<td>150 mL d⁻¹</td>
<td>600 mL d⁻¹</td>
<td>300 mL d⁻¹</td>
<td>150 mL d⁻¹</td>
</tr>
<tr>
<td>TSS</td>
<td>Conc.</td>
<td>62%</td>
<td>81%</td>
<td>81%</td>
<td>77%</td>
<td>87%</td>
<td>89%</td>
</tr>
<tr>
<td>TSS</td>
<td>Mass</td>
<td>69%</td>
<td>84%</td>
<td>85%</td>
<td>82%</td>
<td>91%</td>
<td>94%</td>
</tr>
<tr>
<td>VSS</td>
<td>Conc.</td>
<td>51%</td>
<td>80%</td>
<td>85%</td>
<td>72%</td>
<td>90%</td>
<td>90%</td>
</tr>
<tr>
<td>VSS</td>
<td>Mass</td>
<td>59%</td>
<td>82%</td>
<td>87%</td>
<td>77%</td>
<td>93%</td>
<td>94%</td>
</tr>
<tr>
<td>BOD₅</td>
<td>Conc.</td>
<td>72%</td>
<td>75%</td>
<td>90%</td>
<td>94%</td>
<td>95%</td>
<td>99%</td>
</tr>
<tr>
<td>BOD₅</td>
<td>Mass</td>
<td>77%</td>
<td>79%</td>
<td>93%</td>
<td>95%</td>
<td>97%</td>
<td>99%</td>
</tr>
<tr>
<td>TP</td>
<td>Conc.</td>
<td>71%</td>
<td>95%</td>
<td>89%</td>
<td>72%</td>
<td>98%</td>
<td>98%</td>
</tr>
<tr>
<td>TP</td>
<td>Mass</td>
<td>71%</td>
<td>94%</td>
<td>88%</td>
<td>71%</td>
<td>98%</td>
<td>98%</td>
</tr>
<tr>
<td>OPO₄</td>
<td>Conc.</td>
<td>60%</td>
<td>96%</td>
<td>88%</td>
<td>58%</td>
<td>97%</td>
<td>98%</td>
</tr>
<tr>
<td>OPO₄</td>
<td>Mass</td>
<td>64%</td>
<td>96%</td>
<td>90%</td>
<td>64%</td>
<td>98%</td>
<td>99%</td>
</tr>
<tr>
<td>TN</td>
<td>Conc.</td>
<td>16%</td>
<td>17%</td>
<td>48%</td>
<td>54%</td>
<td>66%</td>
<td>40%</td>
</tr>
<tr>
<td>TN</td>
<td>Mass</td>
<td>27%</td>
<td>25%</td>
<td>60%</td>
<td>62%</td>
<td>74%</td>
<td>68%</td>
</tr>
<tr>
<td>TKN</td>
<td>Conc.</td>
<td>16%</td>
<td>16%</td>
<td>48%</td>
<td>60%</td>
<td>82%</td>
<td>74%</td>
</tr>
<tr>
<td>TKN</td>
<td>Mass</td>
<td>27%</td>
<td>25%</td>
<td>60%</td>
<td>67%</td>
<td>87%</td>
<td>86%</td>
</tr>
<tr>
<td>NH₃</td>
<td>Conc.</td>
<td>10%</td>
<td>13%</td>
<td>49%</td>
<td>60%</td>
<td>95%</td>
<td>93%</td>
</tr>
<tr>
<td>NH₃</td>
<td>Mass</td>
<td>21%</td>
<td>22%</td>
<td>61%</td>
<td>69%</td>
<td>97%</td>
<td>97%</td>
</tr>
</tbody>
</table>
Table 4.5 Statistical summary of contaminant treatment efficiency from total removal over the entire study (10 weeks). Letters denote statistical difference of values.

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Units</th>
<th>Saturated</th>
<th>Unsaturated</th>
<th>p-value</th>
<th>600mL</th>
<th>300mL</th>
<th>150mL</th>
<th>p-value</th>
<th>p-value</th>
<th>Block</th>
</tr>
</thead>
<tbody>
<tr>
<td>TSS</td>
<td>mg/L</td>
<td>79.2%</td>
<td>89.1%</td>
<td>&lt;.0001</td>
<td>75.2%</td>
<td>87.7%</td>
<td>89.5%</td>
<td>&lt;.0001</td>
<td>0.153</td>
<td>0.833</td>
</tr>
<tr>
<td>TSS</td>
<td>mg/L</td>
<td>74.6%</td>
<td>84.2%</td>
<td>&lt;.0001</td>
<td>69.4%</td>
<td>83.9%</td>
<td>84.8%</td>
<td>&lt;.0001</td>
<td>0.179</td>
<td>0.805</td>
</tr>
<tr>
<td>VSS</td>
<td>mg</td>
<td>75.7%</td>
<td>88.3%</td>
<td>0.035</td>
<td>69.1%</td>
<td>85.9%</td>
<td>90.9%</td>
<td>0.0131</td>
<td>0.778</td>
<td>0.505</td>
</tr>
<tr>
<td>VSS</td>
<td>mg/L</td>
<td>71.2%</td>
<td>84.1%</td>
<td>0.0818</td>
<td>62.1%</td>
<td>83.0%</td>
<td>87.9%</td>
<td>0.0221</td>
<td>0.717</td>
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<td>BOD₅</td>
<td>mg/L</td>
<td>83.0%</td>
<td>97.2%</td>
<td>0.0011</td>
<td>86.2%</td>
<td>87.9%</td>
<td>96.2%</td>
<td>0.0007</td>
<td>0.022</td>
<td>0.652</td>
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<td>83.9%</td>
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<td>72.0%</td>
<td>82.5%</td>
<td>88.4%</td>
<td>0.0022</td>
<td>0.754</td>
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<td>TN</td>
<td>mg</td>
<td>37.3%</td>
<td>17.8%</td>
<td>&lt;.0001</td>
<td>44.5%</td>
<td>19.2%</td>
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<td>mg</td>
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<td>47.0%</td>
<td>22.2%</td>
<td>55.6%</td>
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<td>16.4</td>
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<td>16.7%</td>
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<td>38.0%</td>
<td>24.1%</td>
<td>49.3%</td>
<td>0.0001</td>
<td>17.5</td>
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<td>NH₃</td>
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<td>19.8%</td>
<td>&lt;.0001</td>
<td>44.9%</td>
<td>25.6%</td>
<td>59.4%</td>
<td>&lt;.0001</td>
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<tr>
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<td>27.1%</td>
<td>53.7%</td>
<td>&lt;.0001</td>
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<td>&lt;.0001</td>
<td>253%</td>
<td>361%</td>
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<td>&lt;.0001</td>
<td>-353%</td>
<td>466%</td>
<td>-1034%</td>
<td>&lt;.0001</td>
<td>2603</td>
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<tr>
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<td>mg</td>
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<td>12.1%</td>
<td>0.0516</td>
<td>70.8%</td>
<td>7.52%</td>
<td>96.0%</td>
<td>0.0001</td>
<td>6.27</td>
<td>0.182</td>
</tr>
<tr>
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<td>mg/L</td>
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<td>12.1%</td>
<td>0.0392</td>
<td>71.3%</td>
<td>7.37%</td>
<td>96.2%</td>
<td>&lt;.0001</td>
<td>5.88</td>
<td>0.232</td>
</tr>
<tr>
<td>OPO₄</td>
<td>mg</td>
<td>83.5%</td>
<td>17.6%</td>
<td>&lt;.0001</td>
<td>64.2%</td>
<td>14.3%</td>
<td>97.0%</td>
<td>&lt;.0001</td>
<td>7.33</td>
<td>0.632</td>
</tr>
<tr>
<td>OPO₄</td>
<td>mg/L</td>
<td>81.4%</td>
<td>19.3%</td>
<td>0.4809</td>
<td>59.1%</td>
<td>15.9%</td>
<td>56.8%</td>
<td>0.0001</td>
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<td>0.543</td>
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<td>2.49</td>
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<td>8.68</td>
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80
Figure 4.3 Top Panel: Weekly average total suspended solids (TSS) concentration (mg L\(^{-1}\)) of inflow and outflow. Bottom Panel: cumulative mass load of TSS (mg) in inflow and outflow from 600 mL, 300 mL, and 150 mL dosages of saturated (left) and unsaturated (right) filters.
Figure 4.4 Average nitrogen outflow concentration from the entire study for each filter (Sat = saturated, Unsat = unsaturated) and inflow, showing decreasing ammonia (NH$_3$) and increasing nitrate (NO$_3$) from nitrification, and ammonia removal (TKN).
4.4.4 Microscopy

Bright field micrographs (Figure 4.5) show sand particles at 20× objective, with DNA and RNA (in green) overlapped onto the brightfield image. The 150 mL dosed unsaturated filter (Fig. 5, row C) clearly shows more DNA and RNA visible in the sand sample compared to the 600 mL dosed saturated filter (Fig. 5, row B) and the unfiltered sand (Fig. 5, row A), which has the least. The DNA and RNA represent the presence of organic material in the sample. Although we cannot identify what the source was, we suggest that the differences are due to microbiological growth within the filter sand. If the difference in DNA and RNA levels was due to solids caught by the filter, the 600 mL dosed saturated filter would have the most because it had the highest dosage. On the other hand, if the difference was due to microbiological growth, then the observation that the 150 mL dosed unsaturated shows the most is in agreement with our observations of high nitrate production driven by microbial nitrification of ammonia.
Figure 4.5 Brightfield micrographs at 20x objective (left column), green pseudocolourlense highlighting DNA and RNA on the same sample (middle column), overlay of the DNA and RNA green onto the brightfield micrographs (right column). Samples showing are: experimental sand prior to filtration (row A), sand gathered from a saturated 600 mL dosed filter (row B), and sand gathered from unsaturated 150 mL dosed filter (row C).
4.5 Discussion

4.5.1 Water Flow

Most of the clogging was observed in the saturated filters. This suggests that saturated filters would need more upkeep than unsaturated filters, therefore, increasing the manpower and cost of the filter system. Additionally, in most treatment systems, wastewater would not be dosed in one large volume, but closer to continual trickling. The clogging in our system may have been caused by this one large dosage, rather than more customary intermittent loading, especially since clogged filters unclogged after some time. Potato shipping does not usually occur on a daily basis and there are often longer periods of no water movement. This means that during time of no activity the clogged filters would have time to unclog, as they did in the experiment. Although during heavy shipping time (every day for two weeks) the filters could clog and back-up or overflow the system. Future research is needed to determine when unclogging is no longer occurring.

Guar et al. (2010) looked at filtering turkey wastewater through unsaturated sand filters, with a dosage 132 L m$^{-2}$ d$^{-1}$ over 645 d, and found no clogging throughout the experiment. Although they do not mention TSS, they had a BOD$_5$ of 485 ± 151 mg L$^{-1}$ in the inflow wastewater.

Rodgers et al. (2004b) looked at agricultural wastewater loading with a chemical oxygen demand (COD) of 18.2 g m$^{-2}$ d$^{-1}$ and TSS of 2.7 g m$^{-2}$ d$^{-1}$ and found it took 42 d before clogging occurred. Spychala and Blazejewski (2003) found it took 150 d for their filters to clog at 11.6 g m$^{-2}$ d$^{-1}$ COD and 2.7 g m$^{-2}$ d$^{-1}$ TSS in domestic wastewater. Due to the high risk for clogging, the US Environmental Protection Agency (EPA) recommends that sand filters be dosed with no more than 9.3 g m$^{-2}$ d$^{-1}$ COD and 3.9 g m$^{-2}$ d$^{-1}$ TSS (EPA, 1999). Our study had clogging appear much sooner than in the studies mentioned, possibly due to our TSS dosage being much higher than what the EPA limits suggest, with doses of 9 g m$^{-2}$ d$^{-1}$ (150 mL d$^{-1}$), 18 g m$^{-2}$ d$^{-1}$ (300 mL d$^{-1}$), and 36 g m$^{-2}$ d$^{-1}$ (600 mL d$^{-1}$). This suggests that intermittent loading may be better for prevention of clogging. Controlled sand filter loading can be done by potato farmers with the use of a simple pumping system, rather than passive inflow into the treatment system following shipping.
4.5.2 Contaminant Removal

There was a moderate to strong correlation between TSS and BOD$_5$ mass removal ($r^2 = 0.60$), and between TSS and TP mass removal ($r^2 = 69$). This suggests that the solids, measured by TSS, were also the source of most of the phosphorus and organic matter causing the high BOD$_5$ readings. Throughout the study the TSS in the influent water was on average 44% VSS, therefore nearly half of the solids were from organic matter. This suggests that TSS removal was a good indicator of P and organic matter removal. In more practical terms, for farmers, this means that focusing on separating solids from their wastewater will decrease TP and BOD$_5$ proportionally.

