# Biofilm-Enhanced Treatment for Arctic Wastewater Stabilization Ponds Using Geotextile Substrate

by

Evan Bridson-Pateman

Submitted in partial fulfillment of the
requirements for the degree of
Master of Applied Science

at

Dalhousie University
Halifax, NS

August 2013

# Table of Contents

# List of Tables

# List of Figures

# Abstract

Wastewater stabilization ponds (WSPs) are a common method of treating wastewater in the remote communities of the Canadian Arctic. Unfortunately, the construction materials occasionally lead to highly permeable pond berms. These berms allow improperly treated effluent to escape in an uncontrolled manner, creating environmental and regulatory issues. In this thesis, a semi-permeable lining system was proposed to upgrade existing facilities. In this proposal, a geotextile layer would line the WSP berms, acting as a biofilter to provide additional treatment of the exfiltrating wastewater. Although biofilters are commonplace at lower latitudes, the combined effects of cold temperature and short-duration summers on biofilter performance are inadequately studied. The goal of this research was to study the hydraulic and treatment performance of geotextile substrate biofilters under these arctic conditions.

Filtration experiments were conducted over 9 months in a controlled laboratory environment. Municipal wastewater was passed through acrylic columns containing nonwoven geotextiles over 10 cm of gravel, simulating the berm in contact with the exfiltrating wastewater. Three experimental trails were conducted at either 10°C or 2°C, with each lasting 12 weeks. Weekly samples were taken before and after the filtration columns, and were analyzed for a suite of water quality parameters. Also, hydraulic conductivity was monitored weekly using constant head permeameter testing.

Results showed that it is possible to accumulate biomass on geotextile material over a 3 month period. Significant removal of total suspended solids (TSS), 5-day biochemical oxygen demand ($BOD_5$), total nitrogen (TN), and total phosphorus (TP) was observed. However, at the lower temperature of 2°C, less microbial activity was present. Evidence for this includes a decline in the treatment performance of the biofilters with respect to TSS and $BOD_5$. Average TSS removal declined from 59-68% to 20-45% when temperature was changed from 10°C to 2°C. Average $BOD_5$ removal declined from 30-45% to 11-18%. Significant removal of TN was observed only at the higher temperature of 10°C. TP removal was primarily dependent on the gravel layer and not temperature. Hydraulic conductivity decreases of 90% were observed at both temperatures.

This study has demonstrated that geotextiles can be used to enhance biological treatment processes at cold temperatures. However, it is recommended that further experimentation be conducted to determine the effect of freeze-thaw processes on geotextile biofilters, and if seeding the geotextile at the start of the treatment season would shorten the growth period. Both of these recommendations should be tested at the pilot scale before deployment of geotextile-based WSP enhancement strategies in the Canadian Arctic.

# List of Abbreviations and Symbols Used

AANDC – Aboriginal Affairs and Northern Development Canada

AoS – Apparent Opening Size

AS – Activated Sludge

ASTM – American Society for Testing and Materials

BOD – Biochemical Oxygen Demand

$BOD_5$ – 5-day Biochemical Oxygen Demand

CCME – Canadian Council of Ministers of the Environment

CFU – Colony Forming Units, bacterial enumeration technique

COD – Chemical Oxygen Demand

$d_{50}$ –Screen opening size at which 50% of grains pass

DO – Dissolved Oxygen

*E. coli* – *Escherichia coli*

HRM – Halifax Regional Municipality

IFAS – Integrated Fixed-Film Activated Sludge

$NH_4$-N – Ammonium-Nitrogen

$NO_3$-N – Nitrate-Nitrogen

NPS – National Performance Standards

NU – Nunavut

NWB – Nunavut Water Board

PVC – Polyvinyl Chloride

STE – Septic Tank Effluent

TKN – Total Kjeldahl Nitrogen

TN – Total Nitrogen

TOC – Total Organic Carbon

TP – Total Phosphorus

TSS – Total Suspended Solids

TWTP – Timberlea Wastewater Treatment Plant

USEPA – United States Environmental Protection Agency

WSP – Wastewater Stabilization Pond

WSER – Wastewater Systems Effluent Regulations

YT – Yukon Territory

$\Delta H$ – Head difference

# ACKNOWLEDGEMENTS

To begin, I want to thank my supervisors for believing in me and facilitating this project:

Dr. Craig Lake – for noticing the potential in a first year engineering student and providing guidance to that student over 6 years in classes, co-ops, and graduate degrees. Thank you for all the support over the years.

Dr. Rob Jamieson – for his understanding and assistance during the many confused and panicked scenarios I have accidentally initiated. Thank you for the opportunity to explore the Canadian north.

I would like to thank my thesis committee; Dr. Graham Gagnon, and Dr. Jamie VanGulk, for their technical assistance in planning and operating the experiments. I would like to express my appreciation for the financial support of the Government of Nunavut and the Natural Sciences and Engineering Research Council of Canada – specifically the STEWARD program. Thank you Wendy Krkosek and Tarra Chartrand for organizing the program, and all the educational events it included.

Special thanks go to Brian Kennedy in the Civil Engineering shop for constructing my columns, and their replacements. Further thanks go to Ray Dubé in the Environmental Engineering shop for his advice, troubleshooting, sense of humor, and patience. I would further like to thank Rick Lowe, Joe Hazelden, Dan Wesley, and Heather Shea at Halifax Water for letting me use their Timberlea Facility, for answering all my questions, and for hanging precariously to fill wastewater pails. I am thankful for the aid of friends and colleagues for their help both in and out of school, especially Leah Boutilier, Rick Scott, Janeen McGuigan, Greg Piorkowski, Jenny Hayward, Kristin Myette, Ali Hosein, and Erin Mentink.

Finally, I would like to thank my parents, Cheryl Bridson and Chris Pateman, for their love, guidance, and understanding. This thesis is dedicated to the memory of Edmund J. Bridson.

# 1 INTRODUCTION

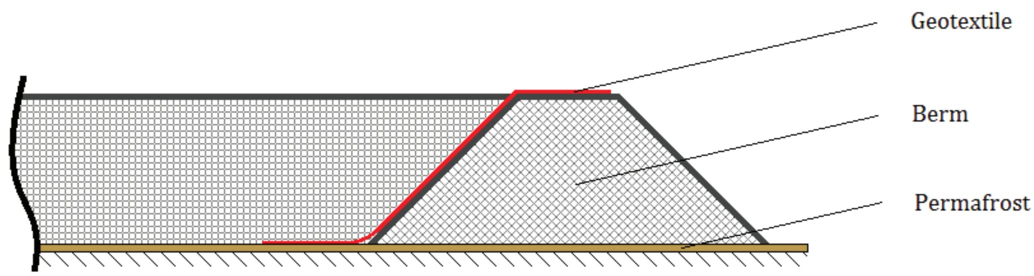
Wastewater stabilization ponds (WSPs) are a common method of treating wastewater in the Canadian Arctic. It is passive treatment systems like WSPs that are the most economical methods of wastewater treatment. Remote communities and restricted transportation methods limit the availability of the materials, equipment, and energy requirements to construct and operate a conventional treatment system (Wootton, et al., 2008). However, the abundance of land makes a constructed lagoon system ideal (Horan, 1990). As such, nearly 50% of wastewater treatment in Nunavut and the Northwest Territories use WSPs (Jamieson, et al., 2012). In this type of system, treatment is accomplished by sedimentation in combination with suspended-growth microbial processes (Crites and Tchobanoglous, 1998), and treatment is highly dependent on temperature and retention time (Krkosek, et al., 2012).

Unfortunately, WSPs are typically constructed of local fill, and in some arctic locations, this fill is highly permeable (Bölter, et al., 2006). The resulting porous WSP walls allow for uncontrolled discharge of wastewater (Hayward, et al., 2012). This is a problem when the exfiltrating wastewater has not resided in the pond for the designed retention time, and may not meet water quality standards. Poorer discharge water quality creates issues of non-compliance with Territorial water permits (Wootton, et al., 2008) or potentially the Environment Canada Wastewater Systems Effluent Regulations (WSER; Government of Canada, 2012).

In order to achieve the treatment performance required, these exfiltrating WSPs will need to be modified or reconstructed to operate under the controlled conditions assumed in their designs. However, reconstruction is an infeasible option when low-permeability material must be shipped in at very high cost.

Geosynthetic materials, on the other hand, are a light-weight and low-volume material that would be comparatively inexpensive to acquire. Lining the WSPs with a geomembrane would be a simple way to limit the permeability of an existing facility. However, the technical training in geomembrane welding and leak detection is uncommon in northern communities (like other skilled trades). If installed with poor quality control measures, leaks in the membrane may result in the formation pockets of anaerobic activity between the geomembrane and the impermeable permafrost, leading to ballooning, stretching, and ultimately failure of the geomembrane. A more suitable liner would be one that does not prevent leaks, requiring less skill in the installation process. Lining the perimeter of the WSP with a hydraulically-permeable geotextile (on the inner wall of the berm; Figure 1a) would retain the impermeability of the permafrost floor, and add a substrate for physical and biological clogging to slow the release of wastewater to a more controlled rate.

(a)




(b)

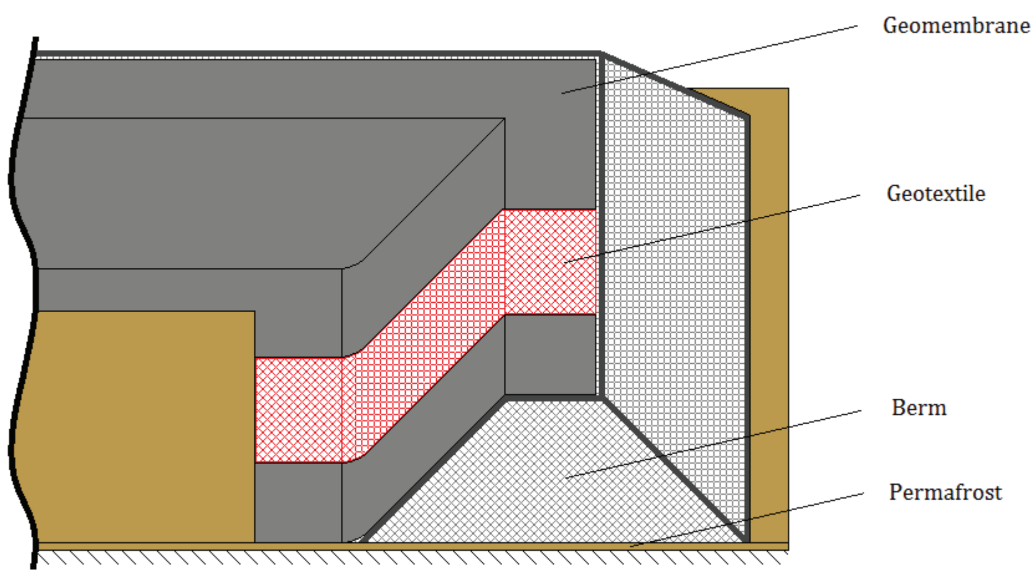



Figure 1: Geotextile liner location on berm (a) and a geotextile section location as part of geomembrane liner (b)

Geotextiles have been used in drainage and soil filtration applications for over 70 years (Bertram, 1940) and enormous resources have been devoted by the solid waste handling industry to study the clogging of geotextile under leachate flow (Cancelli and Cazzuffi, 1987; Koerner and Koerner, 1988; Koerner and Koerner, 1990; Rowe, et al. 1995; Armstrong, 1998; Rowe, et al., 2000; Rowe, et al., 2002; Palmeira, et al., 2008). Using the clogging potential of geotextiles a geotextile layer would serve two purposes: (i) to lessen wastewater seepage rate, increasing WSP retention time; and (ii) to provide tertiary fixed-growth biofiltration as wastewater passes through the geotextile.

However, completely lining the WSP berm walls would not be worthwhile. As the geotextile clogs in a particular zone, wastewater would preferentially flow towards zones of lower permeability. This would leave the overall system in a state of continual microbial development without reaching the optimal treatment potential of steady-state conditions. A more appropriate design would be to use geomembranes to focus wastewater exfiltration through a specific area of the WSP wall, and line that particular segment with geotextile (Figure 1b). Allowing a location for constant escape of wastewater would significantly reduce the volume of wastewater leaving through geomembrane imperfections. The relatively small area of the exfiltration section of the berm would also make adjustments, cleaning, repair, or replacement of the geotextile much easier and more affordable than if the whole berm perimeter was considered. Furthermore, the focused release of wastewater would also allow for better control of direction of discharge, and more concentrated nutrients for fixed-growth bioactivity.

Unfortunately, investigation into hydraulically-similar biofilters operating under low temperature conditions indicates a dramatic decrease in microbial growth (Ratkowsky, et al., 1982; Koerner and Koerner, 1990; Rowe, et al., 1995; Armstrong, 1998) and water quality improvement (Gullicks and Cleasby, 1986; Gullicks and Cleasby, 1990; Moll, et al., 1999) can be expected. The goal of this study is to determine the potential level of wastewater treatment and the expected permeability reduction in a geotextile biofilter under low arctic temperatures and a short arctic summer treatment season. By evaluating the hydraulic performance and treatment potential of a geotextile filters, this study will help improve WSP design to better meet effluent discharge regulations for the protection of environmental and public health in arctic regions.

## 1.1 Objectives

To achieve the research goal and determine the feasibility of using geotextile liners in an arctic WSP, bench-scale models of the geotextile filtration unit were evaluated. Two primary objectives were identified to address the intended effect of a geotextile liner:

1. To determine if significant increases in hydraulic retention time are achievable within the time and temperature constraints of an arctic summer. This was tested by analyzing changes in geotextile hydraulic conductivity.
2. To determine if significant water quality improvement is achievable within the time and temperature constraints of an arctic summer. This was tested by comparing influent and effluent concentrations of standard water quality indicators, including total suspended solids (TSS), 5-day biochemical oxygen demand ($BOD_5$), nitrogen species, and *Escherichia coli* (*E. coli*).

# 2 LITERATURE REVIEW

The purpose of this section is to provide background on the current wastewater management practices in Nunavut so as to identify weaknesses in design and the need for improvement. It will follow with a review of the physical and hydraulic characteristics of geotextiles and their use within wastewater treatment systems. Finally, research gaps and needs with respect to the use of geotextiles within arctic wastewater treatment systems will be identified.

## 2.1 WASTEWATER MANAGEMENT IN NUNAVUT

Wastewater management in Nunavut presents unique challenges not faced by municipalities in southern Canada. Although climate would certainly be an obvious challenge, of greater concern may be the fact that Nunavut communities are extremely remote and isolated. Furthermore, the average population size of communities outside Iqaluit is less than 1,000, with some communities as small as 140 persons (Wootton, et al., 2008). These low populations make it very difficult to justify the high capital costs of wastewater treatment systems.

In Nunavut, all communities are located in the zone of continuous permafrost (Wootton, et al., 2008). This creates a barrier to sub-grade development of conventional wastewater collection and treatment systems (Miyamoto and Heinke, 1979), such as piping networks or infiltration fields. As such, infrastructure must be constructed above-grade. Infrastructure projects are difficult to undertake, as most communities do not have the necessary machinery, construction materials, or engineering expertise on-site (Wootton, et al., 2008). Materials and equipment must be imported at great cost by barge once or twice per year, when the ocean has thawed (Wootton, et al., 2008). Even after construction, it is difficult to recruit the skilled workers required to operate the facilities (Miyamoto and Heinke, 1979). Finally, high energy requirements of conventional treatment systems would be provided by diesel fuel, also imported at high cost.

Due to the aforementioned reasons, wastewater stabilization ponds (WSP) are the most common type of treatment system. A reported 49% of communities in

Nunavut and the Northwest Territories use WSPs, compared to only 7% that use mechanical treatment (Jamieson, et al., 2012). WSPs have been a preferred option as they require little energy or maintenance, and the large land areas required for their construction (Horan, 1990) are readily available outside most communities. Constructing these ponds is relatively straightforward, with fill material generally placed directly onto the permafrost layer to form a bermed enclosure. Furthermore, local fill can generally be used to construct such systems. Operation costs are those associated with the regular collection of wastewater from homes within the community, and transportation by truck to a communal WSP outside of town (Wootton, et al., 2008).

### *2.1.1 Wastewater Stabilization Ponds*

The majority of wastewater stabilization ponds in Nunavut can be best described as facultative; these systems are designed to utilize both aerobic and anaerobic processes in addition to settling of solids by gravity (Crites and Tchobanoglous, 1998).

Contaminant removal in facultative ponds is dependent on hydraulic retention time (HRT), with typical ponds in temperate zones holding water for 4-6 days, and achieving 50-70% reduction in biochemical oxygen demand (BOD; Horan, 1990). WSPs are also capable of achieving up to 99% removal of ammonia (Pano and Middlebrooks, 1982). However, ammonia removal rate decreases with decreasing pH, HRT, and temperature.

Low temperatures negatively affect overall treatment performance of WSPs, making it difficult to apply southern WSP design standards to northern locations (Krkosek, et al., 2012). This is due to slower biological reactions and sedimentation (Metcalf & Eddy, 2003). Furthermore, freezing temperatures impede treatment for much of the year. Winter months in Nunavut can begin as early as September, with spring thaw as late as June (Wootton, et al., 2008). The freezing of WSPs over this (up to) 9 month period prevents the discharge of treated water, and causes newly added wastewater to freeze in place (Heinke, et al., 1991).

With a short treatment season occurring after the spring thaw, followed by an end of season discharge, WSP are generally designed with longer HRTs for wastewater storage (Krkosek, et al., 2012). A 12-month HRT is considered the benchmark (Heaven, et al., 2003), with a minimum retention time of 60 days following the sping thaw (Heinke, et al., 1991). Discharge at the end of the summer treatment season season typically feature effluent concentrations similar to those shown in Table 1.

Table 1: Arctic WSP effluent concentrations and performance

| Parameter | Effluent Concentration | % Removal |
|---|---|---|
| $BOD_5$ (mg/L) | 7 - 276 | 34 - 90 |
| TSS (mg/L) | 7 - 1,090 | 26 - 87 |
| Fecal Coliform (CFU/100mL) | $5.3 \times 10^1$ - $1.3 \times 10^7$ | 56 - 99.9 |

Source: (Jamieson, et al., 2012)

Further concerns arise when the WSPs are constructed of local materials; typically coarse gravel or sand (Bölter, et al., 2006). If unlined, this becomes a problem, as the lack of clay and other fines results in highly permeable berms. Under these conditions, there is reliance on berm core permafrost for containment (Krkosek, et al., 2012). However, the permafrost may experience differential melting, particularly in a global climate of increasing temperatures, allowing wastewater to exfiltrate from multiple locations over the summer season (Hayward, et al., 2012). This wastewater has not spent the intended residence time in the WSP before exfiltrating, and as such, may not meet the discharge requirements. This is of particular importance during the early summer, when large volumes of water exfiltrate the lagoons over a short period as the accumulated wastewater and winter snow melts (Hayward, et al., 2012). Typical seepage concentrations from WSPs across the Canadian arctic are shown in Table 2.

Table 2: Exfiltration pond seepage average concentrations

| Location | $BOD_5$ (mg/L) | TSS (mg/L) | Fecal Coliforms (CFU $100mL^{-1}$) |
|---|---|---|---|
| Kugaaruk, NU | 218 | 1,090 | $3.3 \times 10^6$ |
| Old Crow, YT | 17 | 146 | – |
| Clyde River, NU | - | 103 - 177 | – |
| Coral Harbour, NU | 120 | 100 | $>1.1 \times 10^5$ |

Source: (Jamieson, et al., 2012)

### *2.1.2 Environmental Considerations*

When wastewater enters the environment after less than adequate treatment, the excessive loading of organic material and nutrients that result can lead to severe consequences.

Increased suspended solid concentrations over long periods can lead to physical damage to fish gills or macrophyte biomass, or reduced population size of plant, invertebrate, and salmonid species (Bilotta and Brazier, 2008).

Organic material entering a receiving water system is rapidly oxidized and decomposed. This can lead to depletion of dissolved oxygen (DO) from the water. Further anaerobic digestion of this organic material may lead to odors (Vandevivere, 1999). Oxygen depleted waters will change the relationships between predators and prey (Breitburg, et al., 1997), and likely decrease the population size of certain fish species (Stevens, et al., 2006).

Many studies provide evidence that excessive nutrient loading of water can lead to algal blooms, and the risk of eutrophication that follows (Ryther and Dunstan, 1971). The extended daylight hours of the northern territories amplify this risk, as it provides longer periods of solar energy for photosynthesis.

Finally, in arctic environments, over-fertilization of land from nitrogen and phosphorus introduction can also lead to changes in biodiversity, such as diminished lichen growth, greater ground cover dominance by bryophytes, and altered bryophyte species diversity (Gordon, et al., 2001).

### 2.1.3 Regulatory Issues

Up until 2012, there were no enforceable standards for municipal wastewater discharge on a federal level. Guidelines were suggested by the Canadian Council of Ministers of the Environment (CCME) and Health Canada for the protection of aquatic life and recreational water quality, however these were non-enforceable. Variable enforceable discharge regulations were developed and enforced on a provincial and territorial level.

In Nunavut, the responsibility of issuing and revoking water licenses falls on the Nunavut Water Board (NWB), and it is this body that determines the effluent discharge standards to which municipal facilities must adhere (Jamieson, et al., 2012). Effluent limits in Nunavut as licensed by the NWB typically fall in the ranges given in Table 3.

Table 3: Typical maximum allowable effluent concentrations as licensed by the NWB

| Substance | Range |
|---|---|
| Fecal Coliforms (CFU/100mL) | $10^4$ - $10^7$ |
| $BOD_5$ (mg/L) | 80 - 120 |
| TSS (mg/L) | 100 - 180 |
| Ammonia (mg/L) | – |
| pH | 6 - 9 |

Source: (Wootton, et al., 2008)

Aboriginal Affairs and Northern Development Canada (AANDC) monitors facilities with licenses, and reports non-compliance to the Federal Government under the *Canadian Environmental Protection Act* and the *Fisheries Act* (Jamieson, et al., 2012; AANDC, 2010).

In 2009, the CCME proposed a unified set of wastewater discharge regulations to be used in every province and territory (CCME, 2009). At that point, northern regions (including Nunavut) were granted 5 years by the federal government to research existing treatment facilities to assess performance, and develop appropriate effluent discharge limits that address the technical difficulties of wastewater treatment in Nunavut. In the meantime, communities must continue to adhere to the provincial and territorial regulations currently in place.

In 2012, Environment Canada implemented the strategy recommended by the CCME as regulations of the *Fisheries Act.* These regulations encompass all provinces and territories in Canada (Government of Canada, 2012). The minimum National Performance Standards (NPS) in this act are shown in Table 4.

Table 4: Wastewater systems effluent regulations (WSER) under the *Fisheries Act*

| Substance | Limit[a] (mg/L) |
|---|---|
| TSS | 25 |
| Carbonaceous BOD | 25 |
| Total Residual Chlorine | 0.02 |
| Un-ionized Ammonia | 1.25[b] |

[a]Averaged value over a time span dictated by treatment volume

[b]Maximum observed concentration, at 15°C

Source: (Government of Canada, 2012)

Any effluent discharged from a wastewater treatment facility must meet these standards, and must not be acutely lethal to fish (Government of Canada, 2012). If the facility continues to discharge effluent above these performance standards, the Minister of Fisheries and Oceans has the authority to terminate operation (Government of Canada, 1985). These regulations do not apply to Nunavut, the Northwest Territories, or any wastewater systems located above 54° latitude during the 5-year research process (Government of Canada, 2012). At the time of publication of this report, no performance standards specific to Nunavut have been proposed.

Regardless of whether the new effluent discharge limits come from federal or territorial regulatory bodies, it is imperative that the existing wastewater treatment facilities be updated to provide better treatment in preparation for further effluent concentration restrictions. One option for improving treatment performance could involve retrofitting existing WSPs with a geotextile-substrate biofilter.

## 2.2 GEOTEXTILE CLOGGING FACTORS

To achieve improved treatment of wastewater in a WSP in a passive manner, there are two general approaches: (i) the WSP could be upgraded to provide a longer retention time to increase sedimentation and suspended-growth biological processes (Horan, 1990), or (ii) the exfiltrating water could be further treated by filtration and fixed-film biological processes. Both can result from biomat growth on a geotextile lining system.

By reducing the rate at which wastewater exfiltrates the WSP, the retention time will increase. This section compiles scientific evidence for physical and biological clogging of geotextiles loaded with wastewater, leading to reductions in hydraulic conductivity. Furthermore, factors that significantly influence the rate and extent of clogging will be explored.

There have been several studies that identify geotextile clogging potential when loaded with different types of effluent. For example, geotextile installed at the entrance to an arsenic-precipitating biofilter was found to clog with organic matter when supplied with mine tailings (Germaine and Cyr, 2003). However, the majority of research into clogging was conducted by the waste management industry, which showed reduced hydraulic conductivity in nonwoven geotextiles that physically remove TSS from landfill leachate (Cancelli and Cazzuffi, 1987).

In landfills, severe problems can result from the clogging of leachate collection systems. For example, reduced permeability can result in mounding of water on the liner, which causes leachate flow through the liner, and increases the risk of groundwater contamination (Rowe, et al., 2002). To quantify the possible loss in permeability, sand and gravel leachate collection systems were supplied with continuous leachate flow (Koerner and Koerner, 1988). Under aerobic conditions, reductions in hydraulic conductivity of 12% over 4 months, and 100% over 11 months were observed.

Further study was conducted on the biological reductions in hydraulic conductivity of leachate collection systems. Palmeira, et al. (2008) first passed leachate through a sand filter before passing it through nonwoven geotextiles. This reduced the influence of physical filtration on clogging of the material. Still, hydraulic conductivity was reduced by almost 4 orders of magnitude over 90 days when operating at continuous flow rates of 1.5-5 mL/s and chemical oxygen demand (COD) concentrations of 500-3,000 mg/L

In an effort to minimize biofouling of these systems, the waste management industry has researched clogging mechanisms. The results of these studies can provide useful information for design of geotextile-based wastewater treatment, where greater biological growth is desired.

### *2.2.1 Geotextile Type and Arrangement*

Geotextiles are generally classified by the arrangement of their constituent fibres. The two most common arrangements are woven and nonwoven. In soil drainage applications, woven geotextiles are ideal when strength is paramount, and nonwoven geotextiles are ideal for filtration of fines (Shukla and Yin, 2006). Under landfill leachate loading, nonwoven geotextiles have generally shown greater reductions in hydraulic conductivity than woven geotextiles (Koerner and Koerner, 1990). Nonwoven geotextiles resulted in 74% to 94% reductions in permeability compared to a maximum of 23% reduction in woven geotextiles (McIsaac and Rowe, 2006).

Other researchers have documented similar results with other effluents. On average, nonwoven geotextiles resulted in greater clogging than woven geotextiles in a 1-week recirculation experiment using septic tank effluent; nonwoven geotextiles achieved 24.7% reduction in hydraulic conductivity, versus 22.9% by the woven (Yaman, et al., 2005). Combined with an additional 15.7% effectiveness at nitrification, nonwoven geotextiles appeared to be the best substrate candidates for biofilm growth under less-than-ideal conditions.

Under the category of nonwoven geotextiles, two methods of manufacture are most common: stapled-fibre and continuous filament. Stapled-fiber manufacturing uses many short fibres matted together by physical, chemical or thermal means, while continuous filament geotextiles consist of very long fibres that are tortuously layered (Shukla and Yin, 2006). Continuous filament geotextile accumulated more biomass than stapled-fibre geotextile over a 9 week experiment when acting as the first filtration layer (Yaman, et al., 2005). In another study, a week-long batch reactor test showed similar treatment performance by stapled-fibre and continuous filament geotextiles; providing 90% removal of TSS and $BOD_5$, and 95% removal of ammonia (Korkut, et al., 2006). However, the greatest accumulation of biomass occurred on nonwoven, continuous filament geotextile.

Another factor which influences clogging dynamics within geotextile-wastewater systems is the presence or absence of sand or gravel buffer layers. When sand is placed above the geotextile, it acts as buffer, protecting the geotextile from direct loading of suspended solids (Rohde and Gribb, 1990), resulting in gradual changes in hydraulic conductivity (Koerner and Koerner, 1990). Combined gravel-geotextile leachate collection systems with the geotextile under a layer of gravel reported better clog prevention than systems with the geotextile separating the gravel from the landfill contents (McIsaac and Rowe, 2006). Based on this information, to achieve the greatest change in permeability in a geotextile filter system over the shortest time, the geotextile should be the first layer of filtration. This is further evidence supporting the proposed lagoon retrofit, as exfiltrating water will pass first through the geotextile before the sand/gravel of the berm.

