

# **Development of a Combined Reed Bed – Freezing Bed Technology to Treat Septage in Cold Climates**

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**A Thesis Submitted in Partial Fulfillment of the Requirements for the  
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## **Abstract**

The Government of Ontario plans to ban the land application of untreated septage; however, most town wastewater treatment plants do not have the capacity to accept septage. A combined reed bed – freezing bed technology has been successfully developed to dewater and treat septage. Lab column studies established that freeze-thaw conditioning can restore drainage in clogged sand drying beds dosed with common biological sludges and that septage can be dosed at 10 cm/week for 2.5-5.0 months before clogging is observed. Pilot studies showed that freezing beds can operate without the need for a cover with the applied sludge effectively melting any snow cover in regions with moderate snowfall. Septage freezing was successfully modelled following an accepted model for ice formation on water bodies while septage thawing was modelled using a regression analysis with initial frozen depth and precipitation found to be insignificant and degree days of warming controlling the rate of thawing. Model results were utilized to produce a freezing bed design loading map for North America based on temperature normals. Field scale planted and unplanted reed bed – freezing bed systems were constructed and tested with varying hydraulic loading rates (1.9-5.9 m/y) and solid loading rates (43-144 kg/m<sup>2</sup>/y) over a 5 year period resulting in a recommended design hydraulic loading rate of 2.9 m/y or 75 kg/m<sup>2</sup>/y. Drainage rates doubled after freeze-thaw conditioning compared to during the growing season, suggesting that freeze-thaw conditioning restores filter hydraulic conductivity. No effect of solid loading rate, planted versus unplanted filters and 7 versus 21 d dosing cycles on filter drainage was observed; however, drainage varied significantly with hydraulic loading rate. The filters separated almost all contaminants with filtrate equivalent to a low-strength domestic wastewater which can be easily treated in any municipal or decentralized wastewater system. The dewatered sludge cake had similar nutrient and solid content to a solid dairy manure and met biosolid land application standards in terms of metals and pathogens. The combined reed bed-freezing bed technology can provide a low-cost solution for the treatment and reuse of septage in cold-climate regions.

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## List of Abbreviations

- AIT – Asian Institute of Technology
- AD – anaerobic digestion
- ANOVA – analysis of variation
- B/E3 – benzene to toluene ratio
- BOD<sub>5</sub> – 5 day biochemical oxygen demand (mg/L)
- CFM – cubic feet per minute (ft<sup>3</sup>/min)
- CFU – coliform forming units
- COD – chemical oxygen demand (mg/L)
- CST – capillary suction time (s)
- D10 – soil diameter with 10% sample passing
- DM – dry matter (%)
- E4/E6 – ratio of humic fractions
- EC – electrical conductivity (mS/cm)
- ET - evapotranspiration
- FA – fulvic acids
- FOG – fats, oils and greases
- HA – humic acids:  $C_{HA} / C_{hum} \times 100$ ;
- HI – humification index:  $C_{HA} / C_{org} \times 100$
- HLR- hydraulic loading rate (cm/week)
- HPC – heterotrophic plant count
- HR – humification ratio:  $C_{hum} / C_{org} \times 100$
- K/E3 – acetic acid/toluene ratio
- MC – moisture content (%)
- MMAH – Ontario Ministry of Municipal Affairs and Housing
- MOE – Ontario Ministry of Environment
- OCWA – Ontario Clean Water Agency
- OM – organic matter (%)

OMAFRA – Ontario Ministry of Agriculture, Food and Rural Affairs

O/N – pyrrole to furfural ratio

ORP – oxygen reduction potential (mV)

PE – Person equivalent

PI – polymerisation index:  $C_{HA} / C_{FA}$

PPT – precipitation

PVC – polyvinyl chloride

RBC – rotating biological contactor

SAR – sodium adsorption ratio

sCOD – soluble COD (mg/L)

SLR – solids loading rate (kg TS/m<sup>2</sup>·year)

SVI – sludge volume index (mL)

TKN – total Kjeldahl nitrogen (mg/L)

TN – total nitrogen (mg/L)

TOC – total organic carbon (mg/L)

TP – total phosphorus (mg/L)

TS – total solids (mg/L)

TSS – total suspended solids (mg/L)

TWAS – thickened waste activated sludge

UC – uniformity coefficient for soil (D<sub>60</sub>/D<sub>10</sub>)

USACE – US Army Corp of Engineers

USACERL - US Army Construction Engineering Research Laboratories

USEPA – United States Environmental Protection Agency

VFA – volatile fatty acid

VS – volatile solids (mg/L)

VSS – volatile suspended solids (mg/L)

WAS – waste activated sludge

WWTP – wastewater treatment plant

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## **1 Introduction**

Septage (accumulated solids in septic tanks) has traditionally been land applied without prior treatment to agricultural soils in Ontario as in other jurisdictions throughout North America. However, with increased public concern over environmental issues surrounding the land application of untreated septage, increasingly stringent regulations are coming into force. The Ontario Government committed to banning the land application of untreated septage over a five year period (OMOE, 2008); however, in order to implement the ban, sufficient capacity to treat septage either at rural municipal wastewater treatment plants (WWTPs) or at independent septage treatment facilities is required. There are a variety of options to manage septage including:

- co-treatment at municipal WWTPs, either mixed at the headworks or added directly to the sludge treatment train;
- lime stabilization followed by land application;
- dedicated septage treatment followed by land application. Technologies include aerobic or anaerobic digestion and dewatering technologies.

Table 1-1 compares some of the advantages and disadvantages of various options to treat septage (USEPA, 1984; Martel, 1999). Low capital and operating cost options including lime stabilization, sand drying beds, reed bed filters and freezing beds, which are appropriate for rural areas where land costs are low and proximity to agricultural land for land spreading is practical.

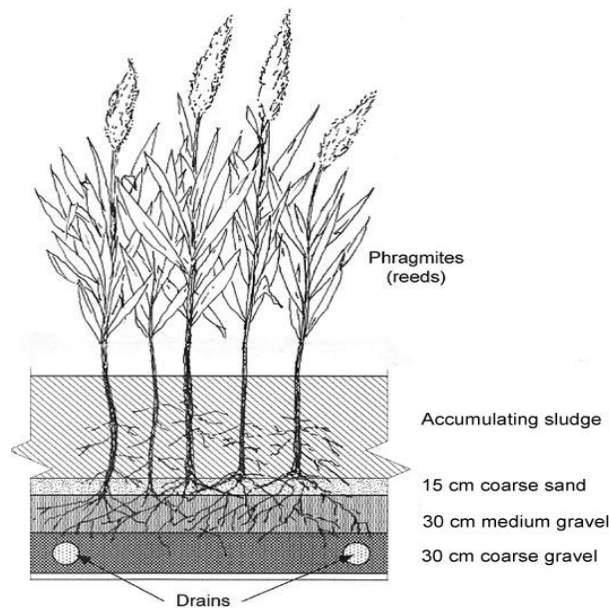
**Table 1-1. Septage Treatment Options**

<b>Treatment Option</b>	<b>Advantages</b>	<b>Disadvantages</b>
Lime stabilization	<ul style="list-style-type: none"> <li>○ Low cost option</li> <li>○ Minimum treatment to meet Ontario regulations</li> </ul>	<ul style="list-style-type: none"> <li>○ Winter storage required</li> <li>○ Reticence from farmers to apply on agricultural soils with existing high pH</li> </ul>
Co-treatment at municipal WWTP	<ul style="list-style-type: none"> <li>○ If capacity exists, works well with WAS and AD technologies</li> <li>○ dedicated solid/liquid separation at WWTP with filtrate returned to headworks</li> </ul>	<ul style="list-style-type: none"> <li>○ Distance to centralized WWTP</li> <li>○ Insufficient capacity at most small WWTP</li> <li>○ Not feasible for most lagoon systems</li> </ul>
Dedicated Aerobic Treatment	<ul style="list-style-type: none"> <li>○ Low footprint</li> <li>○ Moderate capital cost</li> <li>○ Stabilized sludge can be land applied</li> </ul>	<ul style="list-style-type: none"> <li>○ High energy cost</li> <li>○ May require sludge dewatering with further treatment of filtrate</li> </ul>
Dedicated Anaerobic Treatment	<ul style="list-style-type: none"> <li>○ Low footprint</li> <li>○ Low energy costs</li> <li>○ Stabilized sludge can be land applied</li> </ul>	<ul style="list-style-type: none"> <li>○ High capital cost</li> <li>○ May require sludge dewatering with further treatment of filtrate</li> </ul>
Solid/Liquid Separation (belt presses, centrifuges)	<ul style="list-style-type: none"> <li>○ Low footprint</li> <li>○ Stabilized solids can be land applied</li> </ul>	<ul style="list-style-type: none"> <li>○ High capital and operating costs</li> <li>○ Requires further treatment of filtrate</li> </ul>
Sand Drying Beds	<ul style="list-style-type: none"> <li>○ Low capital costs</li> <li>○ Solids can be land applied</li> </ul>	<ul style="list-style-type: none"> <li>○ Does not work in winter</li> <li>○ High footprint</li> <li>○ High operating costs</li> <li>○ Requires further treatment of filtrate</li> </ul>
Reed Bed Filters	<ul style="list-style-type: none"> <li>○ Low capital costs</li> <li>○ Low operating costs</li> <li>○ Solids can be land applied</li> </ul>	<ul style="list-style-type: none"> <li>○ High footprint</li> <li>○ Requires further treatment of filtrate</li> </ul>
Freezing Beds	<ul style="list-style-type: none"> <li>○ Low capital costs</li> <li>○ Utilizes natural freeze-thaw conditioning during winter</li> <li>○ Produces granular high solid dewatered sludge cake</li> </ul>	<ul style="list-style-type: none"> <li>○ Only applied at pilot scale</li> <li>○ Operates solely during winter months</li> <li>○ Requires a roof to avoid snow covering the bed</li> </ul>

According to the 2005 Ontario Provincial Policy Statement new lots serviced by onsite wastewater treatment systems (i.e. not connected to the municipal sewer) can be created only if there is confirmation of sufficient reserve sewage system capacity to treat the septage produced (OMMAH, 2007). Most municipal sewage treatment plants in rural communities, where the majority of septage is being generated, are not equipped to receive and treat septage. The capital cost to upgrade existing facilities can be prohibitive for many

small communities, hence there is a need to develop cost effective solutions for septage management. Reed bed systems were shown to have significantly lower lifecycle costs than comparable mechanical dewatering technologies (Nielsen, 2015).