All parameters had a significant effect of orientation with the exception of TP, OPO$_4$, and VSS concentration. Overall, the unsaturated filters had higher removal rates of TSS, VSS, BOD$_5$, TP, and TN. No significant dosage effect between 300 mL and 150 mL rates was found in all parameters, with the exception of TKN, NH$_3$, and NO$_3$. In all these cases, the 600 mL dosage had significantly lower removal. The best removal of TSS, VSS, BOD$_5$, and TP was through 300 mL or 150 mL dosages in unsaturated filters. When we look at parameters which have a significant interaction between dosage and filter orientation (TSS, VSS, BOD$_5$, TKN, NH$_3$, TP, OPO$_4$), we notice that the 300 mL dosage will be more similar to 150 mL dosage or 600 mL based on the filter orientation. We see this in the outflow volume patterns in Fig 1 as well. We see that in the saturated filters the 300 mL dosage has outflow speed similar to the 600 mL dosage, while in the unsaturated filters it was more similar to the 150 mL outflow speed.

TSS removal was correlated with filter temperature, and a mild negative correlation was found ($r^2 = 0.37$). This suggests that there was little possibility of temperature effect on TSS removal. Additionally, there was a significant effect of filter temperature on block but not TSS removal.

No positive correlation of BOD$_5$ (mg L$^{-1}$) and DO (mg L$^{-1}$) outflow was found. Unlike NO$_3$, BOD$_5$ had no significant orientation and dosage interaction, suggesting that BOD$_5$ removal was not affected by increased DO.

Very few recent studies have looked at basic one-layer, single-pass, sand filtration, unsaturated or saturated; although, they have been used in wastewater treatment for over a hundred years (Healy 2007b). Kang et al. (2007) looked at similar dosages in flow-through filters using turkey processing wastewater and found an average BOD$_5$ removal of 99.7, 98.6, and 89.2% concentration removal for low (66 L m$^{-2}$ d$^{-1}$), medium (132 L m$^{-2}$ d$^{-1}$), and high (264 L m$^{-2}$ d$^{-1}$)
dosages, respectively. They also found average total organic carbon concentration removal of 97.3% (low), 94.3% (medium) and 89.4% (high). Their total organic carbon removal is best compared to our VSS removal, which had similar removal rates. Despite their higher BOD$_5$ inflow, they had a higher percent removal compared to our study. This could be due to the 60min dosing intervals they used, instead of our one volume dosing.

Saturated roughing filters are often used for treating river water for drinking. Dastanaie et al. (2007) looked at using a roughing filter with gravel media for treating river water for drinking. They found a removal efficiency of 89.7% of TSS from an inlet concentration of 18.93 mg L$^{-1}$ at a dosage of 1.8 m h$^{-1}$. This TSS concentration removal is comparable to ours. Similarly, Kalibbala et al. (2012) looked at HRF for treatment of river water for drinking and found dissolved organic carbon removal of 27% at a dosage of 0.8 m h$^{-1}$. These types of filters are used for treatment of drinking water and as such have lower inflow concentrations. Regardless, they are one of the few saturated filters available for comparison. Other studies have also found 60-90% turbidity removal for treatment of drinking water, which is also a measurement of solids present, however, not directly correlated with TSS (Kalibbala et al. 2012; Dome, 2000; Ochieng et al. 2004; Nkwonta et al. 2010). Lastly, the removal efficiency of organic matter in our study is more than what Kalibbala et al. (2012) found. Organic matter removal is measured by VSS and BOD$_5$ removal in our study.

Similar to TSS results, DO and NO$_3$ mass removal was best in dosages of 150 mL and 300 mL filters; however, the 150 mL dosage was significantly different from 300 mL only in NO$_3$ mass removal. Both DO and NO$_3$ have a significant interaction of orientation and dosage. This means that the two variables, dosage and orientation, have a synergistic effect - having a higher removal together than either one alone. DO (mg L$^{-1}$) and NO$_3$ (mg L$^{-1}$) outflow concentrations were found to have a moderately strong correlation ($r^2 = 0.65$). In N removal, NH$_3$ is nitrified to NO$_3$ in an oxygenated environment, followed by denitrification in an anoxic environment where NO$_3$ is converted to nitrite and to N$_2$ gas, which leaves the system. The low volume and high surface area interaction of the wastewater in the 300 mL and 150 mL dosed filters causes more oxygenation, hence the increased DO in the outflow water. The saturated filters, on the other hand, have little opportunity of oxygen reaching the wastewater, because all pores are filled with water. The presence of oxygen leads to more organic matter growth which allows for nitrification. Increased organic matter growth was measure by VS and visually explored through
microscopy. The VS was lower in higher dosages and micrograph images showed more RNA and DNA in the 150 mL dosed filter compared to the 600 mL dosed – hence more biological matter present. The US EPA states usual N removals using intermittent sand filters on residential septic tank effluent to be 40% TN, 90% TKN, 90% NH₃-N, and 99% increase in NO₃ (EPA, 1999). Our 150 mL dosed filters had more removal in all N constituents. Therefore, the 150 mL filters are the best option for farmers with concerns about N in their wastewater, however, other treatments will be necessary to denitrify and fully remove N from the water.

4.5.3 Practical Application

The difference between TSS mass removal of the worst performing treatment (600 mL - saturated) and the best performing treatment (150 mL – unsaturated) was 27%. Similarly, BOD₅ mass removal was only 22% lower in the 600 mL dosed saturated filter. Although these differences were statistically significant, this does not mean that it may make much difference to a farmer. Therefore, we must look at how these dosages transfer to on-farm filter design. The farm, where the wastewater came from, discharged into their treatment system 2,510,000 L y⁻¹ (2013-2014) and their average wash-water production was 18,190 L d⁻¹, occurring on average 13 d a month from September to July. These days are usually random, depending upon the demand for potatoes. This means that the higher dosages that experienced clogging would have time to unclog. Using the dosage rates tested in this study, 34, 68, and 136 L m⁻² d⁻¹, it would require an area of 535, 267, and 134 m², respectively, to filter their water in one day (Table 4.6). As a comparison, the sedimentation basin on the farm’s wash-water treatment system has an area of 340 m². Building an unsaturated filter, which requires a top-dosing system and outlets for the water, of this size would take more engineering and infrastructure. Saturated filters are easier to design and build on such a large surface area, by making a sand filled sub-surface flow wetland, for example. This would be especially hard in the winter, when pumping systems and flow through pipes would freeze and hinder water flow through the system.

As an alternative, a filtration system can be designed which operates daily. Since the farm only ships roughly 13 d mon⁻¹, spreading out the volume into smaller portions throughout the year can lower the area requirements. Therefore, the 2,510,000 L y⁻¹ would now become 7,000 L d⁻¹ inflow into the filter. Based on out dosages, 34, 68, and 136 L m⁻² d⁻¹, the required areas would be 205, 103, and 51 m². This is about 60% less area required compared to dosing the filter only
on days when the storage facility ships potatoes. A system like this would require a pump to dose the filters on a daily basis, instead of a passive flowing system. Making a 205 m$^2$ unsaturated filter, compared to a 535 m$^2$ unsaturated filter would require less work. Another benefit of a pump would be to have an intermittently dosed system, which would minimize clogging. However, intermittent dosing with potato wash-water has yet to be studied and therefore, there are no removal rates available. Although, we can assume that more frequent, smaller dosages will increase the water exposure to the sand surface area and therefore increase treatment. On the other hand, as mentioned in the previous paragraph, pumping systems do not function in the winter when freezing could stop or damage the system. Intermittent dosage systems need to be tested in cold climates to find the treatment efficiency.

A 150 mL dosed filter would use 75% more area to filter the water in one day compared to the 600 mL dosage (535 vs 134 m$^2$), therefore a 75% increase in area, equates to 27% increase in TSS mass removal (69% (600 mL – saturated) vs 94% (150 mL – unsaturated)). Since many farmers have more land available than money for infrastructure, it is more economical for farmers to build a larger saturated filter, than a smaller unsaturated filter, to achieve the same TSS mass removal. At a 75% increase in area used for 150 mL dosed unsaturated compared to the 600 mL dosed saturated filter, other removals would increase by: 22% BOD$_5$, 27% TP, 35% OPO$_4$, 69% TKN, and 78% NH$_3$. Since the BOD$_5$ from the unsaturated filters was below the provincial guideline, the use of the filter in the larger system would be helpful. The existing treatment system on the potato farm has an outflow TP concentration of 2.5 mg L$^{-1}$, therefore, it needs an additional 78% removal to reach the discharge guideline. Most of the filters had more than 78% removal, therefore, it would help reach the guideline. Lastly, although none of the filters were able to reach the TSS discharge guideline, the existing treatment system had no issues with this. Filtration is used most often as either a primary treatment at the beginning for removing large contaminants, or at the end as a tertiary to remove fine contaminants. However, removing solids earlier in the treatment system could allow for further treatment of other parameters further in the system. For example, removing solids could allow for UV light to kill unwanted microorganisms. Additionally, the oxygenation early on would allow for denitrification later in the system. Therefore, the best option is for the filtration system to be used following sedimentation.
The greatest benefit from flow-through filters is the nitrification, which was not observed in the saturated filters. Since N removal in single-pass sand filtration is limited by the lack of a denitrifying environment, many other studies looked at additional strategies for getting further removal (Healy et al. 2007b). These can be adapted for saturated filters to supplement N removal. Healy et al. (2007) looked at the performance of stratified filters with recirculation and dosed with mixed liquor from an activated sludge for extra removal of N and BOD$_5$. With inflow concentrations of 352.9 mg L$^{-1}$ TSS and 2921 mg L$^{-1}$ chemical oxygen demand (COD) and 175.7 mg L$^{-1}$ TN, they found removals of 100% TSS, 99% COD, and 83-91% TN. Additionally, they found that by slowly increasing the hydraulic dosage, they found the best performance was seen at 10 L m$^{-2}$ d$^{-1}$ into the anoxic recirculation tank, therefore having a rate of 14 g m$^{-2}$ d$^{-1}$ of COD, 3.7 g m$^{-2}$ d$^{-1}$ of TSS and 2.1 g m$^{-2}$ d$^{-1}$ of TKN. These removal rates are better than the present study, due to the extra filtration from recirculation. The prolonged filtration allowed nitrification of all the NH$_3$ to complete and deplete the oxygen, creating an anoxic environment for denitrification.

The treatment system present on the potato farm has average concentration removals of 65% NH$_3$, 50% TKN, and 44% TN, and outflow concentrations of 53.7 mg L$^{-1}$ NH$_3$, 101.7 mg L$^{-1}$ TKN, 1.5 mg L$^{-1}$ NO$_3$, and 89.7 mg L$^{-1}$ TN (2013-2014 data). To pass governmental discharge limits, further removal is necessary. Nitrification is limiting within this system for N removal. Therefore, a filtration system with additional nitrification would be the best option. As the NO$_3$ increases from the filtration, the existing system can denitrify to remove N.