### *2.2.2 Contaminant Loading*

In porous media, higher contaminant loading rates of landfill leachate result in greater clogging of the material (Rowe, et al., 2000). Under landfill leachate flow, biological processes will contribute to the media clogging, but more clogging is attributed to precipitation of inorganics (Rowe, et al., 2000). Further investigation has confirmed that physical clogging by suspended solids was a leading contributor to overall clogging (Rowe, et al., 2002). However, under low-flow conditions, bacteria were shown to contribute more to reductions in hydraulic conductivity in nonwoven geotextiles (Chen, et al., 1981). Furthermore, the extent of changes to hydraulic conductivity is similar under both aerobic and anaerobic conditions (Koerner and Koerner, 1990).

### 2.2.3 *Temperature*

A study of a Toronto landfill provided evidence that biological clogging of leachate collection systems is temperature dependent, with mass accumulation 2.6 times higher at 27°C than at 10°C (Rowe, et al., 1995). More evidence was provided by direct experimentation on the effect of temperature in leachate collection systems. A study showed that at higher temperatures, the degree of biologically-induced clogging is greater and develops more rapidly (Armstrong, 1998). The amount of mass accumulation on porous media was approximately 3 times higher at 27°C than at 10°C. The material also clogged 2.5 times faster at 27°C than at 10°C. However, the minimum temperature of this study was 10°C. It is reasonable to predict that geotextiles under arctic conditions will take longer to develop biologic clogging, and have less accumulated mass.

## 2.3 Performance of Comparable Biological Filters

The previous section provides justification for nonwoven geotextiles as the best candidate for overall biomass growth and clogging potential, but does not provide an indication of the level of treatment that would be expected when used as a wastewater biofilter. Unfortunately, limited research exists that examines geotextiles as wastewater biofilters, and none exists examining treatment under arctic conditions. As such, the most comparable biological treatment designs will be discussed in this section, followed by a summary of literature on experimental use of geotextiles as filters in any capacity.

Biological treatment units typically rely on bacteria present in the wastewater to provide either suspended- or attached-growth treatment (Benjes Jr., 1980). Typical saprophytes genera present in municipal wastewater that are responsible for organic degradation include *Achromobacter, Alcaligenes, Flavobacterium, Pseudomonas,* and *Zooglea* (Grady Jr., et al., 1999). Depending on the treatment type, other bacteria may be present, such as *Bacillus* and *Micrococcus* in suspended reactors, and *Sphaerotilus* in fixed-film reactors (Grady Jr., et al., 1999).

Nitrifying (ammonia oxidizing) bacteria are primarily of the genus *Nitrosomonas*, and convert ammonia into nitrite (Grady Jr., et al., 1999). Nitrite is then oxidized to nitrate by bacteria of the genus *Nitrobacter* (Grady Jr., et al., 1999). Additional microorganisms may also be present in municipal wastewater, such as the algae *Scenedesmus*, which has been used to remove both ammonia and orthophosphate (Zhang, et al., 2008). Although phosphorus is used by microorganisms for normal cell processes, phosphorus removal from wastewater primarily occurs when the microorganisms are no longer in a growth phase (Levin and Shapiro, 1965).

As microorganisms come into contact with filtration media, they adsorb to surfaces and each other, forming a biofilm layer (O'Toole, et al., 2000). There is also evidence that bacteria may undergo morphological changes (such as the loss of flagella) upon attachment to the biofilm (O'Toole, et al., 2000). Depending on the availability of oxygen, the biofilm may operate aerobically, or anaerobically. Microorganisms in this layer are responsible for removal of solids and the uptake of dissolved nutrients (Water Environment Federation, 2010).

### *2.3.1 Integrated Fixed-film Activated Sludge*

Integrated fixed-film activated sludge (IFAS) is a hybrid wastewater treatment method that combines the suspended growth and sedimentation processes of an activated sludge (AS) process with the attached growth processes of trickling filters (Water Environment Federation, 2010). Fixed-media in IFAS are commonly baffle-type (similar to geotextile sheets), or hanging rope-type. The media is attached to a frame that is anchored to the tank walls, while the media itself remains submerged in the wastewater. However, unlike the design proposed in this thesis, the fixed-film media is positioned throughout the reactor in an IFAS.

As with activated sludge processes, IFAS systems are able to provide excellent treatment of BOD and COD. Full scale IFAS systems can remove 86.7% of COD (Kim, et al., 2010), significantly more than AS sludge alone. IFAS excels at lower retention times, reaching the same level of treatment 43% faster than activated sludge (Water Environment Federation, 2010). However, at both long and short retention times, nitrification is significantly better than AS, with over 83.6% removal of ammonia (Kim, et al., 2010). Study of a pilot scale rope-type IFAS in

Maryland showed that nitrification rates were 2.25 times greater than activated sludge alone (Randall and Sen, 1996).

When using geotextiles as baffles in IFAS, similar treatment efficiencies are achievable. An 8-week bench-scale geotextile baffle contact system study showed average TSS, $BOD_5$, and ammonia concentration reductions in municipal wastewater of 94.6%, 91.6%, and 97%, respectively (Korkut, et al., 2006). The primary course of biological treatment was found to be at the biofilm surface. Although light penetration was prevented, the system was aerated and vapors were allowed to escape over a 22.5 hour retention time, and it therefore is unclear to what extent volatilization influenced the removal of ammonia. Extending on the geotextile baffling system, study at full-scale achieved biofilm growth on geotextile baffles within 2 months, at water temperatures of approximately 10°C (Korkut, et al., 2006).

Further study of aerated geotextile baffle systems treating contaminated groundwater achieved COD removal efficiencies of 61%, resulting in concentrations of 46-47 mg/L in the effluent (Jechalke, et al., 2010). An average removal of 99.9% of benzene was achieved, along with an average of 38% removal of methyl *tert*-butyl ether. However, limited growth of nitrifying bacteria in this 14 month study resulted in poor ammonia removal. This system ran at average pH of 7.1 ± 0.9 and average temperatures of 12°C ± 3°C.

IFAS using geotextile surfaces is similar to the design proposed in this thesis in that additional fixed film surfaces are incorporated into a suspended growth biological treatment system. However, the geotextile lining of a berm will have lower total surface area for microorganism attachment than an IFAS system, presumably resulting in lower overall treatment.

### *2.3.2 Trickling Filters and Other Fixed-Media Bioreactors*

Trickling filters use porous media to treat wastewater, and are similar to the proposed thesis design in that they promote removal of wastewater constituents by slowly passing water through a static biofilm (Water Environment Federation, 2010). However, unlike trickling filters, the filter media (the geotextile) in the proposed thesis design will be submerged, and not aerated.

Trickling filters have repeatedly shown their proficiency at removing BOD and TSS. Even before their widespread use and optimization, trickling filters at an air force base were able to achieve 70% removal of BOD, and 50% removal of TSS under high (by modern standards) hydraulic loading rates (Hatch, 1943). Gullicks and Cleasby (1986) studied nitrification performance in pilot-scale trickling filters treating wastewater with $BOD_5$ and TSS concentrations below 30 mg/L. They showed ammonia removal efficiencies of 61-92% in Michigan, and removal efficiencies of 74-95% in Illinois.

Trickling filters designed for combined removal of BOD and ammonia are also common. Single stage filters designed for this dual purpose can accept a loading rate of 120 g BOD per $m^3$ of filter media per day before effluent quality declines (Horan, 1990). Higher loading rates of 150 g BOD per $m^3$ can typically be applied if effluent is recirculated; or up to 250 g BOD per $m^3$ if two-stage filters are used. At rates between 100 g and 300 g of BOD per $m^3$ per day, effluent BOD concentrations below 10 mg/L can usually be achieved (Metcalf & Eddy, 2003), corresponding to a removal efficiency of up to 90% for municipal wastewater.

Other types of fixed-media biofilters are also used for smaller wastewater treatment operations, and achieve comparable results. In a scaled-down version of a trickling filter for treating sanitary sewer overflows, average TSS reductions were above 90% (Tao, et al., 2009). Solid organic matter was also removed, and ranged from 55-84% reduction in $BOD_5$ and 74-90% reduction in COD. Hu and Gagnon (2006) tested 4 materials in recirculating packed-bed biofilters for treating residential wastewater. They found that in combination with a septic tank, average removal efficiencies of 89.8%-96.1% for $BOD_5$, 31.3%-79.2% for TSS, 68.6%-81.2% for total nitrogen (TN), and 85.3%-99.4% for ammonia. They also showed up to 4.2 log-removal of *E. coli*.

The most comparable fixed-media biofilter to the proposed thesis design used packed-bed filters consisting of nonwoven geotextile chips. These were able to remove 97% of organic material, and 95% of TSS from wastewater under a hydraulic loading rate of 410 L/m$^2$/d (Leverenz, et al., 2000). These filters also removed 30% of the TN from the system, although it was speculated to be low due to poor mixing in the recirculation tank. Overall, the geotextile beds outperformed sand filters of the same size (Leverenz, et al., 2000), showing the promise of nonwoven geotextiles in other treatment operations.

### *2.3.3 Soil Absorption Fields*

Where larger municipalities tend to use centralized trickling filters for biological wastewater treatment, rural homeowners are likely to use on-site, decentralized systems. These systems typically consist of a settling (septic) tank, followed by a soil absorption field (McCray and Christopherson, 2008). Septic tank effluent (STE) is similar in quality to wastewater that has undergone primary treatment in a WSP (Table 5). In the following soil absorption field, biofilm growth on the porous media is responsible for further wastewater treatment (Van Cuyk, et al., 2001).

Table 5: Comparison of effluent from septic tanks versus WSPs

| | Average Concentration | |
|---|---|---|
| Parameter | Septic Tank Effluent | WSP Effluent |
| TSS (mg/L) | 79 | 112.1 |
| $BOD_5$ (mg/L) | 180 | 168.4 |
| *E. coli* (CFU/100mL) | $1.57 \times 10^6$ | $5.59 \times 10^5$ |
| TN (mg/L) | 57.7 | – |
| Ammonium-nitrogen ($NH_4$-N; mg/L) | 37.2 | 39.4 |
| Nitrate-nitrogen ($NO_3$-N; mg/L) | 0.82 | – |
| TP (mg/L) | 12.2 | – |

Source: (Lowe, et al., 2007; Krkosek, et al., 2012)

Soil absorption fields can have numerous designs; each suited to soil permeability, ground slopes, and depths to the water table (Nova Scotia Environment, 2009). The most common non-recirculating systems are the traditional gravity leachfield - for deep permeable soils with a low groundwater table, and the sloping filter - for shallow soils.

### 2.3.3.1 Vertical Filters

Gravity leachfields are the most common soil absorption fields (Crites and Tchobanoglous, 1998). They use perforated pipe to distribute STE over a large area, and flow through the soil is primarily vertical. Their use in wastewater treatment is supported by numerous volumes of research, from bench to field scale.

At the bench scale, 80 cm vertical sand columns produced a 5 log-removal of *E. coli* at 50 mm/day dosing rates (Stevik, et al., 1999). The greatest *E. coli* removal was observed in the top 10cm of media. Another study showed approximately 1 log-removal in 7.5-30cm sand columns with influent *E. coli* concentrations of 5.89 x $10^5$ CFU/100mL (Amador, et al., 2008). Other removal efficiencies observed in this second study include 81-99% of $BOD_5$, 22-28% of TN, 27-87% of ammonia, and 13-18% of phosphorus (Amador, et al., 2008). However, the high efficiency at the bench scale may not be indicative of performance at the field scale: Lamb et al. (1991) found 71% of ammonia was removed at the lab scale, compared to 58% removal in a full-scale system.

Jenssen and Siegrist (1990) summarized their previous research on field scale sand infiltration systems, concluding that effluent concentrations of $BOD_5$, TSS, total phosphorus (TP), and TN could be as low as 5 mg/L for each parameter. They also showed between 0 and $10^4$ CFU/100mL effluent fecal coliform concentrations. Peeples et al. (1991) also studied of pilot scale filters, finding $BOD_5$ removal efficiencies of 53-75%. Disposal fields studied by Harrison et al. (2000) removed 80% of TN, and produced a 3 log-reduction in fecal coliforms. Rodgers et al. (2005) had even lower effluent concentrations than Jenssen and Siegrist (1990), and achieved removal efficiencies of 99%, 100%, 94%, 88%, and 27% for BOD, TSS, TP, $NH_4$-N, and TN, respectively.

Even in filters without sand media, treatment is still possible at field scale. 82.7% of COD was removed from post-sedimentation sewage by clay–wood infiltration trenches, with 70.0% removal of ammonia, 77.7% removal of TN, and 98% removal of TP (Zhang, et al., 2005).

Finally, although not treating STE, but still relevant, vertical flow sand filters were used in the field to treat WSP effluent with average $BOD_5$, COD, TSS, TKN, and

$NH_4$-N concentrations (mg/L) of 60, 140, 44, 19, and 12, respectively. Sixty-five cm of sand achieved removal efficiencies of 70%, 49%, 69%, 70%, and 73%, respectively (Torrens, et al., 2009).

#### 2.3.3.2 Sloping Filters

Sloping filters employ the same porous media biofiltration aspects of gravity leachfields; however the flow has a degree of lateral movement. Contour trenches were first designed to overcome the topsoil depth-requirements of traditional gravity leachfields (Nova Scotia Environment, 2009). The lateral flow of wastewater in contour trenches, and other sloping media filters can be likened to the seepage flow through an exfiltrating berm. Although not studied as much as vertical filters, sloping filters have proven effective at treating STE, both in the laboratory and in the field.

In the bench scale, lateral flow sand filters achieved removal efficiencies of up to 99.1%, 99.4%, 99.9%, 32.0%, and 99.9% for $BOD_5$, TSS, Orthophosphate, TN, and ammonia, respectively (Check, et al., 1994). Complete removal of fecal coliform was observed.

A full-scale study of lateral flow sand filters found similar results for removal of $BOD_5$, TSS and *E. coli*, with minimum removal efficiencies of 98.5%, 95.5%, and 5.4 log-reduction, respectively (Harvard, et al., 2008). TP removal of 71.2-98% was observed; the greatest removal occurring in filters with media with the largest ratio of surface area to volume. In all filters, a minimum of 96.5% of ammonia was removed. Denitrification led to an average reduction in TN of approximately 63%.

After several more years of operation, the same filters still retained treatment performance, with the exception of TP (Wilson, et al., 2011). $BOD_5$, TSS, TN, Ammonia, and *E. coli* removal remained high, exceeding 97.3%, 89.6%, 43.6%, 96.4%, and 4.3 log units, respectively. Phosphorus removal, however, was significantly lower, with minimum removal falling to 43.7% (Wilson, et al., 2011). This is best explained by Cucarella and Renman (2009), whose literature review showed a trend in on-site disposal fields where phosphorus removal efficiency decreases as sorption sites on the media are exhausted.

### 2.3.4 *Geotextile Filtration Systems*

Although it is beneficial to compare the proposed method to similar biological treatment processes, ultimately, the best way to predict its performance is by comparison with experimental geotextile filters. In this section studies involving geotextile-based filtration of municipal wastewater, and other industrial wastewater streams, are summarized.

#### 2.3.4.1 Physical Filtration

This section will summarize studies showing the applicability of geotextiles as strictly physical filtration units.

Geotextile filtration has been used in dewatering applications on several occasions. Typically, very high solids in the source fluid are removed by physical retention on the geotextile fibres. The resulting fluids show significant reductions in solids, oxygen demand, and nutrients. In an agricultural application, filtration of liquid dairy and swine manure by hanging bag geotextiles was studied in South Carolina (Cantrell, et al., 2008). Geotextile bags were loaded with liquid dairy manure (containing 7,600 mg/L TSS and 90 mg/L ammonia) and liquid swine manure (containing 5,152 mg/L TSS and 210 mg/L ammonia). They found average TSS removal efficiencies of 45.6% and 39.1% for dairy and swine, respectively. Additionally, average ammonia removal efficiencies (by mass) were found to be 19.8% and 15.8% for dairy and swine, respectively. This study did not address removal through biological processes.

In another agricultural application, both woven and nonwoven geotextiles used as generic farm surfaces (i.e. paddock and feeding area linings) were able to retain significant amounts of solids and COD from cattle manure (Bicudo, et al., 2003). Finally, when used as filers to dewater oily sludge in petroleum lubricant production, column tests showed that nonwoven geotextiles layered on gravel removed more COD than traditional sand filters (Mendonça, et al., 2004). Nonwoven polypropylene geotextiles were shown to better absorb petroleum products from water if manufactured to include organosilicon (a water repellent), with sorption potential proportional to repellent percent by weight (Matveev and Gorchakova, 2007).

Outside of dewatering, testing of physical filtration in geotextiles for water treatment purposes is rare. In one study, water was recirculated through geotextile filters until clogged, or turbidity fell below 5 NTU (Mulligan, et al., 2009). The study did not run long enough for biological development. In this test, a nonwoven geotextile with apparent opening size (AoS) of 0.12mm produced the highest removal of suspended solids (approximately 99%). Removal of COD averaged between 65.5% and 71.2% for this geotextile.

Another study showed the applicability of geotextiles in storm water treatment. Nonwoven, stapled-fibre geotextiles with AoSs of 0.18 and 0.15 mm achieved TSS removal efficiencies of 79.5% and 94%, respectively (Franks, et al., 2012). Again, this test was not run long enough to achieve biological filtration, and removal efficiency was dependent on the particle diameter of sediment.

#### 2.3.4.2 Biological Filtration

As described earlier, if microorganisms are present in the influent water stream of a filtration system, or if purposefully introduced to the system, biofilm development can occur, leading to enhanced treatment as compared to solely physical filtration. In one study, a nylon mesh with AoS of 0.1 mm was used as a wastewater filtration bioreactor. It was able to produce a 98.6% reduction in BOD from 200mg/L influent, and 48.2% reduction in TN from 50 mg/L (Kiso, et al., 2000).

In a study on packed-bed geotextile filters, 99% removal of $BOD_5$, 95% removal of COD, 99% removal of TSS, and 89% removal of TKN was observed (Leverenz, et al., 2000). Total nitrification of ammonia was also achieved. A similar study of geotextile packed beds in recirculating biofilters by Hu and Gagnon (2006) was less effective. However, geotextiles were able to remove 80.6% of $BOD_5$, 54.2% of TSS, 52.4% of TN, 83.7% of ammonia, and from septic tank effluent. They also produced a 1.6 log-reduction in *E. coli*.

The most extensive and relevant study on biofiltration of municipal wastewater by geotextiles was conducted by Yaman et al. (2005), due to its use of individual geotextile layers. First, 11.36 L of wastewater was passed through a 90cm gravel column with a single layer of geotextile. The effluent from the column was reapplied to the top of the filter once per day to simulate a recirculating granular filter. After 1 week, columns with woven geotextiles provided 80% removal of ammonia compared to a 90% reduction by non-woven geotextiles. This provides more evidence that non-woven geotextiles will perform best in filtration applications. Both types of geotextile performed similarly with respect to TSS and $BOD_5$ removal; approximately 90% (Yaman, et al., 2005).

Yaman, et al. (2005) also performed a two-month study in which 55cm sand and gravel columns with two separated layers of non-woven geotextiles were loaded with municipal wastewater at 15-18°C. Loading rates started high, at 4,315 $L/m^2 \cdot day$ to prime the columns, and eventually reduced to 365 $L/m^2 \cdot day$. During this time, reductions in hydraulic conductivity of approximately 1 order of magnitude were observed (Yaman, et al., 2005).

The authors concluded that the chemoheterotrophs necessary for organic oxidation developed in the biofilm within 1 week at high loading rates, and the nitrifying bacteria necessary for ammonia oxidation become established after 4 weeks. Results indicated that after one week of loading influent TSS concentrations of 30-70 mg/L were reduced by an average of 86%, regardless of hydraulic loading rate. Similar results were seen for $BOD_5$ removal under 30-80 mg/L influent concentrations. An average of 84% removal of ammonia was also observed, but only after 6 weeks of operation (Yaman, et al., 2005).

## 2.4 Biological Filter Performance Factors

After establishing the applicability of biofilters in wastewater treatment, it is important to understand the factors influencing treatment capacity. If any of these factors inhibit treatment performance under the operating conditions of Arctic communities, alternative design must be considered. This section will explore the effect of the most pertinent factors in order to better estimate treatment performance.

### 2.4.1 *Filter Media*

In biofilters, specific surface area (the area available for biological attachment per unit volume) is paramount for treatment performance (Metcalf & Eddy, 2003). Greater specific surface area is achieved by selecting suitable filter materials, and arranging the material appropriately. It is also important to size the filter to ensure filtration pathways are long enough to provide adequate treatment.

Within trickling filter applications, plastic media have been engineered to have a higher specific surface area than rock media in an attempt to optimize treatment (Water Environment Federation, 2010). At high loading rates, engineered plastic media is recommended as the most appropriate filter media (Metcalf & Eddy, 2003). At loading rates below 1 kg $BOD_5$ per $m^3$ of media per day, plastic media and rock media perform similarly with respect to carbon removal (Grady Jr., et al., 1999). With respect to nitrification, both media perform similarly regardless of loading rate (Parker and Richards, 1986).

The type of media in other fixed-film bioreactors has been shown to significantly influence $BOD_5$ and COD removal from sanitary sewer overflow, with sand performing better than peat and fabric chips (Tao, et al., 2009). In recirculating biofilters treating residential wastewater, peat removed the least amount of $BOD_5$, ammonia, and TN (Hu and Gagnon, 2006). It also would have resulted in the lowest TSS removal, if not for sand media, which resulted in a net increase in TSS.

Treatment performance is not only affected by the type, but also the orientation of plastic media. In pilot-scale trickling filters, cross-flow plastic media removed ammonia more efficiently than vertical-flow plastic media with the same surface area (Parker and Richards, 1986). Vertically oriented trickling filter media were considered obsolete by 1989, replaced by cross-flow media (Parker, et al., 1989).

In sand filters, longer filter lengths are associated with improved filter performance for filter lengths less than 1 meter (Torrens, et al., 2009). Sloping sand filters with lengths of 3 and 5.5 meters showed similar effluent concentrations of $BOD_5$, TN, TP, and *E. coli*, however TSS removal was still greater in longer filters

(Wilson, et al., 2011). This indicates that for some parameters, there is a law of diminishing return for extending filter length.

In packed filter beds, separating the total filter media volume out into individual layers of smaller volume had little effect on the overall treatment of wastewater (Leverenz, et al., 2000). When installing filter media, it is also important to not pack the filter bed too tightly, because dry areas may form in densely-packed media, reducing nitrification rates (Parker, et al., 1989).

When using geotextiles for filtration purposes, the factors affecting porous media filter performance should still be considered. For example, geotextile chips were ranked as the best candidate for TN removal, and performed in the top echelon of four materials tested for TSS, $BOD_5$, and ammonia removal (Hu and Gagnon, 2006). This is because geotextiles have a large specific surface area for biofilm growth, and a high porosity, so wastewater application rates can be higher than in sand filters without concern over clogging (Leverenz, et al., 2000). Orientation is also important in geotextile systems. In filters with the same specific surface area of geotextile, better wastewater treatment occurs in flow through packed bed filters than flow down vertical hanging sheets, particularly at high flow rates (Leverenz, et al., 2000).

### *2.4.2 Contaminant Loading*

Filter performance and biological activity are affected by contaminant loading, whether it is the actual concentration of nutrients for metabolism, or the retention time necessary to incorporate these nutrients into microbial cells in the biofilm. As with WSPs, retention time in filters is dependent on flow rate. The effect of reduced retention time was shown by tripling flow rates through geotextile packed bed filters, resulting in higher effluent concentrations of solids, organics, and nutrients (Leverenz, et al., 2000). The negative effect of tripling flow rate was more prominent in hanging geotextiles sheets under the same dosing regimen, due to the vertical orientation of the geotextile diminishing its ability to retain water (Leverenz, et al., 2000).

Further study of vertical sand filters showed that higher hydraulic loading rates (producing lower hydraulic retention times) resulted in lower log-removal of *E. coli* (Stevik, et al., 1999). Examination of clogged porous media filtering landfill leachate showed that biological growth was only responsible for clogging at lower rates (Rowe, et al., 2000). At increased flow rates, physical filtration became the dominant clogging method.

Although high flow rates may result in lower retention times, they also result in greater mass flux of nutrients through the biofilter. In a biological filter, more nutrients allow for more biological growth. A fixed-media bioreactor achieved higher organic removal efficiencies when organic loading was increased (Tao, et al., 2009). Furthermore, in trickling filters, BOD removal efficiencies were correlated more with BOD loading rate than hydraulic loading rate (Bruce and Merkens, 1970; Bruce and Merkens, 1973).

While higher influent BOD concentrations result in better removal of BOD, nitrogen removal suffers. This is because *Nitrosomonas* and *Nitrobacter* only grow in areas with low organic concentrations (Grady Jr., et al., 1999). In a pilot-scale plastic media trickling filter, when organic loading was above 0.64 kg $BOD_5/m^3$/day (or 20 mg/L), heterotrophic bacteria outcompeted nitrifying bacteria for oxygen (Parker and Richards, 1986). As such, nitrification was only possible in the last 25% of a 4.88 m tall filter tower, once organic load was reduced to $BOD_5$ below 20 mg/L.

Comparable results were seen also in rock media trickling filters, where loading rates less than 160-190 g $BOD_5$ per cubic meter of media per day are required for 75% ammonia removal (USEPA, 1975). This range comes from a consolidation of performance data from 8 full- and pilot-scale trickling filters. When the proportion of total COD attributed to nitrifiers is low, nitrification is inhibited (Ling and Chen, 2005). In moving-bed biofilm reactors, nitrification was eliminated at biodegradable soluble COD concentrations above 4.5 g per $m^2$ of specific surface area per day (Rusten, et al., 1995). Nitrification inhibition by organic loading was also present in IFAS, where negligible nitrification occurred above COD loading rates of 1.68 kg per $m^3$ of fixed-film media per day (Ye, et al., 2009).

Wastewaters with high oxygen demand, whether carbonaceous or nitrogenous, require high dissolved oxygen (DO) to achieve total nitrification. Nitrification in trickling filters was diminished below 60-65% DO, and non-existent below 45-50% DO (Gullicks and Cleasby, 1990). Crites and Tchobanoglous (1998) also state that a DO concentration of above 1 mg/L is necessary for nitrification.

### *2.4.3 Temperature*

Temperature has a large effect on biological treatment. It has been long known that lower temperatures result in slower biological growth (Ratkowsky, et al., 1982). The internal enzymatic reactions of bacteria are also temperature dependent (Grady Jr., et al., 1999). For example, temperatures below 2°C result in inactivation of carbon-oxidizing bacteria (Metcalf & Eddy, 2003). Furthermore, the diffusion rate of nutrients through the biofilm is reduced at lower temperatures (Grady Jr., et al., 1999). However, the solubility of oxygen increases at lower temperatures (Metcalf & Eddy, 2003). Also, the dominant bacteria in the biofilm might change, since *Pseudomonas* becomes more competitive at lower temperatures (Water Environment Federation, 2010).

The combined effects of retarded growth, slow metabolism, and poor diffusion can lead to significantly lower treatment performance by biofilters. Overall less biomass was produced in biofilters at 5°C than at 20°C, resulting in 22%-38% lower organic removal efficiency (Moll, et al., 1999). The effect of temperature is even greater on nitrifying bacteria. According to (Metcalf & Eddy, 2003), nitrifying bacteria are inactivated below 5°C. Even naturally occurring polar cyanobacteria have difficulty growing at temperatures of 5°C, and are poor candidates for nitrogen and phosphorus removal from wastewater streams (Chevalier, et al., 2000).

Empirical evidence supports the nitrification trends demonstrated in pure culture analysis. At pilot-scale trickling filters in Midland, Michigan, average ammonia removal efficiencies were reduced from 85% to 75% at temperatures below 10°C (Gullicks and Cleasby, 1986). Similarly, average ammonia removal efficiencies in Bloom Township, Illinois, were reduced from 90% to 74% at temperatures below 10°C. This evidence was corroborated in Ames, Iowa, where a plastic, cross-flow trickling filter showed a decline in ammonia removal from 87.5% at 23.5°C to an average of 40% at 12.5°C (Gullicks and Cleasby, 1990).