It is proposed to develop a combined reed bed / freezing bed technology to dewater septage. Reed beds combine sand drying bed and constructed wetland technology and can be applied to dewater and *in-situ* stabilize sludge. Reed beds consist of lined basins with layers of gravel and coarse sand planted with water tolerant plant species (typically *Phragmites Australis*) (Figure 1-1). The key difference between reed bed filters and sand drying beds is that the dewatered sludge is frequently removed from drying beds (typically after each sludge application), whereas dewatered sludge is only removed from a reed bed filter after 5-10 years of operation; as the plants act to maintain filter drainage through stem movement and rhizome development. Freezing beds are simply sand drying beds where layers of sludge are consecutively applied to the bed during freezing conditions and allowed to freeze completely prior to adding a subsequent layer. The freezing process is a very effective solid-liquid separation technique with water draining freely as the frozen sludge thaws in the spring (Martel, 1999).



**Figure 1-1. Reed Bed Schematic**

(Credit: Ontario Rural Wastewater Centre, University of Guelph)

Sludge is applied periodically to the filter surface and is dewatered by gravity drainage and through evapo-transpiration (ET) during the growing season and through freeze-thaw conditioning from winter to spring. The underdrains are connected to aeration stand pipes which provide passive bed aeration. Sludge volumes are also reduced over time through the decomposition of organic matter and mineralization of the sludge. The stabilized sludge is removed at the end of the cycle and can be land applied as an organic fertilizer assuming metal and pathogen regulatory limits are met. The percolate can be discharged to a municipal WWTP, collected and treated by an onsite wastewater technology before subsurface discharge or land applied as a source of irrigation water and nutrients for crop growth.

The Reed Bed technology has been widely applied throughout Europe to dewater municipal waste activated sludge as well as anaerobic sludges (Nielson, 2003; Troesch *et al.*, 2009); however, very limited work has been done applying reed beds to treat septage and only empirical observation of winter operation have been made (Mellstrom and Jager, 1994). Freezing beds have been successfully applied at the pilot scale to dewater a number of biological and chemical sludges, but not septage (Martel, 1993). Combining reed bed and freezing bed technologies can potentially provide a complete solution for septage management in cold-climate regions.

### 1.1 Hypothesis and Research Objectives

It is hypothesized that reed bed and freezing bed technologies can be combined to treat septage under Canadian climatic conditions taking advantage of plant development during the growing season and freeze-thaw conditioning during winter.

The goal of this research project is to develop a combined reed bed / freezing bed technology to provide a low energy and low-cost treatment solution for septage management under Canadian climatic conditions.

#### **The Research Objectives are:**

**Objective 1** – Characterize the impact of freeze-thaw conditioning on drying bed operation treating septage and other common types of sludge;

**Objective 2** – Model septage freezing and thawing as a function of the depth of sludge layer applied and average daily temperature and evaluate the impact of snow cover on freezing bed operation;

**Objective 3** – Evaluate the hypothesis that a combined reed bed – freezing bed technology can effectively treat septage year round under Canadian climatic conditions. Determine design hydraulic and solid loading rates for septage treatment. Determine relationships between filter drainage and filtrate quality with solid and hydraulic loading rates, dosing frequency, plant development and season;

**Objective 4** – Characterize dewatered sludge quality for agricultural reuse.

## 1.2 Thesis Layout

The thesis is organised in manuscript style. Chapter 2 provides a comprehensive literature review. Chapter 3 describes a laboratory scale study which addresses Objective 1. Chapter 4 describes a pilot scale study which addresses Objective 2. Chapters 5 and 6 describe a field scale study which addresses Objectives 3 and 4. Chapter 7 provides a global summary of the thesis findings. Appendix A describes the design and construction of the field scale drying bed and reed bed filters while Appendix B provides a detailed description of the analytical methods used in the study.

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## 2 Annotated Literature Review

### 2.1 Sludge Freeze-thaw Conditioning - Laboratory Studies

The freezing process has long been known to improve sludge dewatering, although the actual mechanisms of freeze-thaw dewatering are not well understood (Vesilind and Martel, 1990). Mechanisms involved in sludge freezing are complicated since phase change occurs in a multi-phase porous medium (Chu *et al.*, 2002). Sludge freezes at the same rate as water even though it contains electrolytes, organic and inorganic particles as well as microorganisms. The dewatering process occurs as particulate matter is rejected during ice crystal formation and consolidated into solid particles along the crystal boundary (Reed *et al.*, 1986). Clements *et al.* (1950) conducted pioneering work on sludge freezing. He concluded that: 1) complete freezing is essential to create granular particles, 2) freezing rate must be slow enough to reject solid particles from the forming ice crystal front and 3) freeze-thaw improves the dewaterability of all sludges.

It is generally agreed that water in sludges can be classified based on how bound the water is to the sludge solids. A general definition describes sludges in terms of bulk water and bound water, where bound water is the water associated with the sludge after 30 minutes of settling using the sludge volume index (SVI) test (Vesilind and Hsu, 1997). Another definition of bound water is the water that does not freeze in sludge at a given temperature, usually -20°C. Vesilind and Martel (1990) classify water in sludges as:

**Free water:** Water that surrounds the sludge flocs and does not move with the sludge flocs. *Free water* can be easily drained from the sludge.

**Interstitial water** - Water that is contained within the sludge floc or is held by capillary forces between particles. Breaking the sludge floc releases *interstitial water*. Some *interstitial water* can be removed through mechanical dewatering.

**Surface water**, or water that is held to the surface of particles through surface forces.

**Bound water:** Water that is chemically bound to individual particles. *Bound water* can only be released by destroying the individual particle.

It was hypothesized by Hoekstra and Miller (1967) that a thin layer of surface water on solid particles allows particles to slide along the ice crystal boundary, pushing solid particles along the ice front and colliding with other particles, forming larger flocs. Halde (1980) showed that freeze-thaw is most effective with smaller particles and does not work as well for primary sludge as for WAS, as primary sludge consists of larger particles. Vesilind and Martel (1990) suggest that larger particles greater than 100  $\mu\text{m}$  are trapped in the ice, while colloidal particles of less than 10  $\mu\text{m}$  are pushed along the developing ice front. Logsdon and Edgerley (1971) found that freezing rate was critical to final sludge dewaterability and that freezing rates above 43-65 mm/h decreased sludge dewaterability. Hung *et al.* (1996) noted a freezing rate of slower than 10 mm/h was effective for activated sludge, while Chu and Lee (1998) showed that the critical freezing rate for sewage sludges range from 2-7 mm/h. Visual observations of sludge freezing by Vesilind and Martel (1990) show ice needles descending into the sludge pushing aside solid particles and ultimately enveloping the solid flocs. With time, the trapped solids are dehydrated within the ice crystals into more compact particles.

Baskerville (1971) carried out a series of sludge dewaterability experiments following freeze-thaw conditioning. Samples were frozen overnight at  $-28^{\circ}\text{C}$ , thawed in a water bath at  $+28^{\circ}\text{C}$ , and the capillary suction time (CST) was measured. Raw sludge, activated sludge and alum sludge CST was reduced from 2550, 133 and 142 s before freezing to 182, 16 and 8 s, respectively, after freeze-thaw.

Using a CST apparatus a series of freezing experiments with AD sludge and WAS was carried out by Vesilind and Martel (1990). In the first experiment, the sludges were frozen at  $-6^{\circ}\text{C}$  then stored at temperatures down to  $-30^{\circ}\text{C}$ . No significant difference in filterability was observed between  $-6$  to  $-20^{\circ}\text{C}$ ; however, samples at  $-30^{\circ}\text{C}$  showed significantly better filterability for both types of sludge. When freezing the samples at different temperatures from  $-6$  to  $-30^{\circ}\text{C}$ , a marked decrease in filterability was observed for both types of sludge as temperature decreased. This suggests that if the sludge freezes too quickly, particles are

not able to migrate and consolidate. Finally storage time, from 1-8 days, showed a linear increase in filterability of the sludge, suggesting that surface water is extracted from sludge flocs slowly over time.

In a second set of CST experiments, the hypothesis that the mechanism of freeze-thaw conditioning is related to double layer compression was tested (Vesilind *et al.*, 1991). It was hypothesised that the dissolved solids expelled by the advancing ice layer would increase the ionic strength of the water, compress the double layer and neutralise the charges between particles, which would lead to increased aggregation. Salinities of 0-20,000 mg/L as NaCl were tested on alum sludge, WAS, chemical sludge and AD sludge. No increase in dewaterability with increasing salinity was observed with any of the four freeze-thawed sludges, suggesting that double layer compression is not a significant factor in freeze-thaw conditioning.

Wang *et al.* (2001) carried out freeze-thaw experiments on flotation-thickened WAS. A series of 3L containers of sludge were frozen at -10, -20 and -80°C for 24h then thawed at room temperature. Samples of the thawed sludge (150mL) were dewatered using an air pressure filtration apparatus at 10 N/cm<sup>2</sup>. Dewatering rates increased as freezing temperature increased, with similar dewatering rates at -10 and -20°C, which were significantly better than dewatering rate at -80°C; however, all freeze-thawed samples had better dewatering rates than the unfrozen control sample. Sludge samples frozen at -10°C exhibited significant increases in dewatering rate as the freezing time increased from 1 to 3 to 7 d. In a similar experiment, Kawasaki *et al.* (1991) found that freezing and thawing greatly improved gravitational settling of activated sludge due to floc compaction during the freezing process. Slow frozen sludge (1.9 mm/h) had superior solid-liquid separation to fast frozen sludge (9.1 mm/h).