*Table 4.6 Area needed to create filters based on-farm wash-water production and dosages tested.*

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<tr>
<th>Wash-water production (L d$^{-1}$)</th>
<th>Dosage (L m$^{-2}$ d$^{-1}$)</th>
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4.6 Conclusion

Unsaturated filters were found to be significantly better at contaminant removal for all contaminants. TSS mass removal was an average 89.1% from unsaturated filters and 74.6% from saturated filters. Overall, the smaller dosages were found to have more mass removal than the larger dosages. TSS mass removals were found to be 89.5, 87.7, and 75.2% from 150 mL, 300 mL, and 600 mL dosed filters, respectively. BOD\(_5\) mass removal was found to range from 77% to 99% and TP removal from 71% to 98%. There was almost no NO\(_3\) increase in the saturated filters. NO\(_3\) increase from 150 mL dosed filters was significantly better than 300 mL dosed in the unsaturated filters. Micrographs confirm increased DNA and RNA presence in 150 mL dosed unsaturated filters compared to 600 mL dosed saturated filters, indicating more biological growth which increases nitrification. The existing treatment system on the potato farm lacks nitrification for N removal, but has the proper environment for denitrification, therefore, the addition of the unsaturated filter could help remove the N to the discharge limit. Although smaller doses provide better treatment, they require a larger surface area to support the same amount of flow compared to the larger dosages. The unsaturated filters performed better, although due to the nature of top-dosing, unsaturated filters need more infrastructures compared to saturated filters. This is especially important in winter when pumping systems would not function when water freezes. However, the large surface area required for treating at these dosages means unsaturated filters would require much more infrastructure and cost than a saturated filter. Therefore, the two options left are, lower cost without N removal, or higher cost with N removal. On-farm studies are necessary to explore the performance of a full-sized system and the cost analysis of building an unsaturated system.
Chapter 5 On-Farm Treatment of Potato Wash-Water: Supplementary Treatment Through Coagulation

ABSTRACT

Processing wash water from a potato farm storage facility was coagulated using 4 proprietary Nalco blend polymers, aluminium sulphate, and ferric chloride, to assess the removal efficiency of suspended solids and pH adjustment. Additionally, a cost-benefit analysis of coagulation treatments was performed in hopes of aiding potato farmers in choosing the best on-farm coagulant option. The Nalco polymers all had minimal effects in changing pH levels and high treatment efficiency. Ultrion 8185 had the best removal efficiency and cost-effectiveness at 50%, 75% and 90% solids removal. At 99% removal of solids, FeCl$_3$ was the most cost-effective, although high doses of coagulant may cause re-stabilization of solids, as well as, the low pH following FeCl$_3$ addition could hinder further removal of contaminants in the treatment system. Therefore, Ultrion 8185 was observed to be the most cost-effective option of solids removal from potato processing wash water.

Key words: Coagulation, alum, ferric chloride, polymer, potato wash-water, vegetable wash-water, horticulture, agriculture

5.1 Introduction

The management of agricultural crop wash water is a growing concern for farmers and in particular potato growers, since the processing industry is increasingly requiring the washing and pre-processing of crops. To pass governmental discharge guidelines, wash water often needs to be treated to an acceptable level (Ontario Ministry of Environment, 2011). This study sets out to test several coagulants as possible treatment options to reduce high total suspended solids (TSS) concentrations in potato wash water and find the most cost-effective coagulant. Agricultural treatment systems tend to be small and decentralized systems that can accommodate for large deviations in water quantity and quality (Crites and Tchobanoglous, 1998). Due to their organic source, these wastewaters are high in nutrients like nitrogen and phosphorus, and organic carbon. Potatoes usually contain 18% starch, 1% cellulose, and 81% water with dissolved organic matter (Hung et al., 2006; Gottormsen and Carlson, 1969). Therefore, we expect to see these contaminants in potato wash water, as well as, soils which are being washed off the potato
surface. Potatoes are harvested in the fall and growers store their potatoes for shipping/processing throughout fall, winter, and spring. This means that the majority of shipping is performed during cold months when biological treatment in outdoor treatment systems is least effective. As a result, chemical and physical treatment, including coagulation, are appealing possible means of treating this wastewater.

Coagulation is commonly used in combination with other wastewater treatment techniques to remove solids where sedimentation fails and allows for fast settling of solids which decreases the total suspended solids (TSS). Coagulation is the destabilization of colloidal particles that are smaller than 1 µm. Their small size hinders normal sedimentation because the forces which attract them to other particles are smaller than the repelling forces of their electrical charge. Colloidal particles can be clay, organic molecules, proteins, or carbohydrates. Agricultural wastewaters contain large concentrations of any or all of these, because the sources of the contaminants are soil and plant and animal matter. It is important to remove these not only to pass governmental guidelines, but also because solids can host pathogens, minimize treatment of other contaminants and be aesthetically displeasing. Some studies have already found positive results with coagulation of agricultural wastewaters. Coagulation of abbatoir wastewater and winery wastewater found TSS removal of 75-95% (Amunda et al., 2006; Braz et al., 2010). During coagulation, particles attract each other and grow in size, which makes them more suitable for sedimentation and filtration (Tchobanoglous and Schroeder, 1987). The four coagulation mechanisms which exist are: double-layer compression, charge neutralization, inter-particle bridging, and precipitate enmeshment. There are multiple types of coagulants, each with a specific range of functionality based on the wastewater pH. The coagulant may use one or more of those mechanisms depending on the particle type, molecular weight and pH (Huang and Shiu, 1996).

Coagulants are tested using “jar tests” in specific stirring machines (Tchobanoglous and Schroeder, 1987). This is a useful way to evaluate the efficacy of a coagulant for a given wastewater. More than one coagulant can be tested, in different coagulant concentrations. It is common practice when testing wastewater, including agricultural wastewaters, to alter the water pH to optimize the coagulant activity (Amunda and Alade, 2005; Braz et al., 2010; Wang et al., 2011). However, this necessitates pH always be altered during treatment, adding a step to the process. Given that pH of agricultural wastewater does not drastically fluctuate, it is better to
choose the coagulant based on the existing pH of wastewater. Additionally, coagulants may alter the pH of the wastewater following coagulation (Tchobanoglous and Schroeder, 1987; Song et al., 2004). This may cause problems with further wastewater treatment, where specific ranges are necessary for biological activity. Al-Mutairi et al. (2004) found the pH of wastewater dropped to as low as 3.6 using alum (aluminium sulphate) for coagulation of slaughterhouse wastewater. A pH this low could hinder future biological treatment which requires a pH between 6 and 7.

Alum has optimal functioning around pH 7, however, depending on the coagulation needed, a pH between 5-9 can be used. Ferric chloride has optimal coagulation around pH of 6.5, but can be used over a range of 5-7, which is the lowest functioning range of all the coagulants that will be tested. These metals are multivalent cations, which coagulate by precipitation and enmeshment. The bonding of OH⁻ from water to the metal creates a precipitant, which creates a sweeping floc that gathers colloidal particles (Tchobanoglous and Schroeder, 1987).

Polymers with high molecular weight have been used as coagulant aids and have several advantages: require lower dose, create less sludge, do not need pH adjustment, and produce flocs more resistant to breakage (Tyagi et al 2010). Polymer-blend coagulants are available from many companies, such as Nalco®, as proprietary, convenient, products for industrial use. Although they may be better performing than traditional coagulants, their prices are higher (per L) and it is unclear if they are more cost-efficient.

Research in coagulation of agricultural wastewaters has covered a broad range of agricultural sources (e.g. abbatoir, pulp mill, winery); however research has been limited to larger-scale operations with money and resources. There has been no research on potato wash-water coagulation with the possibility of small-scale, on-farm usage, while being convenient and economical for farmers. Currently available options for farmers are often costly and inconvenient and require labor and maintenance. Therefore, it is important to study not only the effectiveness of the coagulants, but also the relative, comparative cost-effectiveness of the coagulants at the existing pH of the wastewater, to determine the best coagulation option for potato farmers.

### 5.2 Materials and Methods

#### 5.2.1 Characterization of Source Water

Potato wash-water was collected from a potato storage facility in Alliston, ON. Prior to shipping, clean water from a groundwater well was used for washing and transporting potatoes (within the
facility). The wash-water was pumped out of the facility at the end of each shipping day. Water was collected from the potato storage facility immediately after shipping had finished. Wash-water quality varied due to the quantity and type of potato shipped, type of soil potatoes were grown in, and amount of rot. The colloidal particles in the wash-water were found to be too small for removal through sedimentation. The wash-water passed through a passive land-based treatment system located on the farm, containing a sedimentation basin, one aeration pond, and two wetland cells. The sedimentation basin removed >90% of the TSS concentration; however, colloidal particles persisted. The outflow from the entire system, with several months of retention time, was not able to remove the TSS to an acceptable level.

Raw wash-water was analyzed for total nitrogen (TN), total phosphorus (TP), TSS, 5-day biochemical oxygen demand (BOD₅), and pH. Due to its high concentration, ease of analysis, and high correlation with BOD₅ removal, TSS was chosen as the focus contaminant in this study.

5.2.2 Coagulants and Flocculation Procedure

The coagulation of potato wash-water was tested using alum (Acros Organics, Aluminium Sulfate octadecahydrate extra pure), ferric chloride (MP, Ferric Chloride hexahydrate), both obtained from Fisher Scientific (Cat. No. AC400590025 and 15350080), and four Nalco® brand polymer blend coagulants (Nalco Holding Company, Naperville, IL, USA). The polymer blends used were recommended by the manufacturer: Nalco 8158 (8158), Ultrion 8186 (8186), Ultrion 8187 (8187), and Ultrion 8185 (8185). Prices of the products were supplied by Nalco©, and a local chemical redistributor Alphachem limited.

Preliminary tests were performed in order to find the best range of concentrations for testing and length of time for settling. Experimental methods were standardized for all coagulants. Following manufacturer recommendation, the polymer blend coagulants were first prepared as a 1% solution of demineralized water, immediately prior to each experiment. Jar testing was performed with 300 mL of the wash-water poured into beakers and slowly mixed using a glass stirring rod for ~10 seconds following coagulant addition to facilitate flocculation. Following flocculation, the beakers were left in a closed fume hood at room temperature and allowed to settle overnight. The following day, 40 mL of the supernatant was extracted using a 10 mL pipette at four locations at different depths, leaving one quarter of the liquid at the bottom
undisturbed so as not to disturb the accumulated sediments. The samples of supernatant were bottled and kept at 4-7 °C prior to being analyzed for TSS within 14 d.

5.2.3 Experimental Design

Potato wash-water was tested in triplicate at 7 or 8 different doses, including the control, of each coagulant (a total of 144 samples; Table 5.1). The control consisted raw wash-water that had no coagulant, although otherwise methods were the same. Each coagulant was tested on a different day, over the course of two weeks. Regression analysis was done on means of all replicates for each dose on TSS concentrations and pH.