## 2.5 Research Gaps

In biofiltration systems, media type and contaminant loading play important roles in biological development and water quality improvement. However, the performance factor with the most control over biofilm development and stability is temperature. Temperature is an issue of priority when designing passive biological wastewater treatment systems in the arctic. In the research most similar to the proposed design (geotextile biofilters or baffling systems), the effects of temperature were not addressed. Currently, there has been no research into low-temperature biofiltration with geotextile sheet substrates.

Although wastewater treatment has been observed in other biofilters at low temperatures, research into the effect of cold-temperature start-up and operation is limited. In most cases, it was only after biofilm was established that temperature was lowered. In an arctic exfiltration WSP, contaminant loading will begin at wastewater temperatures of 1-2°C, and will typically continue to operate at temperatures below 10°C. This research will provide insight into biofilm treatment performance under such arctic temperature conditions.

# 3 METHODOLOGY

Biomat development on geotextile in a simulated arctic environment and its wastewater treatment capacity was analyzed in two experiments. Experiment 1 involved analysis of treatment in a batch reactor at 10°C. Experiment 1 was designed to examine biomat development potential at arctic temperatures, as well as provide preliminary information for the design of Experiment 2.

Experiment 2 had two trials to analyze temperature effects on the filters. In both trials, treatment was analyzed under low flow conditions, similar to the flow rates observed from exfiltrating WSPs. The first trial was conducted at 10°C and the second was conducted at 2°C.

Water quality improvement and geotextile hydraulic conductivity reduction were used as primary response variables. This section describes in detail the sampling and analytical methodology used in each experiment.

Literature on geotextile clogging and treatment was first consulted to determine the most appropriate candidate for an arctic biofilm substrate. Referring to Section 2.2.1, nonwoven geotextiles were shown to provide better water treatment than the woven variety (Yaman, et al., 2005; McIsaac and Rowe, 2006). Within the nonwoven category, continuous filament and stapled fiber geotextiles performed similarly with respect to treatment (Korkut, et al., 2006). However, continuous filament geotextiles resulted in greater accumulation of shear biomass than stapled fiber, a characteristic that will be favorable in northern climates, where less overall growth is predicted due to temperature constraints (Ratkowsky, et al., 1982; Rowe, et al., 1995). As such, nonwoven geotextiles were chosen for experimentation in this study.

Samples of nonwoven, continuous filament geotextiles were acquired from Terrafix® Geosynthetics Inc. from Toronto, Ontario. Geotextile products with the numbers 400R and 600R by Terrafix® were chosen, because they encompassed the most common apparent opening sizes, and were of similar thickness. Properties of these geotextiles are shown in Table 6, and photographs are shown in Figure 2.

Table 6: Characteristics of nonwoven geotextiles used

| | 400R | 600R |
|---|---|---|
| Quoted by Terrafix® | | |
| Apparent Opening Size (mm) | 0.212 | 0.15 |
| Weight (mg/cm$^2$) | 23.7 | 33.9 |
| Permittivity (sec$^{-1}$) | 1.5 | 1.2 |
| Measured | | |
| Thickness (mm) | 2 | 3 |
| Weight (mg/cm$^2$) | 25.7 | 40.8 |
| Hydraulic Conductivity (cm/s) | $2.4 \times 10^{-1}$ | $2.5 \times 10^{-1}$ |

Source: (Terrafix, 2011)

Figure 2: Nonwoven geotextile samples used in experiments (400R on left, 600R on right)

Duplicates of the 400R variety were given the labels “4A” and “4B,” while duplicates of the 600R geotextile were given the labels “6A” and “6B.” Two control filters were used, without geotextiles, and given the labels “gA” and “gB” (i.e. they contained only gravel).

## 3.1 Experimental Apparatus

ASTM standard testing method D1987-07 is often used to compare biological clogging of geotextile and soil-geotextile systems. It outlines the procedures and apparatus dimensions to quantify changes in permeability and permittivity. This standard was used as a basis to design the experimental apparatus. Figure 3 shows the column designed for this study. Detailed dimensions are presented in Appendix A.

Figure 3: Filtration column with gravel in lower half

In total, six columns were manufactured out of clear acrylic. Each column was designed to hold a circular cutout (coupon) of geotextile between the two halves of the column. The lower half of all columns was filled with gravel to simulate the contact of the geotextile and the WSP berm. Coupons of 400R geotextile were cut for columns “4A” and “4B,” and coupons of 600R geotextile were cut for columns “6A” and “6B.” These geotextiles were placed over top of the gravel in all columns, with the exception of “gA” and “gB,” which received no geotextile over the gravel. The top half was of the columns was then secured in place with screws. An O-ring provided the seal between halves, and prevented the short-circuiting of wastewater around the geotextiles.

In Experiment 1, the gravel was unwashed, which was an experimental issue, as disturbing the columns would generate a release of inorganic sediments, impairing accurate suspended solid measurements. Only 1 week of TSS data from Experiment 1 was elevated by disturbed columns. This was remedied in Experiment 2; the gravel was rinsed thoroughly beforehand through a 6.3 mm sieve to remove all fines. A sample of berm material was collected from an exfiltrating WSP in Coral Harbor, NU. A grain size analysis determined that 50% of the material passed a 4 mm screen ($d_{50}$ = 4 mm). The gravel used in this experiment had a $d_{50}$ of 7.1 mm. The measured porosity of the unpacked gravel filling the bottom half of the column averaged 0.58.

The experimental apparatus was designed to be similar to the one suggested in ASTM Standard D1987-07. An elevated distribution tank (Figure 4) was used to feed wastewater to the columns, and maintain a constant head of wastewater. Outflows from each of the columns were collected in individual receptacles. The receptacle elevations were individually controlled to adjust the difference in head pressure across the column, thus controlling flow rate individually for each column. A general schematic of the wastewater path through the experimental system is shown in Figure 5.

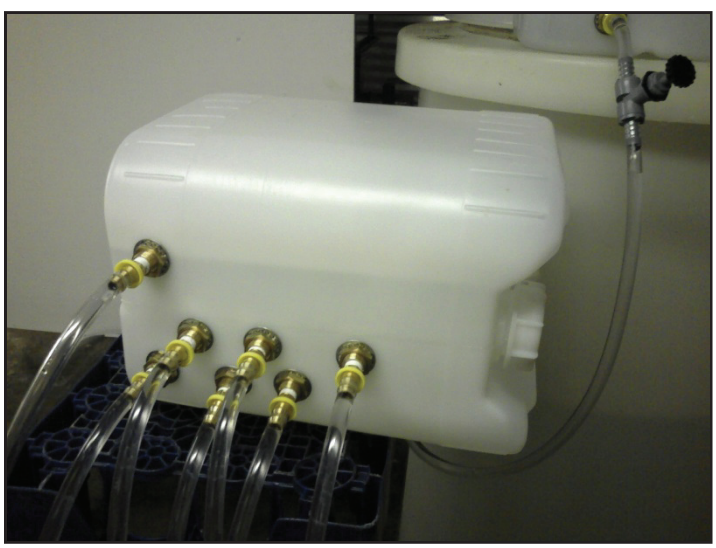


Figure 4: Distribution tank with overflow

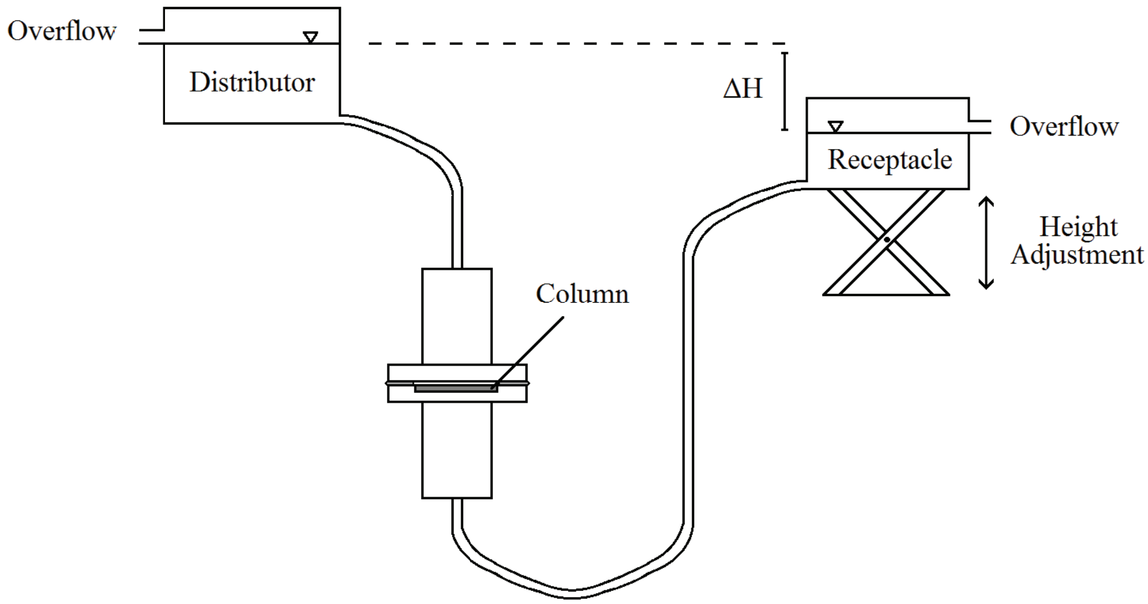



Figure 5: Schematic showing placement of column within experimental system

The difference in total head was used to drive water through the apparatus. A pumping system was not used for this task, as those capable of providing the necessary low flow rates required tubing too narrow to allow passage of suspended solids. The six columns were connected in parallel to the distribution tank to ensure each column experienced the same quality of wastewater. A reservoir tank supplied wastewater to the distribution tank by a combination of gravity feed and pumping, and only overflow from the distributor was recirculated. The overall system is shown in the flow diagram in Figure 6.

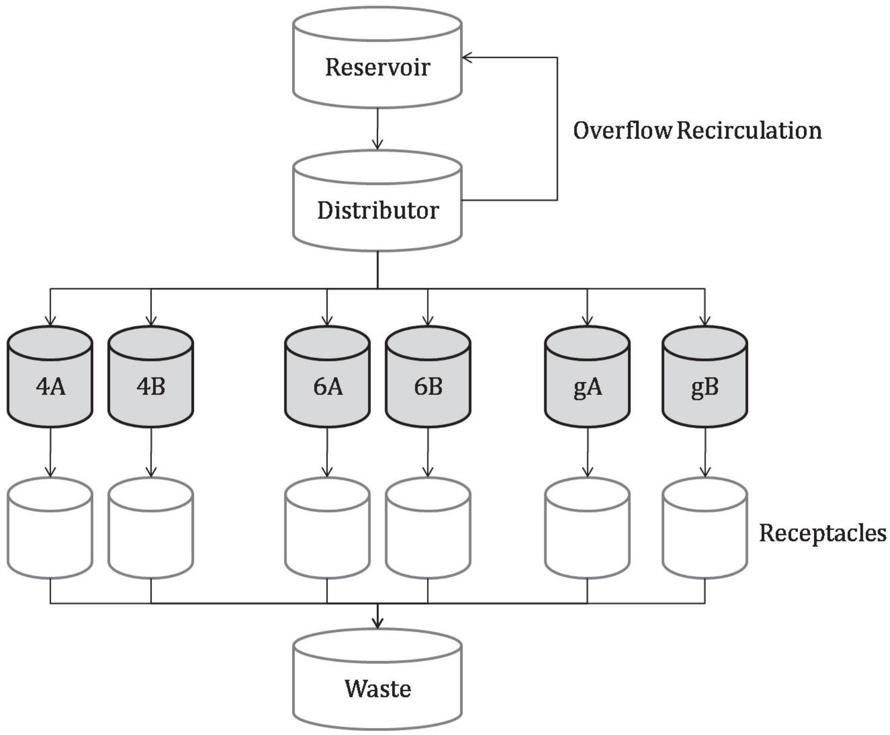



Figure 6: Flow diagram showing the configuration of distributor, columns, and waste receptacles

The reservoir and distribution vessels were each constructed from an 18.9L polyethylene pail (Figure 4). The six post-column receptacles were 3.8L polyethylene pails. Their elevations were controlled by six adjustable-height laboratory jacks. Connecting the reservoir, distributor, columns and receptacles was 0.953 cm inch inner diameter flexible PVC tubing. Connections to all vessels and columns were 0.635 cm inner diameter brass fittings, and all inline valves were PVC or nylon. The entire system is shown Figure 7.

The system was housed in a refrigerated room at Dalhousie University. The temperature of this room was adjustable, and remained within ±1°C of the set temperature for the full duration of the experiment. To inhibit the growth of chemoautotrophs, the columns, reservoir, and distributor were covered between sampling runs.

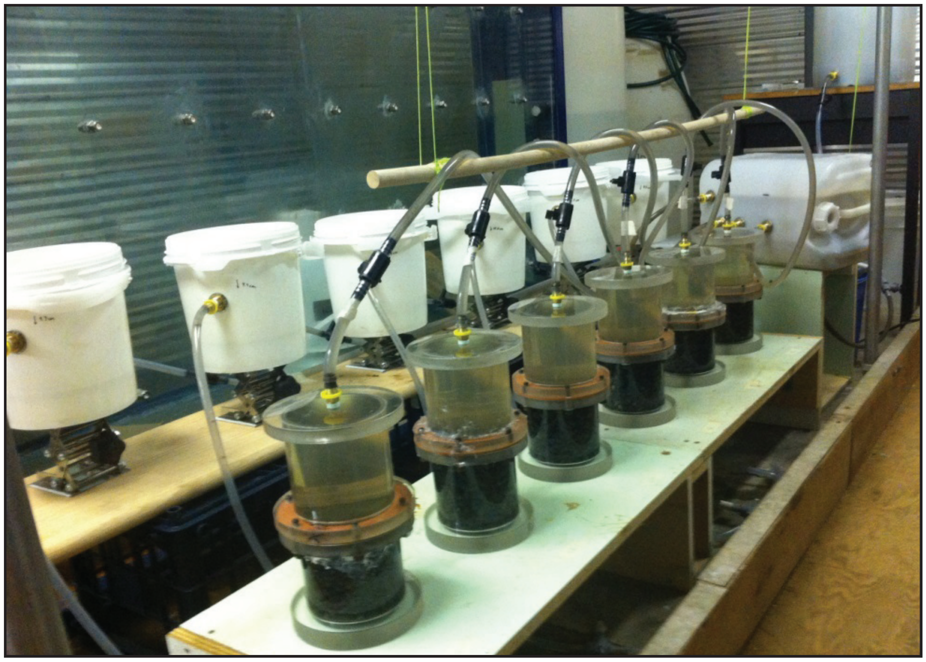


Figure 7: Photograph of the experimental apparatus in operation

## 3.2 Experimental Procedure

### *3.2.1 Experiment 1*

In Experiment 1, wastewater was applied to the columns once per week. This experiment was performed with the goal of assessing biofilm growth potential (if any) and to test the experimental design. The weekly sampling routine began with filling the columns from the distributor tank. An "influent" water sample was also taken from the distributor for water quality analysis. After one week in contact with the filter material, the water was extracted from the columns for analysis as an "effluent" sample. This weekly sampling routine was continued for 3 months.

During Experiment 1, the low volumes of wastewater passing through the filters per week were identified as a major factor contributing to the limited success of the filters (to be discussed further in Chapter 4). Furthermore, starting brand new geotextile coupons at low flow rates is unrepresentative of the high seepage rates observed during spring thaw in an actual exfiltrating WSP. As such, the filters were primed with a high flux of wastewater before starting of Experiment 2.

### 3.2.2 *EXPERIMENT 2*

In order to create the priming effect, a pump was used to drive a known volume of wastewater through each filter column prior to starting the experiment. First, to determine the appropriate priming volume, a column with a sample geotextile and gravel was prepared. Wastewater was then pumped continuously through the column until reduced flow rate was observed. Flow rate was measured at 5 minute intervals. It was found that after approximately 30L of wastewater passed the filter, a noticeable decrease in flow rate was observed.

After cleaning and preparing all columns with new geotextile coupons and gravel, each column was individually primed with 30L of wastewater. The columns were then connected back to the experimental system.

As in Experiment 1, wastewater was applied to the filters once per week. However, the goal of these trials was to operate the system under very low flow rates, representing the seepage of water through a WSP berm. The weekly sampling routine for Experiment 2 was more complex than in Experiment 1. First, the water that had been stagnant in the columns since the previous week was drained at 1 – 2 mL/s. After draining, an "influent" sample was then taken from the distributor for analysis. Water from the distributor was then allowed to flow through all columns at 1 – 2 mL/s. Initial flows through the filter media were laminar, shown by the low Reynolds numbers in Table 7. After passing through the filters, this water was collected for analysis as the "effluent" sample.

Table 7: Reynolds numbers for new geotextiles/filter media at 1 – 2 mL/s flow rate

| Flow Rate (mL/s) | 10°C | | | 2°C | | |
|---|---|---|---|---|---|---|
| | 400R | 600R | Gravel | 400R | 600R | Gravel |
| 1 | 0.019 | 0.014 | 1.447 | 0.015 | 0.011 | 1.130 |
| 2 | 0.038 | 0.029 | 2.895 | 0.030 | 0.022 | 2.261 |

## 3.3 Water Quality Monitoring

In Experiment 1 wastewater for the experiment was acquired bi-weekly. Collection frequency was increased for Experiment 2 for two reasons. First, it was important to limit possible bacterial die-off over 1 week of stagnant conditions at 10°C (Easton, et al., 2005). Second, the latter experiment required more wastewater to pass through the filters per week, and collecting this larger volume of water on one day was logistically challenging. Wastewater collected was stored until use in the refrigerated room at the temperature required for the current experiment. This water was added to the experimental system through the "reservoir" tank.

Wastewater was acquired from Timberlea Wastewater Treatment Plant (TWTP), located in the Halifax Regional Municipality (HRM) Nova Scotia. The TWTP uses a 3-stage biological treatment process. Wastewater is screened as it enters the plant before entering a primary settling unit with a 3-4 hours hydraulic retention time. Secondary treatment is provided by a rotating biological contactor. Tertiary oxidation and clarification is provided by aeration tanks before discharge to Nine Mile River system, which empties into Shad Bay. The wastewater collected for use in these experiments was taken between the primary and secondary treatment stages to mimic the level typical in arctic WSPs.

TWTP was chosen as the source of wastewater to ensure the best approximation to the wastewater of small northern communities. Water collected by trucks in Nunavut is almost exclusively residential wastewater, with limited industrial inputs (Krkosek, et al., 2012). The TWTP receives primarily residential wastewater from the community of Timberlea, Nova Scotia. Unlike some other wastewater treatment facilities in the HRM, the TWTP does not receive industrial wastewater such as landfill leachate. Although there was high variability in TWTP water quality, the comparison between the primary-treated wastewater collected from TWTP and typical arctic WSP effluent shown in Table 8 shows average concentrations are similar.

Table 8: Comparison of primary-treated wastewater from TWTP and typical WSP effluent

| Parameter | Average Concentration TWTP Wastewater | Average Concentration WSP Effluent |
|---|---|---|
| TSS | 58.6 | 112.1 |
| $BOD_5$ | 53.7 | 168.4 |
| *E. coli* | $7.40 \times 10^4$ | $5.59 \times 10^5$ |
| $NH_4$-N | 33.8 | 39.4 |
| pH | 8.4 | 7.9 |

Source: (Krkosek, et al., 2012)

Although pH is higher at TWTP, this is not a major concern. Algae growth in facultative lagoons can remove $CO_2$ from the water, causing the pH to rise, occasionally up to 9.5 (Horan, 1990). Excessive algae growth is common in WSPs during the extended summer daylight hours in Nunavut.

In both experiments, various water quality parameters were analyzed at Dalhousie University in Halifax, Nova Scotia. Standard Methods for the Examination of Water and Wastewater (Clescerl, et al., 1998) were used to determine TSS (Std. Method 2540D) and $BOD_5$ (Std. Method 5210B). *E. coli* was quantified using membrane filtration onto m-ColiBlue24 culture media (HACH Company, 2012a).

$NH_4$-N concentrations were determined using TNTplus™ 832 kits (HACH Company, 2012b), while $NO_3$-N concentrations were determined with TNTplus™ 835 kits (HACH Company, 2012c). TN was analyzed with Test 'n Tube kit 0-25 mg/L (HACH Company, 2012d). TP was analyzed with Test 'n Tube kit 0-100 mg/L (HACH Company, 2012e). Final concentrations were determined with a DR 5000 spectrophotometer (HACH Canada, Mississauga, Ontario). Total organic carbon (TOC) was measured with a TOC-VCSH analyzer (Shimadzu America Inc., Columbia, Maryland) which uses a high-heat combustion and catalytic oxidation method. Finally, specific conductivity, pH, and DO were measured using a 600R multi-parameter sonde (YSI Inc., Yellow Springs, Ohio).

## 3.4 Hydraulic Conductivity Testing

The second method of assessing filter performance was observing changes in hydraulic conductivity resulting from physical and biological clogging. The benefit of designing the experimental apparatus to approximate the specifications of ASTM standard D1987 −07 was that it incorporates the designs and methodology of ASTM standards D5856−95 and D5493−06. It was therefore not necessary to remove the columns from the experiment apparatus to attach them to a permeameter, minimizing logistical barriers to repeated measurements.

As such, hydraulic conductivity was measured before, after, and at weekly intervals during each experiment. After completing the weekly sampling of wastewater for water quality monitoring, hydraulic conductivity would be determined. Using height adjustments on the effluent receptacles, it was possible to accurately control and measure the head difference (ΔH) across the columns. The system was then allowed to flow to fill a known volume of water, and the length of time was recorded. This timed fill was repeated three times, and the average time was calculated. The measured parameters were entered into Darcy's Law describing finite one-dimensional flow in porous media (Equation 1) to calculate hydraulic conductivity of the filters.

Equation 1:

$$\frac{V}{t} = A \cdot K \cdot \frac{\Delta H}{L}$$

Where $V$ is the known volume (mL), $t$ is the average time (s), $A$ is the area of the filter ($cm^2$), $K$ is hydraulic conductivity (cm/s), $\Delta H$ is the difference in total head across the filter (cm), and $L$ is the length of the filter (cm).

Friction and minor head losses were calculated based on the characteristics of the tubing and valve fittings used, as well as the flow rate through the system and the temperature of the water. On average, combined head dynamic losses resulted in between 0% and 38% of the total head difference observed, depending on fluid velocity and turbulence. The resulting losses were subtracted from the total ΔH before calculating hydraulic conductivity using Equation 1. Sample hydraulic conductivity calculations are shown in Appendix B.

## 3.5 Statistical Analysis

Once determined, influent and effluent wastewater concentrations were compared using paired student's *t*-tests. These were compared on the 95% confidence level to determine if significant concentration reductions were observed across the filters.

Average effluent concentrations between the duplicates of each geotextile type were compared against the average effluent concentrations from the columns with gravel only. These tests were conducted at 95% confidence using *t*-tests that assume unequal variances. In all cases, arithmetic means were used for statistical analysis, with the exception of bacteria counts, where geometric mean was used.

## 3.6 Biomat Analysis

The USEPA (2002) defines biomat as "a layer of organic and inorganic material and bacteria." This layer is responsible for the chemical, physical, and biological manipulation of wastewater composition, and controls wastewater flow rate through porous media biological filters (USEPA, 2002).

Biomat development was analyzed by determining biomat dry weight per unit area. After completion of the experiments, the geotextile coupons were removed from the columns and placed on sterilized aluminum foil sheets. The sheets were baked at 60°C for 24 hours to remove moisture without volatilizing solids. The dry geotextile coupons were cut into rectangles representative of the whole biomat. The weight of each cutting was recorded and the rectangle area was measured.

# 4 Results and Discussion

## 4.1 Experiment 1

In Experiment 1, columns holding geotextile coupons and gravel were operated at 10°C for 11 weeks. To reiterate, the columns operated under batch flow conditions such that the filters were in contact with the same wastewater for 7 days. Weekly hydraulic conductivity measurements were taken, shown in Figure 8.

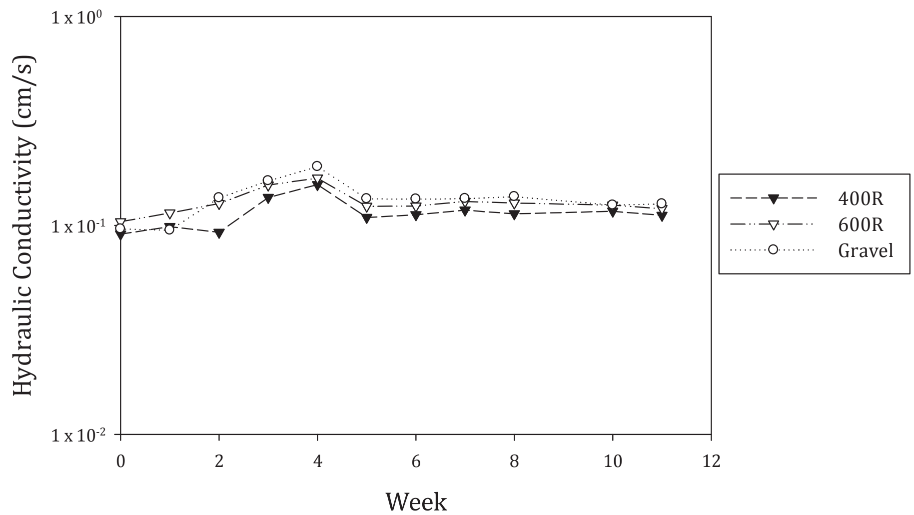



Figure 8: Experiment 1 hydraulic conductivity

The total change in hydraulic conductivity observed over the 3-month study was minimal. A slight increase in hydraulic conductivity occurred over the first month, followed by an abrupt return to original values. It is speculated that this was caused by a preferential flow path developing through the filters before biomat development began to clog the filters. Although both geotextile types appear to have lower hydraulic conductivity than the gravel control, only the 400R was statistically lower.

At the conclusion of the 3 month experiment, the columns were disassembled to recover the geotextile coupons. After drying, the coupons were cut, weighed, and measured. A comparison of average dry coupon weight per unit area is shown in Table 9.

Table 9: Dry weight of geotextile coupons before and after Experiment 1

| Dry Weight ($mg/cm^2$) | 400R | 600R |
|---|---|---|
| Initial | 23.65 | 41.81 |
| Average Final | 30.09 | 47.94 |
| Average Increase | 27.2% | 14.6% |

After receiving the same quality of wastewater for 3 months, both geotextiles increased in dry weight. Tallying the volume of wastewater passing through each filter, it was discovered one of the 400R duplicates received approximately 10% more wastewater than the other columns. This is likely the factor responsible for the greater biomat development by the 400R geotextile on average.

Wastewater quality was tested before and after dosing the batch reactor, with paired influent and effluent readings separated by a week of stagnation. One parameter tested was pH, monitored to identify anomalies that might influence microbial growth and water treatment. After the first month of the experiment, lower raw wastewater pH was observed (Figure 9). Consulting with TWTP, it was learned that a temporary switch to lime pH adjustment was responsible. However, pH remained between 7 and 8.7; a range consistent with the pH of arctic WSPs. Overall, no statistically significant change in pH between influent and effluent was observed.

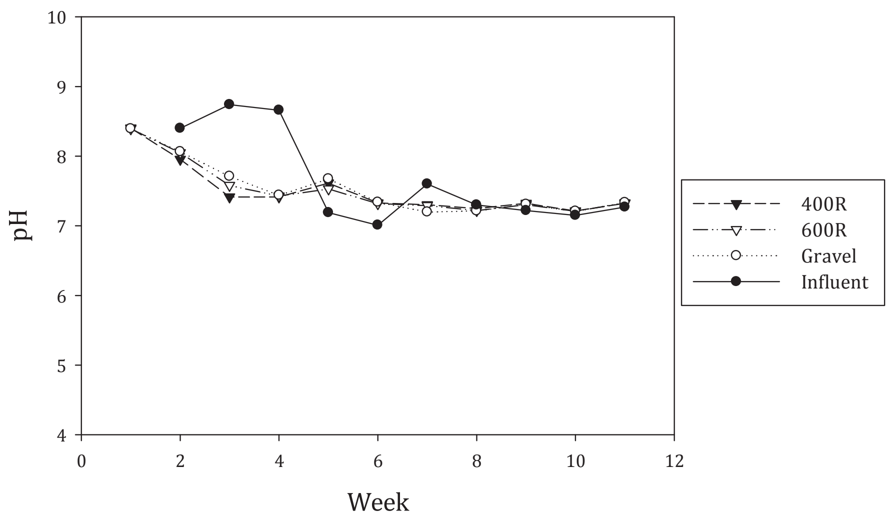



Figure 9: Experiment 1 influent and effluent pH levels

In addition to pH, TSS, $BOD_5$, *E. coli*, $NH_4$-N, TN, TP, and TOC were measured weekly. A statistical summary of these results are shown in Table 10.