Intracellular water can also be released during the freeze-thaw process. The first mechanism is the development of ice crystals damaging the cell wall and the second mechanism is the increase in osmotic pressure as high solute concentrations build up outside the cell as extracellular water freezes. The difference in osmotic pressure between the inside and outside of the cell causes the cell to dehydrate. Proteins and carbohydrates,

two major components in intracellular liquid, were measured by Wang *et al.* (2001) before and after freeze/thaw of thickened WAS. Both protein and carbohydrate concentrations in the sludge liquid increased dramatically after freeze-thaw and increased with decreasing freezing temperature. Bacterial plate counts decreased from  $1.3 \times 10^8$  CFU/mL in the unfrozen sludge to  $2.1 \times 10^7$ ,  $8.8 \times 10^6$  and  $5.0 \times 10^6$  CFU/mL in the sludge samples frozen at -80, -20 and -10°C, respectively.

In a series of experiments using a dilatometer, Vesilind and Hsu (1997) showed that in an aerobically digested sludge of 0.5 percent solids unfrozen water declined from 10 to 5 g/g dry solids as temperature was decreased from -2 to -20°C. Below -20°C no further decline was observed. In another experiment at a constant freezing temperature of -8°C, unfrozen water increased from <1 g/g dry solids to 5 g/g dry solids as sludge percent solids increased from 0.5 to 2.2 percent. These results suggest that as sludge particles dehydrate, interstitial water trapped within the sludge flocs increases in solute concentration which in turn decreases the freezing point. With higher sludge solids, more interstitial water is trapped between sludge flocs as the flocs are compressed when freezing, reducing sludge dewaterability.

Knocke and Trahern (1989) investigated rapid freezing of sludges with butane evaporated in the sludge as well as the addition of Freon 12 under the appropriate temperature and pressure conditions. Results showed that conventional freeze-thaw conditioning at -10°C provided much better sludge dewatering characteristics. Particle size distribution was measured using a HIAC PC-320 Particle Sizing and Analysis System and showed that rapid freezing actually produced very fine size particles (5-10  $\mu\text{m}$ ), while freezing at -10°C greatly increased the amount of solids retained on the 200  $\mu\text{m}$  sieve, indicating good particle agglomeration.

Martel (1989) conducted a series of column tests to evaluate the dewaterability of freeze-thawed sludge. Frozen layers of 30, 60, 120 and 200 cm of water treatment plant alum sludge, AD sludge and WAS were evaluated. Once thawed, drainage times for the three 200 cm sludge columns were 6.5, 11.1 and 18.5 min, respectively yielding 30, 37 and 17 percent solids, while the unfrozen control sludge samples had not drained after several days.

Pathogen attenuation in sludges will depend upon the type of pathogenic microorganism (virus, bacteria, protozoa) as well as the environmental conditions present (moisture content, temperature, time, exposure to sunlight, predation, adsorption). Freeze-thaw conditioning is one potential method of reducing pathogen numbers in sludges. Freezing can destroy microorganisms through two possible mechanisms. Under slow cooling conditions, osmotic pressure dehydrates the cell and intracellular solute concentrations become toxic whereas under high cooling conditions, ice nucleation within the cell could destroy the cell (Mazur, 1965). Martel (1989) reports that temperature and duration of freezing affect microbial die off with typhoid bacilli reductions of 50% by freezing 7 days, 90% in 14 d and 100% in 84 d.

Sanin *et al.* (1994) conducted a study of the effect of freeze/thaw on pathogen attenuation in sludges. In the first phase of the study both the effect of temperature and duration of freezing were evaluated. Aerobic sludge samples were frozen at -7°C and stored for 7 d at -7°C, -18°C and -25°C. As well, samples were frozen at -7°C, -18°C and -25°C and thawed after 1, 7 and 28 d. The samples were analysed for fecal coliforms, *Salmonella* and plaque forming units. Fecal coliform reduction of greater than 0.6 log were observed with storage times of 7 d or greater. *Salmonella* reductions of 0.7 to 1.1 log were observed at all storage times and temperatures except for 1 day at -7°C. Plaque forming units showed complete inactivation (5 log removal) at a minimum storage time of 7 d and -18°C and -25°C and at all temperatures at a 28 d storage time. In a second phase of the study, aerobic and anaerobic sludge samples were frozen at -25°C for 7 d and analysed for seven microbial indicator and pathogen species: fecal coliforms, fecal streptococci, plaque forming units, *Salmonella*, poliovirus, *Ascaris suum* ova and *Cryptosporidium parvum* oocysts. Very similar results between aerobically digested and anaerobically digested sludges were observed, respectively: a 1.1 and 1.9 log removal of fecal coliform, a 0.2 log removal of fecal streptococci, a 0.5 and 0.7 log removal of *Salmonella*, only 0.8 and 0.9 log removal of plaque forming units (contrary to the results of the first phase of the study), no reduction in *Ascaris* ova but complete inactivation of *Cryptosporidium parvum* oocysts. (Sanin *et al.*, 1994)

A study by Gao *et al.* (2006) considered the effect of freezing temperature (-5°C, -15°C, -35°C), storage time and freeze-thaw cycles on *E. coli* reduction in wastewater. Greater inactivation was achieved at warmer freezing temperatures (-5°C) and longer storage times. The number of freeze-thaw cycles greatly increased *E. coli* inactivation, from 0.5-1.0 log at one freeze-thaw cycle to 1.8-3.5 log at 5 freeze-thaw cycles. Spray freezing proved to be very effective, with a 4-5 log inactivation of *E. coli* in wastewater observed.

Chu *et al.* (1999) studied the effect of freeze-thaw on total coliform and heterotrophic plate count (HPC). HPC was reduced from 90-95% to 70-80% as freezing rate increased from 1.8 – 50.4 mm/h, while total coliform numbers were reduced from 70-85% to 50-80% as freezing rate increased from 1.8-18 mm/h, while no reduction was observed at a freezing rate of 50.4 mm/h. Total coliform numbers in sludges at a freezing rate of less than 21.6 mm/h contained less than  $3.2 \times 10^4$  CFU/ml DM, meeting the EPA class B sludge regulation of  $2 \times 10^6$  MPN DM. Freezing in liquid nitrogen resulted in no HPC or total coliform reductions.

### 2.1.1 Discussion

Many studies have demonstrated the effectiveness of freeze-thaw conditioning of sludges including: primary, WAS, TWAS, AD, chemical coagulant and water plant sludges (Clements *et al.*, 1950; Logsdon and Edgerly, 1971; Vesilind and Martel, 1990; Vesilind and Hsu, 1997; Chu and Lee, 1998). Freezing temperature plays an important role in dewaterability, with dewaterability increasing at freezing temperature increases. The most effective freezing temperatures generally ranged between -2 to -10°C and dewaterability decreased rapidly below -30°C (Logsdon and Edgerly, 1971; Baskerville, 1971; Hung *et al.*, 1996; Chu and Lee, 1998; Vesilind and Martel, 1990; Wang *et al.*, 2001). This is due to the freezing rate, as small particles are pushed along the ice front as the sludge freezes. If the freezing rate is too great, particles will become entrapped in the ice front and do not agglomerate into larger particles. The more slowly the sludge freezes, the more effective is the consolidation of solids. The period of time the sludge is frozen also plays an important role in increasing dewaterability. Dewatering rates were shown to increase as freezing time increased from 1-8 d (Vesilind and Martel, 1990; Wang *et al.*, 2001). It has been hypothesised that interstitial water within sludge flocs dehydrate over time, increasing dewatered sludge dry

matter. Freeze-thaw conditioning works best in agglomerating colloidal particles of less than 10 µm while particles greater than 100 µm tend to become trapped in the advancing ice front (Vesilind and Martel, 1990). Several authors have reported on pathogen and pathogen indicator attenuation from freeze-thaw conditioning, with increased log reductions at lower freezing rate (higher freezing temperature) and longer time frozen (Martel, 1989; Sanin *et al.*, 1994; Chu *et al.*, 1999). The number of freeze-thaw cycles of a wastewater was shown to increase pathogen kill (Gao *et al.*, 2006).

## **2.2 Freezing Bed Sludge Treatment – Pilot and Field Studies**

Sludge freezing has been applied at several wastewater plants by incorporating sludge freezing into their operating practices, either by leaving sludge in a drying bed over winter to freeze, or dedicating a sludge lagoon to winter freeze-thaw conditioning (Martel, 1999). For example, the City of Winnipeg utilised sludge freezing during the 1970s (Penman and Van Es, 1973). AD sludge was applied to drying beds, left to settle for several days, and the supernatant was pumped back to the treatment plant. This process continued through the fall then the thickened sludge remaining in the bottom of the drying beds (20% DM) was left to freeze. The frozen sludge was then scraped from the beds and spread on near-by agricultural fields during the winter. Farmers incorporated the sludge into the soil in the spring as soon as farm machinery had access to the fields.

A pilot freezing bed test site was established at the US Army Cold Regions Research and Engineering Laboratory in Hanover, New Hampshire where pilot studies were conducted over four winters from 1987-1990 (Martel and Diener, 1991; Martel, 1993). The pilot unit consisted of a 13.1 x 2.6 x 2.4 m (LxWxD) concrete basin with a sand filter media covered by a corrugated fibreglass roof. AD sludge was applied during the first two winters, WAS during the third winter and water treatment alum sludge during the fourth winter. During the first winter, 580 mm of sludge was frozen. The average temperature increased to above zero on March 20<sup>th</sup> and all sludge was thawed by May 11<sup>th</sup>. During the second winter, no sand layer was used and a total of 1.14m of sludge was applied to the bed. The average temperature increased to above zero on March 23<sup>rd</sup> and all sludge was thawed by June 6<sup>th</sup>. The lack of a sand layer impeded drainage of the bed. During the third winter,

890 mm of sludge was frozen. The average temperature increased to above zero on March 24<sup>th</sup> and all sludge was thawed by May 23<sup>rd</sup>. The pilot experiments produced 39% DM in the first year with AD sludge and 25% DM in the third year with WAS. During the second season, drainage problems were encountered due to the lack of a sand layer and anaerobic conditions developed in the spring creating odour problems. Effluent quality was of similar strength to domestic wastewater with average BOD<sub>5</sub> and TSS concentrations of 310 and 100 mg/L, respectively. During the fourth winter, 99 cm of water treatment plant alum sludge was frozen in the bed. Excellent results were achieved, with sludge solid concentration increasing from 0.5 to 82% after dewatering. The stabilized solids had a consistency similar to coffee grounds with an average size of 0.5 mm, a uniformity coefficient of 4.6 and a hydraulic conductivity of 50.4 cm/h, which is similar to a highly permeable soil. A full scale system was put into operation in 1990 at Fort McCoy, Wisconsin. During the first year of operation 1.0 m of AD sludge was applied and dry matter content in the sludge increased from 4.5% in the raw sludge to 78% in the reed bed after dewatering (Martel, 1993).