*Table 5.1 Number of samples of each coagulant at different doses. Concentrations in samples resulting from these dosages ranged from 0.01 – 0.5 mL L⁻¹ for the Nalco polymers, 25 – 1540 mg L⁻¹ for ferric chloride (FeCl₃), and 1.5 – 90 mg L⁻¹ alum.*

<table>
<thead>
<tr>
<th>mL of 1% solution in 300 mL</th>
<th>Polymers</th>
<th>Non-Polymers</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>8158</td>
<td>8185</td>
</tr>
<tr>
<td>Control</td>
<td>3</td>
<td>3</td>
</tr>
<tr>
<td>0.25</td>
<td>3</td>
<td>3</td>
</tr>
<tr>
<td>0.5</td>
<td>3</td>
<td>3</td>
</tr>
<tr>
<td>1</td>
<td>3</td>
<td>3</td>
</tr>
<tr>
<td>2</td>
<td>3</td>
<td>3</td>
</tr>
<tr>
<td>5</td>
<td>3</td>
<td>3</td>
</tr>
<tr>
<td>8</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>10</td>
<td>3</td>
<td>3</td>
</tr>
<tr>
<td>15</td>
<td>3</td>
<td>3</td>
</tr>
</tbody>
</table>

5.2.4 Laboratory Analysis

5.2.4.1 Total Suspended Solids

TSS was measured using modified Standard Method 2540D (APHA, 2005). Prior to analysis, the filters (1.5 µm glass microfiber, Whatman, Buckinghamshire, UK) were washed using demineralized water and dried at 105±3 °C for at least 4 h in an Isotemp™ 500 series oven (Fisher Scientific, Ottawa, ON). Filters were cooled in a desiccant chamber for 15 min prior to
being weighed using a Sartorius analytical balance CPA225D (Sartorius AG, Goettingen, Germany) scale to find the initial filter weight. Water samples were filtered using a Buchner funnel and vacuum line. The amount of sample filtered was dependent upon the visible turbidity of the sample (i.e., 10 mL of untreated samples were filtered, while up to 30 mL of less turbid samples were filtered). The filters were then dried again at 105±3 °C for at least 4 h and cooled in a desiccant chamber for 15 min prior to weighing to determine the final sample weight. The concentration of the TSS in the sample was found using the following equation:

\[
TSS = \frac{(Final \ weight - Initial \ filter \ weight)}{Volume \ of \ sample} \times 1000
\]

where TSS = TSS concentration of the sample (mg L\(^{-1}\)); final weight = sample + filter after drying(g); initial filter weight = filter weight after drying (g); volume of sample = volume of the sample filtered (L).

5.2.4.2 pH

The pH of the wash-water was not adjusted in this study, in order to study the coagulant behavior at the natural pH of the wash-water. The pH was measured and recorded using a calibrated YSI 556 multimeter (YSI556, YSI Inc., Yellow Spring, Ohio, USA) prior to each set of experiments and before bottling each sample. Calibration was done using the manufacturer recommendations. The pH measurements were taken of the raw wash-water and the supernatant after coagulation, where pH measurements of the supernatant were taken following TSS sampling. The multimeter was allowed to stabilize for approximately 5 minutes before the measurement was recorded. Prior to each experiment, a subsample homogenized wash-water of this was poured into the multimeter probe attachable cup to take measurements. The next day the supernatant was carefully poured into the YSI 556 multimeter attachable cup and allowed to stabilize (~5min).

5.2.5 Microscopy

Microscopy images of the floc formations were conducted for trials using coagulant 8185 at three different doses and for the control. Doses were calculated based on estimated 50, 75, and 90% reductions of TSS as determined in coagulation tests. Once coagulants were added to the wash-water and allowed to mix, pipettes were used to collect the flocs before they were able to settle. Pipette tips were cut to create a wider opening and prevent floc damage. The flocs were
then transferred to a microscopy slide. Samples were placed in depression slides. Cover slips were not used due to condensation on the inside of the cover slip. All samples had sedimentation in the center of the depression, although there was a marked difference in appearance between the coagulated solids and settled solids (no coagulant). The coagulated flocs appeared ‘fluffy’ in comparison to the more densely settled solids in the control. Images were taken using an Axioplant 2 imaging light microscope and the associated Axiovision computer program (Carl Zeiss AG, Jena, Germany). Images were taken at 5× magnification with no more than 10 stacks to create a Z-stack image.

A coagulation video was filmed on a dissection microscope, Olympus SZX16 (Olympus©, Tokyo, Japan), equipped with a Luminera® Infinity 3 digital camera (Luminera Corp., Ottawa, Canada). The video was captured using Infinity Capture Software 6.1 and edited using Adobe Premiere Pro. For ease of viewing, the addition of coagulant 8185 was chosen to have 99% TSS removal. The coagulation procedure followed the experimental protocol except that a 5 mL volume of wash-water was used.

5.2.6 Statistical Analysis

The TSS removal efficiency was calculated with the following equation:

\[
\text{\% Removal} = \frac{C_i - C_f}{C_i} \times 100
\]  

[5.2]

where \% Removal is the percentage of TSS concentration removal due to coagulation; \(C_i\) is initial concentration of the wash-water (mg L\(^{-1}\)), \(C_f\) is final TSS concentration of supernatant following treatment(mg L\(^{-1}\)).

Dose-response curves were created for each coagulant using regression analysis in SigmaPlot 12.0 (SigmaPlot, San Jose, CA, USA). Best fits were selected from several options (linear, linear-log, and polynomial) and selected based on \(r^2\), p-value, and residuals. The curves had the following equation:

\[
y = b_0 + b_1 \times \ln(x)
\]  

[5.3]

Where \(b_0\) = natural log transformed intercept; \(b_1\) = slope of the logarithmic equation.

Using the regression equation we found the estimated doses needed for 50, 70, and 90% TSS removal. We defined the effective dose, ED50, as the point 50% removal of the contaminant.
from the wastewater is achieved. From these values we found the total amount of coagulant needed per year and price per year. Prices were obtained from Nalco and Alphachem, a local redistributor of chemicals.

\[
Coagulant_{yr} = \frac{Coagulant_{exp} \times 3\,000\,000 \times .01}{.3}
\]

where \( Coagulant_{yr} \) = amount of coagulant needed for a year of shipping (L y\(^{-1}\) or kg y\(^{-1}\)); \( Coagulant_{exp} \) = dose of coagulant in 1% solution (L300 mL\(^{-1}\)), found with the regression equation, needed to achieve the wanted TSS removal (mg L\(^{-1}\)); 3 000 000 = amount of wash-water produced in a year on a potato farm (calculated as an average of two years of observation; L); .01 = accounting for 1% solution of coagulant doses; .3 = accounting for 300 mL wash-water volumes used during experimental trials (L).

5.3 Results and Discussion

5.3.1 Characteristics of Raw Potato Wash-Water

Potato wash-water quality was monitored for two years at the Sunrise Potato storage facility in Alliston, ON as part of a larger study (Table 5.2). The mean TSS concentration of the raw wash-water was found to be 5400± 6098 mg L\(^{-1}\) for Y1 and 3600 ± 2829 mg L\(^{-1}\) for Y2. The TSS values show high variability, which had been observed throughout the larger study as an inherent part of the potato wash-water.

The high BOD\(_5\) and TSS concentrations in the raw wash-water suggest a significant presence of organic matter (Table 5.2). Dissolved organic molecules can be large enough to be considered colloidal particles and will be coagulated along with suspended solids. Micrograph images show the control wash-water did not contain lot of particles that exceeded 50 µm in size, although it had many colloidal particles (<1 µm) suspended in the supernatant (Figure 5.2). The composition of the raw wastewater will alter the coagulation. For example, hydrophobic compounds can be more easily destabilized than hydrophilic compounds (Samrani et al., 2004) and polysaccharides, proteins, and amino-sugars are poorly coagulated by alum (Franceschi et al., 2002). For this reason, each wastewater must be first tested for its optimal dose and coagulant before usage.

The Ontario Ministry of the Environment and Climate Change (MOECC) recommends a discharge concentration of <25 mg L\(^{-1}\) TSS, as stated in the Provincial water quality guidelines.
(PWQG) (Ontario Ministry 1994). This means a 99% reduction in TSS is needed. The existing treatment system at the potato farm reduces TSS by 91% in the primary treatment cell, with an overall reduction of 97% in TSS within the entire system.

Table 5.2 Average contaminant concentration present in raw wash-water from observations of potato storage facility in Y1 and Y2 as part of a larger study (see Chapter 3).

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Y1</th>
<th>Y2</th>
</tr>
</thead>
<tbody>
<tr>
<td>TSS</td>
<td>5400</td>
<td>3600</td>
</tr>
<tr>
<td>BOD₅</td>
<td>1200</td>
<td>716</td>
</tr>
<tr>
<td>TN</td>
<td>200</td>
<td>160</td>
</tr>
<tr>
<td>TP</td>
<td>36</td>
<td>42</td>
</tr>
</tbody>
</table>

5.3.2 TSS Removal
TSS removal from all doses and coagulants ranged from -154% to 99.8% removal. Negative removals occurred at very low coagulant doses and were attributed to the high variability in the raw wastewater. The high variability was also seen in the control wash-water during the experiment, which had sedimentation without coagulation, with an average TSS of 721±332 mg L⁻¹. The best coagulant was Nalco 8185, with a median removal of 87% TSS, with a significantly higher removal than any other coagulant. Ultrion 8186 and 8187 performed similarly (49 and 41% TSS removal, respectively); however, 8186 performed better at higher doses, while 8187 performed better at lower ones. These coagulants were not found to be significantly different from each other. Coagulant 8158 had significantly lower performance than the other Nalco coagulants with a median removal of only 33%. Nalco coagulants cannot be easily compared to alum and FeCl₃ TSS removals, since the Nalco coagulants were used on a 1% (v v⁻¹) dilution, while alum and FeCl₃ had 2% (w v⁻¹) or 1.2% (vv⁻¹) and 3% (vv⁻¹) or 1.7% (vv⁻¹) dilutions, respectively. FeCl₃ performed better than alum due to its smaller dilution (54.9 ± 38.9% vs. 57.3 ± 35.8%).

The removal of TSS using coagulation was only sufficient at doses of 0.01 mL L⁻¹ (1 mL of 1% coagulant solution/300 mL) or higher. All other doses were not high enough to produce supernatant TSS concentrations that could meet the governmental recommended discharge limit.
(<25 mg L\(^{-1}\)). Coagulation following this sedimentation cell could have greater overall removal of TSS and reach the limits. Removing solids early in the treatment system could allow for the rest of system to remove dissolved contaminants or reduce the system size. Further on-farm testing is necessary to find the best way to dose and mix wash-water.

All coagulants had a dose dependent response. Micrograph images (Figure 5.2) of coagulated wash-water using coagulant 8185 show that while the coagulated samples all appeared to have a clear supernatant, images of 50%, 75%, and 90% TSS removal had fewer and larger flocs (>100 \(\mu\)m) at higher doses, as well as fewer residual solids or what could have been 'tiny flocs' surrounding the center floc (<100 \(\mu\)m). Dosage responses were all found to be linear-log, therefore linear regressions were performed on log-transformed dose data. The untransformed curves had two sections, one of rapid growth, where small additions of coagulant made a big improvement in TSS removal and a section of stabilization where additional coagulant had little effect. Therefore, as the coagulant dose increases particle attraction and floc size also increase. The larger flocs are created from the attraction of smaller flocs. (This can be seen in the video segment created by a dissecting microscope with camera attachment.) This behavior suggests that smaller coagulant doses may need more time for sedimentation because of the smaller size of flocs. Most coagulants resulted in little or no treatment at the lowest dose when standard deviations were taken into consideration (< 10% TSS removal). Coagulants 8185 and 8187 were the exceptions, where TSS removal was 61.0 ± 13.2% and 31.3 ± 6.3%, respectively, at the lowest concentration (0.01 mL L\(^{-1}\)).

The stabilization phase denotes that the majority of coagulation process has been completed and the further addition of coagulants will result in insignificant changes in TSS removal. The point where the curve changes from rapid growth to stabilization is the optimum coagulation concentration (OCC) (El Samarani et al., 2004). The OCC of most coagulants lies at 50-60% TSS removal, with the exception of 8158 which was closer to 40% and 8185 which was closer to 80%. At the OCC TSS is removed at the optimal coagulant dose : TSS removal ratio. El Samrani et al. (2004) looked at FeCl\(_3\) coagulation of municipal sewage water. They found a similar logarithmic curve as found in this study; however, they continued testing at even higher concentrations and found a re-suspension of solids when the dose was increased past the stabilization of the curve. As a result, the turbidity increased drastically, to values even higher than the raw sewage; demonstrating that over-dosing hinders the coagulation process and
ultimately TSS removal. The coagulant doses used for this study did not demonstrate any re-suspension of solids.