With this data, average removal rates for each parameter were calculated; shown in Table 11. Columns which produced a statistically significant ($p < 0.05$) difference between influent and effluent concentrations in both duplicates are indicated with an asterisk beside its removal efficiency.

Table 10: Statistical summary of treatment performance in experiment 1

| Parameter | | Influent Concentration | Effluent Concentration | | |
|---|---|---|---|---|---|
| | | | 400R | 600R | Gravel |
| TSS (mg/L) | Mean | 69.3 | 20.7 | 25.2 | 31.1 |
| | Median | 69.7 | 18.7 | 20.8 | 21.1 |
| | Min | 52.0 | 8.6 | 9.3 | 15.8 |
| | Max | 109.6 | 48.8 | 75.3 | 115.0 |
| | St. Dev. | 18.2 | 10.2 | 17.0 | 27.2 |
| $BOD_5$ (mg/L) | Mean | 67.3 | 33.0 | 37.2 | 40.3 |
| | Median | 72.6 | 33.5 | 38.0 | 41.4 |
| | Min | 22.5 | 14.3 | 10.5 | 12.8 |
| | Max | 103.3 | 64.0 | 75.0 | 72.0 |
| | St. Dev. | 22.4 | 12.8 | 17.7 | 16.1 |
| *E. coli* (CFU/100mL) | Mean | 1.6E+05 | 8.0E+03 | 1.9E+04 | 2.1E+04 |
| | Median | 2.5E+05 | 1.3E+04 | 1.9E+04 | 2.4E+04 |
| | Min | 6.7E+03 | 1.0E+00 | 3.9E+03 | 3.0E+03 |
| | Max | 6.5E+05 | 8.9E+04 | 1.0E+05 | 1.3E+05 |
| | St. Dev. | 6.8E+05 | 8.0E+04 | 4.5E+04 | 5.3E+04 |
| $NH_4$-N (mg/L) | Mean | 40.0 | 39.1 | 39.6 | 39.9 |
| | Median | 38.3 | 40.2 | 39.7 | 39.8 |
| | Min | 27.7 | 23.7 | 22.6 | 24.3 |
| | Max | 49.4 | 49.3 | 50.7 | 52.1 |
| | St. Dev. | 6.8 | 7.4 | 8.0 | 7.6 |
| TN (mg/L) | Mean | 36.5 | 29.6 | 30.1 | 30.8 |
| | Median | 36.4 | 29.6 | 30.2 | 31.0 |
| | Min | 22.1 | 19.2 | 19.7 | 19.2 |
| | Max | 44.0 | 39.2 | 40.8 | 40.8 |
| | St. Dev. | 6.7 | 5.6 | 5.6 | 6.4 |
| TP (mg/L) | Mean | 3.4 | 1.3 | 1.4 | 1.4 |
| | Median | 3.4 | 1.1 | 1.4 | 1.4 |
| | Min | 2.12 | 0.00 | 0.10 | 0.26 |
| | Max | 4.8 | 2.7 | 2.7 | 2.9 |
| | St. Dev. | 1.0 | 0.85 | 0.84 | 0.91 |
| TOC (mg/L) | Mean | 25.5 | 26.7 | 26.7 | 28.2 |
| | Median | 27.4 | 25.0 | 26.5 | 27.1 |
| | Min | 12.4 | 17.9 | 15.1 | 17.5 |
| | Max | 42.1 | 42.6 | 43.8 | 43.3 |
| | St. Dev. | 9.1 | 7.0 | 7.6 | 7.4 |

Table 11: Summary of wastewater constituent concentration reduction performance

| Parameter | 400R | 600R | Gravel |
|---|---|---|---|
| TSS | 69.7%* | 63.3%* | 55.0%* |
| $BOD_5$ | 46.6%* | 39.8%* | 38.4%* |
| *E. coli* | 1.37* | 1.01* | 0.92* |
| $NH_4$-N | 1.26% | -1.42% | -1.54% |
| TN | 14.5%* | 13.2%* | 11.1% |
| TP | 62.3%* | 57.4%* | 57.0%* |
| TOC | -8.67% | -7.79% | -15.2% |

*Average reduction in concentration was statistically significant ($p < 0.05$)

An in-depth analysis of each of the above parameters is discussed in the next sections.

### *4.1.1 Total Suspended Solids*

Total suspended solid removal was anticipated, based on the history of physical clogging in landfill leachate systems (Section 2.2). Table 10 provides the average effluent concentrations of TSS from the gravel control columns as 31.1 mg/L, which did not meet the 25 mg/L Environment Canada regulation. The 400R geotextile met this regulation at 20.7 mg/L, while the 600R geotextile produced 25.2 mg/L on average. TSS concentrations were statistically lower in the effluent than in the influent (Figure 10).

TSS removal efficiency improved with time for both the geotextiles and control (Figure 11). This may be caused by narrowing of the fluid pathways through the filter, either by biological growth or physical clogging. The 400R and 600R geotextile columns achieved a maximum removal efficiency of 82.7% and 84.1%, respectively. The gravel control achieved a maximum of 72.2%.

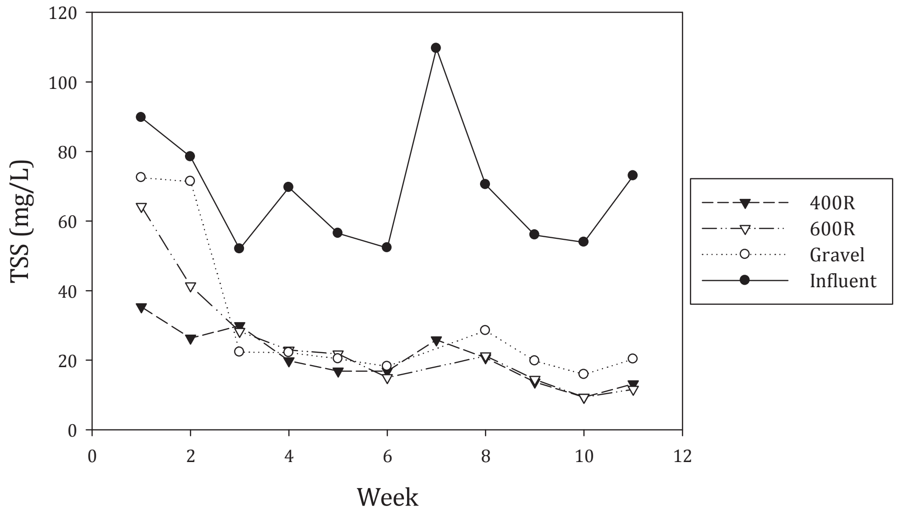



Figure 10: Experiment 1 influent and effluent TSS concentrations

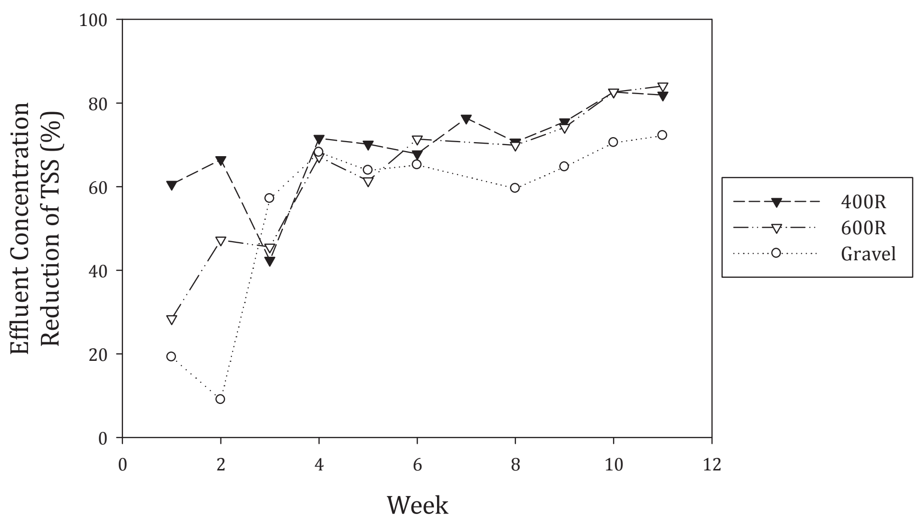



Figure 11: Experiment 1 effluent concentration % reduction of TSS

Average removal efficiencies of 69.7% and 63.2% were achieved by the 400R and 600R columns, respectively. The gravel control averaged 55.0% removal of TSS. In all columns, this removal of TSS was statistically significant.

Both geotextile sizes produced significantly lower effluent TSS concentrations than the gravel control. There was, however, no statistical difference between the two geotextiles with respect to TSS removal.

### *4.1.2 5-day Biochemical Oxygen Demand*

The average effluent concentrations of $BOD_5$ were 33.0 mg/L, 37.2 mg/L, and 40.3 mg/L for the 400R, 600R and control columns, respectively (Table 10). Although lower than the influent (Figure 12), average effluent levels from all columns were higher than the 25 mg/L Environment Canada regulation.

$BOD_5$ removal efficiency removal did not change over time in both geotextile and control columns, as shown in Figure 13. Removal efficiencies averaged 46.6% and 39.8% by the 400R and 600R columns, respectively. To compare, the gravel control averaged 38.4% removal of $BOD_5$. In all columns, $BOD_5$ effluent concentrations were statistically significant lower than influent.

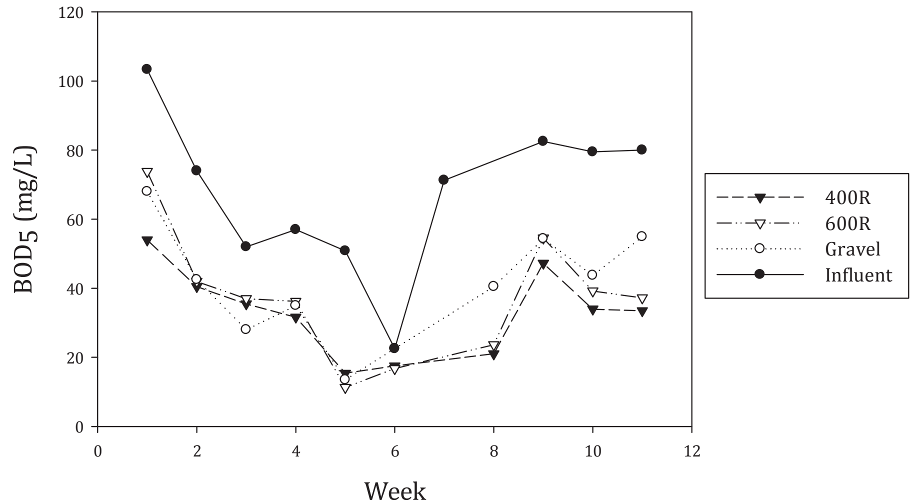



Figure 12: Experiment 1 influent and effluent $BOD_5$ concentrations

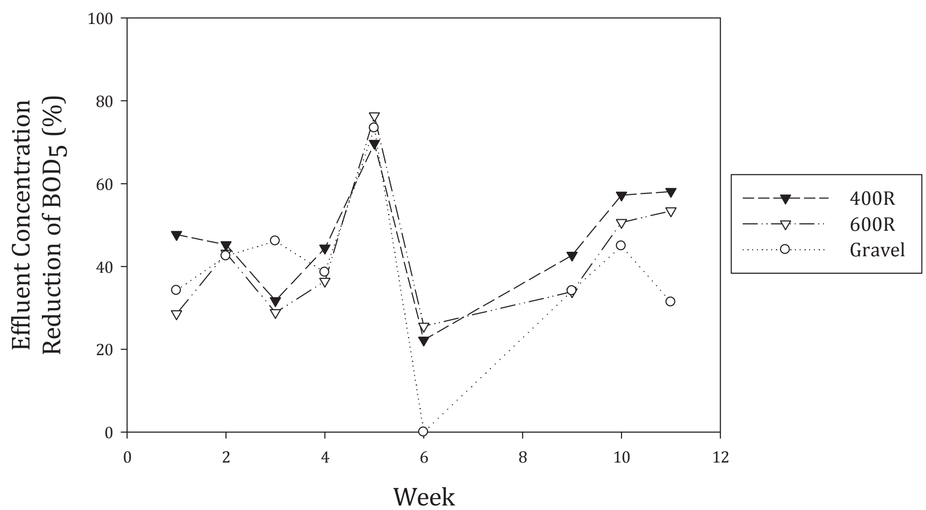



Figure 13: Experiment 1 effluent concentration % reduction of $BOD_5$

Statistically, the 400R geotextile produced significantly lower effluent $BOD_5$ concentrations than the 600R. In fact, no significant difference between the 600R effluent and the gravel control effluent was found. Only the 400R geotextile produced significantly lower effluent concentrations than the gravel. Based on this information, the gravel was likely a major contributor to the removal of $BOD_5$.

When comparing the individual columns, it was apparent that the column that received 10% more wastewater outperformed the others. This column (4A) resulted in better average removal than its counterpart, compared to similar performance by duplicates of the 600R geotextile (Table 12).

Table 12: $BOD_5$ concentrations and removal efficiency in Experiment 1 column duplicates

| | Influent | 4A | 4B | 6A | 6B |
|---|---|---|---|---|---|
| Mean (mg/L) | 67.3 | 32.6 | 33.5 | 37.1 | 37.3 |
| Reduction (%) | | 47.9 | 45.3 | 38.5 | 40.9 |

The improved performance of the 4A column when receiving more wastewater is consistent with the literature studied in Section 2.4.2. Tao et al. (2009) showed that filters receiving higher $BOD_5$ loading achieved greater $BOD_5$ removal. This is likely due to the increased availability of substrate causing greater microbial growth. A similar result was observed in column 4A under greater loading. As such, it is reasonable to predict that the filters have the capacity to treat higher loading rates. This meant the columns would not be overloaded when increasing the volume of wastewater filtered in Experiment 2 to better represent actual WSP exfiltration rates.

### 4.1.3 *E. coli*

In Experiment 1, *E. coli* removal was low – averaging approximately 1 order of magnitude – but statistically significant for all columns (Figure 14). Average *E. coli* removal was 1.37, 1.01, and 0.92 log units for the 400R, 600R, and control columns, respectively (Figure 15).

Even though the 400R column outperformed the 600R column, performance by both geotextiles was not statistically different from the gravel control.

Although a component of bacteria removal in Experiment 1 may have been biofiltration, it is likely that the die-off under cold temperatures and stagnant water conditions also contributed to overall removal. This is consistent with the literature, in which study of bacterial die-off at low temperatures also showed a 1 log-reduction in *E. coli* over 7 days at 9°C (Easton, et al., 2005).

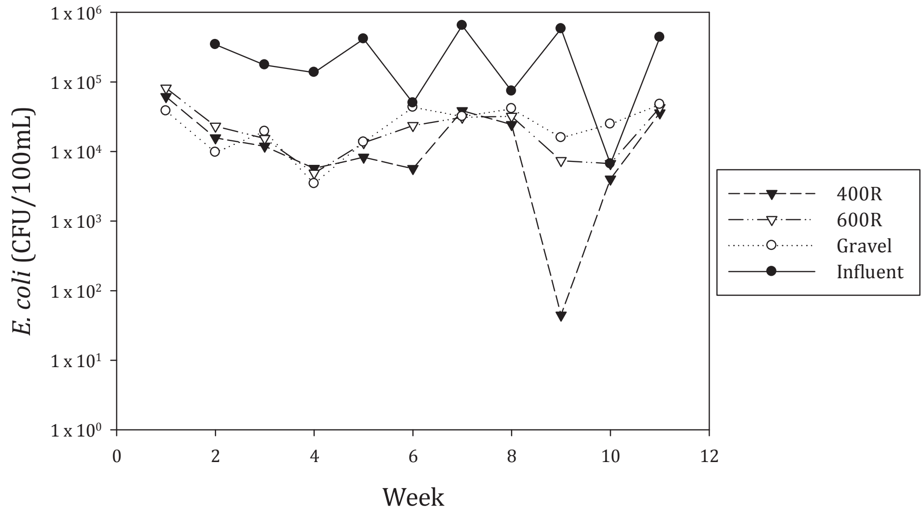



Figure 14: Experiment 1 influent and effluent *E. coli* concentrations

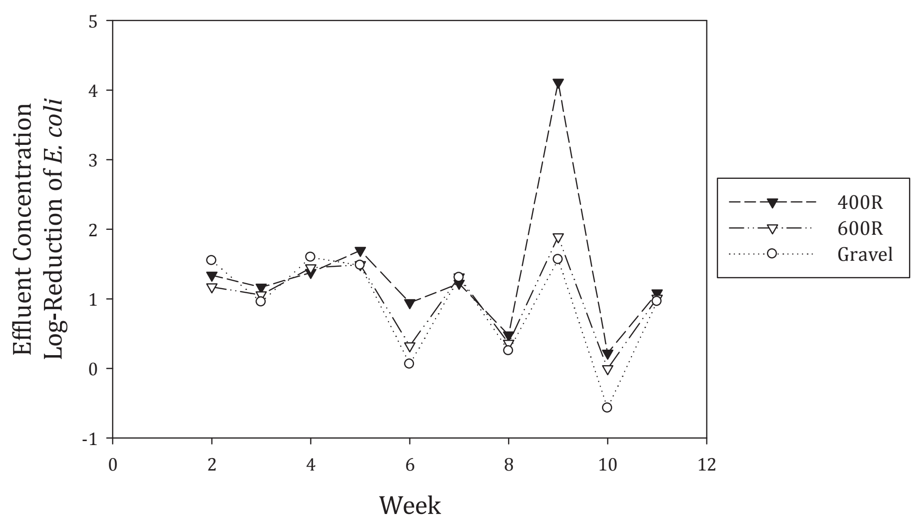



Figure 15: Experiment 1 effluent concentration log-reduction of *E. coli*

### 4.1.4 *Nitrogen*

Overall, no statistically significant ammonia removal occurred in any column in Experiment 1 (Table 11).

This is presumably due to a combination of two factors: (i) 10°C temperatures reduced the growth and metabolism rate of nitrifiers; and (ii) $BOD_5$ concentrations above 25mg/L in 80% of measurements. As a result, nitrifiers were not able to compete with heterotrophic bacteria (Parker and Richards, 1986), and minimal ammonia removal was achieved. Additionally, aerobic conditions are necessary for nitrification. Under stagnant conditions for 7 days, it is likely anaerobic conditions developed, limiting ammonia oxidation.

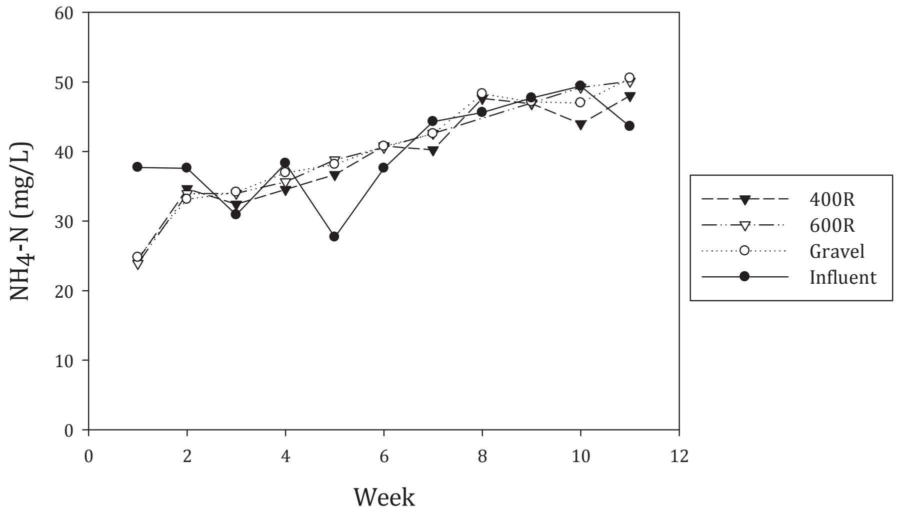



Figure 16: Experiment 1 influent and effluent $NH_4$-N concentrations

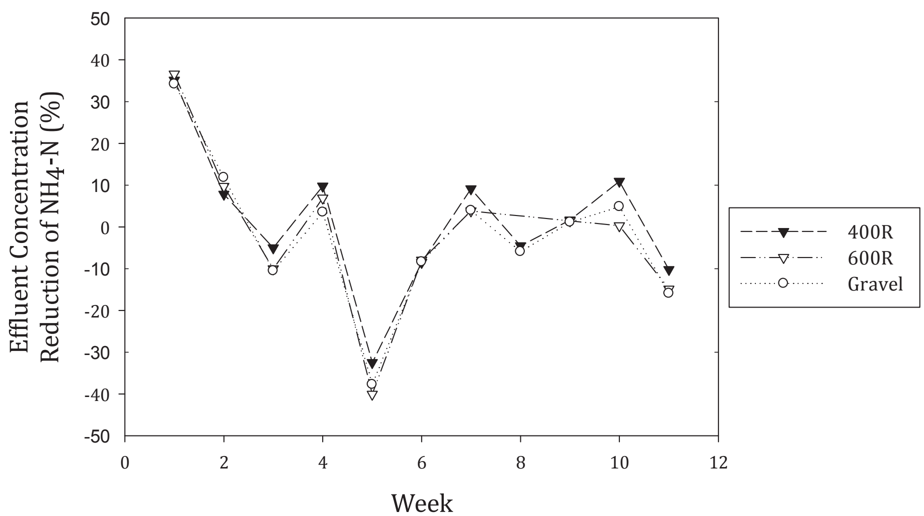



Figure 17: Experiment 1 effluent concentration % reduction of $NH_4$-N

The conclusion that ammonia oxidation was limited was supported by examining nitrate dynamics. During the 11 week study, only 8 weeks had detectable influent nitrate concentrations, shown in Figure 18. Non-detect values are represented at 50% of the 0.23 mg/L detection limit.

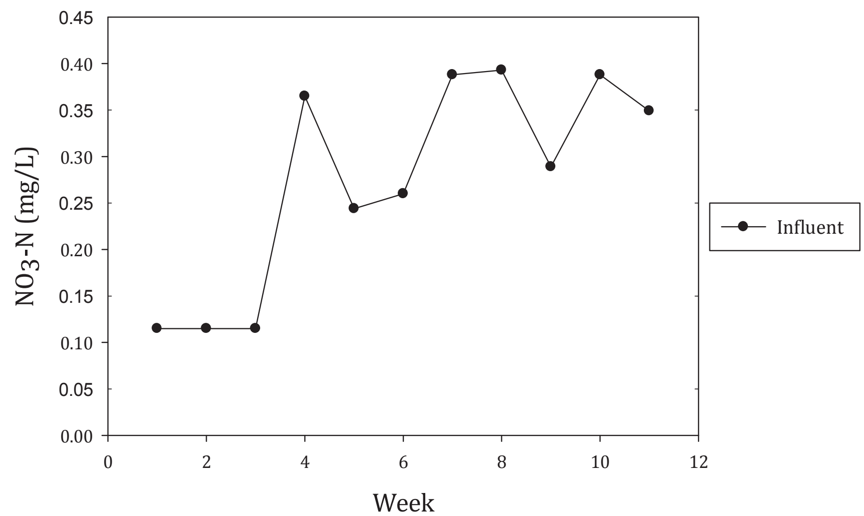



Figure 18: Experiment 1 influent $NO_3$-N concentrations.

However, no effluent samples had detectable nitrate concentrations. To achieve complete denitrification, anaerobic conditions were necessary. It is unknown if aerobic conditions occurred beforehand, since dissolved oxygen levels were not monitored. If aerobic nitrification of ammonia was present, the evidence (in the form of elevated nitrate concentrations) was completely removed.

Although there was no statistically significant ammonia removal, there was an overall decrease in TN in the system. In all columns, with the exception of one of the gravel controls, statistically lower effluent concentrations were observed (Table 11; Figure 19).

Average TN reductions were 14.5%, 13.2%, and 11.1% for the 400R, 600R, and control, respectively. Reduction in TN may be the result of denitrification, or more likely, the result of incorporation into the cells of the biofilm. Analysis showed that only the 400R geotextile column statistically outperformed the gravel control column (Figure 20).

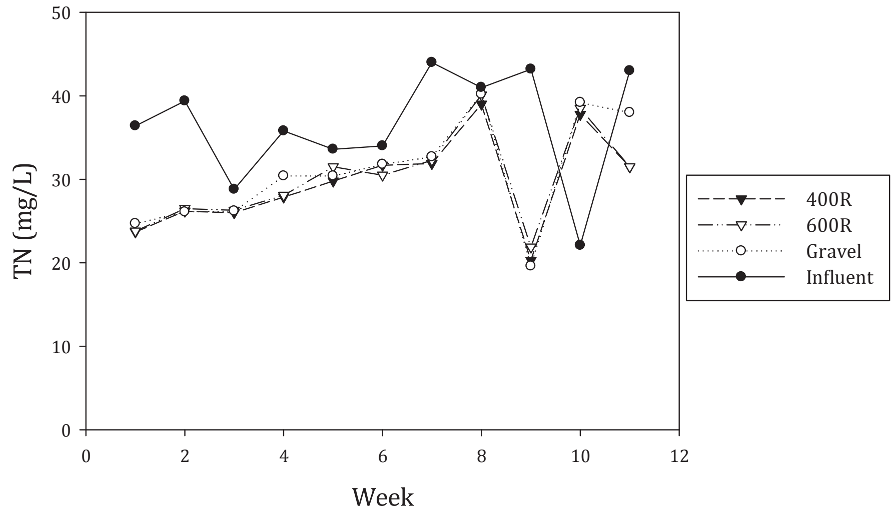



Figure 19: Experiment 1 influent and effluent TN concentrations

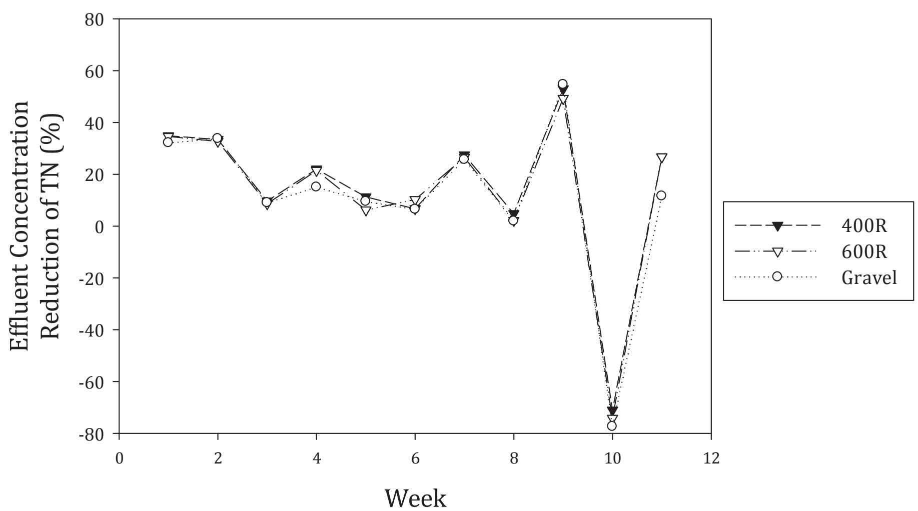



Figure 20: Experiment 1 effluent concentration % reduction of TN

### 4.1.5 *Phosphorus*

In Experiment 1, all columns produced a statistically significant reduction in TP. The gravel control columns averaged a 57.0% reduction in TP. The 600R geotextile performed similarly at 57.4% removal, and the 400R removed 62.3% of TP. Only the 400R geotextile removed significantly more TP than the gravel alone. It is likely that the gravel was the main contributor to TP removal, as only a slight improvement was observed by adding the 400R geotextile (Figure 21).

Over the course of the experiment, the TP removal rate declined. This is consistent with the literature, in that only a limited number of phosphorus sorption sites are available in the filters (Cucarella and Renman, 2009). As more wastewater is applied, these sites become occupied, leaving fewer for future phosphorus treatment. This resulted in much lower removal rates at the end of the study (Figure 22), and higher effluent TP concentrations.