A pilot scale freeze-thaw experiment was conducted on RBC sludge (2-3% solids) from a northern mining exploration facility (Diak *et al.*, 2011). The pilot unit consisted of a stainless steel box with a 10 cm layer of medium-coarse sand. Sludge was applied in 10 cm layers for a total of 8 doses over a 3 month period. The unit was maintained at -10<sup>o</sup>C. Cake solids increased to 19% after thawing at room temperature for 15 days. Melt water COD averaged 3000 mg/L and TN averaged 400 mg/L, while TSS averaged only 70 mg/L; suggesting a high concentration of dissolved organics in the effluent.

A full scale freezing bed was established to treat the sludge from a WWTP in Sweden utilising primary precipitation, thickening and polymer addition (Hellstrom and Kvarnstrom, 1997). The freezing bed design consisted of 0.3-0.5 m of filter sand over a gravel layer with underdrains and 1.5 m of freeboard to accommodate the sludge during winter. Sludge is applied in 10 cm layers, allowing each layer to freeze before adding the next layer. The pilot continued over two winter seasons. During the first season, which was consistently below the freezing point, average DM increased from 6-10% to 53%, while during the second season, average DM increased from 2-8% to 26%. During the

second season, cores of the sludge showed layers of anaerobic sludge between layers of more porous sludge which had undergone complete freezing. Fully freeze-thawed sludge had a porous and fluffy structure and high DM compared with unfrozen sludge which had a compact structure and lower DM content. The thawing time is a limiting design parameter in many regions as successive layers of frozen sludge can be accumulated in a bed (Hellstrom, 1997).

In the US, a survey of reed bed filters report that three of the eighteen systems operate during the winter and report the accumulation of 0.3-0.6 m of frozen sludge which dewater very well in the spring producing a “layer of fluffy, friable sludge” (Mellstrom and Jager, 1994).

An experimental freezing bed was established in Sweden to dewater domestic septic tank waste (Hedstrom and Hanaeus, 1999). Septage from a household was transferred in November to a lined freezing bed. The septage depth was only 2-15cm and froze within a day. In the middle of May, the dried sludge was removed and composted. The DM of the septage applied to the bed had a DM content of 4-6%. After the bed thawed in May DM had increased to 11-21% and further increased to 25-95% after a further three weeks of drying. The lack of a drainage system limits the use of this concept to individual isolated dwellings as the surface loading rate is very small. However, excellent dewatering of septage was observed during the experiment.

A comprehensive investigation of freeze-thaw conditioning of aerated facultative lagoon sludges using column and pilot drying beds was conducted in Quebec at the École polytechnique de Montréal and at the Mont-Laurier WWTP by Desjardins and Brière (1996). The column experiments evaluated successive applications of sludge to a drying bed and another method, whereby unfrozen sludge is pumped from the bottom of the lagoon to the frozen surface. The cycle is continued until the frozen layer descends to the bottom of the lagoon. Conventional sand and gravel filtration (400mm) was compared with a Wedgewater drain system (50 mm) with no significant differences between the two types of drainage layers. The columns were loaded between 52 and 204 kg TSS/m<sup>2</sup> while the pilot beds were dosed 20-22 kg TSS/m<sup>2</sup>. Once thawed, all systems drained very quickly

at 5-13 min for chemical sludge and 10-37 min for biological sludge, with dry matter ranging from 33-37%. When a layer of sludge was only partially frozen, the drainage rate decreased dramatically from 54 to 0.06 cm/h for biological sludge and from 821 to 0.52 cm/h for chemical sludge. The authors recommended either removing snow cover or melting it with spray irrigated water. (Desjardins and Brière, 1996)

### Modeling Freezing Depth

Martel (1989) and Reed *et al.* (1986) proposed models to determine the design depth of sludge in a sludge freezing bed. Both approaches were based upon the differential equation describing steady heat flux through a composite slab, which is commonly used to determine ice formation on lakes and streams (USACE, 2002):

$$\frac{\partial h}{\partial t} = \frac{1}{\rho\lambda} \frac{(T_m - T_a)}{\left(\frac{h}{k_i} + \frac{1}{H_{ia}}\right)} \quad \text{(Equation 2.1)}$$

Where:

h = ice thickness

$T_m = 0^\circ\text{C}$

$T_a$  = air temperature, °C

t = time

$\rho$  = ice density

$k_i$  = thermal conductivity of ice

$H_{ia}$  = heat transfer coefficient from the ice surface to the atmosphere

$\lambda$  = latent heat of ice

If the heat conduction through the ice is the controlling rate of energy flux, then the  $H_{ia}$  term can be ignored and the equation solved as:

$$h = m(\Delta T \cdot t)^{1/2} \quad \text{(Equation 2.2)}$$

Where

$$m = \sqrt{\frac{2k_i}{\rho\lambda}}$$

h=depth of freezing, cm

m=proportionality coefficient, which depends on thermal conductivity, density and latent heat of material being frozen, cm (°C·d)<sup>1/2</sup>

ΔT= average negative daily temperature, °C

t=time period, d

Typical values for m are described in Table 2-1 (USACE, 2002).

**Table 2-1. Proportionality Constant m for Various Snow Covered Conditions**

Ice Cover Condition	m
Windy lake with no snow	2.7
Average lake with snow	1.7-2.4
Average river with snow	1.4-1.7
Sheltered small river	0.7-1.4

Reed *et al.* (1986) applied Equation 2.2 to a sludge freezing bed. He found that m = 2.01-2.14 cm (°C·d)<sup>1/2</sup> for sludges of less than 8% solids and suggests a design value of m=2.04 (°C·d)<sup>1/2</sup>.

Rearranging the expression and using a sludge layer of 8cm gives the following expression, which allows the operator to calculate the time required to freeze a layer based on actual, average or forecasted temperature:

$$\Sigma(\Delta T \cdot t) = 15.38^\circ\text{C} \cdot d \quad \text{(Equation 2.3)}$$

Equation 2.3 was successfully validated in a field trial in Duluth, Michigan, where successive 20 cm layers of sludge were applied to a lagoon during the winter of 1981, with field observations indicating similar total frozen sludge depth to design calculations based upon Equation 2.3. Ideal freezing layer depth was determined by both Martel (1989) and Reed *et al.* (1986) to be 8 cm for the northern half of continental United States. As average temperatures decline, the freezing layer depth increases. Layers of 23 cm were successfully frozen in Duluth, Minn. and a 46 cm layer was frozen in Fairbanks, Alaska (Reed *et al.*, 1986). The model suggests a potential for freezing 1.8 m of sludge during a winter season in the US north east, which would be similar to climatic conditions in eastern Ontario.

### 2.2.1 Discussion

A number of pilot studies have focused on optimizing the design and operation of sludge freezing beds and this technology has been proven to effectively dewater a variety of sludges including WAS, AD sludge, alum sludge and aerated lagoon sludge (Martel, 1999; Penman and Van Es, 1973; Farrell *et al.*, 1970; Reed *et al.*, 1986; Martel, 1989; Martel and Diener, 1991; Martin, 1993; Hellstrom and Kvarnstrom, 1997; Hellstrom, 1997; Desjardins and Brière, 1996). It is suggested that sludge be applied in 8-10cm layers and allowed to fully freeze before the next layer is applied (Farrell *et al.*, 1970; Reed *et al.*, 1986, Martel, 1989). Dry matter after freeze-thaw varies from 39% for an AD sludge and 25% for WAS (Martel and Diener, 1991) or up to 78% after a drying period (Martel, 1993). At a pilot facility in Sweden, DM increased to 53% during the first season and 26% during the second season, when temperatures were not consistently below the freezing point (Hellstrom and Kvarnstrom, 1997). Dewatered sludge will have a solids concentration of 17–35% right after freeze-thaw and can increase to greater than 50% within several weeks of drying (Reed, 1987). One pilot study evaluated the freeze-thaw of septage (Hedstrom and Hanaeus, 1999), although the system consisted of a lined pond without drainage.

The effectiveness of freeze-thaw conditioning at the pilot and full scale has been demonstrated, although the technology has not been applied specifically to dewater septage. As will be described in the next chapter, reed bed filters have not been specifically designed to operate as freezing beds during winter. By combining the benefits of reed bed

sludge dewatering in the growing season and freeze-thaw conditioning in winter, it should be possible to develop a technology which maximizes dewatering capacity throughout the years, resulting in a more efficient technology which can accept higher annual loading rates compared with a system designed either as a reed bed filter or as a freezing bed alone.

The potential depth of frozen sludge will depend primarily upon average winter temperature (degree days of freezing). Wind convection will increase the freezing rate, while snow cover and the reed stands will act to insulate the sludge bed. Ground temperature and the depth at which the system is installed (in-ground versus raised) will also affect the freezing rate, as will the temperature of the applied sludge. The thickness of the applied layer of sludge will impact the freezing rate as the top layer will freeze first and will act as an insulating layer to the sludge below.

Several authors have commented that snow should not be allowed to accumulate on the filter, as it will insulate the bed and hinder freezing (Martel, 1996; Reed *et al.*, 1986, Desjardins and Brière, 1996). However, covering the bed is not practical for large scale systems as it is costly and reduces air flow. The option of mechanically removing the snow is an operational headache and risks machinery becoming stuck in a partially frozen sludge bed. Melting the snow cover with irrigation water is feasible, but involves working with water outdoors during winter with the likelihood of pipes and pumps freezing. As a layer of sludge is applied, it should melt and incorporate the bottom layer of snow cover, thus reducing the insulation effect. Allowing more time to freeze a layer of sludge is more practical than trying to remove the snow. However, as Desjardins and Brière (1996) showed, it is essential that each layer of sludge fully freezes before the next layer is applied. Determining the proportionality constant ( $m$ ) of Equation 2.2 for conditions of snow covered freezing beds would be useful to evaluate the potential of using reed beds as freezing beds without complicating operations with snow removal.