The regression analysis of all coagulants had higher $b_0$ had smaller $b_1$ and vice versa. Coagulant 8185 had the highest $b_0$ of 77.3 (% TSS removal) and the smallest $b_1$ of 9.8 ($r^2 = 0.96, p < 0.001$; Table 5.3). Since the x-axis is natural log-transformed, when the concentration is 0.1 mL L$^{-1}$ the $x = 0$. Therefore, the $b_0$ is the % TSS removal at concentration 0.1 mL L$^{-1}$ (dose 1 mL of 1% solution in 300 mL sample). The $b_1$ indicates the amount of %TSS reduction achieved for a given coagulant dose increase. Coagulant 8185 had the best treatment throughout all doses and the highest $b_0$ and the smallest $b_1$. Therefore coagulant 8185 had the lowest increase TSS % removal for a given concentration increase. Coagulant 8158 had the largest $b_1$, 40.9, although the lowest $b_0$ of -6.7(%TSS reduction), and the lowest reduction throughout all concentrations ($p<0.001$). Due to the high $b_1$, we would expect that eventually 8158 would have greater removal than 8185; however, due to the logarithmic nature of the x-axis the higher concentrations have a smaller change in % TSS reduction. Therefore, 8158 does not match the removal of 8185. Coagulant 8187 had the next highest $b_0$ of 43.1 (% TSS removal) and a $b_1$ of 18.1, which was the second lowest ($p=0.002$). Coagulant 8186 had similar TSS removal to 8187, although the $b_0$ was 33.3 (% TSS removal) and the $b_1$ was 25.9 ($p<0.001$). Alum (2% solution w/v) had a $b_0$ of 35.6 (%TSS removal) and $b_1$ of 25.8 and eventually passed 8186 and 8187, having higher removal at concentrations higher than 0.5 mL L$^{-1}$ (dose of 5 mL of 1% solution in 300 mL), this is due to the higher $b_1$ ($p<0.001$). FeCl$_3$ (3% solution w v$^{-1}$) behaved similarly, having a $b_0$ of 40.8 (% TSS removal) and $b_1$ of 25.0 ($p<0.001$). It had higher removals than 8186 and 8187 at doses higher than 5 mL and removals higher than Alum at doses higher than 10 mL.

Other agricultural studies have found similar removal of TSS as our study (Table 5.3). Braz et al. (2010) found 83% removal of TSS from alum (20 mL of 5% wv$^{-1}$ solution), and 76.8% removal from FeCl$_3$ (10 mL of 5% wv$^{-1}$ solution) in winery wastewater. Karim and Sistrunk (1985) found 91% removal of TSS from abrasive-peeled potato processing water when treated with ferric chloride at a pH of 7. They found their best treatment was when they used a mixture of FeCl$_3$ and a polymer, which increased the removal to 99%. Other studies suggest that mixtures of coagulants and polymers had superior removal. Although results specific to these Nalco coagulants were not found in the literature, we can compare our results to those found by polymer-mixed coagulants. Tyagi et al. (2010) found that the addition of a polymer produced
consistently better TSS removal over all doses tested. They found an additional 26% removal when a polymer was added to alum and 31% when added to FeCl₃. Our study found similar results, with >30% higher mean TSS removal from 8185, a polymer blend, compared to using alum and FeCl₃ alone.

Table 5.3 Results of regression analysis of TSS % reduction \((Y)\) using doses of 1% solution \((\times)\) of coagulant Ultrion 8158, 8158, 8186, 8187, 2% alum, and 3% ferric chloride to treatment potato wash-water collected from the flume before discharge into treatment wetland.

<table>
<thead>
<tr>
<th>Coagulant</th>
<th>Equation</th>
<th>(r^2)</th>
<th>p-value</th>
<th>(b_0)</th>
<th>(b_1)</th>
</tr>
</thead>
<tbody>
<tr>
<td>8158</td>
<td>(Y = -22.5 + (287 \times \ln(\times)))</td>
<td>0.98</td>
<td>&lt;0.001</td>
<td>-22.5</td>
<td>287</td>
</tr>
<tr>
<td>8185</td>
<td>(Y = 73.9 + (66.1 \times \ln(\times)))</td>
<td>0.96</td>
<td>&lt;0.001</td>
<td>73.9</td>
<td>66.1</td>
</tr>
<tr>
<td>8186</td>
<td>(Y = 22.1 + (189 \times \ln(\times)))</td>
<td>0.95</td>
<td>&lt;0.001</td>
<td>22.1</td>
<td>189</td>
</tr>
<tr>
<td>8187</td>
<td>(Y = 32.2 + (151 \times \ln(\times)))</td>
<td>0.88</td>
<td>0.002</td>
<td>32.2</td>
<td>151</td>
</tr>
<tr>
<td>Alum</td>
<td>(Y = 25.4 + (183 \times \ln(\times)))</td>
<td>0.94</td>
<td>&lt;0.001</td>
<td>25.4</td>
<td>183</td>
</tr>
<tr>
<td>FeCl</td>
<td>(Y = 23.2 + (265 \times \ln(\times)))</td>
<td>0.95</td>
<td>&lt;0.001</td>
<td>23.2</td>
<td>265</td>
</tr>
</tbody>
</table>
**Table 5.4 Removal efficiency of TSS from literature.**

<table>
<thead>
<tr>
<th>Study</th>
<th>Wastewater</th>
<th>Coagulant</th>
<th>Concentration (mg L⁻¹)</th>
<th>TSS % Removal</th>
</tr>
</thead>
<tbody>
<tr>
<td>Karim and Sistrunk 1985</td>
<td>potato processing</td>
<td>FeCl₃</td>
<td>150</td>
<td>91</td>
</tr>
<tr>
<td>Karim and Sistrunk 1985</td>
<td>potato processing</td>
<td>FeCl₃ + polymer</td>
<td>150 + 20</td>
<td>99</td>
</tr>
<tr>
<td>Tyagi et al. 2010</td>
<td>municipal</td>
<td>FeCl₃</td>
<td>300</td>
<td>80</td>
</tr>
<tr>
<td>Tyagi et al. 2011</td>
<td>municipal</td>
<td>FeCl₃ + polymer</td>
<td>300 + 0.2</td>
<td>99</td>
</tr>
<tr>
<td>Tyagi et al. 2012</td>
<td>municipal</td>
<td>Alum</td>
<td>300</td>
<td>78</td>
</tr>
<tr>
<td>Tyagi et al. 2013</td>
<td>municipal</td>
<td>Alum + polymer</td>
<td>300 + 0.2</td>
<td>98</td>
</tr>
<tr>
<td>Amunda and Alade 2006</td>
<td>abattoir</td>
<td>Alum</td>
<td>2000</td>
<td>70</td>
</tr>
<tr>
<td>Amunda and Alade 2007</td>
<td>abattoir</td>
<td>Alum + polymer</td>
<td>750 + 10</td>
<td>95</td>
</tr>
<tr>
<td>Guida et al. 2007</td>
<td>municipal</td>
<td>Alum</td>
<td>150</td>
<td>&gt;75</td>
</tr>
<tr>
<td>Al-Mutairi et al. 2004</td>
<td>slaughterhouse</td>
<td>Alum</td>
<td>100-1000</td>
<td>98-99</td>
</tr>
<tr>
<td>Al-Mutairi et al. 2004</td>
<td>slaughterhouse</td>
<td>Praestol® polymer</td>
<td>60-90</td>
<td>95-96</td>
</tr>
</tbody>
</table>
Figure 5.1 Reduction of TSS using 1% solution of coagulant Ultrion 8185, 8158, 8186, 8187, 2% alum, and 3% ferric chloride to treatment potato wash-water collected from the flume before discharge into treatment wetland.
Figure 5.2 Micrographs\(^4\) of flocs created with Ultrion 8185 coagulant at 5 × magnification: 1 = control; 2 = 50% TSS removal; 3 = 75% TSS removal; and 4 = 90% TSS removal.

5.3.3 Cost Effectiveness

A comparative cost-effectiveness was calculated using TSS removal efficiency and prices supplied by local chemical distributors. The Nalco coagulant prices were $15.7, $11.9, $14.2, and $12.3 L\(^{-1}\) for 8185, 8158, 8186, and 8187, respectively. Alum and FeCl\(_3\) had quotes from

\(^4\)For full image follow this link: https://drive.google.com/file/d/0B9xM7DxOQ6l9ZW1PbW5JaVpwOTA/view?usp=sharing
Alphachem at $2.86 \text{ L}^{-1}$ of a 50% alum solution and $4.62 \text{ L}^{-1}$ of a 40% FeCl$_3$ solution (Table 5.5).

While the Nalco 8185 was the most expensive (per L) of all the coagulants tested, it had the best value overall due to its high efficiency. At ED50, only 10 L $y^{-1}$ of 8185 is needed, while the other coagulants need 150-600 L to achieve the same removal. The price to achieve 50% removal was found to be $150 \text{ y}^{-1}$ for 8185 and $1,000 - 4,700 \text{ y}^{-1}$ for all the other coagulants, which is at least an 85% price increase. To achieve ED 75, 77 L $y^{-1}$ of 8185 is required at $1200 \text{ y}^{-1}$. The other coagulants required a minimum of 88% more volume resulting in 50% more cost. At ED90, coagulant 8185 was still the most effective on a dose basis, and most cost-effective. Coagulant 8185 required 271 L $y^{-1}$, while the other coagulant required at minimum 65% more volume and would cost $4,300 y^{-1}$. At ED90, coagulant 8186 was the next most efficient with 790 L $y^{-1}$ required, while FeCl$_3$ was the next cheapest option at $5,400 \text{ y}^{-1}$. Due to the nature of the logarithmic treatment, however, it takes much more coagulant 8185 to treat past 90% removal. At 99% removal, FeCl$_3$ becomes a cheaper option priced at $7,700 \text{ y}^{-1}$, while 8185 was priced at $9,000 \text{ y}^{-1}$. The quantity of FeCl$_3$ needed is still much higher, but the lower price makes it more cost effective. Tyagi et al. (2010) found that the addition of a polymer decreased the price of coagulation. They found that to reach 75% BOD$_5$ removal was 20% cheaper when a polymer was added to FeCl$_3$. The polymer aided in settling due to its additional weight and allowed for an extra 20% TSS removal.

At higher doses, 8185 and FeCl$_3$ seem to be the two most cost-effective coagulants. Due to the dangers of over-dosing (which hinders coagulation), striving for a removal of 99% could be unwise, especially with the highly variable nature of the wash-water. Additionally, dosing beyond the OCC becomes less efficient, that is, using a lot more coagulant for a small change in TSS removal. For FeCl$_3$, the OCC is around 50-60% removal, which is lower than 8185 around 80%. Dosing up to the OCC guarantees the best removal for the coagulant quantity, any further increase in the dose would disproportionally increase the price. At ED75, the most cost-effective coagulant is 8185 at a volume requirement of 77 L $y^{-1}$ and cost of $1200 \text{ y}^{-1}$. Although this does not pass the governmental TSS discharge limit, the remaining treatment system may be able to treat to the appropriate concentration (<25mg L$^{-1}$ TSS). More studies are needed to test the applicability of the coagulant on-farm.
Table 5.5 The doses and cost per year to remove 50% (ED50), 75% (ED75), and 90% (ED90) of the TSS in the wastewater using the coagulants 8185, 8186, 8187, 8158, alum (99% purity), and ferric chloride (99% purity). All calculations based on treating 3,000,000L of wash-water annually.