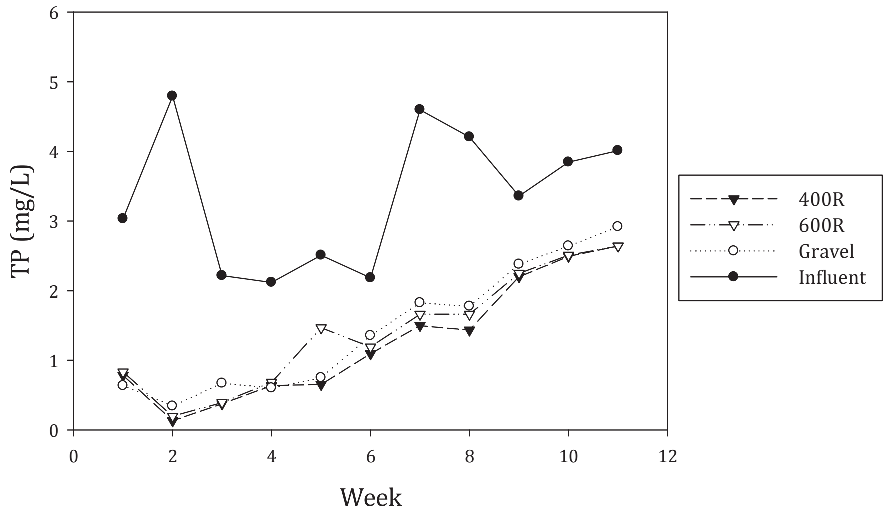



Figure 21: Experiment 1 influent and effluent TP concentrations

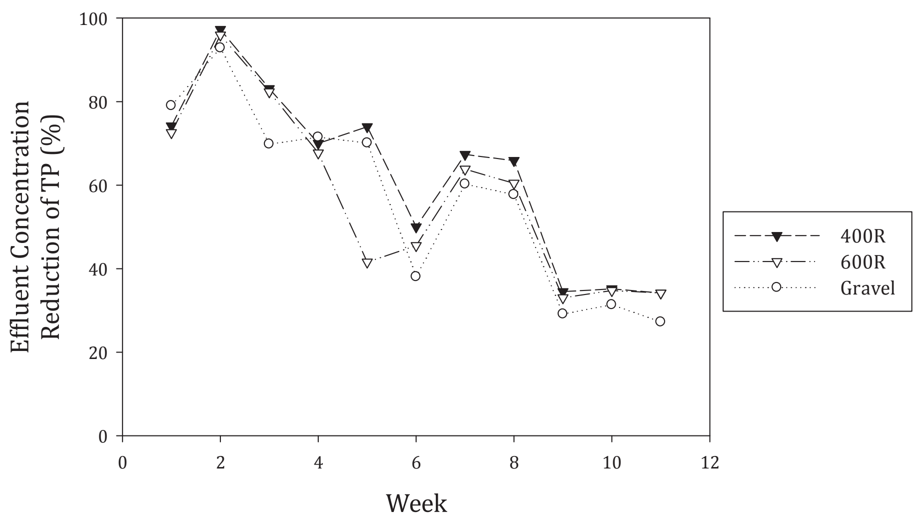



Figure 22: Experiment 1 effluent concentration % reduction of TP

### 4.1.6 *Total Organic Carbon*

Influent and effluent TOC concentrations were not statistically different (Table 11). There was also no statistical difference between the two geotextiles (Figure 23), although both had lower effluent TOC concentrations than the gravel control.

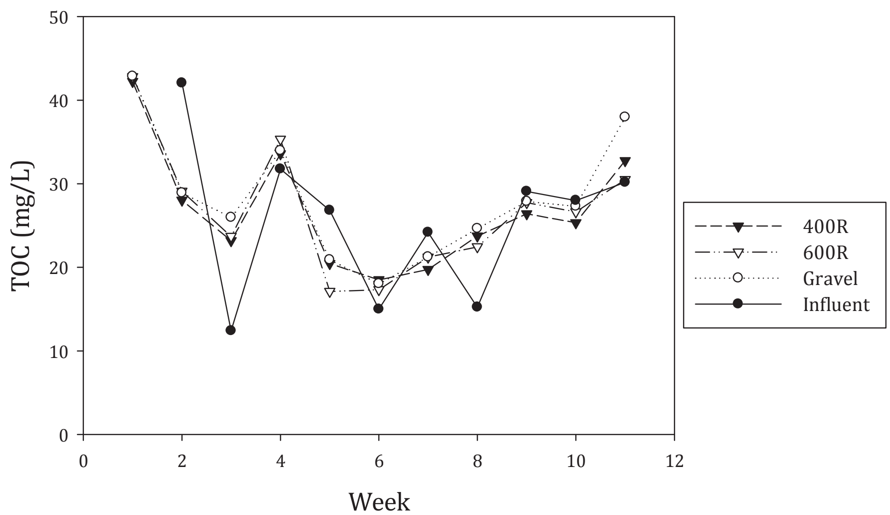



Figure 23: Experiment 1 influent and effluent TOC concentrations

## 4.2 EXPERIMENT 2

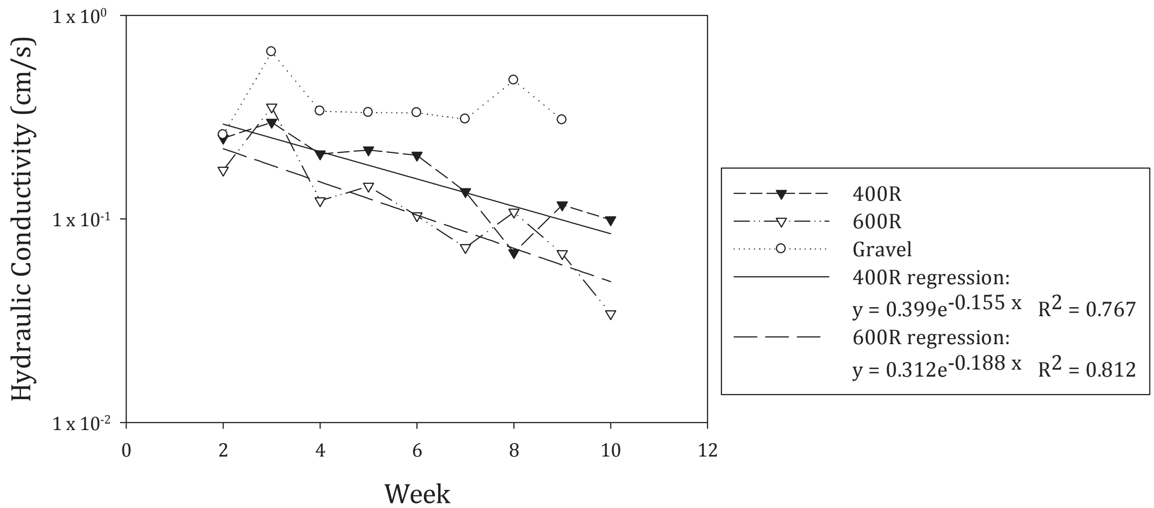
Several outcomes from Experiment 1 were of note, particularly for moving forward and designing an improved experimental process. These include:

1. Biomat development was achieved given the time and temperature constraints of the experiment.
2. Statistically significant removal of TSS, $BOD_5$, *E. coli*, TN and TP from the wastewater justified further experiments at similar and lower temperatures.
3. Better treatment and biomat development occurred in a column receiving a slightly larger volume of wastewater.

With this information in mind, and the goal of better representing the flow characteristics of actual exfiltrating WSPs, the procedure for Experiment 2 was developed (Section 3.2.2).

In Experiment 2, two trials were conducted. The first trial used primed geotextile coupons and gravel columns tested at 10°C for 12 weeks. New geotextile coupons and gravel columns were primed for the second trial, and tested at 2°C for 12 weeks. Each week, approximately 10L of wastewater flowed through each filter at 1-2 mL/s. Weekly hydraulic conductivity measurements were taken, shown in Figure 24. Exponential regressions of the hydraulic conductivity time series are also shown.

(a)




(b)

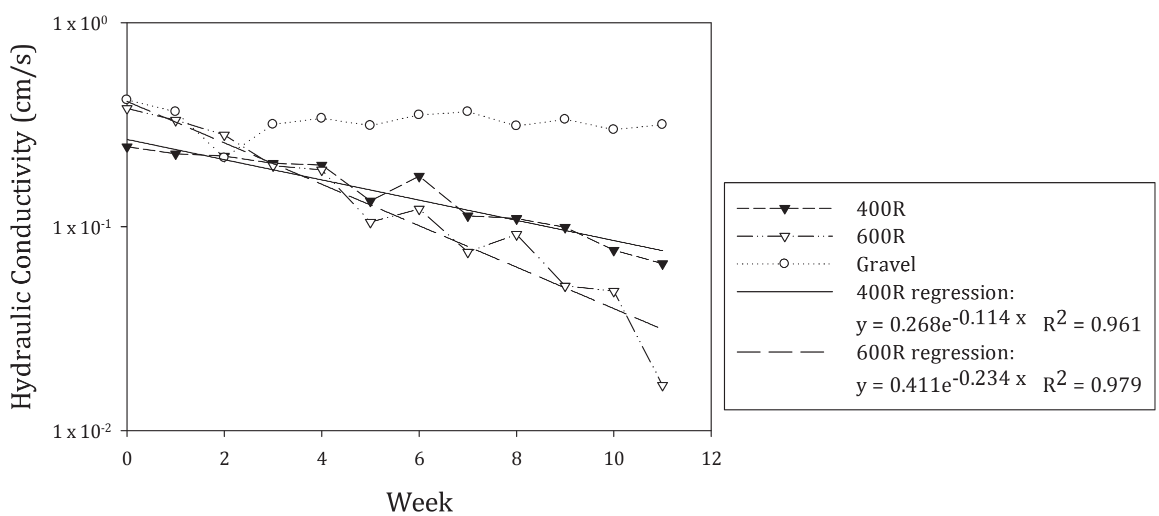



Figure 24: Experiment 2 hydraulic conductivity at 10°C (a) and 2°C (b)

At 10°C, no change in hydraulic conductivity was observed in the gravel control from weeks 2 to 10. However, during the same period, a 0.92 log-reduction in hydraulic conductivity was observed in the 400R columns and a 1.62 log-reduction was observed in the 600R columns.

Over the same period of time (week 2 to week 10) at 2°C, no change in hydraulic conductivity was observed in the gravel control. However, a 1.06 log-reduction in hydraulic conductivity was observed in the 400R columns, and a 1.76 log-reduction was observed in the 600R columns.

Overall, there was no statistical difference between hydraulic conductivity measurements at 10°C and 2°C. The decrease in hydraulic conductivity in the geotextiles appeared to follow exponential functions, which is consistent with the literature (Rowe, et al., 2000). This may be an indication of microbial biomat formation, as microorganisms are known to follow an exponential growth curve.

After 3 months of hydraulic conductivity and water quality data collection, the columns were disassembled and the geotextile coupons were recovered. After drying, the coupons were cut, weighed, and measured. A comparison of average dry coupon weight per unit area at 10°C and 2°C is shown in Table 13. Greater biomat was accumulated at the warmer temperature.

Table 13: Dry weight of geotextile coupons before and after Experiment 2

| | 10°C | | 2°C | |
|---|---|---|---|---|
| Dry Weight ($mg/cm^2$) | 400R | 600R | 400R | 600R |
| Initial | 27.77 | 39.71 | 27.77 | 39.71 |
| Average Final | 38.59 | 50.31 | 33.67 | 48.70 |
| Average Increase | 38.9% | 26.7% | 21.3% | 22.6% |

In addition to the hydraulic conductivity, the quality of the wastewater was analyzed on a weekly basis. As the system operated under low-flow conditions, the wastewater was analyzed before and after application to the columns. General water quality parameters of pH, DO, and specific conductivity were collected starting in Week 2 to determine if their variation would have a significant influence on treatment performance.

During the 10°C trial, influent pH fluctuated between 7.5 and 9, averaging approximately 8.5 (Figure 25). At 2°C, pH reached a maximum of 10.1 during one week, but returned to its average of 9.3 shortly thereafter. The slightly higher pH is still consistent with the higher pH levels produced by algae in aerobic lagoons (Crites and Tchobanoglous, 1998).

At 10°C, a statistically significant pH decrease was observed in all columns. This may be due to nitrification of ammonia; a process that produces excess hydrogen ions which acidify the water (Crites and Tchobanoglous, 1998). This decrease in pH was not observed at 2°C.

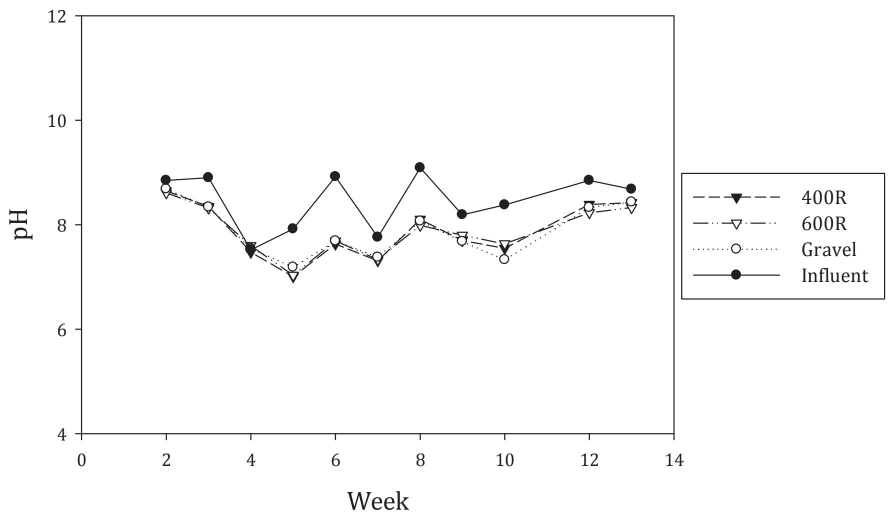
An additional parameter added to the Experiment 2 analysis was DO. This was done to develop a better understanding of the oxygen state of the wastewater, and if oxygen limitations inhibited biological growth. Weekly DO measurements pre- and post-filter are shown in Figure 26. As expected, higher DO was observed at lower temperature (Metcalf & Eddy, 2003).

For the most part, influent DO concentrations remained above 1 mg/L. It was only in the last 3 weeks of the 10°C trial that influent DO dropped below 1 mg/L. It was only during these last weeks at 10°C that effluent DO concentrations also fell below 1 mg/L. Otherwise, oxygen was never entirely depleted by the biofilters.

Specific conductivity measurements at 10°C and 2°C are shown in Figure 27. At 10°C, there was no statistical difference between influent and effluent. Specific conductivity was higher during the 2°C trial, and a small, but statistically significant increase of approximately 40 μS/cm was observed in all columns.

In addition to pH, DO, and specific conductivity, TSS, $BOD_5$, *E. coli*, $NH_4$-N, TN, TP, and TOC were also measured weekly. The results from both trials are shown consolidated in Table 14.

(a)




(b)

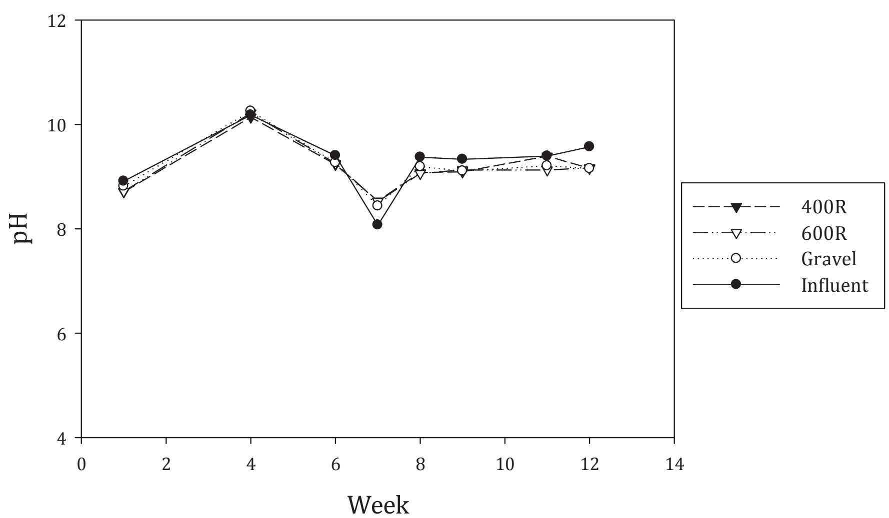
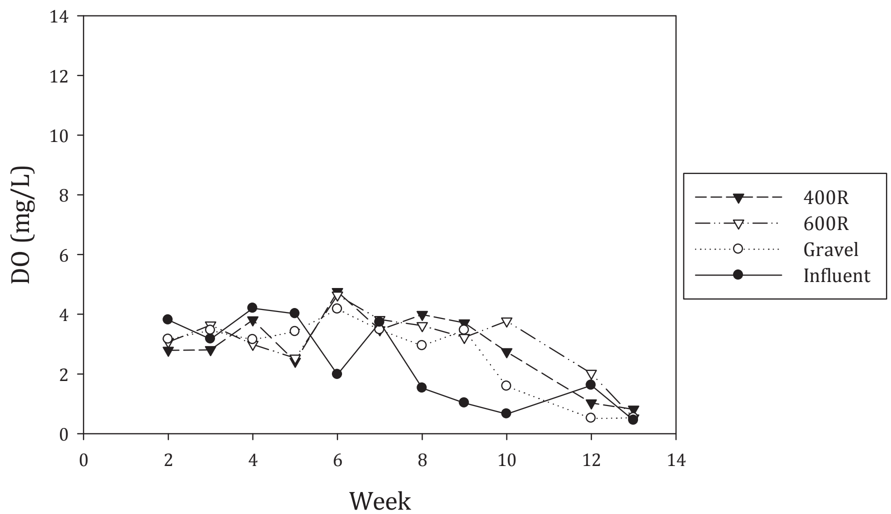



Figure 25: Experiment 2 influent and effluent pH levels at 10°C (a) and 2°C (b)

(a)




(b)

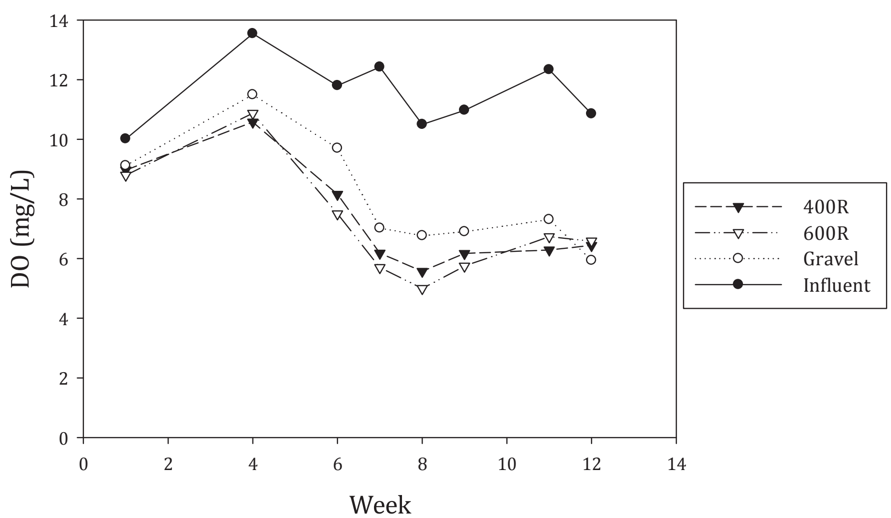
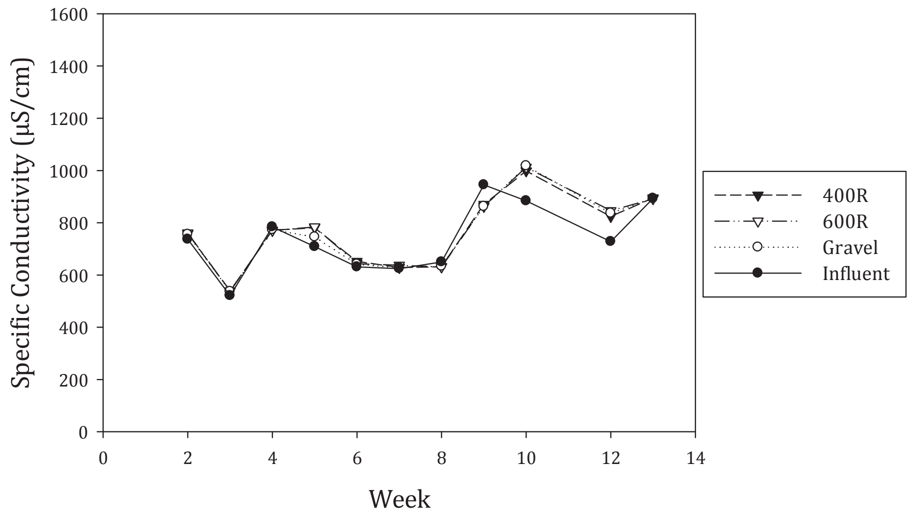



Figure 26: Experiment 2 influent and effluent DO concentrations at 10°C (a) and 2°C (b)

(a)




(b)

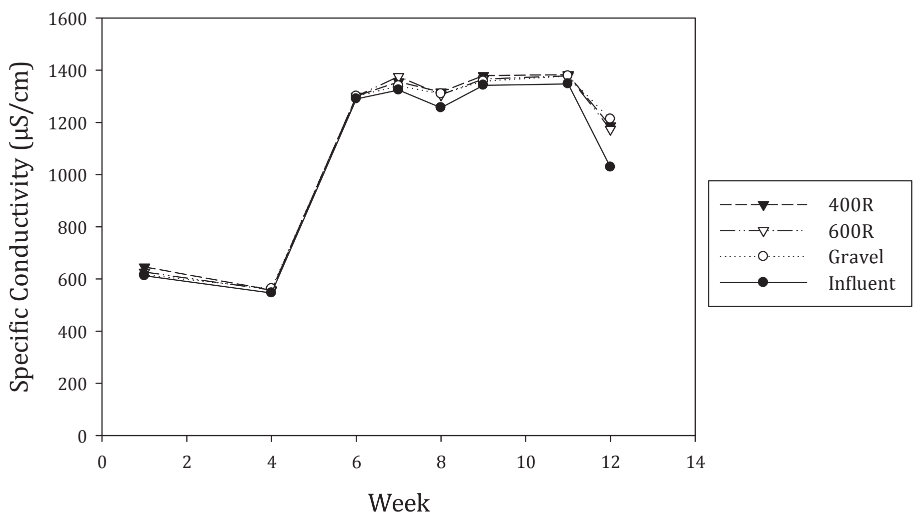



Figure 27: Experiment 2 influent and effluent specific conductivity concentrations at 10°C (a) and 2°C (b)

Table 14: Statistical summary of treatment performance in experiment 2 at 10°C (a) and 2°C (b)

(a)

| Parameter | | Influent Concentration | 400R | 600R | Gravel |
|---|---|---|---|---|---|
| TSS (mg/L) | Mean | 64.7 | 20.5 | 18.6 | 23.8 |
| | Median | 68.3 | 13.3 | 13.3 | 18.9 |
| | Min | 23.0 | 8.7 | 7.7 | 5.5 |
| | Max | 95.5 | 54.7 | 53.6 | 62.8 |
| | St. Dev. | 27.5 | 14.5 | 13.7 | 15.8 |
| $BOD_5$ (mg/L) | Mean | 47.9 | 29.3 | 31.6 | 27.5 |
| | Median | 43.1 | 20.0 | 19.6 | 16.7 |
| | Min | 9.4 | 4.5 | 6.0 | 3.8 |
| | Max | 111.0 | 109.0 | 107.0 | 109.0 |
| | St. Dev. | 29.1 | 28.5 | 28.3 | 28.9 |
| *E. coli* (CFU/100mL) | Mean | 1.7E+04 | 1.3E+04 | 6.6E+03 | 1.6E+04 |
| | Median | 2.7E+04 | 8.3E+03 | 6.0E+03 | 1.2E+04 |
| | Min | 7.8E+02 | 1.0E+03 | 1.1E+02 | 1.6E+03 |
| | Max | 1.9E+06 | 1.6E+06 | 1.6E+06 | 1.3E+06 |
| | St. Dev. | 2.1E+05 | 8.4E+04 | 6.5E+04 | 9.5E+04 |
| $NH_4$-N (mg/L) | Mean | 36.4 | 34.3 | 34.5 | 32.8 |
| | Median | 34.8 | 36.2 | 36.6 | 35.8 |
| | Min | 13.1 | 12.8 | 13.2 | 12.4 |
| | Max | 55.8 | 48.4 | 47.8 | 48.3 |
| | St. Dev. | 10.2 | 10.6 | 11.3 | 12.1 |
| TN (mg/L) | Mean | 38.3 | 32.0 | 33.9 | 32.0 |
| | Median | 42.2 | 34.5 | 32.4 | 33.6 |
| | Min | 19.0 | 13.8 | 13.6 | 14.0 |
| | Max | 56.8 | 39.6 | 76.8 | 48.8 |
| | St. Dev. | 11.5 | 8.1 | 14.0 | 9.6 |
| TP (mg/L) | Mean | 2.5 | 1.8 | 1.7 | 1.7 |
| | Median | 2.1 | 1.7 | 1.3 | 1.5 |
| | Min | 0.85 | 0.55 | 0.55 | 0.65 |
| | Max | 5.4 | 4.5 | 4.4 | 4.8 |
| | St. Dev. | 1.3 | 1.1 | 1.1 | 1.1 |
| TOC (mg/L) | Mean | 28.9 | 23.6 | 23.4 | 21.2 |
| | Median | 28.5 | 20.1 | 21.8 | 20.1 |
| | Min | 14.6 | 13.6 | 13.4 | 12.4 |
| | Max | 42.0 | 44.6 | 41.2 | 44.6 |
| | St. Dev. | 9.1 | 8.6 | 8.4 | 8.1 |

(b)

| Parameter | | Influent Concentration | 400R | 600R | Gravel |
|---|---|---|---|---|---|
| TSS (mg/L) | Mean | 41.9 | 25.2 | 22.9 | 33.3 |
| | Median | 41.7 | 24.9 | 23.4 | 34.0 |
| | Min | 30.7 | 18.2 | 14.3 | 24.6 |
| | Max | 50.2 | 31.0 | 28.7 | 41.2 |
| | St. Dev. | 6.4 | 3.8 | 3.4 | 5.4 |
| BOD5 (mg/L) | Mean | 46.0 | 39.0 | 36.6 | 40.1 |
| | Median | 44.6 | 38.1 | 36.0 | 42.3 |
| | Min | 21.8 | 14.4 | 18.0 | 20.4 |
| | Max | 70.0 | 57.5 | 45.9 | 53.0 |
| | St. Dev. | 12.4 | 10.8 | 7.4 | 9.1 |
| *E. coli (CFU/100mL)* | Mean | 4.2E+04 | 2.0E+04 | 1.6E+04 | 2.3E+04 |
| | Median | 3.9E+04 | 1.9E+04 | 1.6E+04 | 2.5E+04 |
| | Min | 3.6E+03 | 3.8E+03 | 1.3E+03 | 1.1E+03 |
| | Max | 6.4E+05 | 1.8E+05 | 1.9E+05 | 2.3E+05 |
| | St. Dev. | 1.6E+05 | 6.0E+04 | 5.2E+04 | 9.2E+04 |
| NH4-N (mg/L) | Mean | 25.9 | 25.7 | 25.3 | 25.2 |
| | Median | 28.6 | 28.6 | 26.9 | 26.8 |
| | Min | 10.6 | 9.7 | 9.7 | 9.4 |
| | Max | 32.4 | 32.1 | 33.3 | 32.5 |
| | St. Dev. | 6.9 | 7.6 | 7.1 | 7.3 |
| TN (mg/L) | Mean | 26.7 | 24.6 | 24.5 | 24.8 |
| | Median | 27.9 | 25.5 | 24.1 | 24.7 |
| | Min | 17.8 | 13.0 | 13.6 | 13.0 |
| | Max | 33.8 | 34.0 | 36.8 | 33.2 |
| | St. Dev. | 5.6 | 5.6 | 5.6 | 6.2 |
| TP (mg/L) | Mean | 1.1 | 0.61 | 0.52 | 0.88 |
| | Median | 1.0 | 0.57 | 0.46 | 0.86 |
| | Min | 0.62 | 0.29 | 0.13 | 0.42 |
| | Max | 1.8 | 1.1 | 1.0 | 1.4 |
| | St. Dev. | 0.36 | 0.24 | 0.25 | 0.31 |
| TOC (mg/L) | Mean | 26.9 | 28.6 | 27.3 | 26.6 |
| | Median | 26.3 | 27.8 | 26.2 | 25.1 |
| | Min | 15.0 | 16.3 | 16.5 | 16.7 |
| | Max | 37.8 | 41.4 | 36.8 | 37.0 |
| | St. Dev. | 6.4 | 5.7 | 5.3 | 5.5 |

With this data, average removal rates for each parameter were calculated, shown in Table 15. Furthermore, those columns which produced a statistically significant ($p < 0.05$) concentration difference between influent and effluent in both duplicates are indicated with an asterisk beside the average effluent concentration.