### **2.3 Reed Bed Sludge Treatment – Pilot and Field Studies**

There are numerous pilot and full scale studies considering sludge dewatering with reed bed filters. Much of the development and research work comes from Europe, although the US also has a significant number of reed bed filters in operation. This Chapter presents a

thorough review of the scientific literature relating to sludge dewatering with reed bed filters. The papers are described by country or region, with the few papers dealing specifically with septage treatment presented at the end of the Chapter. Table 2-2 summarises the design and operating parameters of reed bed filters described in the literature.

### 2.3.1 Plant Effects

Plant species plays multiple roles in the filters including breaking apart the sludge matt and maintaining bed drainage, increasing evapo-transpiration and adding oxygen to the root zones.

With the exception of several studies in tropical climates, all sludge dewatering beds reviewed use the common reed (*Phragmites australis*). As *Phragmites* is an invasive species in Ontario, other plant species should be considered. In constructed wetland applications, reeds (*Phragmites*), cattails (*Typha*), bulrushes (*Scirpus*) and willows (*Salix*) have all been used with success (De Maeseneer, 1997). Cattails have root depths of 30-40 cm compared with 50-60 cm for reeds. Willows are of interest due to their high evapo-transpiration potential; however, the root depth could potentially lead to root clogging of the drainage system.

De Maeseneer (1997) provides a good summary of the advantages of using the common reed (*Phragmites*) for sludge dewatering. Broad leafed cattail (*Typha latifolia*) has the advantage that its initial growth is faster than *Phragmites* and can grow easily from seeds; however, the cattail stems fall down in the autumn, which can lead to die off when covered in sludge. The advantages of *Phragmites* are:

- Fast growth under diverse conditions.
- High transpiration capacity, typically 1500 mm/y in Western Europe limited to the summer months where daily ET can exceed 2.5 cm.
- Tolerance to varying water levels and drought conditions.
- Tolerance to low and high pH and salinity.
- Deep rhizome and root system.
- New root growth on nodes as they become enveloped by organic matter.
- Upright stems in winter time.
- Ready commercial availability and ease of planting.

Reeds are typically planted from pots, root balls or rhizomes. Plants establish themselves fully within 1-2 growing seasons (Nielson, 2003). Weed control is typically carried out after planting by flooding the bed with treated wastewater. The leaves of the plant must remain above the water level. In general flooding is carried out one month after planting and again the following spring. Alternatively a 5 cm layer of sludge application can inhibit weed development.

The effect of plant species was studied at the pilot scale (mesocosm trial) treating a fish farm sludge during summer in Montreal, Canada (Gagnon *et al.*, 2012; Gagnon *et al.*, 2013). Sludge was applied to planted reed bed filters (*Phragmites australis*, *Typha angustifolia*, *Scirpus fluviatilis*) as well as unplanted controls at 30 kgTS/m<sup>2</sup>/y. Planted systems outperformed the unplanted control in terms of filtrate quality, with *Phragmites* performing better than both *Typha* and *Scirpus*. Average system ET varied from 2.8 mm/d in the unplanted control to 10.3 mm/d (*Phragmites*), 5.9 mm/d (*Typha*) and 3.3 mm/d (*Scirpus*). Dewatered sludge DM was similar for all experiments, ranging from 28±1 and 28 ±5% for *Typha* and unplanted control, respectively to 31±3 and 33±15% for *Phragmites* and *Scirpus*, respectively; suggesting that sludge dewatering was efficient with or without plants. In a vertical reed bed study conducted in Poland by Bialowiec *et al* (2014), ET rates ranged from 1.0-3.0 mm/d in the first year of operation and from 2.6 to 4.6 mm/d in the second year.

### 2.3.2 Danish Studies – Biosolids Treatment

Sludge dewatering reed beds have been in operation in Denmark since 1988, with 105 systems in operation as of 2003 (Nielsen, 2003). Nielsen (2015) has shown that reed bed systems have much lower life cycle costs than equivalent chemical and mechanical dewatering technologies. Systems are typically designed for a 10-year operating cycle once the plants have become established. Systems are designed with mass surface loading rates of 50 kg TS/m<sup>2</sup>/y for waste activated sludge (WAS) and 60 kg TS/m<sup>2</sup>/y for WAS mixed with AD sludges. However, Nielsen (2011) reported on long term operating conditions at a number of reed bed facilities and found that dewatering issues, anaerobic conditions and plant die-off were encountered when sludge VS exceeded 65% or when fats exceeded 10,000 mg/kg DM. Maximal drainage rates of 0.008-0.020 L/S/m<sup>2</sup> were observed with VS

between 50-60%, while maximal drainage rates were 5-10 times lower when VS exceeded 65%.

Reed bed systems in Denmark treat sludge with dry matter from between 0.5 to 3-5%. There must be a minimum of 8 basins for a 10 year operating period. Maximum basin size should be 4000 m<sup>2</sup> to account for sludge distribution and percolate collection. A maximum of 250 m<sup>2</sup> per sludge applicator is recommended with pumps sized to deliver 0.15 m of sludge within one hour.

Each filter consists of an impermeable liner, an aeration / drainage network (10 cm / 4 inch PVC) at 3m centres, 0.55-0.60 m of filter material consisting of 30-45 cm pea gravel; 15 cm filter sand and a growth layer on top. Basin depths must be no less than 1.70 m from the filter surface.

The system design life should be at least 30 years, typically divided into three 10 y operating cycles. The commissioning period comprises the system construction and planting as well as the first and second growing seasons. Basins are dosed 1-3 times per day from a few days during commissioning to a maximum of 2 weeks for systems of 5-10 y. The majority of systems are loaded between 4-10 d with rest periods of 50-60 d. Dosing schedules are determined based upon annual solids surface loading rates (kg TS/m<sup>2</sup>/y) and number of basins in operation. Dosing to basins is stopped ½ - 1 year before emptying to allow for final dewatering and sludge stabilization. Gravity dewatering can achieve 20% DM, while long rest periods can increase DM to 40% through evapo-transpiration. (Nielsen, 2003)

Dominiak *et al.* (2011a) developed a sludge settling column test to determine drainage properties of activated sludge with gravity drainage called the Specific Resistance to Drainage (SRD). In a follow-up study they compared dewatering efficiency at several Danish reed bed facilities and found sheer forces, oxygen depletion and long-distance transportation impacted WAS dewaterability and found large variations between SRD measurements at 7 plants studied suggesting the need for sludge characterisation to size reed bed systems. They also found that hydraulic loading impacts drainage as the applied

layer can become anaerobic which can lead to the development of a compacted layer reducing drainage efficiency (Dominiak *et al.*, 2011b).

Iannelli *et al.* (2013) studied the drainage patterns during two dosing cycles at a mature reed bed system in Denmark treating WAS. The reed bed was dosed daily for 7-11 days followed by a 44-48 d rest period with an annual loading rate of close to 60 kg TS/m<sup>2</sup>/y, which is typical of Danish system operation. It was observed that most of the water drained from the bed over the 6-9 h following each 1 h loading event indicating that good hydraulic conductivity was maintained in the reed bed system. Water content, water soluble carbon and volatile solids declined with sludge depth while little change with time was observed following the initial 6-9 h drainage period. This suggests that the 45 d rest period could possibly be reduced with little impact on filter performance.

### 2.3.3 French Studies - Biosolids

There are approximately 300 sludge drying reed bed systems installed in France with a treatment capacity of between 200-26,000 person equivalents (PE) (Troesch *et al.*, 2009). The performance of a reed bed system established in 2003 was evaluated. The filters were of standard design with 55-60 cm of gravel and sand media and drainage / passive aeration pipes spaced at 2 m intervals. Each bed is loaded for a 2 week period, followed by a 14 week rest period at 30-40 kg TS/m<sup>2</sup>/y. Core samples at three depths showed high variability in DM, from 10-55% DM, with an average of 25% DM and 8% organic matter (OM) content. Dewaterability was determined through CST measurements and bound water measurements following the method of Kipp and Dichtl (2001). Redox potential in the reed beds was used to assess the oxidation level in the sludge. Weekly measurements of O<sub>2</sub>, CO<sub>2</sub> and CH<sub>4</sub> were taken under the filtration layer. Sludge humification and stabilization were assessed using OM, biologic stability indicator and biochemical organic matter characterization. It was found that 78±12% of the volume drains within 24 h to obtain a DM content of 8±4% in the top sludge layer. Beds achieved an average 30% DM in the top layer by the end of summer and 17% DM in winter. The decomposition of OM takes place principally in the top 10cm as ORP increases to above 200 mV. Organic matter in the bottom layer of sludge had stabilized at 60% VS (4yr old) from an initial VS of 80% in the sludge. Sludge stability indices show that the sludge stability was similar to that of

vegetable compost. Blocking every 2<sup>nd</sup> aeration stack had no significant observed impact on sludge stabilization, dewaterability or oxygen content in the filter media, suggesting that aeration stacks at 4 m intervals are sufficient. (Troesch *et al.*, 2009a)

A pilot study was carried out with 8 pilot filters (2m<sup>2</sup> ea.) consisting of a 15 cm drainage layer (30-60 mm pea stones), a 10 cm medium gravel layer (15-25 mm), a 20 cm fine gravel layer (2-6 mm) and a 5-10 cm filtration layer of either coarse sand (d<sub>10</sub>=0.35, UC=3.2) or compost. Six of the pilot filters received extended aeration WAS while 2 filters received extended aeration TWAS. The pilots were planted with 9 clumps/m<sup>2</sup> of one year old *Phragmites australis* plantlets. For the first 6 months the filters were dosed with treated wastewater only, followed by a 1.5 year acclimatization period at half sludge solid loading rate of 25-30 kg TSS/m<sup>2</sup>/y. All beds were dosed over a 3.5 d period. A rest period of 17.5 d versus 31.5 d did not impact sludge accumulation rates in the beds receiving WAS (5-6 cm/y accumulation); however, the beds receiving TWAS showed signs of clogging and had sludge accumulation rates of 9.3 and 19.6 cm/y with 31.5 and 17.5 d rest period, respectively. This suggests that reed beds are not effective at further dewatering TWAS. Reed density (stocks/m<sup>2</sup>) was measured over the two year commissioning period. During the 2007 growing season, reed stock density increased to 200-280 stocks/m<sup>2</sup>, suggesting that a 1 season acclimatization period is sufficient. Clogging was observed during the 3<sup>rd</sup> day of dosing to the sand filter but not to the compost filter, suggesting either better reed development or better capillary connection between the sludge and filtration media. The ORP drops to below 200 mV (anoxic conditions) when oxygen demand spikes in spring as the increasing temperature stimulates biological activity in the filters while ORP values approached 450 mV during the summer. (Troesch *et al.*, 2009a)

#### 2.3.4 US Studies - Biosolids

Mellstrom and Jager (1994) surveyed 18 reed bed systems in operation in northern New England. Most systems were sand drying beds which had been converted into reed beds simply by increasing the height of the berms or by adding sideboards. System commissioning takes a few months to a full growing season. *Phragmites* are planted at a density of 11 plantlets/m<sup>2</sup>. The bed is flooded with wastewater or watered continually for a few months to a full growing season, during which time little or no sludge is applied.