<table>
<thead>
<tr>
<th>Coagulant</th>
<th>r²</th>
<th>Price ($ L⁻¹)</th>
<th>ED50 Quantity (L y⁻¹)</th>
<th>Cost per Y ($)</th>
<th>ED75 Quantity (L y⁻¹)</th>
<th>Cost per Y ($)</th>
<th>ED90 Quantity (L y⁻¹)</th>
<th>Cost per Y ($)</th>
</tr>
</thead>
<tbody>
<tr>
<td>8158</td>
<td>0.98</td>
<td>11.9</td>
<td>400</td>
<td>4,766</td>
<td>737</td>
<td>8,779</td>
<td>1,063</td>
<td>12,664</td>
</tr>
<tr>
<td>8185</td>
<td>0.99</td>
<td>15.7</td>
<td>10</td>
<td>149</td>
<td>77</td>
<td>1,213</td>
<td>271</td>
<td>4,265</td>
</tr>
<tr>
<td>8186</td>
<td>0.94</td>
<td>14.2</td>
<td>182</td>
<td>2,586</td>
<td>454</td>
<td>6,467</td>
<td>788</td>
<td>11,208</td>
</tr>
<tr>
<td>8187</td>
<td>0.82</td>
<td>12.3</td>
<td>154</td>
<td>1,884</td>
<td>679</td>
<td>8,335</td>
<td>1,658</td>
<td>20,344</td>
</tr>
<tr>
<td>Alum</td>
<td>0.93</td>
<td>2.9</td>
<td>620</td>
<td>1,775</td>
<td>1,443</td>
<td>4,126</td>
<td>2,393</td>
<td>6,844</td>
</tr>
<tr>
<td>FeCl</td>
<td>0.94</td>
<td>4.6</td>
<td>237</td>
<td>1,095</td>
<td>642</td>
<td>2,971</td>
<td>1,169</td>
<td>5,408</td>
</tr>
</tbody>
</table>

5.3.4 pH Effects

Regression analysis of coagulant pH had the largest change due to dose increase in alum and FeCl₃. There was a strong correlation between the concentration of alum and pH, while FeCl₃ had a moderately strong correlation. Both had a similar intercept of 7.87 and 7.84 (pH) and similar slope of 8.0 and 5.45, respectively. In comparison, all the Nalco products had a slope between -1 and 1. This means that the linear curve was mostly horizontal (Figure 5.3). They also all had a higher intercept between 7.82 and 8.01 (pH), which is closer to the untreated control pH. However, each Nalco coagulant had different correlation strength. Coagulant 8185 had a strong correlation ($r^2 = 0.91; p < 0.001$), 8158 a moderate correlation ($r^2 = 0.53; p = 0.06$), 8186 a moderately strong correlation ($r^2 = 0.79; p = 0.01$), and 8187 had a slight to no correlation ($r^2 = 0.09; p = 0.52$).

The pH was found to be a significant variable when all the coagulants and doses were compared ($p < 0.0001$). The pH of the supernatant was mostly unaltered by the Nalco coagulants. It increased from 7.8 (control) to 8.0 in the highest doses. Coagulant 8185 was found to be statistically different from the other Nalco coagulants, with a mean of 8.0 (vs. 7.9). In comparison, alum and ferric chloride decreased the pH substantially from 7.8 to 4.3 and 5.3,
respectively, in the highest doses. Each was statistically different, with means of 6.6 (alum), and 7.1 (FeCl₃). The alum and FeCl₃ correlation of TSS removal and pH was a moderately strong negative linear relationship ($r^2 = 0.74$ in both). The decrease in pH is caused by alum and FeCl₃ hydrolysis. The metals (iron and aluminum) capture OH⁻ and create polymeric species which allow for the coagulation and solid removal to occur (El Samrani et al., 2004). For each hydroxyl group acquired, however, a hydrogen ion is released and therefore lowering the pH (Tyagi et al., 2010):

$$2Al^{3+} + 6H_2O \rightarrow 2Al(OH)_3 + 6H^+ \quad [5.5]$$

$$2Fe^{3+} + 6H_2O \rightarrow 2Fe(OH)_3 + 6H^+ \quad [5.6]$$

Lowering the pH during coagulation could hinder contaminant removal further along in the treatment process. This would be especially relevant in biological treatment (e.g. nitrification requires 7.5 – 8.5 and denitrification requires 6.5 – 7.5) which requires specific pH values for microorganisms to grow (Barnes and Bliss 1983). This means that frequent pH testing and additional chemicals would be necessary to bring the pH back to a biologically neutral level when using alum and FeCl₃. This would add additional cost and labor, complicate the process, and create more areas for possible errors. This would be especially hard for farmers which may not have the man-power and funding.

Table 5.6 Results of regression analysis of pH using 1% solution of coagulant Ultrion 8185, 8158, 8186, 8187, 2% alum, and 3% ferric chloride to treatment potato wash-water collected from the flume before discharge into treatment wetland.

<table>
<thead>
<tr>
<th>Coagulant</th>
<th>Equation</th>
<th>pH reduction</th>
<th>r²</th>
<th>p-value</th>
<th>intercept</th>
<th>slope</th>
</tr>
</thead>
<tbody>
<tr>
<td>8158</td>
<td>$Y = 7.92 - (0.288 \times Ln(x))$</td>
<td>0.53</td>
<td>0.06</td>
<td>7.92</td>
<td>0.29</td>
<td></td>
</tr>
<tr>
<td>8185</td>
<td>$Y = 8.01 - (0.241 \times Ln(x))$</td>
<td>0.91</td>
<td>&lt;0.001</td>
<td>8.01</td>
<td>0.24</td>
<td></td>
</tr>
<tr>
<td>8186</td>
<td>$Y = 7.82 + (0.413 \times Ln(x))$</td>
<td>0.79</td>
<td>0.01</td>
<td>7.82</td>
<td>0.41</td>
<td></td>
</tr>
<tr>
<td>8187</td>
<td>$Y = 7.94 - (0.08 \times Ln(x))$</td>
<td>0.09</td>
<td>0.52</td>
<td>7.94</td>
<td>0.08</td>
<td></td>
</tr>
<tr>
<td>Alum</td>
<td>$Y = 7.87 - (8.00 \times Ln(x))$</td>
<td>0.83</td>
<td>0.00</td>
<td>7.87</td>
<td>8.00</td>
<td></td>
</tr>
<tr>
<td>FeCl</td>
<td>$Y = 7.84 - (4.54\times Ln(x))$</td>
<td>0.66</td>
<td>0.03</td>
<td>7.84</td>
<td>4.54</td>
<td></td>
</tr>
</tbody>
</table>
Figure 5.3 Reduction of pH using 1% solution of coagulant Ultrion 8185, 8158, 8186, 8187, 2% alum, and 3% ferric chloride to treatment potato wash-water collected from the flume before discharge into treatment wetland.
5.3.5 Applicability to Treatment System

In order to assess the applicability of our results, the best performing coagulant, 8185, was used to coagulate effluent collected from the outflow from the primary sedimentation cell of the treatment system. The effluent from the sedimentation cell had a TSS of 300 mg L\(^{-1}\), which is less than half the TSS concentration found in the raw water. The effect of coagulation on TSS removal was lower at all doses in this wastewater than in the flume water used in the main study. At the highest concentrations 1.5 mL L\(^{-1}\) of coagulant the removal was only 80\% of the TSS, leaving 60 mg L\(^{-1}\) of TSS. The results from the flume water coagulation found highest removal of 99\%, leaving 5 mg L\(^{-1}\) of TSS. The regression analysis found an intercept of 27 (% TSS removal) and a slope of 20.5. This equation found a lower intercept and higher slope than the 8185 equation for the flume water coagulation. The cost-effectiveness analysis found that ED50 would cost almost $5,000 y\(^{-1}\), ED75 $16,000 y\(^{-1}\), and ED99 $33,500 y\(^{-1}\). Using coagulant 8185 on the water following the sedimentation pond would be more costly and less effective than using any of the other coagulants for the flume water.

Since there are many processes that take place within the primary treatment system, the change in the wastewater may not only be a decrease in TSS, but also change in the types of solids and their polarities and reactivity. Therefore, we cannot conclude that the lower TSS concentration due to sedimentation was the only cause for lower TSS removal from coagulation. Other possible effects on coagulation may be temperature, biological activity, dilution due to precipitation, etc.

These results exemplify the need to test every wastewater prior to usage. This also shows that the ED pricing calculated in this study are important in a comparative manner; however, should not be used as a guideline replacing additional testing.

5.4 Conclusion

The best performing and most cost-effective coagulant was 8185 at ED50, ED75, and ED90. FeCl\(_3\) was most cost-effective at ED99, although it required more coagulant than 8185 to achieve the same treatment. At 99\% TSS removal (ED99) FeCl\(_3\) required 1675 L y\(^{-1}\) and cost $7750 y\(^{-1}\). Due to the nature of the logarithmic treatment of the coagulants, the optimal treatment was between 50-60\% in most coagulants with the exception of 8185 which had an optimal treatment at 80\%. Therefore, at its peak performance coagulant 8185 can reach higher TSS removal, at the lowest cost, than any other coagulant. At 75\% TSS removal (ED75), 8185 required only 77 L y\(^{-1}\)
and cost $1200 y^{-1}. A smaller volume of liquid is easier for farmers to work with, making treatment more convenient. Although at this removal governmental TSS guidelines were not met, as part of another treatment system, additional on-farm treatment should be able to achieve the required TSS outflow concentrations. Coagulation using 8185 on wash-water following sedimentation had lower treatment and higher treatment cost to achieve ED50, ED 75, and ED90. This suggests that each wastewater should be tested and prices are only applicable in a comparative manner. Analysis of pH found that the Nalco polymers did not have any effect on pH, however alum and FeCl₃ lowered the pH below 6 at a dose of 10 mL and higher. Changing pH will make other treatments more complicated by requiring pH adjustment. Therefore, the Nalco polymers are easier and more convenient for farmers. Future research should evaluate the performance of using more than one type of coagulant as well as conducting on-farm trials.
Chapter 6 Conclusion

6.1 Conclusion

Due to lack of guidance and information, farmers washing and processing crops on-farm are left with large volumes of wash-water that are either expensively or poorly managed. Finding ways to help farmers starts with recognizing how water is used and how much water is used. The whole storage facility used 5500 m$^3$ of water in one year (February 2013-February 2014), of which 44% was used for shipping potatoes. Throughout the first shipping season a total of 4044 m$^3$ of water was used, while the second season used only 2345 m$^3$. In Y2 water usage was statistically lower in FM_pump and FM_tank, although the water to FM_flume remained the same. Since the farm had different piping, it was easy to separate the water used for each process, however, this may not be the same for every farm. There is need to assess other potato and other root vegetable farms in order to better understand how water is being used and ways other farmers can conserve water. Additionally, promoting the usage of finger roller to remove more solids prior to water entry would benefit all washing/storage facilities by decreasing the amount of contaminants entering the wash-water.

When the water usage was calculated as the difference between the intake and the outflow, it was 1829 m$^3$ in Y1 and an increase to 2292 m$^3$ in Y2. Due to increased leaching into the soil in the second year, there was less discharge and therefore 20% more water usage by the farm as would be calculated by an industrial system. Although more water was used in terms of water conservation, for the farm this has benefits. When there is no discharge, there is no need for treatment systems and no need to deal with governmental guidelines. There appears to be a dichotomy between water conservation and inexpensive techniques of protecting surface waters from on-farm wastewaters. Therefore, there is need to resolve this dichotomy with a clear definition of water usage, education of farmers and increased incentives for farmers to treat and discharge their wastewater. The best way to eliminate this dichotomy would be to research potential for on-farm water reuse, which will create no-discharge systems and yet at the same time conserve water.

The treatment system had mass removals of 95 and 99 % of BOD$_5$, 98 and 99 % TSS, 77 and 99 % TN, 77 and 99 % TP, and 71 and 99 % log CFU $E. coli$ in Y1 and Y2, respectively. The outflow concentrations were 135 and 25 of BOD$_5$, 176 and 91.2 of TSS, 114 and 43.1 of TN,
13.8 and 1.35 of TP, and 4.46 and 2.39 of log CFU $10^0$ E. coli in Y1 and Y2, respectively. Although most of the contaminants were removed, the outflow concentrations were not low enough to pass governmental discharge guidelines. More research into removing solids earlier in the treatment system should be a priority in future designs. The TSS was not low enough to pass governmental guidelines and many other contaminants are associated with solids, therefore, adding supplementary treatment to the system will be able to pass guidelines.