Table 15: Average water quality improvement as percent reduction, or log-reduction where indicated with "†"

| | 10°C | | | 2°C | | |
|---|---|---|---|---|---|---|
| Parameter | 400R | 600R | Gravel | 400R | 600R | Gravel |
| TSS | 65.7%* | 68.0%* | 58.9%* | 39.6%* | 44.9%* | 20.4%* |
| $BOD_5$ | 44.0%* | 29.8%* | 44.6%* | 12.2%* | 17.6%* | 11.1%* |
| †*E. coli* | 0.19 | 0.35 | 0.17 | 0.28 | 0.36 | 0.22 |
| $NH_4$-N | 7.72% | 2.91% | 9.59% | 0.04% | 1.71% | 3.01% |
| TN | 32.0%* | 33.9%* | 32.0%* | 7.62% | 6.47% | 7.00% |
| TP | 26.3%* | 30.1%* | 29.5%* | 43.1%* | 51.5%* | 17.6%* |
| TOC | 23.6%* | 23.4%* | 21.2%* | -8.42% | -2.92% | 0.10% |

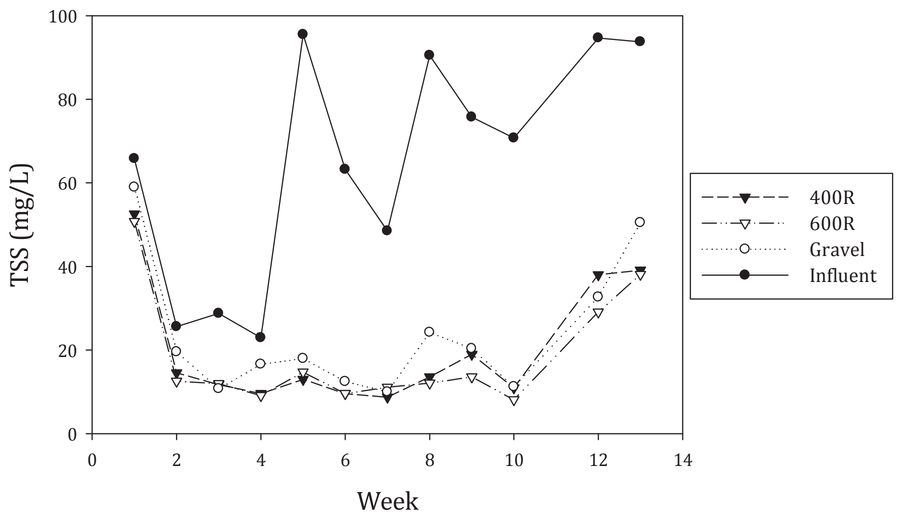
* Average reduction in concentration was statistically significant ($p < 0.05$)

### *4.2.1 Total Suspended Solids*

Based on Experiment 1 observations, TSS removal was expected in Experiment 2 at 10°C. All columns run at 10°C showed statistically lower effluent concentrations (Figure 28). In fact, average effluent concentrations from all columns at 10°C met the 25 mg/L Environment Canada regulation (Table 14). The same could not be said at 2°C, as only the 600R column was below 25 mg/L.

However, all columns at both 10°C and 2°C produced statistically significant reductions in TSS, shown in Table 15. At 10°C, TSS removal efficiency appeared to improve with time; achieving over 80% removal 5 weeks into the experiment (Figure 29). On the whole, both geotextiles removed significantly more TSS than the gravel controls. Of the two, the 600R geotextile was statistically more efficient.

(a)




(b)

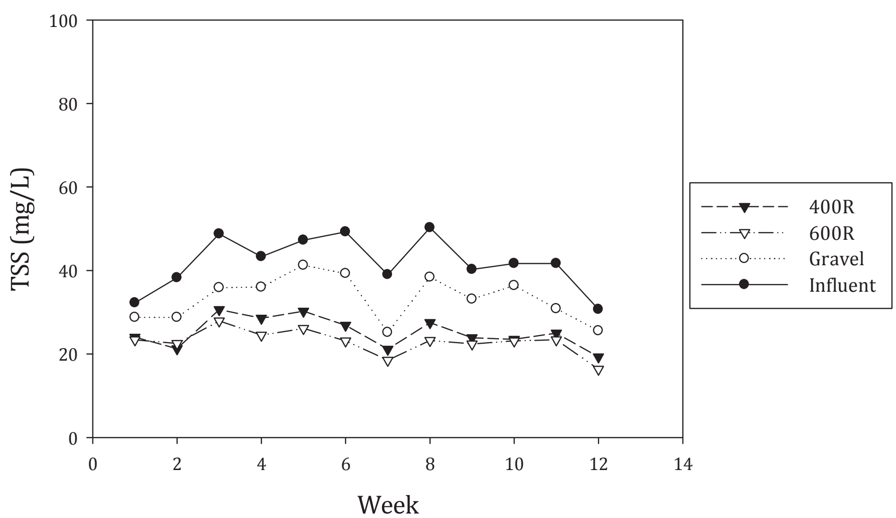
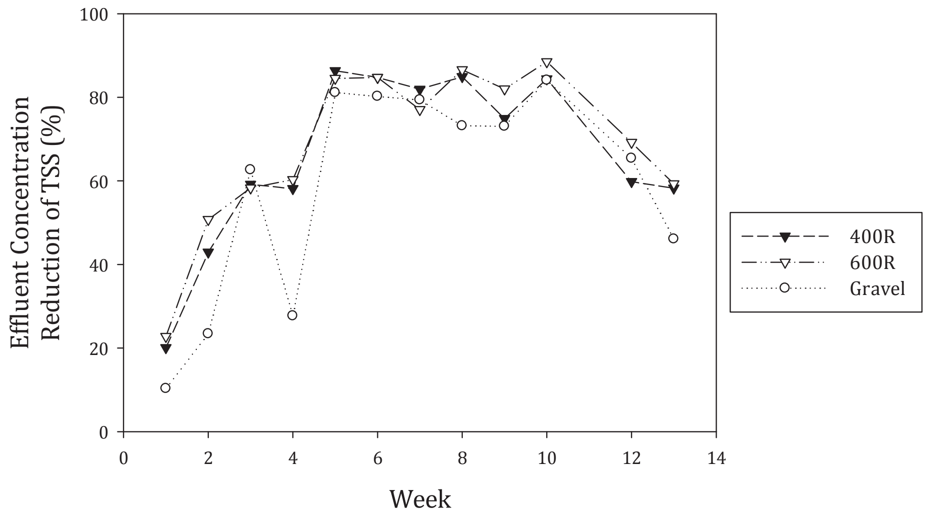



Figure 28: Experiment 2 influent and effluent TSS concentrations at 10°C (a) and 2°C (b)

(a)




(b)

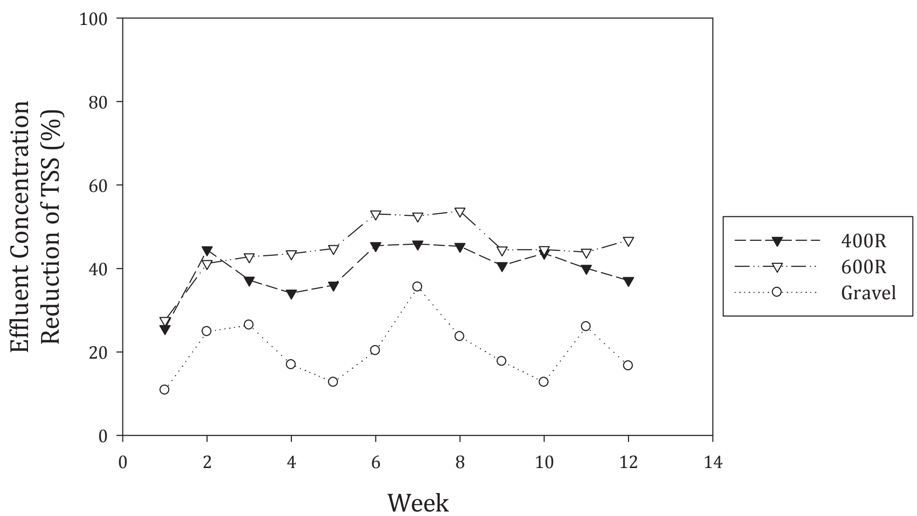



Figure 29: Experiment 2 effluent concentration % reduction of TSS at 10°C (a) and 2°C (b)

At 2°C, TSS removal increased in the first week, followed by relatively constant removal between 35-45% in the 400R column, and 40-50% in the 600R (Figure 29) In contrast, the gravel controls were more unstable, and averaged 21% after the first week. A statistical comparison showed that both geotextiles produced significantly lower effluent concentrations than the gravel controls. Statistical comparison also confirmed that the 600R column performed better at 2°C.

Comparing the two temperatures, TSS removal was more consistent – albeit lower – at 2°C. This may be attributable to physical filtration by the geotextiles; without substantial biomat growth. The observable improvement over time (above the baseline removal rate) further indicates that more biological development occurred in the 10°C trial.

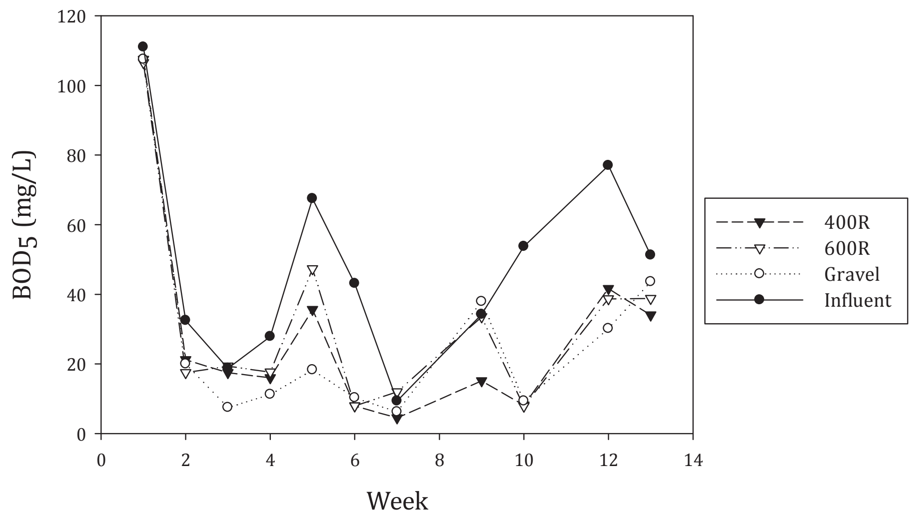
### *4.2.2 5-day Biochemical Oxygen Demand*

Table 14 shows the performance of the columns at 10°C and 2°C. Overall, no column at either temperature was able to achieve average $BOD_5$ concentrations below the WSER limit of 25 mg/L. However, at 10°C, 7 weekly measurements of 400R column effluent, 6 weekly measurements of 600R column effluent, and 7 weekly measurements of the control column effluent were below the 25 mg/L limit for $BOD_5$ (Figure 30). However, on half of these low effluent concentration days, influent concentrations were below 30 mg/L. At 2°C, only the samples from week 12 were below the limit.

Although not below the regulatory limit, statistically significant reductions in $BOD_5$ were observed in all columns at all temperatures (Table 15). At 10°C, the geotextile and control columns performed similarly, and none were shown to perform any better than the others. All columns achieved 30-45% removal of $BOD_5$ on average, with 17% of geotextile column samples above an 80% removal rate.

At 2°C, the 600R performed statistically better than the 400R and control columns. However, the 600R column achieved only a maximum removal efficiency of 38% (Figure 31). The filters averaged 12.2%, 17.6%, and 11.1% removal of $BOD_5$ for the 400R, 600R, and control columns, respectively.

(a)




(b)

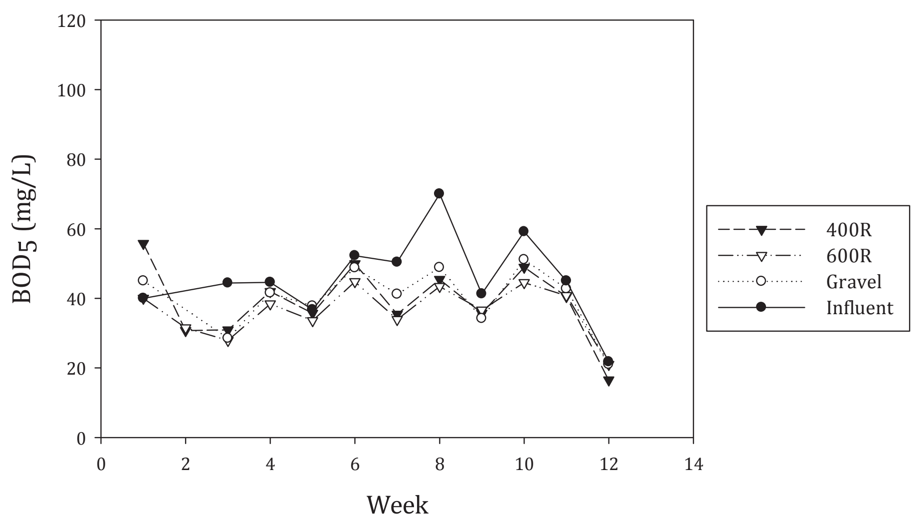
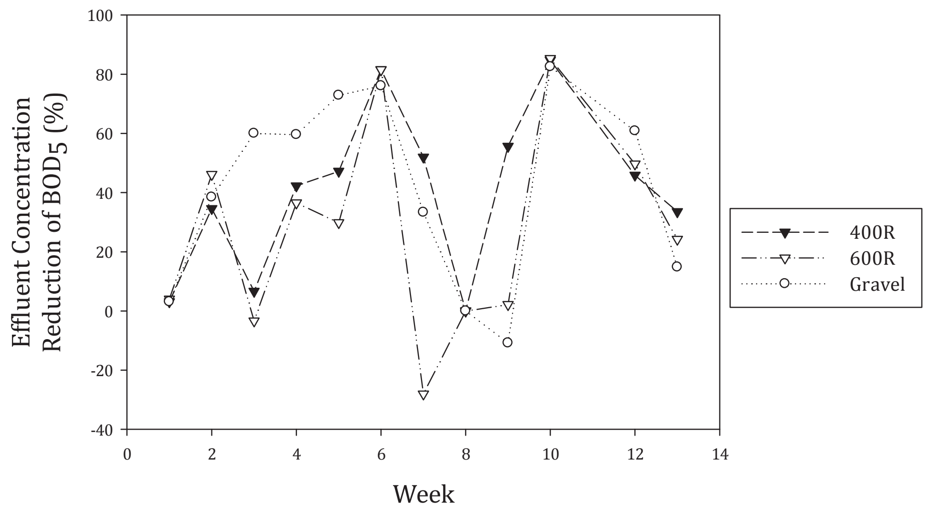



Figure 30: Experiment 2 influent and effluent concentrations of $BOD_5$ at 10°C (a) and 2°C (b)

(a)




(b)

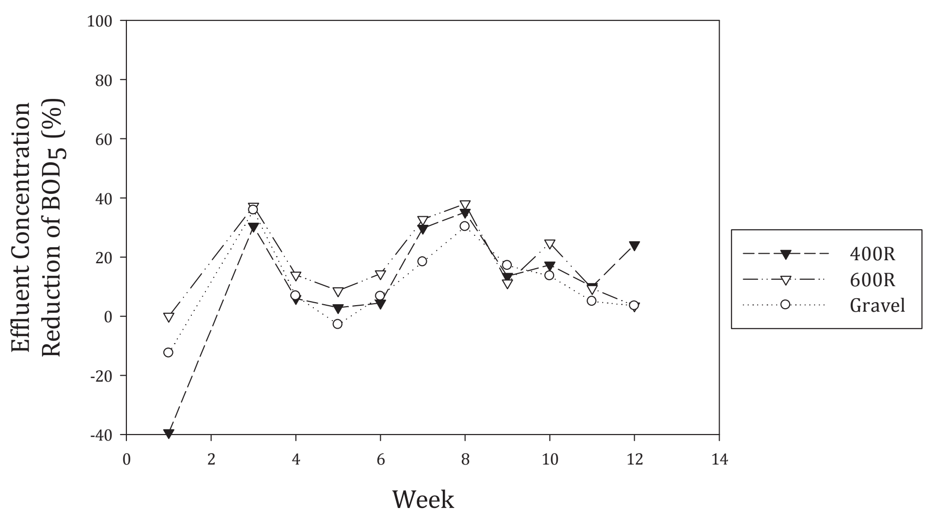



Figure 31: Experiment 2 effluent concentration % reduction of $BOD_5$ at 10°C (a) and 2°C (b)

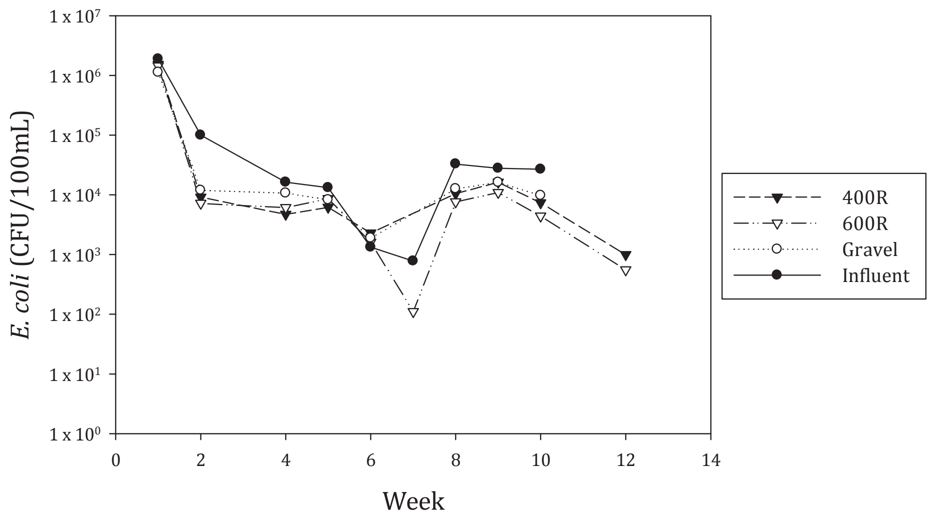
Comparing the results from the two temperatures, again, better treatment was observed at higher temperature. Over a 50% reduction in removal efficiency was observed at the lower temperature. However, like TSS, less variability was evident at the lower temperature. Additionally, comparing the high removal rate of $BOD_5$ by the 600R column at 2°C to similar removal in the 600R and control columns at 10°C suggests that the effect of the geotextile is more important at lower temperatures.

### *4.2.3 E. coli*

Influent and effluent concentrations of *E. coli* from Experiment 2 are shown in Table 14. Average effluent *E. coli* concentrations were statistically similar to influent concentrations in all columns at both temperatures. This is shown in Figure 32 with the tight grouping of data pre- and post-filter.

All filtered columns performed statistically similarly in the 10°C trial. In the 2°C trial, however, both geotextile types outperformed the gravel control (Figure 33). Comparing *E. coli* removal at both temperatures, similar performance is observed, suggesting removal is not entirely dependent on biofilm development. This also supports the hypothesis that the primary method of *E. coli* removal in Experiment 1 was bacterial die-off (Section 4.1.3).

(a)




(b)

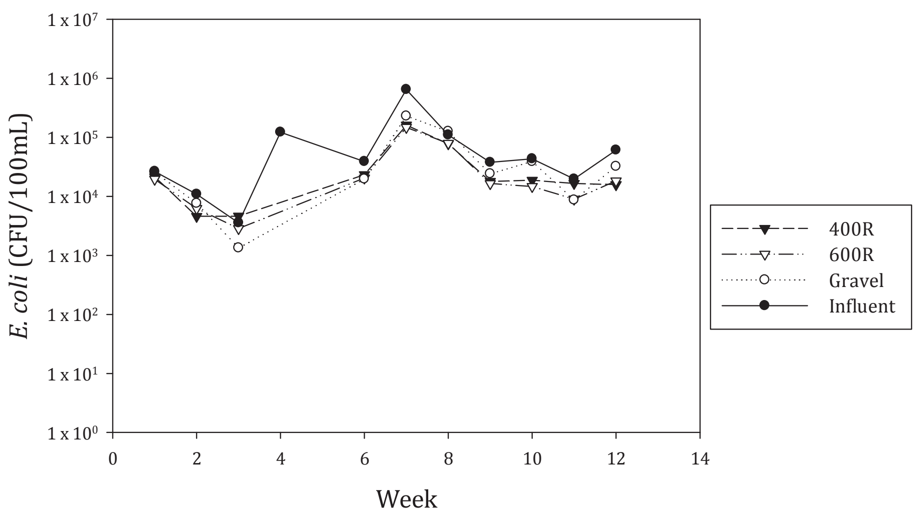
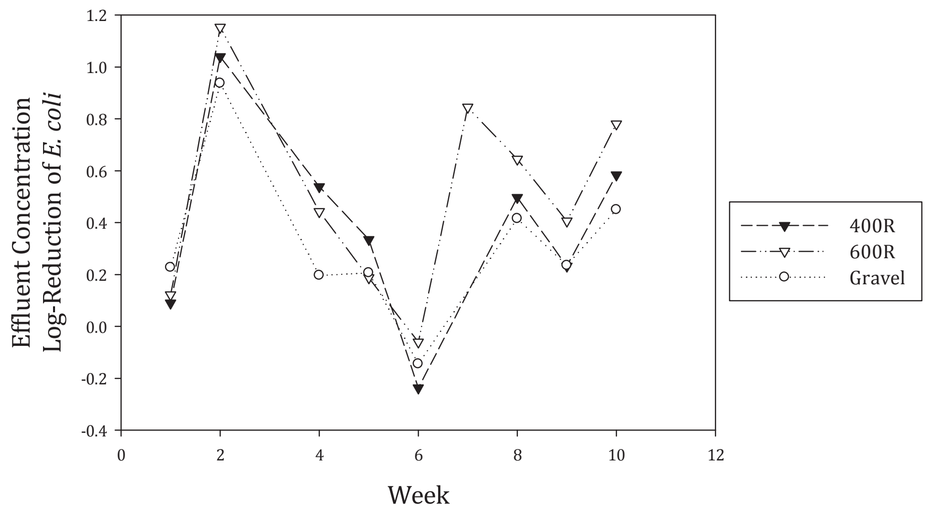



Figure 32: Experiment 2 influent and effluent concentrations of *E. coli* at 10°C (a) and 2°C (b)

(a)




(b)

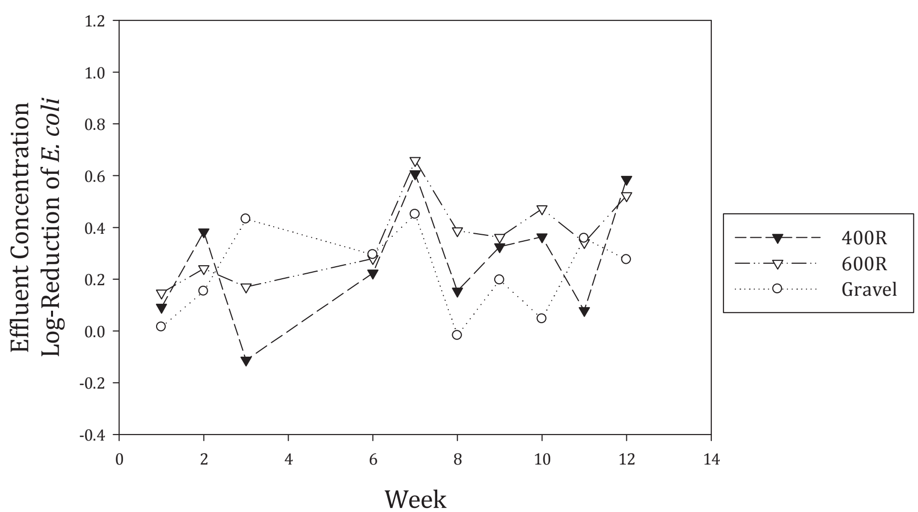



Figure 33: Experiment 2 effluent concentration log-reduction of *E. coli* at 10°C (a) and 2°C (b)

### 4.2.4 *NITROGEN*

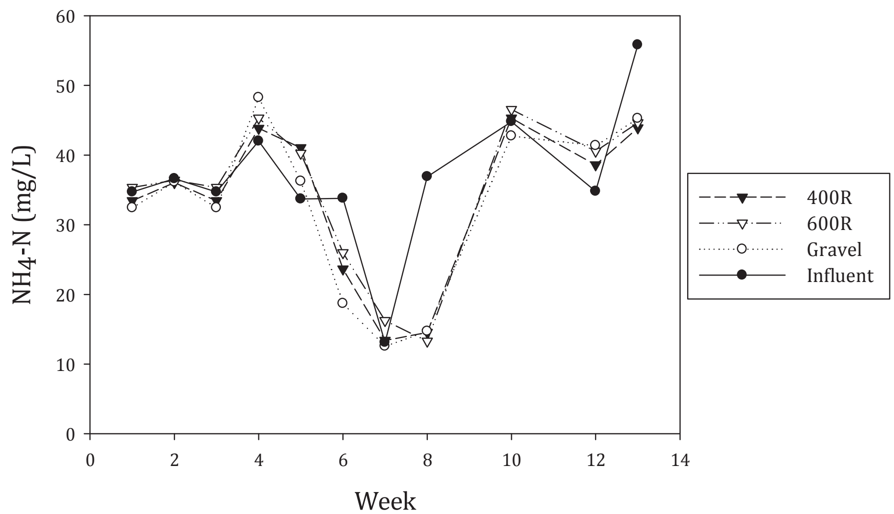
Similar to Experiment 1, no statistically significant ammonia removal occurred in Experiment 2 (Table 15). However, at 10°C, there were weeks with up to 64% removal (week 8 - Figure 34); this was still not enough to produce a statistically significant change overall. All columns at both 10°C and 2°C were statistically similar to the control (Figure 35). At neither temperature did ammonia concentrations fall below 10 mg/L in the effluent.

Again, the lack of treatment is likely associated with the amount of $BOD_5$ remaining in the wastewater. During week 7, low influent $BOD_5$ concentrations would allow nitrifiers to temporarily compete for oxygen, and briefly establish an ammonia oxidizing colony by week 8. Temperature is also likely playing a role in nitrifying bacteria growth, as evidenced by the decline in removal rate to nearly zero at 2°C. As noted before, dissolved oxygen levels did not reach anoxic conditions, eliminating the issue of lack of available oxygen.

The abundance of oxygen (Figure 26) did however limit the anaerobic conditions necessary for denitrification. No statistically significant difference in nitrate concentration was observed between influent and effluent, at either temperature. Figure 36 shows nitrate concentrations over the course of the experiment.

At 10°C, the spike in nitrate concentration between weeks 6 and 8 is likely due to two factors: (i) increased concentrations in the influent; (ii) two weeks of nitrification of ammonia under aerobic conditions (denitrification inhibited). At 2°C, very high dissolved oxygen limited denitrification of nitrate. In combination with low influent concentrations, very low effluent concentrations resulted.