After each sludge dosing, the bed is rested for 1-2 weeks, or longer. Once sludge accumulation has reached approximately 1m, the bed is allowed to rest for a growing season to 1 year before the bed is desludged. Reeds are harvested annually before they shed their leaves (unlike systems in Europe which leave the plants in the filter to degrade and compost). Three of the eighteen systems operate during the winter and report the accumulation of 1-2 ft of frozen sludge which dewater very well in the spring producing a compost like material. Of the 18 systems surveyed, 15 received sludge from extended aeration plants, 1 from an aerated lagoon and 2 from RBCs. Several systems co-treated septage in digesters with municipal sludge. Hydraulic loading rates (HLR) varied from 1841-2659 L/m<sup>2</sup>/y while solids loading rates (SLR) varied from 18-82 kg TSS/m<sup>2</sup>/y, with an average HLR of 2150 L/m<sup>2</sup>/y and SLR of 41.6 kg TSS/m<sup>2</sup>/y. Sludge accumulation rates varied from 3 - 35 cm/y with an average of 13.1 cm/y. Six of the eighteen systems were closed, mostly relating to problems related to plant establishment, while reed growth was reported as good to excellent in 13 of 18 facilities. A common characteristic with successful systems was that they treated well digested sludges including a 4-5 year old lagoon sludge and a system which holds aerobic digester sludge throughout the winter. Most successful systems provided a 2 week rest period between sludge applications. The systems which applied sludge during the winter reported odours in the spring when the frozen sludge thawed and that the high organic loading may negatively impact plant growth in the spring.

A reed bed system was installed in the Northwestern US to dewater WAS from a vegetable processing industry (14,100 mg/L COD; 14,600 mg/L TSS; 970 mg/L TKN; 126 mg/L NH<sub>4</sub>) (Burgoon *et al.*, 1997). The filter consists of a layer of drain rock and a layer of coarse sand. Root cuttings were potted in 3.8 L containers, grown in greenhouses for six months then planted in the reed beds in the fall. Regrowth of the plants was greater than 95% the following spring. Solid loading was increased from 9.8 to 65 kg TS/m<sup>2</sup>/y over a four month acclimatization period. Solids were applied weekly until the end of November, stopped for the winter, and resumed in April of the following year. Sludge was applied weekly to maintain moisture for the reeds. After 1.5 years of operation the average sludge depth was 15.2 cm and solids content varied between 10-12%.

The US Army Construction Engineering Research Laboratories (USACERL) evaluated various sludge dewatering technologies for military bases and selected reed bed filters as the best alternative for sludge dewatering based on economic and technical feasibility and technology demonstration at Fort Campbell, KY, USA (Kim and Smith, 1997). Sand drying beds are a common sludge dewatering technology for small WWTPs. They are simple to operate and maintain. However, sand drying beds can have long dewatering times (2-4 weeks), intensive labour requirements to remove dewatered sludge, and can experience clogging. Reed beds can easily be modified from existing sand drying beds by adding 1-1.5 m sidewalls above the existing sidewalls of the sand filter and planting reeds in the filter at a density of 9 plants/m<sup>2</sup>. USACERL sent a questionnaire to 44 WWTPs that used sludge dewatering reed beds and received information from 24 WWTPs including both aerobic and anaerobic stabilized sludge. The facilities treating anaerobic sludge had hydraulic loading rates of 0.16 – 0.98 m/y, while facilities treating aerobic sludge had hydraulic loading rates of 0.73 – 7.3 m/y. Solids content of the anaerobic sludges varied between 2-10% with solids loading rates of 13-60 kg TS/m<sup>2</sup>/y while those of the aerobic sludges varied between 1-5% with solids loading rates of 16-106 kg TS/m<sup>2</sup>/y. USEPA (1987) reported an average of 81 kg TS/m<sup>2</sup>/y for 16 facilities operating in New Jersey, New York and North Carolina. By means of comparison, recommended design loading for sand drying beds varies between 100-160 kg TS/m<sup>2</sup>/y. A cost comparison of mechanical dewatering (new), Wedgewater (retrofit), reed bed (retrofit), composting (new) and sand filter (existing) showed the reed bed to be the low cost option with an annualized cost of half of the existing sand bed, due to higher O&M costs experienced with operating a sand filter. After 3 years, residual sludge was analysed at the demonstration reed bed system. Solids increased from 15% in the top layer (0- 10 cm) to 47% in the bottom layer (50-71 cm). VS declined from 49-46% of TS from the top to the bottom of the sludge layer. Sludge accumulated at 24 cm/y in the reed bed. Volume reductions were smaller in a warm climate (Kentucky) than systems in a colder climate (New England), likely due to the freeze-thaw conditioning in winter. The authors commented that loading rates are empirically based and likely do not fully take advantage of reed bed treatment capacity and that science based research is needed to optimize the technology (Kim and Smith, 1997).

A reed bed system located at the Buckland WWTP, Buckland, MA was studied over a 7 year period, from 1992-1999 (Begg *et al.*, 2001). Aerobically digested sludge (HRT 25-30 d) from the extended aeration plant reduces VS by 60-70% before the sludge is applied to the reed beds (1.2% TSS). Each bed is comprised of: 20-30 cm pea stone (3-6 mm) or gravel (>2 mm) and 30 cm of coarse sand (0.5-1 mm). After 6 years of operation, the dewatering efficiency is 92.6%. Measured water loss was 39 cm/month during the growing season and 4.7cm/month during the winter. Sludge accumulated at 12.5 cm/y. Effluent quality was excellent (TSS=14 mg/L; BOD<sub>5</sub>=6 mg/L). Average removal efficiency for metals was 87%, with metals accumulating in the sludge as opposed to the plant tissue. All metal concentrations except for copper met the Department of Environmental Protection standards for Type 1 sludge suitable for unrestricted land application.

### 2.3.5 Various Other Studies - Biosolids

A pilot reed bed system was established in the Gaza Strip, Palestine to compare sludge dewatering to that of a sand bed filter (Nassar *et al.*, 2006). The filter media is comprised of: 20 cm coarse gravel, (30-50 mm), 20cm medium gravel (10-30 mm), 10 cm fine gravel (5-10 mm) and 20cm sand. A sludge application frequency of every two weeks was observed to be better than a weekly application both in terms of reed health and sludge drying. Infiltration rates were calculated during the first 24 h after sludge application. The hydraulic loading rate varied between 1.4-3.2 cm/day over the 27 month study period. Infiltration stopped in the planted bed 4-5 d after dosing, while it took 7 d in the unplanted bed. The most suitable time to apply more sludge is when cracks in the sludge reach 10 cm depth, which typically occurred 10 d after infiltration had stopped. Both beds had an infiltration rate of 60 mm/day during the winter; however, during the summer the planted bed had an infiltration rate of 200 mm/day compared with only 150 mm/day for the unplanted bed. The infiltration rate correlated with the hydraulic loading rate for the planted bed ( $R^2=0.45$ ) but not for the unplanted bed ( $R^2=0.01$ ) indicating that the plants maintain drainage pathways through the filter media. In a follow-up paper (Nassar *et al.*, 2009), accumulated sludge volume (as a percent of sludge applied) was 4.2% for the planted bed and 4.5% for the unplanted bed; however, on a mass basis, 38% of the applied solids mineralized in the planted bed while only 24.9% mineralized in the unplanted bed.

In other words, the sludge density was greater in the unplanted bed which corresponds to the higher infiltration times observed in the unplanted bed.

A pilot study was conducted in Northern Italy where two reed bed filters were dosed with TWAS (3.1% DM) from Feb-April and WAS (0.5% DM) from May-July (Barbieri *et al.*, 2003). Each filter was dosed 8.3 cm followed by a 25 day rest period with TWAS loading and a 13 day rest period with WAS loading. Dried sludge increased from 13 to 20% during the TWAS loading period and from 20 to 30% during the WAS dosing period. Increasing DM was likely due to climatic effects (winter versus summer) as opposed to the type of sludge loaded to the filters. Sludge breakthrough was observed in bed 2 during the WAS loading period, otherwise filtrate water quality was good with COD values around 200 mg/L and TSS less than 30 mg/L. TN in the filtrate remained stable at 300-400 mg/L during the first phase of the study while it climbed to 500 mg/L during the second phase, likely due to concentration effect from higher ET.

A pilot study in England evaluated the dewatering of an agricultural slurry with reed bed filters and found that the planted beds performed better than the unplanted sand filter in terms of % DM and a greater reduction in the height of the residual sludge layer (Edwards *et al.*, 2001). The study found that high loadings of raw sludges in the spring can seriously damage newly emerging shoots of reed plants. It was observed that loading rates could be increased during the summer time to twice the suggested annual loading rate of 60 kg TS/m<sup>2</sup>/y.