The supplementary treatments explored were sand filtration and coagulation. Unsaturated filters were found to be significantly better at contaminant removal for all contaminants, with the exception of nitrate and TN due to nitrification of ammonia. TSS mass removal was an average 89.1% from unsaturated filters and 74.6% from saturated filters. Overall, the smaller dosages were found to have more mass removal than the larger dosages, with the exception of nitrate and TN. TSS mass removals were found to be 89.5, 87.7, and 75.2% from 150 mL, 300 mL, and 600 mL dosed filters, respectively. BOD$_5$ mass removal was found to range from 77% to 99% and TP removal from 71% to 98%. There was almost no nitrate increase in the saturated filters, suggesting that nitrification was only present in unsaturated filters. Micrographs confirm increased DNA and RNA presence in 150 mL dosed unsaturated filters compared to 600 mL dosed saturated filters, indicating more biological growth which increases nitrification. The existing treatment system on the potato farm lacks nitrification for N removal, but has the proper environment for denitrification, therefore, the addition of the unsaturated filter could help remove the nitrogen to the discharge limit. Additionally, with the combined TSS removal of both systems the outflow concentrations would be low enough to pass governmental guidelines. On-farm studies are necessary to explore the performance of a full-sized system and the cost analysis of building an unsaturated system.

The best performing and most cost-effective coagulant was 8185 at ED50, ED75, and ED90. Due to the nature of the logarithmic treatment of the coagulants, the optimal treatment was between 50-60% in most coagulants with the exception of 8185 which had an optimal treatment at 80%. Therefore, at its peak performance it can reach higher TSS removal than any other coagulant at the lowest price. At 75% TSS removal (ED75), 8185 required only 77 L y$^{-1}$ and cost $1200$ y$^{-1}$. A smaller volume of liquid is easier for farmers to work with, making treatment more convenient. As part of the larger treatment system TSS outflow concentrations will pass
guidelines. Future research should evaluate the performance of using more than one type of coagulant as well as conducting on-farm trials.

6.2 Recommendations

1. Simple techniques can be implemented in other potato/root crop farms to decrease water used for washing and processing. More research is necessary to understand larger patterns of water usage.

2. Changes to the treatment system were not as effective as increasing the removal of contaminants using finger rollers. Promoting increased crop dry cleaning will decrease the contaminants in the wash-water and the subsequent discharge to surface bodies.

3. Wash-water contaminants were persistent due to colloidal particles that were hard to remove. Future treatment should focus on removal of these particles in a larger treatment system or supplemental treatments.

4. Removal of solids using sand filtration was higher in unsaturated filters, although not by a large amount. The greatest benefit of using unsaturated filters was increased nitrification. Therefore, when choosing filter orientation, saturated filters will work well for solids removal, but to achieve more N removal, saturated filters are superior.

5. Coagulation was most effective using Nalco brand polymer 8185. It was comparatively the most cost effective and used the smallest volume to reach a given removal. Although none of the dosages tested were able to achieve governmental discharge guidelines, as a supplementary treatment system, coagulation would be helpful to remove colloidal particles.
7 Bibliography


8 Appendix

8.1 Tipping Buckets

8.1.1 Methods
Water movement through the wetland was monitored by continuous measurement of calibrated tipping buckets set up at the outflow of C3 and the final wetland outflow. Hourly counts were recorded by a Campbell Scientific CR21X datalogger (Campbell Scientific, Edmonton, AB, Canada). This datalogger was replaced by a CR3000 May 2013 (9 months after the start of monitoring). Data was downloaded on a regular basis through either a manual USB connection or wirelessly using two Campbell Scientific RF401-series spectrum radios (Campbell Scientific, Edmonton, AB, Canada). One radio was attached to the CR3000, while the other was attached to a laptop located inside the storage facility. Through wireless internet access supplied by the farm, the laptop uploaded the data for easy retrieval from anywhere.

Data was converted into volume and summarized into monthly totals:

\[
Total Volume_{\text{mon}} = \sum (Count_h \times 1.5L)_{\text{mon}}
\]  

[8.1]

where Total Volume$_{\text{mon}}$ is the total volume passing through the tipping bucket in that given month; Count$_h$ is the recorded number of tips; 1.5 L is the average volume of water for each tip.

8.1.2 Problems with tipping buckets

Problems with the tipping buckets arose in Y1 of the study. There were two ways issues arose: water was clogged and did not flow through bucket, or water flowed regularly although it was not registered by the tipping bucket. The first issue was due to large solids or debris filling and clogging the outflow pipe that led to the tipping bucket. Clogging of the pipe would lead to slowing or stopping of flow from the treatment cell. In extreme cases this caused leakage out of the cell in unwanted places. This was solved by placing a metal screen on the outflow pipe to keep large debris from entering. Clogging of pipes and tipping bucket was also caused by ice. Freezing of the tipping buckets was fixed by placing insulated huts over the tipping buckets with one heat generating light bulb in each. Finally, there were times when the water was flowing, but the tipping buckets did not register the tips, either due to issues with the cable that extended to the datalogger, or magnetic switch that generated the pulse. Because of excessive issues, the data
in the second tipping bucket in Y1 is not considered to be reliable and therefore, the data from the first tipping bucket will be used as the outflow for that year.

8.1.3 Flow Volume

The first tipping bucket (TB1) was located at the inflow into the fourth cell, which was the last cell in Y1. In Y2 this cell was followed by two more. In Y1 TB1 had flow right from the start the shipping season (October), while in Y2 TB1 did not register flow until December. This was due to the larger size of the wetland in Y2 and the lower inflow volume. The total volume of water passing through TB1 in Y1 was 2215 m$^3$ and 1780 m$^3$ in Y2. Monthly total flow through TB1 was not found to be statistically different from Y1 to Y2 (p = 0.341; Water use in Y1 was highest in January and July (Figure 2.1). Similarly, in Y2, the highest water usage was July and December. This suggests that January/December and July are seasonally busier times for shipping.

The month with the highest volume through TB1 was 433 m$^3$ in January of Y1 and 416 m$^3$ in July of Y2 (Chapter 2 Figure 2.1). In both years this corresponded to the months with highest volume inflow.

The total water flow through TB1 in Y1 was 55% of the total water input into the treatment system (the total of all three flow meters). In Y2, the total water through TB1 was 75% of the total water input into the treatment system. In both years, TB1 flow was statistically different than the inflow water (P <.001). Linear regression of monthly total volume from TB1 and inflow total found strong correlation in Y1 ($r = 0.72; r^2 = 0.51$). In Y2 there was no correlation found ($r=0$). The lack of a correlation in Y2 is most likely due to the increased size of the wetland. Because there was no water flow through TB1 until December, the pattern of flow was distorted by the pooling in the first three cells of the treatment system.

The second tipping bucket (TB2) was located at the outflow of the treatment system for both years. This means that in Y2 the tipping bucket was moved to account for the two new cells created. The total volume of water passing through TB2 was 473 m$^3$ in Y1 and 53 m$^3$ in Y2. Monthly flow through TB2 was not statistically different from Y1 and Y2 (p = 0.055, Water use in Y1 was highest in January and July (Figure 2.1). Similarly, in Y2, the highest water usage was
July and December. This suggests that January/December and July are seasonally busier times for shipping.

The highest monthly total volume through TB2 was 170 m$^3$ in December of Y1 and 59 m$^3$ in July of Y2. For Y1 this does not correspond to the highest inflow month. The outflow from TB2 does not correspond to the flow through TB1 and inflow measured by the flow meters. Therefore, we suggest that there were measurement errors from malfunctioning of TB2 in Y1.

The total water through TB2 was 12% of the total water input into the treatment system for Y1. In Y2, the total water through TB2 was only 2% of the total water input into the treatment system. In both years, TB2 flow was statistically different than the inflow water (p<.001). The dramatic decrease in outflow in Y2 is due to water leaching into the soil in the newly built cells. Flow into the cells, and pooling from rain and snow melt, was observed, however, each time the water would slowly disappear over the course of a few days. The leaching was so large that only in July were tips registered, when a large amount of shipping was done over consecutive days. Linear regression of monthly total volume from TB1 and inflow total was found to have no correlation in both Y1 and Y2 (r<0). This lack of correlation in Y1 is further evidence of the poor quality of the TB2 data in Y1. Considering there was only one treatment cell between TB1 and TB2 in Y1, there should be little difference between them.
Figure 8.1 Tipping bucket daily outflow from Y1 and Y2 for the shipping season (October – July).
8.2 Tracer study

8.2.1 Methods

Potassium bromide salt tracer was injected once during each shipping season. In both years, salt was obtained from Quadra chemicals Ltd. (Q06901-025KB, Burlington, ON). Salt was mixed with water in a 900 L tote and allowed to dissolve for several days. In both years, salt was injected into the flume prior to discharge into the wetland, however, in Y2 the water was also injected into C4, due to the larger volume of the system. Washwater in the flume and the salt solution were sampled prior to the injections. The solution was added at several locations to spread out the liquid and the flume was allowed to circulate for an hour to mix the wash-water with the salt solution. The flume water was sampled once more prior to discharge into the treatment system. Injection into C4 followed similarly. The wetland cell and the salt solution were sampled prior to injection and the liquid was slowly poured into the wetland at the location of the inflow pipe. In the first year, 20 kg of salt was injected into the wetland, and in Y2 25 kg was injected into each of the two starting location. Grab samples of water were taken from each treatment cell, at the location of the outflow pipe.

Bromide concentration was measured using a bromide ion selective probe (Br 48-0001, Analytical Sensors Inc.) and meter (Accumet XL25). A standard curve was made by measuring known concentrations ranging from 1-100 ppm KBr. The logarithmic equation was only accepted if the $r^2 < 0.99$. Samples were measured undiluted and the equation of the standard curve was used to calculate the concentration. Example equation:

$$\left(\frac{mg}{L}\right) = e^{\frac{Br(mV)-170.54}{-26.87}}$$  

where $Br$ (mV) is the reading from the electrode in mV and $Br$ is the final concentration of bromide (mg L$^{-1}$).

8.2.2 Results

The bromide salt (Br) tracer was used once per year to study the flow patterns in the treatment system. The treatment system in both Y1 and Y2 had a sedimentation pond followed by one or two aeration cells. Therefore, it was expected to see a plug-flow graph for the first cell, a bell-shaped curve with a time skew, and a diluted curve for the rest due to the mixing action of the
aeration cells. In Y1 the Br recovery from each cell was 100% from C1, 90% from C2, 80% from C3, and 72% from the C4. The time for 10% of Br to leave the system was ~10 d. The first cell (C1) had a high starting Br concentration of 52.2 mg L\(^{-1}\), although in day 2 the concentration spiked to 221 mg L\(^{-1}\). On day 3 the concentration once again dropped to 63.9 mg L\(^{-1}\). The Br mass outflow from C1 in day 2 was 87% of the total Br kg mass injected throughout the whole study. The theoretical hydraulic retention time (HRT) for C1 was 6 d, while 87% of the tracer outflowed in 2 d. This suggests there was preferential flow in the cell that allowed the water to flow faster and decreased the retention time. The outflow from the remaining three cells never went above 50 mg L\(^{-1}\) and the cumulative Br mass outflowed in a gradual manner, as expected from the mixed cell (Figure 8.2). Due to the low constant outflow of Br from these cells, it is difficult to determine the HRT. The decreasing % Br recovery throughout the system suggests that continuing measurements could have seen more Br outflow. Therefore, it can only be said that the HRT will be >45d for each cell and the entire system.