(a)




(b)

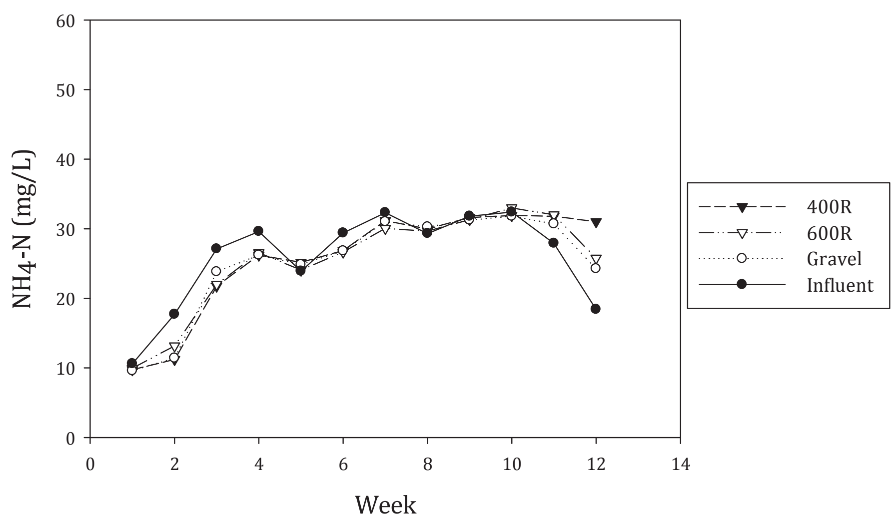
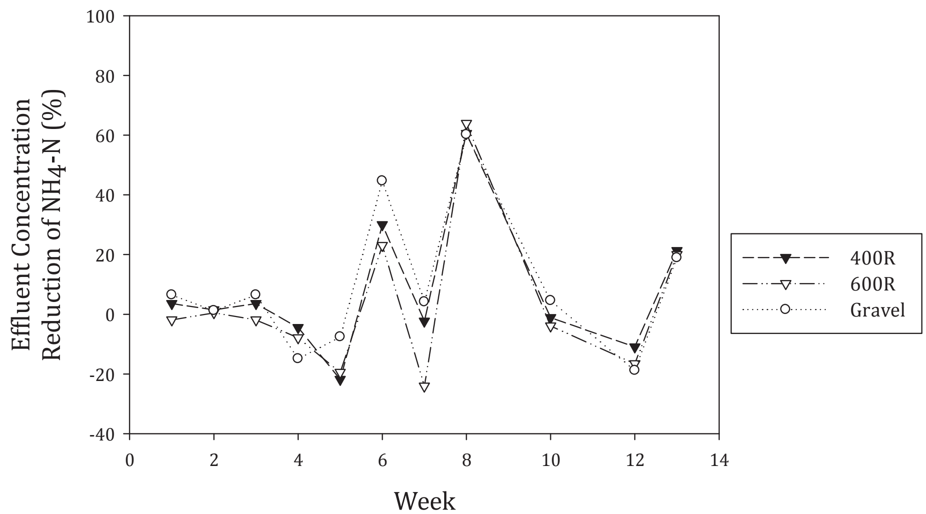



Figure 34: Experiment 2 influent and effluent concentrations of $NH_4$-N at 10°C (a) and 2°C (b)

(a)




(b)

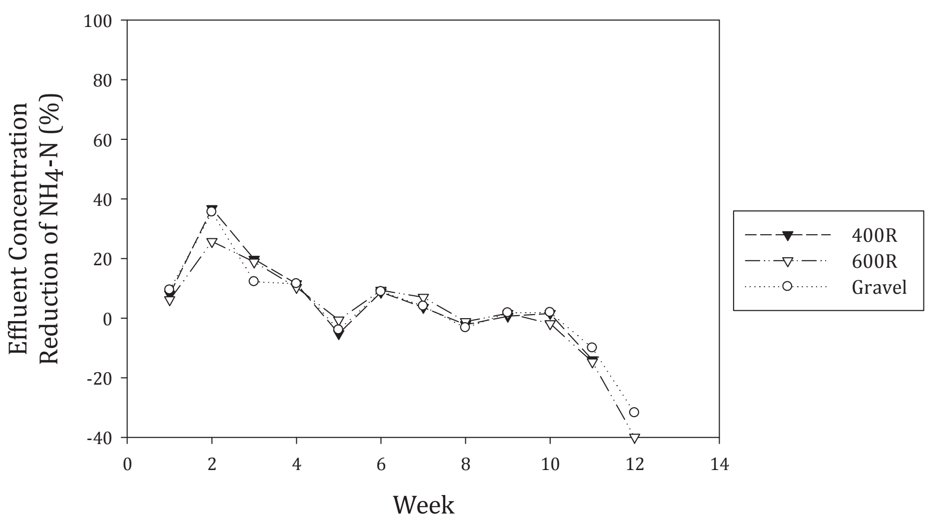
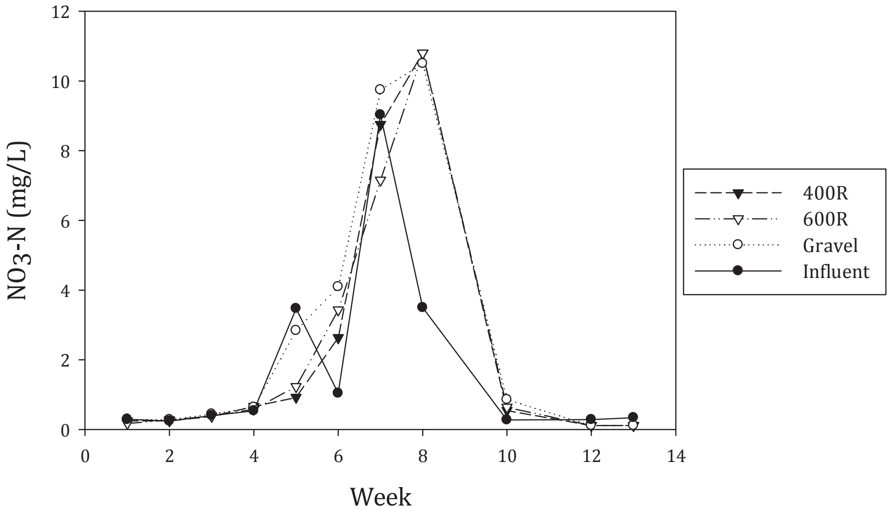



Figure 35: Experiment 2 effluent concentration % reduction of $NH_4$-N at 10°C (a) and 2°C (b)

(a)




(b)

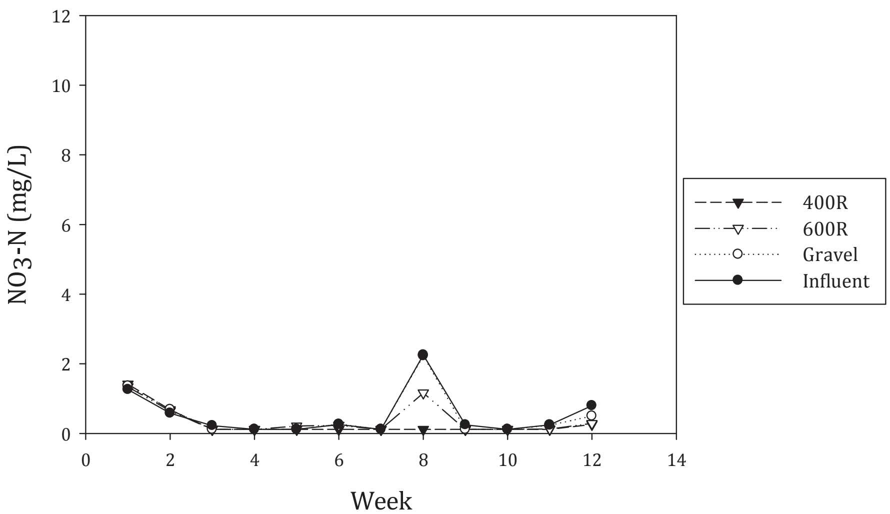


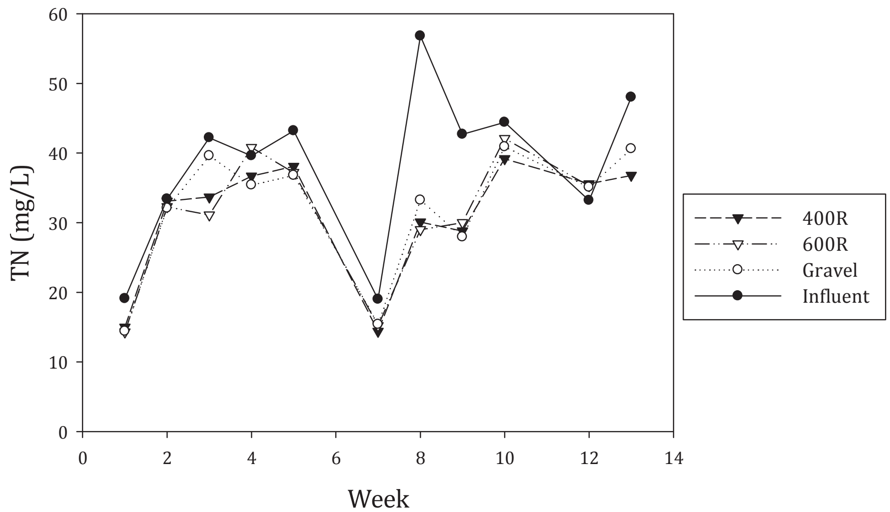

Figure 36: Experiment 2 influent and effluent concentrations of $NO_3$-N at 10°C (a) and 2°C (b)

Although ammonia removal was limited, TN removal was statistically significant at 10°C (Table 14). Like ammonia, however, concentrations did not fall below 10 mg/L in the effluent (Figure 37).

At 10°C, TN removal efficiencies averaged 32%, 33.9%, and 32% for the 400R, 600R, and gravel columns, respectively (Figure 38). However, it could not be shown that any column or control performed statistically better than the others. At 2°C, average removals were much lower, and not statistically significant. Again, all columns and controls performed similarly.

Although some TN reduction may be attributable to those few weeks with significant ammonia nitrification, it is more likely that the dominant removal mechanism was bacterial incorporation. This is evidenced by the lower removal rate under lower temperature conditions, where biological growth would be challenged.

(a)




(b)

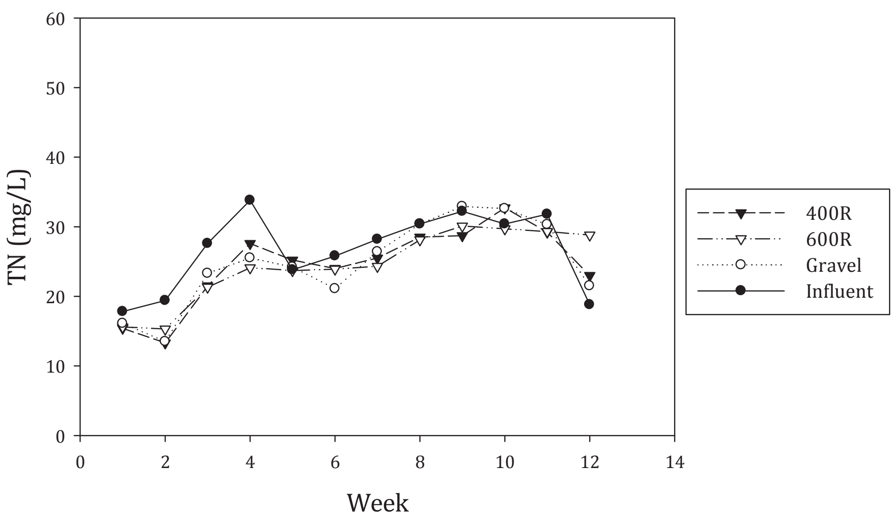
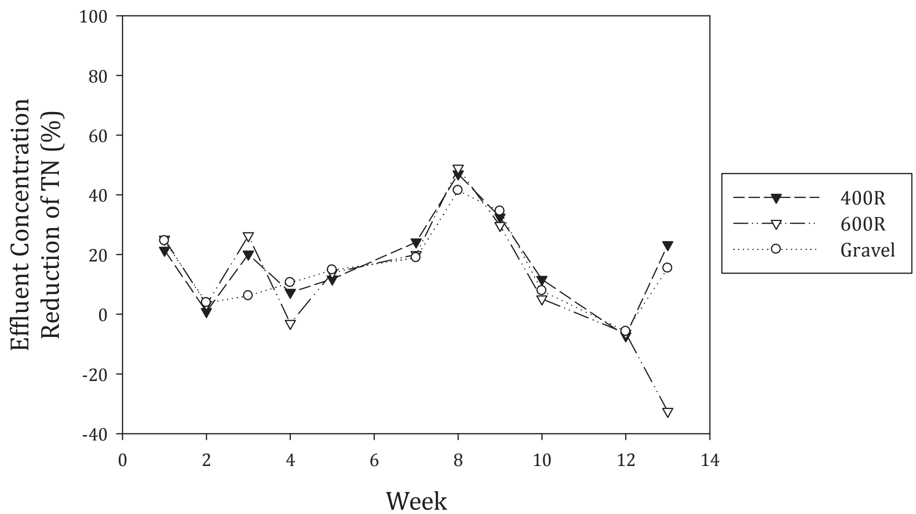



Figure 37: Experiment 2 influent and effluent concentrations of TN at 10°C (a) and 2°C (b)

(a)




(b)

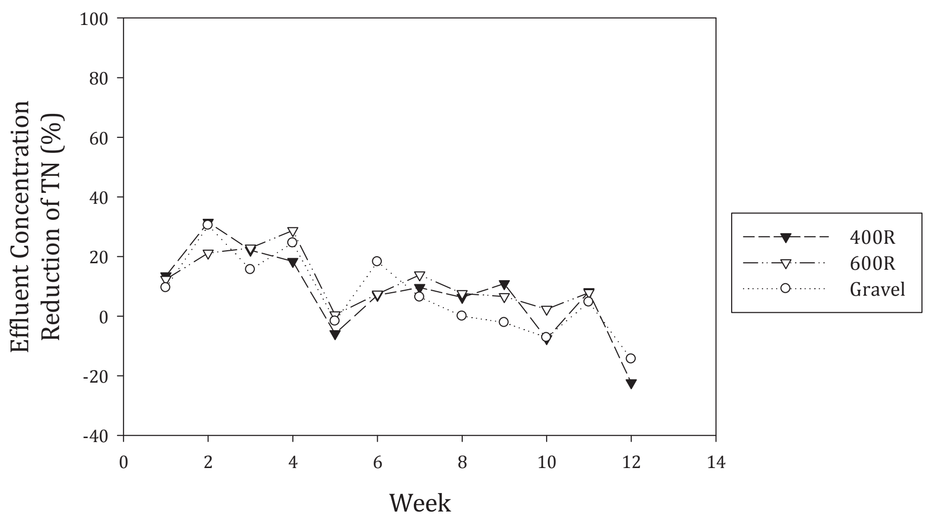



Figure 38: Experiment 2 effluent concentration % reduction of TN at 10°C (a) and 2°C (b)

### 4.2.5 *PHOSPHORUS*

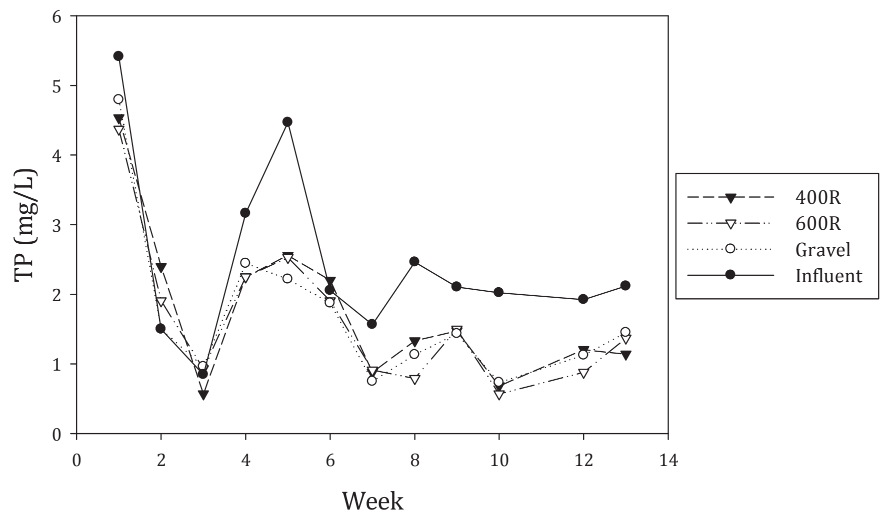
Statistically significant TP removal occurred at both 10°C and 2°C (Table 15). Although receiving lower concentrations of phosphorus, the columns at 2°C exhibited similar treatment performance as those at 10°C.

At 10°C, both of the geotextile columns showed similar treatment to the control column. This is similar to the findings from Experiment 1, showing the importance of the gravel layer in phosphorus removal. However, at 2°C (and under lower contaminant loading) both geotextiles outperformed the gravel control. The 600R geotextile performed statistically better than the 400R (Figure 40).

Figure 40 also shows the major role gravel played in phosphorus removal at 2°C. As concentrations were lower in this trial, the added surface area from the geotextiles had a more pronounced effect. Furthermore, the lower concentrations would also mean less competition for available sorption sites, reducing the amount of TP that could not adsorb under flowing conditions.

Like Experiment 1, there was indication of decline in treatment performance with time. However, it is likely that some degree of microbial uptake was responsible for overshadowing this effect until the final month of the 10°C trial. Limited microbial growth at 2°C may have resulted in sorption sites filling faster, making the decline effect noticeable after the first month.

(a)




(b)

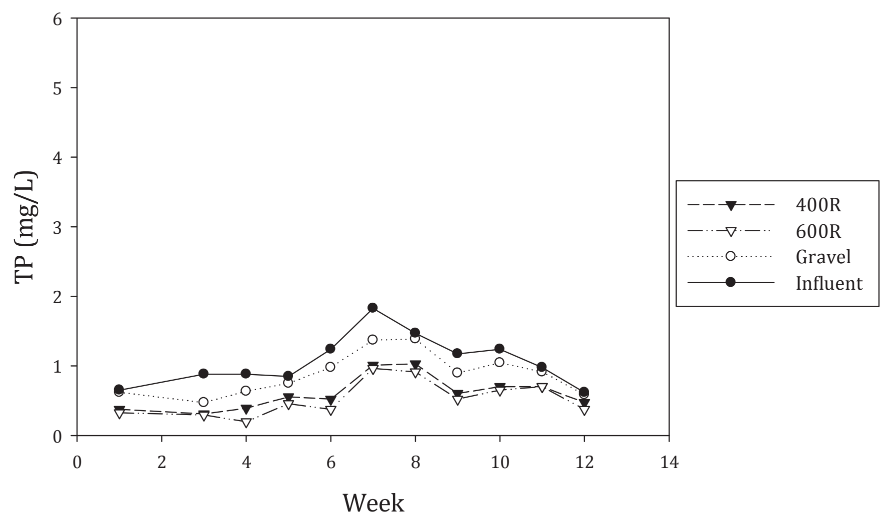
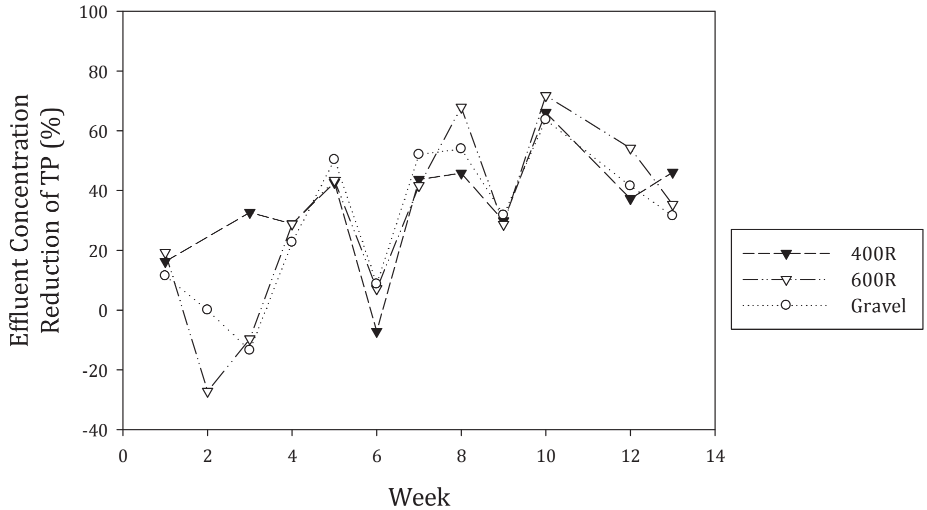



Figure 39: Experiment 2 influent and effluent concentrations of TP at 10°C (a) and 2°C (b)

(a)




(b)

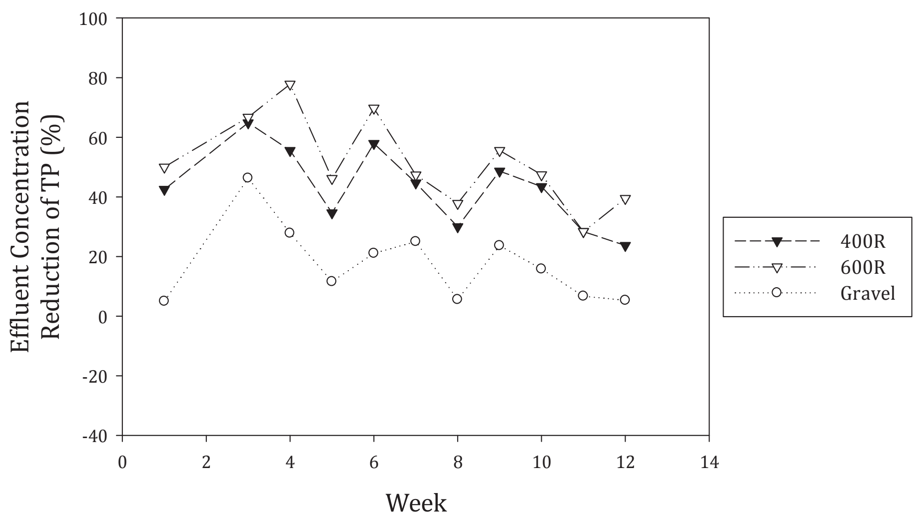
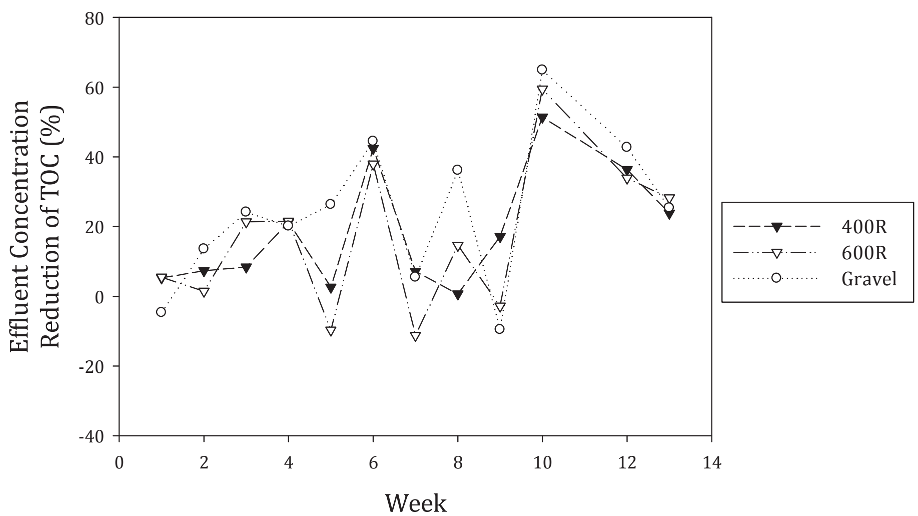



Figure 40: TP Experiment 2 effluent concentration % reduction of TP at 10°C (a) and 2°C (b)

### 4.2.6 *Total Organic Carbon*

Even though influent concentrations of TOC were statistically similar at both temperatures, TOC removal was only statistically significant at 10°C (Table 14; Figure 41). In this test, however, both geotextile columns and the control columns performed similarly. At 2°C, no significant removal of TOC was observed.

(a)




(b)

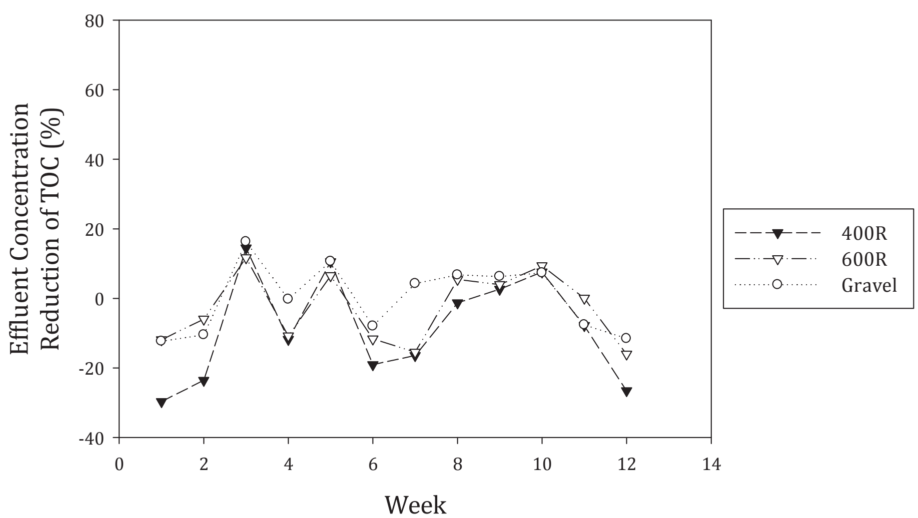



Figure 41: Experiment 2 influent and effluent concentrations of TOC at 10°C (a) and 2°C (b)

# 5 Recommendations

Based on the results shown in Chapter 4, several recommendations were identified for future study and final implementation of a geotextile liner system for arctic WSPs. Further testing of the proof of concept should include:

1. Repetition of the second experiment using a similar method with the addition of initial seeding of the geotextiles with sludge or sludge slurry.
2. Adjustment of the experimental procedure to incorporate both diurnal and seasonal temperature fluctuations over a 3-month experiment.
3. Experiments on the effect of biomat rebound and redevelopment after a freeze-thaw cycle, with the possibility of incorporating geotextile integrity assessments after multiple freeze-thaw cycles.

If continuing experimentation with a similar design, it is recommended that the filter columns lengths be extended to better understand nitrification in the gravel berm after full BOD removal. If moving forward with pilot scale experimentation, the recommendations above should also be incorporated into the experimental design and monitoring strategy. For example, seeding the geotextile with sludge or slurry would have to occur either manually before installation or in-situ using hydroseeding.

The results and conclusions of this study indicate that an exfiltrating berm segment would be highly effective at removing TSS and BOD when modified with a geotextile on the inner wall. If the geotextile and the top 10 cm of the berm are used primarily for TSS and BOD removal, BOD would be reduced to a level where nitrifying bacteria could dominate the remainder of the 2 – 3 meter thick berm (USEPA, 1975; Parker and Richards, 1986; Rusten, et al., 1995; Grady Jr., et al., 1999; Ye, et al., 2009).

Finally, geotextile filtration technology could also be used in a multiple-cell WSP system. These systems typically require a seasonal decant between cells. An exfiltration section would allow continuous flow from one cell to the next, reducing operational costs. Furthermore, if the geotextile provides a high level of BOD treatment, the second cell could be devoted to nitrification processes.

# 6 CONCLUSIONS

The objective of this study was to evaluate the clogging and water treatment potential of geotextiles at low temperatures for use in WSP improvement. Bench scale testing of single-layer geotextile filters over 10 cm of gravel was conducted in a series of 3-month trials at 10°C and 2°C.

Interpretation of the results of these experiments led to several conclusions about low-temperature geotextile biofilter performance, specifically:

1. Biomat development was achievable over a 3-month period. A greater mass of biomat developed at 10°C than 2°C.
2. Hydraulic conductivity followed an exponential decline. Overall, a 90% reduction was observed over 3 months at both temperatures. The 600R geotextile on average resulted in a lower hydraulic conductivity than the 400R geotextile, indicating smaller opening sizes will clog easier in wastewater filtration applications.
3. At colder temperatures, biological filtration of constituents was significantly reduced, as evidenced by the dramatic decline in TSS, $BOD_5$, TN, and TOC removal efficiency. The primarily physical sorption processes responsible for TP removal were unaffected by temperature changes.
4. The 600R geotextile filters resulted in better water quality improvements than the 400R geotextile filters in all cases where there was a significant difference between the geotextiles.
5. Increasing hydraulic loading of wastewater to the filters resulted in improved treatment of TSS and $BOD_5$.
6. Even at 2°C, TSS and $BOD_5$ removal by geotextiles was still significantly better than the control columns, although the gravel was also responsible for significant treatment at both temperatures.
7. Effective removal of *E. coli* was not achieved under seepage conditions at either temperature.