Three full scale reed bed systems were evaluated in Catalonia Spain, with three sampling campaigns carried out in fall 2007, and spring and summer 2008 (Uggetti *et al.*, 2009). Each filter consists of 30 cm gravel (1-3 cm) and 25 cm sand (0.3-1 mm). Two systems treat extended aeration sludge at an SLR of 55 and 51 kg TS/m<sup>2</sup>/y at a loading cycle of 2 days dosing followed by 2 and 10 day rest periods, while the third system treats contact-stabilization sludge at an SLR of 125 kg TS/m<sup>2</sup>/y with no regular dosing pattern. Composite sludge samples were collected from three locations in each filter from the top and bottom layers and analysed for: pH, EC, TS, VS, COD, BOD<sub>5</sub> (easily biodegradable OM) and BOD<sub>21</sub> (slowly biodegradable OM), TKN, TP, heavy metals, *E. coli* and *salmonella*. The sludge

accumulation rate was much higher in the system receiving 125 kg TS/m<sup>2</sup>/y (33 cm/y) compared with a system receiving 55 kg TS/m<sup>2</sup>/y (7 cm/y); however, it was observed that the sludge layers actually decreased by 1/3-1/2 over the summer suggesting that it is important to measure sludge accumulation over an entire year cycle. Sludge DM was 7-26% in the top layer and 20-30% in the bottom layer of the two beds receiving ~50 kg TS/m<sup>2</sup>/y, compared with DM of 18-23% in the top layer and 11-13% in the bottom layer of the bed receiving 125 kg TS/m<sup>2</sup>/y. This suggests that the higher SLR (125 kg TS/m<sup>2</sup>/y) negatively impacts sludge dewatering.

Many studies of sludge reed bed systems describe dewatering efficiency empirically considering loading frequency, areal loading, drying efficiency and life expectancy. However, dewatering can be considered as comprised of two distinct phases; a relatively quick drainage phase, where pore water is drained from the filter, and a longer ET phase, where capillary water is extracted from the sludge. Giraldo and Iannelli (2009) conducted a side-by-side study of short-term dewatering comparing full scale reed bed and sand drying bed filters. The study was conducted over a 9 week period in the summer and winter in central Italy with WAS dosed to each bed every 3 weeks. Water content was measured daily from sample cores, while ET was estimated. Precipitation was collected by an onsite rain gauge throughout the study. A double exponential equation was developed to express water content, with the first term representing drainage and the second term ET. The equation was calibrated using nonlinear least squares regression on the DM data. Dewatering continued over the 3 week rest period during the summer months, increasing DM to 40-49%. During the winter, the planted filter dewatered sludge over a 1 week period to a maximum 8% DM compared with no drainage effect in the sand filter, indicating that the plants helped to maintain drainage pathways through the sludge. The study found an optimum 11 day dosing / rest cycle.

A study of 13 pilot reed bed filters treating WAS was conducted in Greece comparing media, loading rate and plant effects (Stefanakis and Tsihrintzis, 2011; Stefanakis and Tsihrintzis, 2012). A water balance was conducted on the systems with ET calculated to be the main mechanism of water loss with plants significantly enhancing ET. Annual ET values ranged from 938 to 2434 mm in the systems. For Mediterranean climates it is

recommended to follow a loading cycle of one week loading and three weeks rest during winter and one week loading and two weeks rest during the summer to avoid plant water stress. Common reeds were more resilient than cattails in the systems. Fine grained material appears to enhance treatment with a recommended design of 3 x 15 cm layers of fine, medium and coarse stones. The pilot filters were capable of dewatering 75 kg TS/m<sup>2</sup>/y with no signs of clogging.

### 2.3.6 Fecal Sludge Treatment in Tropical Climates

The Asian Institute of Technology (AIT) have been experimenting with reed bed treatment of septage since 1997 (Koottatep *et al.*, 2005). Three pilot beds planted in *Typha* (cattails) were loaded at a rate of 250 kg TS/m<sup>2</sup>/y. The filters consist of: 40 cm coarse gravel (55mm), 15 cm medium gravel (25mm), 10 cm fine sand (1mm) with a 1 m freeboard for solids accumulation. *Typha augustifolia* were collected from a natural wetland and planted in the cells at a density of 10-15 shoots/m<sup>2</sup>. Waste characteristics were: 15,400 mg/L TS; 2,300 mg/L BOD<sub>5</sub>; 17,000 mg/L COD; 1,100 mg/L TKN; 415 mg/L NH<sub>4</sub><sup>+</sup>. Septage was loaded at a HLR of 0.32 m/week, corresponding to a SLR of 210-260 kg TS/m<sup>2</sup>/y. Solids accumulated at 12 cm/y to a total of 80 cm after 7 years of continuous septage loading. No bed clogging was observed during this period. Nitrate concentrations in the filtrate increased from 8 to 180-320 mg/L due to nitrification in the filter bed.

A pilot system consisting of 6 x 1 m<sup>2</sup> filters were established in Cameroon to test the treatment of fecal sludge with dewatering beds planted with 2 native species: papyrus (*Cyperus papyrus*) and antelope grass (*Echinochloa pyramidalis*) (Kengne *et al.*, 2008). After 6 months acclimatization and plant growth, the filters were dosed at SLRs of 100, 200 and 300 kg TS/m<sup>2</sup>/y for a six month period. Above and below ground growth characteristics of the plants were studied (density, size, root length, and above-ground and below-ground biomass were measured over time and as a function of sludge loading. Both plants colonised the filters at all loading rates with 150 and 20 dry tons/ha/y potential harvest from *E. pyramidalis* and papyrus, respectively. In a similar study in Burkina Faso, pilot filters planted with *Oryza longistaminata* and *Sporobolus pyramidalis* were studied treating fecal sludge (Kouawa *et al.*, 2015). SLRs of 63-84 kg TS/m<sup>2</sup>/y were applied to the

filters during a 4 month study with the study concluding that both species were not appropriate for sludge dewatering applications as both plant species died.

Panuvatvanich *et al.* (2009) studied nitrogen transformations in a cattail bed filter treating fecal sludge. The pilot study was conducted at AIT, in Pathumtani, Thailand. Pilot filters (1 m<sup>2</sup>) planted in *Typha augustifolia* were loaded at 250 kg TS/m<sup>2</sup>/y. The filters consisted of a hollow block drainage system, 35 cm large gravel (25-50 mm), 15cm small gravel (10-25 mm) and 10-40 cm fine sand (0.3-0.75 mm). Sludge was applied weekly to the filters. The effects of sand depth (10-40 cm) and percolate impoundment (outlet valve was closed) were evaluated on nitrogen transformation within the filters. On average, 55% of TN was retained in the top sludge layer. N loss from plant uptake and volatilization was negligible. TN in the filtrate varied between 8-13%, while 5-13% can be attributed to denitrification. Less N was lost to denitrification with a 10 cm sand layer compared with a 20 or 40 cm layer. N loss through denitrification was lower when the filtrate drained freely (4.8%) versus when the filtrate was impounded (7.8-12.8%). All N which nitrified during the first day of dosing was lost to denitrification by the sixth day of impoundment. Nitrification-denitrification were the main factors controlling N transformations within the wetland system.

Pilot and field-scale systems were studied in Brazil treating fecal sludge (Calderon-Vallejo *et al.*, 2015). The system consisted of layers of coarse to fine gravel (bottom to top) with no sand layer and planted in *Cynodon* spp. Sludge was applied weekly; however, concentrations were weak compared with typical septage values resulting in a low SLR of 18 kg TS/m<sup>2</sup>/y. Removal efficiencies ranged from 67% for BOD<sub>5</sub> and 44% for TS. The plants propagated well under tropical conditions.

A pilot-scale experiment was conducted in Kathmandu, Nepal comparing fecal sludge dewatering with both planted and unplanted reed bed systems (Pandey and Jensen, 2015). The authors recommend initial loading rates of 100 kg TS/m<sup>2</sup>/y with a gradual increase to 250 kg TS/m<sup>2</sup>/y. The planted filters performed better than the unplanted filters in terms of both sludge dewatering and VS destruction.

### 2.3.7 French Studies - Septage

Two septage treatment reed bed systems were put into operation in France in 2001 and 2002 with solids loading rates of 46 and 109 kg TS/m<sup>2</sup>/y, respectively (Paing and Voisin, 2005). Septage strength is consistent with North American values (28,000 mg/L TS; 8,000 mg/L BOD<sub>5</sub>; 33,000 mg/L COD; 1,100 mg/L TKN; 360 mg/L TP). The high COD/BOD ratio of between 3-5 indicates that a large fraction of the readily degradable organics have already decomposed in the septic tanks. The first system consists of 6 beds fed for 7 days followed by a 35 day rest period, while the second system consists of 3 beds fed for 3 days followed by a 24 day rest period. The first system had a 90 cm media depth comprised of: 20 cm coarse gravel (20-40 mm), 20 cm medium gravel (10-20 mm), 50 cm fine gravel (2-8 mm), while the total depth of filter media in the second system was only 45 cm. In the first system, solids accumulated at 12 cm/y and dry solids reached 38% while in the second system, solids accumulated at 36 cm/y while only reaching 20% dry matter. The difference is due to the higher loading rate of the second system (109 vs 46 kg TS/m<sup>2</sup>/y) and insufficient number of basins (3 versus 6) to allow for sufficient dewatering. Filter percolate quality in the first system was very good with removal efficiencies of 99% for TSS and BOD<sub>5</sub> and 94% for TKN and TP (TSS=143 mg/L; BOD<sub>5</sub>=104 mg/L, COD=463 mg/L, TKN-N=70 mg/L, NO<sub>3</sub>-N=49 mg/L; TP=9 mg/L). *Phragmites Australis* were not sensitive to septage dosing with low redox potential and high ammonia content, likely due to the air vent stacks throughout the filter (Paing and Voisin, 2005).

A pilot experiment with reed bed treatment of septage was commenced in France in 2006 (Troesch *et al.*, 2009b). Each pilot filter (2m<sup>2</sup>) consists of: 15 cm coarse gravel (30-60mm), 10 cm medium gravel (15-25 mm), 20 cm fine gravel (2-6 mm) and 5 cm of sand. One year old *Phragmites australis* were planted in May 2006 at a density of 9 pots/m<sup>2</sup> and had increased to 250 stems/m<sup>2</sup> by May 2008. A two year commissioning period consisted of the filters being dosed with treated wastewater for the first 6 months to allow plant establishment, followed by a 1.5 year period where the filters were dosed at a solids loading rate of 30 kg TS/m<sup>2</sup>/y. A 3.5 day feeding period followed by a 17.5 d rest period was utilized. Sludge characteristics which are thought to influence dewaterability include: particle size distribution, organic matter, cationic salts and extra cellular substances.