In Y2, the tracer was injected in two different locations due to the larger size of the system. A second dose was injected at the inflow of C4, thereby splitting the system into C1, C2, C3 and C4, C5, C6. Br recovery for each cell was 100% for C1, 100% for C2, 77% for C3, 87% for C4, 79% for C5, and 59% for C4. The time for 10% of Br to leave C3 was 35d and to leave C6 was 36 d, therefore, the time for 10% Br to leave the system was ~71 d. The first cell, C1, displayed a plug-flow patter. Similarly to Y1, the day following injection the outflow, Br concentration spiked to 63.3 mg L\(^{-1}\) from the starting concentration of 13.4 mg L\(^{-1}\). In Y2, however, the spike was not as high and the consequent decrease in Br concentration was more gradual. Only 10% of the total Br mass was in the outflow on the second day, compared to the 72% in Y1. This could be due to the smaller quantity of wash-water used for shipping during the Br injection, or the removal of preferential flow. As a consequence of this, the Br outflow from C1 was more gradual (Figure 8.2). The following two cells, C2 and C3 had very gradual outflow concentrations throughout the study. C2 had a spike on day 66 of the study, which caused 12% of the total Br mass to outflow. This may correspond to the spike in C1, although this would suggest that the mixing action of the aerator was not enough to fully mix all the water. There was no such spike seen in C3. The second part of the tracer from C4, C5, and C6 had very similar results to that of Y1. On day 3 of the study C4 had a spike to 64 mg L\(^{-1}\) from a background of 7.66 mg L\(^{-1}\), although the Br mass corresponded to only 2% of the total outflowed mass. The
following day the Br outflow concentrations fell to 10.1 mg L-1 and remained <20 mg L-1 for the rest of the study. The last two cells, C5 and C6 had Br concentrations on average <20 mg L-1 and the outflows were gradual.
Figure 8.2 Cumulative bromide mass each treatment cell in Y1 and Y2.
8.3 Additional Methods

8.3.1 Annual cleaning
Annual cleaning of the treatment system occurred in late summer (August - September) of each year. Cleaning included pumping out the water from cells C1, C2, and C3 and spreading the wastewater onto grass fields as irrigation. The empty cells were dug out to the full capability of the farm-owner excavator (Figure 8.10). The excavator had limited reach and for the larger cells (C3) it would be unable to reach the middle of the cells. Therefore, C3 was left with a make-shift island in its centre. Since each year the same procedure was executed, the dirt dug up by the excavator was mostly sediment from the previous year of water treatment. Additional work during this period included adding soil to the channel separating C1 and C2, which was narrow and eroded over the course of the year.

8.3.2 Depth sampling
The depth of wetland was measured in the spring of each sampling year. Depth sampling was done using an inflatable boat, rope, and a measuring tape. The measuring tape was either weighted to sink or attached to a sampling pole with tape. The boat was attached to a rope with a person holding each end. The sampling points were taken to fill a grid of 5 x 5, or 6 x 6 for larger treatment cells. Therefore, each cell had at least 25 depth measurements. The people holding the rope would move the boat to the appropriate sampling spot visually estimating the grid. A person in the boat would suspend the measuring tape till it reached the sediment and document the reading. Only cells C1, C2, and C3 were measured in this way. Cells C4 was too tickly overgrown to be measured in this way, therefore measurements were done as best possible from the shore to not disturb sediment.

8.4 Climate

8.4.1 Methods
A small weather station was set up and connected to the same data logger as the tipping buckets (Campbell Scientific CR21X datalogger, later replaced by a CR3000 in May 2013 (Campbell Scientific, Edmonton, AB, Canada)). It comprised of a Series 525 rainfall sensor (Texas Electronics, Dallas, TX, USA), a LI-200SA Pyranometer sensor (LI-COR, Lincoln, NE, USA)
for solar radiation, and a thermocouple with solar radiation shield for air temperature. An anemometer for wind speed and velocity was added in May 2013.

In September 2012, two Hobo® pendant temperature loggers (Onset®, Bourne, MA, USA) were suspended in each cell to record daily average water temperature. Due to trouble retrieving the sensors, thermocouples wired to the datalogger were submersed into C3 and C4 in May 2013.

8.4.2 Results

Both Y1 and Y2 followed expected seasonal trends and saw higher volumes of rainfall in spring and fall (Figure 8.3). Total rainfall alone was 577 mm in Y1 and 554 mm in Y2. This is lower than the historical yearly total for the area (682 mm 1981-2010; Government of Canada, 2014). Since the area in Y2 was larger, the volume of rainfall into the system was higher – 656 m³ in Y2 and 732 m³ in Y1. The difference between Y1 and Y2 was statistically different (p = 0.758; Water use in Y1 was highest in January and July (Figure 2.1). Similarly, in Y2, the highest water usage was July and December. This suggests that January/December and July are seasonally busier times for shipping.

Air temperature in Y1 ranged from 26 to -17.1 °C and 38 to -23 °C in Y2 (Figure 8.3). Average air temperature was 5.6 °C in Y1 and 3.3 °C in Y2. This was lower than the historical average daily temperature for the area (7.7 °C 1981-2010; Government of Canada, 2014). The temperature difference between Y1 and Y2 was significantly different (p = 0.028; Water use in Y1 was highest in January and July (Figure 2.1). Similarly, in Y2, the highest water usage was July and December. This suggests that January/December and July are seasonally busier times for shipping.

Water temperature was measured below the ice level and therefore remained above 0 °C. The highest water temperature in Y1 was 21 °C and in Y2 24 °C. Air and water temperatures in both years followed normal seasonal temperature pattern.

Solar radiation followed expected seasonal trends, with highest solar radiation in the summer, and lowest in the winter. Y1 had monthly average solar radiation of 123 w m⁻², while Y2 was slightly lower, 117 w m⁻². Solar radiation was not significantly different from Y1 and Y2 (p

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5 Y1 rainfall data was replaced with Environment Canada rainfall data due to consistently unusually low numbers and clogging of the rainfall gauge with debris.
Water use in Y1 was highest in January and July (Figure 2.1). Similarly, in Y2, the highest water usage was July and December. This suggests that January/December and July are seasonally busier times for shipping.

Figure 8.3 Climate data from the entire study, daily temperature (°C), water temperature (°C), and total precipitation (mm).

8.5 *E.coli*

Due to the unexpectedly large concentrations of *E.coli* in the wash-water, other locations were sampled in order to discover the source of the contamination. The theory behind using *E.coli* as an indicator species suggests that *E.coli* will only be able to survive in the presence of faecal matter (Tchobanoglous and Schroeder, 1987). Once *E.coli* has left the animal environment it will die off as the faecal matter is degraded. Therefore, there should not be any *E.coli* present, since the only possible source of faecal matter would be from spring application onto potato fields. Any residual matter on the potatoes would not be enough to support *E.coli* growth. Sampling of
several locations the storage facility found that the wash-water from the flume following shipping had the highest concentration of *E. coli*. The flume wash-water had 8.84 log CFU 100 mL\(^{-1}\). A piece of rotten potato had 8.15 log CFU *E. coli* 10 g\(^{-1}\) sample, while a non-rotten potato with dirt had 8.19 log CFU 10 g\(^{-1}\) sample, therefore the rotting of the potato did not seem to be linked to *E. coli* values. The smallest concentration of *E. coli* was found on soil/dirt removed from potatoes (6.62 log CFU 10 g\(^{-1}\) soil). This was 19% less *E. coli* (on the log scale). Therefore, the presence of potatoes appears to have a correlation with *E. coli* concentrations; however more research is needed to determine the effect potatoes may have on *E. coli* survival and growth.

Since manure application onto potatoes fields is the only known source of faecal contamination, soil sampling of three different fields was done over the course of a growing season. The fields sampled were a potato field that had received manure in spring, a soybean field that received manure in spring, and soybean field without manure to test *E. coli* presence due to manure fertilization and crop type. Sampling occurred three times: spring following planting, summer during growth, and fall following harvest. All fields had their lowest concentrations in the summer and both soybean fields had their highest concentrations in the spring. The potato field had only a slightly higher concentration in the fall than spring. The highest overall average concentration of *E. coli* was in the soybean field with manure (2.37 log CFU 10 g\(^{-1}\)), while the soy bean field without manure had the smallest concentration (1.91 log CFU 10 g\(^{-1}\)). This is 19% less *E. coli* with in the absence of manure fertilizer. The potato field had an average concentration of 2.24 log CFU 10 g\(^{-1}\) as well as the smallest deviation throughout the season. This suggests that there may be a difference in *E. coli* presence due to manure fertilization and crop type, although due to the lack of other measurements, such as temperature and moisture, which can influence *E. coli* growth no conclusions can be made. Additionally, there have been many studies suggesting that *E. coli* is naturally occurring in soils and not all strains need faecal matter to survive (Jamieson et al. 2002). Therefore, *E. coli* is a poor indicator of faecal contamination. Further research would be necessary to better explore these topics.

Bacteria mortality in freshwater environments is usually 30 d (Kadlec and Wallace, 2009, Crites and Tchobanoglous, 1998), although there was still *E. coli* presence in the outflow of Y1 and C6 of Y2. Without the use of dangerous chemicals, *E. coli* is removed naturally from wastewater through UV light, predation, and filtration/settling (Kadlec and Wallace, 2009). We would
expect that the lower TSS and shallow water or C5 and C6 in Y2 would allow UV to kill more \textit{E. coli}; however there was no visible increased removal rate.

A side project has been undergoing to find the source of the \textit{E. coli} and preliminary results suggest that the results of the mFC media plating may be a false positive. DNA extractions and gel chromatography from the same samples as those plated found zero presence of \textit{E. coli} DNA. Further research will continue to confirm and explain these results (Personal Communication: E. LeRiche).

8.6 Supplementary Figures and Images

\textit{Table 8.1} Outflow contaminant values from the Alliston treatment system, sand filtrations, and coagulation, as well as discharge guidelines needed to achieve.

<table>
<thead>
<tr>
<th>Contaminant</th>
<th>Units</th>
<th>Guideline</th>
<th>Treatment system</th>
<th>Sand Filtration</th>
<th>Coagulant 8185 (0.5 mL L(^{-1}))</th>
</tr>
</thead>
<tbody>
<tr>
<td>TSS</td>
<td>mg L(^{-1})</td>
<td>&lt;25</td>
<td>176</td>
<td>91.2</td>
<td>43 unsaturated, 29.2 saturated, 28 unsaturated</td>
</tr>
<tr>
<td>BOD(_5)</td>
<td>mg L(^{-1})</td>
<td>&lt;25</td>
<td>135</td>
<td>25</td>
<td>77.5 saturated, 15.4 unsaturated, n/a unsaturated</td>
</tr>
<tr>
<td>NH(_3)</td>
<td>mg L(^{-1})</td>
<td>&lt;1</td>
<td>24.4</td>
<td>6.65</td>
<td>87.3 saturated, 5.89 unsaturated, n/a unsaturated</td>
</tr>
<tr>
<td>TP</td>
<td>mg L(^{-1})</td>
<td>log CFU 100m L(^{-1})</td>
<td>13.8</td>
<td>1.35</td>
<td>0.93 saturated, 0.37 unsaturated, n/a unsaturated</td>
</tr>
<tr>
<td>\textit{E. coli}</td>
<td>log CFU 100m L(^{-1})</td>
<td>&lt;2</td>
<td>4.46</td>
<td>2.39</td>
<td>1.61 saturated, 2.15 unsaturated, n/a unsaturated</td>
</tr>
</tbody>
</table>
Figure 8.4 Potatoes entering the flume for transport and washing (B), following finger roller dirt removal (A), as part of pre-shipping processing.
Figure 8.5 Wet potatoes being sorted and checked by storage workers during pre-shipping.
Figure 8.6 Lab-scale saturated (top) and unsaturated (bottom) sand filters used for treating wastewater.
Figure 8.7 Raw (left) and coagulated (right) wastewater – part of initial dosage testing.
Figure 8.8 Treatment system in Alliston, ON, showing weather station and fiberboard, insulated hut covering the first tipping bucket (TB1).
Figure 8.9 Third cell of the treatment system following annual cleaning of sediment prior to the start of Y2 of measurements. Soil in the centre could not be reached by the farm-owned excavator.
Figure 8.10 Farm owned tractor and loader adding material to a berm. Also showing aerator (bottom) irritating water to increase oxygen transfer into the water in the second cell.
Figure 8.11 Micrographs of raw wash-water showing sizes of various particles (Colloidal particles are particles <1μm).