8. Removal of ammonia was limited at both temperatures. This was primarily attributed to elevated concentrations of $BOD_5$ and competition for oxygen by carbonaceous oxidizing bacteria.
9. Reduced biomat development at 2°C also impacted TN removal. All statistically significant TN removal at 10°C was eliminated at 2°C.
10. TP removal efficiency declined over the 3 month experiments, likely due to filling of phosphorus sorption sites in the gravel. However an average reduction 1 mg/L of TP was maintained at both 10°C and 2°C.

# BIBLIOGRAPHY


ASTM D1987–07. (2012). *Standard test method for biological clogging of geotextile or soil/geotextile filters*, American Society for Testing and Materials, West Conshohocken, Pennsylvania, USA.

ASTM D5493-06. (2011). *Standard test method for permittivity of geotextiles under load*, American Society for Testing and Materials, West Conshohocken, Pennsylvania, USA.

ASTM D5856-95. (2007). *Standard test method for measurement of hydraulic conductivity of porous material using a rigid-wall, compaction-mold permeameter*, American Society for Testing and Materials, West Conshohocken, Pennsylvania, USA.

Aboriginal Affairs and Northern Development Canada. (2010). *Protocol for centralised wastewater systems in First Nations communities.* Gatineau, Quebec: Indian and Northern Affairs Canada.

Amador, J., Potts, D., Patenaude, E., & Görres, J. (2008). Effects of sand depth on domestic wastewater renovation in intermittently aerated leachfield mesocosms. *Journal of Hydrologic Engineering , 13* (8), 729-734.

Armstrong, M. (1998). *Laboratory program to study clogging in a leachate collection system.* The University of Western Ontario, Faculty of Graduate Studies. Ottawa: National Library of Canda.

Benjes Jr., H. (1980). *Handbook of biological wastewater treatment.* New York, New York, USA: Garland Publishing, Inc.

Bertram, G. (1940). An experimental investigation of protective filters. *Harvard University Soil Mechanics Series Publication 267* (7).

Bicudo, J., Goode, G., Workman, S., & Turner, L. (2003). Vertical water movement through geotextile and gravel pads used in heavy livestock traffic areas. *Ninth International Animal, Agricultural and Food Processing Wastes Symposium* (pp. 364-371). Research Triangle Park, North Carolina: American Society of Agricultural and Biological Engineers.

Bilotta, G., & Brazier, R. (2008). Understanding the influence of suspended solids on water quality and aquatic biota. *Water Research , 42* (12), 2849-2861.

Bölter, M., Blume, H., & Wetzel, H. (2006). Properties, formation, classification and ecology of arctic soils: Results from the tundra northwest expedition 1999 (Nunavut and Northwest Territories, Canada). *Polarforschung , 73* (2/3), 89-101.

Breitburg, D., Loher, T., Pacey, C., & Gerstein, A. (1997). Varying effects of low dissolved oxygen on trophic interactions in an estuarine food web. *Ecological Monographs , 67* (4), 489-507.

Bruce, A., & Merkens, J. (1973). Further studies of partial treatment of sewage by high-rate biological filtration. *Water Pollution Control , 72* (5), 499-527.

Bruce, A., & Merkens, J. (1970). Recent studies of high rate biological filtration. *Water Pollution Control , 69*, 113-148.

Canadian Council of Ministers of the Environment. (2009). *Canada-wide strategy for the management of municipal wastewater effluent.* Retrieved January 16, 2013, from Municipal Wastewater Effluent: http://www.ccme.ca/assets/pdf/cda_wide_strategy_mwwe_final_e.pdf

Cancelli, A., & Cazzuffi, D. (1987). Permittivity of geotextiles in the presence of water and polluted fluids. *Geosynthetics 1987. 2*, pp. 471-481. St. Paul: Industrial Fabrics Association International.

Cantrell, K., Chastain, J., & Moore, K. (2008). Geotextile filtration performance for lagoon sludges and liquid animal manures dewatering. *Transactions of the ASABE , 51* (3), 1067-1076.

Check, G., Waller, D., Lee, S., Pask, D., & Mooers, J. (1994). The lateral-flow sand-filter system for septic-tank effluent treatment. *Water Environment Research , 66* (7), 919-928.

Chen, Y., Simons, D., & Demery, P. (1981). Hydraulic testing of plastic filter fabrics. *Journal of the Irrigation and Drainage Engineering , 107* (3), 307-324.

Chevalier, P., Proulx, D., Lessard, P., Vincent, W., & Noüe, d. l. (2000). Nitrogen and phosphorus removal by high latitude mat-forming cyanobacteria for potential use in tertiary wastewater treatment. *Journal of Applied Phycology , 12* (2), 105-112.

Clescerl, L., Greenberg, A., & Eaton, A. (Eds.). (1998). *Standard methods for the examination of water and wastewater* (20th ed.). Washington, DC: American Public Health Association, American Water Works Association, Water Environment Federation.

Crites, R., & Tchobanoglous, G. (1998). *Small and decentralized wastewater management systems.* New York: McGraw-Hill.

Cucarella, V., & Renman, G. (2009). Phosphorus sorption capacity of filter materials used for on-site wastewater treatment determined in batch experiments – A comparative study. *Journal of Environmental Quality , 32* (2), 381-392.

Easton, J., Gauthier, J., Lalor, M., & Pit, R. (2005, October). Die-off of pathogenic *E. coli* O157-H7 in sewage contaminated wastewaters. *Journal of the American Water Resources Association* , 1187-1193.

Franks, C., Davis, A., & Aydilek, A. (2012). Geosynthetic filters for water quality improvement of urban storm water runoff. *Journal of Environmental Engineering , 138* (10), 1018-1028.

Germaine, D., & Cyr, J. (2003). Evaluation of biofilter performance to remove dissolved arsenic: Wood cadillac. *Sudbury 2003 Mining and the Environment Conference.* Sudbury, Ontario: Laurentian University Centre for Environmental Monitoring.

Gordon, C., Wynn, J., & Woodin, S. (2001). Impacts of increased nitrogen supply on high arctic heath: The importance of bryophytes and phosphorus availability. *New Phytologist , 149* (3).

Government of Canada. (1985). *Fisheries act.* Retrieved January 16, 2013, from Justice Laws Website: http://laws-lois.justice.gc.ca/PDF/F-14.pdf

Government of Canada. (2012). Wastewater systems effluent regulations. *Canada Gazette , 146* (15).

Grady Jr., L., Daigger, G., & Lim, H. (1999). *Biological wastewater treatment* (2nd Edition ed.). New York, New York, USA: Marcel Dekker, Inc.

Gullicks, H., & Cleasby, J. (1990). Cold-climate nitrifying biofilters: Design and operation considerations. *Research Journal of the Water Pollution Control Federation , 62* (1), 50-57.

Gullicks, H., & Cleasby, J. (1986). Design of trickling filter nitrification towers. *Journal of the Water Pollution Control Federation , 58* (1), 60-67.

Gullicks, H., & Cleasby, J. (1990). Nitrification performance of a pilot-scale trickling filter. *Research Journal of the Water Pollution Control Federation , 62* (1), 40-49.

HACH Company. (2012a). *Coliforms-total and E. coli, membrane Filtration method 10029.* Retrieved June 18, 2013, from Water Analysis Handbook: http://www.hach.com/wah

HACH Company. (2012b). *Nitrogen, ammonia-salicylate HR TNT method 10205.* Retrieved June 18, 2013, from Water Analysis Handbook: http://www.hach.com/wah

HACH Company. (2012c). *Nitrate, dimethylphenol LR method 10206.* Retrieved June 18, 2013, from Water Analysis Handbook: http://www.hach.com/wah

HACH Company. (2012d). *Nitrogen, total-persulfate digestion LR method 10071.* Retrieved June 18, 2013, from Water Analysis Handbook: http://www.hach.com/wah

HACH Company. (2012e). *Phosphorus, total molybdovanadate method with acid persulfate digestion HR TNT method 10127.* Retrieved June 18, 2013, from Water Analysis Handbook: http://www.hach.com/wah

Harrison, R., Turner, N., Hoyle, J., Krejjsl, J., Tone, D., Henery, C., et al. (2000). Treatment of septic effluent for fecal coliform and nitrogen in coarse-textured soils: Use of soil-only and sand filter systems. *Water, Air, and Soil Pollution , 124* (1), 205-215.

Harvard, P., Jamieson, R., Cudmore, D., Boutilier, L., & Gordon, R. (2008). Performance and hydraulics of lateral flow sand filters for on-site wastewater treatment. *Journal of Hydrologic Engineering , 13* (8), 720-728.

Hatch, B. (1943). Sewage treatment at military installations. *Sewage Works Journal , 15* (5), 839-846.

Hayward, J., Jamieson, R., Boutilier, L., Lam, B., Gagnon, G., & Krkosek, W. (2012). Hydrological characterization and treatment performance assessment of a natural tundra wetland receiving effluent from a single-cell wastewater treatment exfiltration lagoon. *15th International Conference on Cold Regions Engineering.* Quebec: Canadian Society for Civil Engineering.

Heaven, S., Lock, A., Pak, L., & Rspaev, M. (2003). Waste stabilisation ponds in extreme continental climates: a comparison of design methods from the USA, Canada, northern Europe and the former Soviet Union. *Water Science & Technology , 48* (2), 25-33.

Heinke, G., Smith, D., & Finch, G. (1991). Guidelines for the planning and design of wastewater lagoon systems in cold climates. *Canadian Journal of Civil Engineering , 18*, 556-567.

Horan, N. (1990). *Biological wastewater treatment systems: Theory and Operation.* West Sussex, England: John Wiley & Sons Ltd.

Hu, Z., & Gagnon, G. (2006). Impact of filter media on the performance of full-scale recirculating biofilters for treating multi-residential wastewater. *Water Research , 40* (7), 1474-1480.

Jamieson, R., Krkosek, W., Boutilier, L., & Gagnon, G. (2012). *Literature review of wastewater treatment desgin and performance in the far north.* Draft Report, Dalhousie University, Centre for Water Resource Studies, Halifax, NS.

Jechalke, S., Vogt, C., Reiche, N., Franchini, A., Borsdorf, H., Neu, T., et al. (2010). Aerated treatment pond technology with biofilm promoting mats for the bioremediation of benzene, MTBE and ammonium contaminated groundwater. *Water Research , 44* (6), 1785-1796.

Jenssen, P., & Siegrist, R. (1990). Technology assessment of wastewater treatment by soil infiltration systesms. *Water Science Technology , 22* (3/4), 83-92.

Kim, H., Gellner, J., Boltz, J., Freudenberg, R., & Gunsch, C. (2010). Effects of integrated fixed film activated sludge media on activated sludge settling in biological nutrient removal systems. *Water Research , 44* (5), 1553-1561.

Kiso, Y., Jung, Y., Ichinari, T., Park, M., Kitao, T., Nishimura, K., et al. (2000). Wastewater treatment performance of a filtration bio-reactor equipped with a mesh as a filter material. *Water Research , 34* (17), 4143-4150.

Koerner, G., & Koerner, R. (1990). Biological activity and potential remediation involving geotextile landfill leachate filters. In R. Koerner (Ed.), *Geosynthetic Testing for Waste Containment Applications* (pp. 313-334). Philadelphia: ASTM International.

Koerner, G., & Koerner, R. (1988). Biological clogging in leachate collection system. *2nd Seminar on Durability and Aging of Geosynthetics.* Philadelphia: Geosynthetics Research Institute.

Korkut, E., Martin, J., & Yaman, C. (2006). Wastewater treatment with biomass attached to porous geotextile baffles. *Journal of Environmental Engineering , 132* (2), 284-288.

Krkosek, W., Ragush, C., Boutilier, L., Sinclair, A., Krumhansl, K., Gagnon, G., et al. (2012). Treatment performance of wastewater stabilization ponds in Canada's far north. *15th International Conference on Cold Regions Engineering.* Quebec: Canadian Society for Civil Engineering.

Lamb, B., Gold, A., Loomis, G., & McKiel, C. (1991). Nitrogen removal for on-site sewage disposal: Field evaluation of buried sand filter/greywater systems. *Transactions of the ASAE , 34* (3), 883-889.

Leverenz, H., Darby, J., & Tchobanoglous, G. (2000). *Evaluation of textile filters for the treatment of septic tank effluent.* University of California, Department of Civil and Environmental Engineering. Davis, California: Centre for Environmental and Water Resources Engineering.

Levin, G., & Shapiro, J. (1965). Metabolic uptake of phosphorus by wastewater organisms. *Journal of the Water Pullution Control Federation , 37* (6), 800-821.

Ling, J., & Chen, S. (2005). Impact of organic carbon on nitrification performance of different biofilters. *Aquacultural Engineering , 33*, 150-162.

Lowe, K., Rothe, N., Tomaras, J., DeJong, K., Tucholke, M., Drewes, J., et al. (2007). *Influent constituent characteristics of the modern waste stream from single sources: Literature review.* Water Environment Research Foundation. London: IWA Publishing.

Matveev, Y., & Gorchakova, V. (2007). Nonwoven textiles for treatment of wastewaters to remove petroleum products. *Fibre Chemistry , 39* (3), 231-233.

McCray, J., & Christopherson, S. (2008). On-site wastewater systems and interactions with the environment. *Journal of Hydrologic Engineering , 13* (8), 653-654.

McIsaac, R., & Rowe, R. (2006). Effect of filter-separators on the clogging of leachate collection systems. *Canadian Geotechnical Journal , 43* (7), 674-693.

Mendonça, M., Cammarota, M., Freire, D., & Ehrlich, M. (2004). A new procedure for treatment of oily slurry using geotextile filters. *Journal of Hazardous Materials , 110* (1), 113-118.

Metcalf & Eddy, Inc. (2003). *Wastewater engineering: Treatment and reuse* (4th Edition ed.). New York, New York, USA: McGraw-Hill.

Miyamoto, H., & Heinke, G. (1979). Performance evaluation of an arctic sewage lagoon. *Canadian Journal of Civil Engineering , 6* (2), 324-328.

Moll, D., Summers, R., Fonseca, A., & Matheis, W. (1999). Impact of temperature on drinking water biofilter performance and microbial community structure. *Environmental Science & Technology , 33* (14), 2377-2382.

Mulligan, C., Davarpanah, N., Fukue, M., & Inoue, T. (2009). Filtration of contaminated suspended solids for the treatment of surface water. *Chemosphere , 74* (6), 779-786.

Nova Scotia Environment. (2009, April). *On-site sewage disposal systems technical guidelines.* Retrieved January 21, 2013, from Regulations and Technical Guidelines: http://www.gov.ns.ca/nse/wastewater/regulations.tech.guidelines.asp

O'Toole, G., Kaplan, H., & Kolter, R. (2000). Biofilm formation as microbial development. *Annual Reviews in Microbiology , 54* (1), 49-79.

Palmeira, E., Remigio, A., Ramos, M., & Bernardes, R. (2008). A study on biological clogging of nonwoven geotextiles under leachate flow. *Geotextiles and Geomembranes , 26*, 205-219.

Pano, A., & Middlebrooks, E. (1982). Ammonia nitrogen removal in facultative wastewater stabilization ponds. *Journal of the Water Pollution Control Federation , 54* (4), 344-351.

Parker, D., & Richards, T. (1986). Nitrification in trickling filters. *Journal of the Water Pollution Control Federation* , 896-902.

Parker, D., Lutz, M., Dahl, R., & Bernkopf, S. (1989). Enhancing reaction rates in nitrifying trickling filters through biofilm control. *Journal of the Water Pollution Control Federation , 61* (5), 618-631.

Peeples, I., Manel, K., & Widrig, D. (1991). An examination of the role of sand depth on the treatment efficiency of pilot scale intermittent sand filters. *6th National Symposium on Individual and Small Community Sewage Systems.* St. Joseph: American Society of Agricultural Engineers.

Randall, C., & Sen, D. (1996). Full-scale evaluation of an integrated fixed-film activated sludge process for enhanced nitrogen removal. *Water Science and Technology , 32* (12), 155-162.

Ratkowsky, D., Olley, J., McMeekin, T., & Ball, A. (1982). Relationship between temperature and growth rate of bacterial cultures. *Journal of Bacteriology , 149* (1), 1-5.

Rodgers, M., Healy, M., & Mulqueen, J. (2005). Organic carbon removal and nitrification of high strength wastewaters using stratified sand filters. *Water Research , 39* (14), 3279-3286.

Rohde, J., & Gribb, M. (1990). Biological and particulate clogging of geotextile/soil filter systems. In R. Koerner (Ed.), *Geosynthetic Testing for Waste Containment Applications* (pp. 299-312). Philadelphia: ASTM International.

Rowe, R., Armstrong, M., & Cullimore, D. (2000). Mass loading and the rate of clogging due to municipal solid waste leachate. *Canadian Geotechnical Journal , 37* (2), 355-370.

Rowe, R., Fleming, I., Cullimore, R., Kosaric, N., & Quigley, R. (1995). *A research study of clogging and encrustation in leachate collection systems at municipal solid waste landfills.* The University of Western Ontario, Department of Civil and Environmental Engineering. London Ontario: Geotechnical Research Centre.

Rowe, R., VanGulck, J., & Millward, S. (2002). Biologically induced clogging of a granular medium permeated with synthetic leachate. *Journal of Environmental Engineering and Science , 1* (2), 135-156.

Rusten, B., Hem, L., & Ødegaard, H. (1995). Nitrification of municipal wastewater in moving-bed biofilm reactors. *Water Environment Research , 67* (1), 75-86.

Ryther, J., & Dunstan, W. (1971). Nitrogen, phosphorus, and eutrophication in the coastal marine environment. *Science , 171* (975), 1008-1013.

Shukla, S., & Yin, J.-H. (2006). *Fundamentals of geosynthetic engineering.* London, United Kingdom: Taylor & Francis Group.

Stevens, P., Blewett, D., & Casey, J. (2006). Short-term effects of a low dissolved oxygen event on estuarine fish assemblages. *Estuaries and Coasts , 29* (6), 997-1003.

Stevik, T., Ausland, G., Jenssen, P., & Seigrist, R. (1999). Removal of *E. coli* during intermittent filtration of wastewater effluent as affected by dosing rate and media type. *Water Research , 33* (9), 2088-2098.

Tao, J., Mancl, K., & Tuovinen, O. (2009). Treatment of sanitary sewer overflow with fixed media bioreactors. *Applied Engineering in Agriculture , 25* (1), 39-43.

Terrafix. (2011). *Terrafix geotextiles.* Retrieved August 15, 2011, from Geotextiles: http://www.terrafixgeo.com/assets/pdf/geotextiles1.pdf

Torrens, A., Molle, P., Boutin, C., & Salgot, M. (2009). Impact of design and operation variables on the performance of vertical-flow constructed wetlands and intermittent sand filters treating pond effluent. *Water Rasearch , 43* (7), 1851-1858.

USEPA. (1975). *Process design manual for nitrogen control.* Office of Technology Transfer. Washington, D.C.: US Environmental Protection Agency.

USEPA. (2002). Renovation/restoration of subsurface wastewater infiltration systems (SWIS). In *USEPA Onsite Wastewater Treatment Systems Manual* (pp. TFS-77). U.S. Environmental Protection Agency.

Van Cuyk, S., Siegrist, R., Logan, A., Masson, S., Fischer, E., & Figueroa, L. (2001). Hydraulic and purification behaviors and thier interactions during wastewater treatment in soil infiltration systems. *Water Research , 35* (4), 953-964.

Vandevivere, P. (1999). New and broader applications of anaerobic digestion. *Critical Reviews in Environmental Science and Technology , 29* (2), 151-173.

Water Environment Federation. (2010). *Biofilm Reactors WEF MOP 35.* Alexandria, Virginia: Water Environment Federation Press.

Wilson, J., Boutilier, L., Jamieson, R., Havard, P., & Lake, C. (2011). Effects of hydraulic loading rate and filter length on the performance of lateral flow sand filters for on-site wastewater treatment. *Journal of Hydrologic Engineering , 16* (8), 639-649.

Wootton, B., Durkalec, A., & Ashley, S. (2008). *Canadian Council of Ministers of the Environment draft Canada-wide strategy for the management of municipal wastewater effluent Nunavut regional impact analysis.* Fleming College, Centre for Alternative Wastewater Treatment, Lindsay, Ontario.

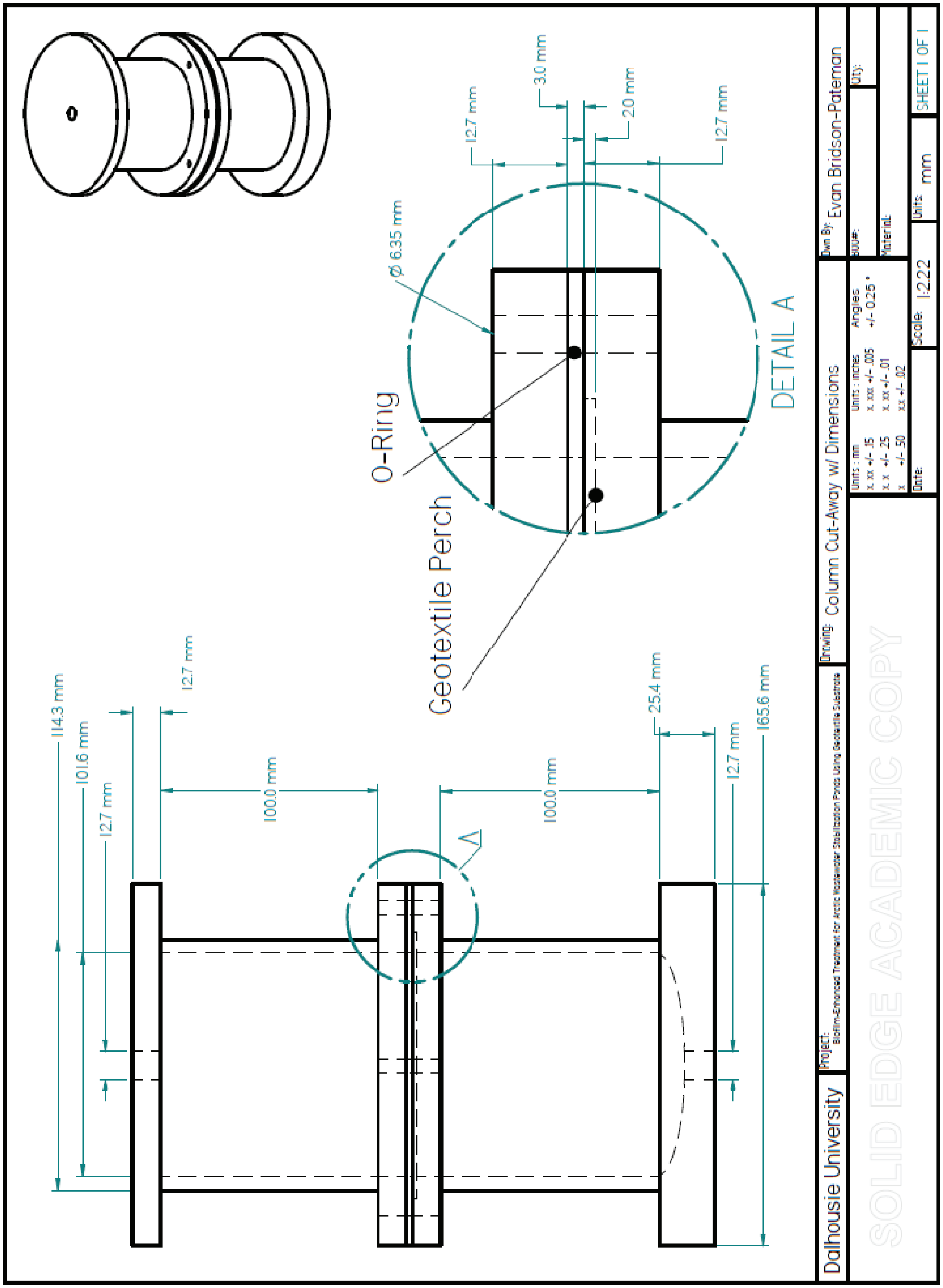
Yaman, C., Martin, J., & Korkut, E. (2005). Use of layered geotextiles to provide a substrate for biomass development in treatment of septic tank effluent prior to ground infiltration. *Journal of Environmental Engineering , 131* (12), 1667-1675.

Ye, J., McDowell, C., Koch, K., Kulick, F., & Rothermel, B. (2009). Pilot testing of structured sheet media IFAS for wastewater biological nutrient removal. *Water Environment Federation Technical Exhibition Conference* (pp. 4427-4442). Orlando: Water Environment Federation.

Zhang, E., Wang, B., Wang, Q., Zhang, S., & Zhao, B. (2008). Ammonia–nitrogen and orthophosphate removal by immobilized *Scenedesmus sp.* isolated from municipal wastewater for potential use in tertiary treatment. *Bioresource technology , 99* (9), 3787-3793.

Zhang, J., Huang, X., Liu, C., Shi, H., & Hu, H. (2005). Nitrogen removal enhanced by intermittent operation in a subsurface wastewater infiltration system. *Ecological Engineering , 25* (4), 419-428.

# APPENDIX A



114.3 mm
101.6 mm
12.7 mm
12.7 mm
100.0 mm
100.0 mm
25.4 mm
12.7 mm
165.6 mm
A
O-Ring
Geotextile Perch
Ø 6.35 mm
12.7 mm
3.0 mm
2.0 mm
12.7 mm
DETAIL A
Dalhousie University
Project: Biofilm-Enhanced Treatment for Arctic Wastewater Stabilization Ponds Using Geotextile Substrate
Drawing: Column Cut-Away w/ Dimensions
Dwn By: Evan Bridson-Pateman
Qty:
Material:
Scale: 1:2.22
Units: mm
SHEET 1 OF 1
Date:
SOLID EDGE ACADEMIC COPY

# APPENDIX B

| Date | ID | Elevation ΔH (cm) | Avg. Fill Time (s) | Fill Volume ($cm^3$) | Q ($cm^3/s$) | Total Head Losses (cm) | Net Head Difference (cm) | Length of Filter - L (cm) | $i = \Delta h/\Delta L$ | Filter Area ($cm^2$) | k = q/*i*A (cm/s) |
|---|---|---|---|---|---|---|---|---|---|---|---|
| 25-Feb | 4A | 13.7 | 32.50 | 225 | 6.92 | 0.36 | 13.34 | 10.6 | 1.26 | 89.9 | 0.06 |
| 25-Feb | 6A | 12.1 | 16.77 | 225 | 13.42 | 1.42 | 10.68 | 10.3 | 1.04 | 80.1 | 0.16 |
| 25-Feb | gA | 13.7 | 13.29 | 225 | 16.93 | 3.02 | 10.68 | 10.4 | 1.03 | 89.9 | 0.18 |

| | | | | | | | Friction Losses: | | | Bend Losses: | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Velocity (cm/s) | $V^2/2g$ | Temp (°C) | μ (kg/m·s) | ρ ($kg/m^3$) | ν ($m^2/s$) | Re | *f* | Tube Length (m) | Friction Head Loss (cm) | $f_T$ | Loss Coeff. (total) $K_T$ | Head Loss (cm) |
| 9.72 | 4.8E-04 | | | | | 708.0 | 0.090 | 2.23 | 1.0E-04 | 0.028 | 3.81 | 0.18 |
| 18.83 | 1.8E-03 | 10 | 1.31E-03 | 999.7 | 1.31E-06 | 1372.3 | 0.047 | 1.55 | 1.4E-04 | 0.028 | 4.19 | 0.76 |
| 23.77 | 2.9E-03 | | | | | 1731.7 | 0.037 | 0.78 | 8.7E-05 | 0.028 | 6.83 | 1.96 |

| Minor Losses: | | | | | | Column Losses: | | System Entrance Losses: | | |
|---|---|---|---|---|---|---|---|---|---|---|
| Ball Valve Loss Coeff. | Head Loss (cm) | T-Junction Loss Coeff. | Head Loss (cm) | Expansion Loss Coeff. | Head Loss (cm) | Contraction Loss Coeff. | Head Loss (cm) | Entrance Loss Coeff. | Head Loss (cm) | Total Head Losses (cm) |
| | 0.00 | | 0.08 | | 0.05 | | 0.02 | | 0.02 | 0.36 |
| 0.084 | 0.02 | 1.68 | 0.30 | 0.98 | 0.18 | 0.42 | 0.08 | 0.5 | 0.09 | 1.42 |
| | 0.02 | | 0.48 | | 0.28 | | 0.12 | | 0.14 | 3.02 |