Septage is composed of many non-flocculated particles with many dissolved salts and VFAs resulting in a high electrical conductivity and a decrease in dewaterability. CST of septage was shown to improve when mixed with WAS or WWTP effluent. SVI was optimum at 10-30% septage mixed with WAS. CST and bound water of septage were  $398\pm 153$  sec and  $31\pm 19\%$ , respectively. The CST was high compared with WAS (CST=7 sec). Particle size distribution was measured using a laser granulometer and shows a large percent volume of the septage is comprised of particles less than  $10\mu\text{m}$  while most of the volume is comprised of particles smaller than  $100\mu\text{m}$ . Sludge accumulated at  $11.5\text{ cm/y}$  at a loading rate of  $29\text{-}33\text{ kg TSS/m}^2\text{/y}$ . Desiccation rates were higher under summer conditions ( $17\pm 3^\circ\text{C}$ ;  $32\%$  DM) compared to winter conditions ( $7\pm 4^\circ\text{C}$ ;  $24\%$  DM). Plant ET was measured to be  $3.6\text{ mm/d}$  in summer and  $1.8\text{ mm/d}$  in winter. ORP was consistently below  $200\text{ mV}$  (anoxic conditions) during the winter season and rose above  $200\text{ mV}$  only 2-3 weeks after temperatures rose in the spring and the easily degraded carbon from winter storage had been oxidized. During summer, ORP remains high as the sludge layer cracks and the reeds add oxygen to the root zone. A strong correlation between reed density and ORP was exhibited during the 2<sup>nd</sup> growing season ( $R^2=0.815$ ). VS was reduced from  $68\%$  in the raw septage to  $57\%$  in the reed beds. High indices of sludge stability were measured after 1.5 years in the filters. Pathogen indicators were reduced by 2-3 log for *E. coli* and *Enterococci* in the filters after a 25 day rest period. Septage, with a high percent of colloidal particles, takes longer to drain than WAS ( $22\text{h}$  vs  $8\text{h}$ ) and results in a higher residual humidity level in the sludge. High CST was shown to negatively correlate with maximum drainage flow and final DM. A 3 day dosing period followed by a 20 day rest period with a 6 bed configuration is recommended to avoid plant stress in the summer time.

The leachate quality was highly variable and fluctuated with raw septage quality. It was hypothesized that colloidal particles pass through the filter, which constitute a large proportionate volume of the septage. The reed bed with a sand filtration layer produced better effluent quality than the reed bed with a compost filtration layer, with the sand filter providing a  $95\%$  reduction in TSS to  $952\pm 997\text{ mg/L}$ , a  $93\%$  reduction of COD to  $2212\pm 2439\text{ mg/L}$  and a  $90\%$  reduction of TKN to  $117\pm 117\text{ mg/L}$  (Troesch *et al.*, 2009b).

In a follow-up study using the same pilot beds, Vincent *et al.* (2011) compared WAS to septage treatment with a 29 day cycle consisting of 5 feeding days followed by 24 rest days. Septage was characterized as having high CST ( $360 \pm 142$  s), fine particles ( $D_{10} = 3.3 \pm 0.2$   $\mu\text{m}$ ) and high fat content ( $7638 \pm 2718$  mg/L), suggesting challenges to dewaterability. Septage was loaded in 3 cm doses, while WAS was loaded in 35 cm doses due to the varying solids content of the two sludge types. Septage removal rates varied from 90-93% for COD, TSS and  $\text{NH}_4^+\text{-N}$  at a loading rate of 30 kg/m<sup>2</sup>/y and 82-87% at a loading rate of 50 kg/m<sup>2</sup>/y. Dewatered septage DM was always greater than 23% in winter and reached as high as 70% in summer after 24 rest days, which was considerably higher than was observed with dewatered WAS. This study demonstrated the feasibility of septage dewatering in reed bed filters at loading rate of up to 50 kg TSS/m<sup>2</sup>/y. A further study by Vincent *et al.* (2012) studied the hydraulic properties of both septage and WAS at varying loading rates. The study concluded that air transfer to WAS reed beds started to be impeded at 50 kg TS/m<sup>2</sup>/y, while no impediment to septage loaded reed beds was observed at up to 70 kg TSS/m<sup>2</sup>/y at a loading cycle of 5 days dosing and 24 d rest. It was hypothesized that the high hydraulic loading (due to low sludge solids) to the WAS beds was the reason for the limitation of air transfer observed.

Table 2-2. Filter Configurations for Sand Drying Beds and Planted Filters

Type of Filter	Type of Biosolids Treated	Filter Configuration (top to bottom)	SLR (kg TS·m <sup>-2</sup> ·yr <sup>-1</sup> )	Dosing Period in days (feeding/rest)	Dry Matter (%)	Solids Accumulation (cm / year)	Reference
Sand Bed	Septage	15 cm fine sand 7.5 cm coarse sand 7.5 cm fine gravel 7.5cm medium gravel 7.5-15cm coarse gravel					USEPA, 1984.
Sand Bed	Municipal Biosolids	30 cm coarse sand 20-46 cm gravel layer	49-195 100-160				Crites and Tchobanoglous, 1998. USEPA, 1987
Reed Bed	Municipal Biosolids	10 cm filter sand 20 cm gravel	60-100	1 / 7-10			Crites and Tchobanoglous, 1998.
Reed Bed	Municipal Biosolids (300 sites)	growth layer (sandy-loam) 15 cm filter sand 30-45 cm gravel	50-60	4-10 / 50-60	40%	10	Nielson, 2003.
Reed Bed	Municipal Biosolids	5-10 cm coarse sand 20 cm fine gravel (2-6 mm) 10 cm medium gravel (15-25 mm) 15 cm pea stone (30-60 mm)	30-40	14 / 98	30% summer 17% winter	10	Troesch <i>et al.</i> , 2008a
Reed Bed	Municipal Biosolids (24 sites) Fort Campbell		13-106		15% - top 47% - bottom	24	Kim and Smith, 1997
Reed Bed	Extended aeration (18 sites)	converted sand drying beds	18-82* Avg. 41.6*	1 / 14	-	3-35 Avg. 13.1	Mellstrom and Jager, 1994
Reed Bed	Aerobic digested	30 cm coarse sand (0.5-1 mm) 20-30 cm pea stone (3-6 mm)			14%	12.5	Begg <i>et al.</i> , 2001
Reed Bed	Vegetable WAS	15 cm coarse sand 45 cm coarse gravel	65	1 / 7	10-12%	10.1	Burgoon <i>et al.</i> , 1997

\* based on TSS

Table 2-2 (Cont'). Filter Configurations for Sand Drying Beds and Planted Filters

Type of Filter	Type of Biosolids Treated	Filter Configuration (top to bottom)	SLR (kg TS·m <sup>-2</sup> ·yr <sup>-1</sup> )	Dosing Period in days (feeding/rest)	Dry Matter (%)	Solids Accumulation (cm / year)	Reference
Reed Bed	WAS TWAS	-	11 38	1 / 13 1 / 25	30% summer 20% winter	-	Barbieri <i>et al.</i> , 2003
Reed Bed	Extended aeration	25 cm coarse sand (0.3-1 mm) 30 cm coarse gravel (1-3 cm)	51 55 125	1/2 1/10 -	20-30 20-30 11-13	7 - 33	Uggetti <i>et al.</i> , 2009
Cattail Filter	Septage	10 cm fine sand 15 cm small gravel 40 cm large gravel	250	1 / 7		12	Koottatep <i>et al.</i> , 2005
Papyrus & Antelope grass	Septage/ pit latrines		100-300	1/7			Kengne <i>et al.</i> , 2008
Reed Bed	Septage	50 cm fine gravel (2-8 mm) 20 cm medium gravel (10-20 mm) 20 cm coarse gravel (20-40 mm)	46	7 / 35	38%	12	Paing and Voisin, 2005
Reed Bed	Septage	45 cm	109	3 / 24	20%	36	
Reed Bed	Septage	5 cm sand 20 cm fine gravel (2-6 mm) 10 cm medium gravel (15-25 mm) 15 cm coarse gravel (30-60 mm)	30 (startup) will be 50	3.5 / 17.5	24-32%	11.5	Troesch, S. <i>et al.</i> , 2009b

### 2.3.8 Discussion

Many studies have evaluated both pilot and full scale reed bed systems to treat both aerobic and anaerobic biosolids (Nielson, 2003; Troesch *et al.*, 2009a; Kim and Smith, 1997; Mellstrom and Jager, 1994; Begg *et al.*, 2001; Burgoon *et al.*, 1997; Barbieri *et al.*, 2003) , while several studies have evaluated planted filters for septage dewatering in tropical climates (Koottatep *et al.*, 2005; Kengne *et al.*, 2008) and reed bed filters in France (Paing and Voisin, 2005; Troesch *et al.*, 2009b).

All studies in temperate climates report using *phragmites*, while several studies in tropical climates use typha and other plant species (Koottatep *et al.*, 2005; Kengne *et al.*, 2008). Several reed bed systems have been abandoned in the US due to problems encountered with plant establishment (Mellstrom and Jager, 1994), while various authors state that it takes 1-2 seasons for plant establishment, during which time no loading to reduced loading of sludge is recommended (Nielson, 2003; Troesch *et al.*, 2009a). As well, *phragmites* is an invasive species, which pose problems for certain applications such as the establishment in a park. The use of an alternative plant species such as bush willow should be investigated.

The solids loading rate (SLR) is the critical design parameter in filter sizing and will greatly impact both the area requirement and construction cost. Most systems in Europe are designed for 50 kg TS/m<sup>2</sup>/y for WAS reed beds and 60 kg TS/m<sup>2</sup>/y for WAS mixed with AD sludge (Nielson, 2003) while lower rates of 30-40 kg TS/m<sup>2</sup>/y are reported in France (Troesch *et al.*, 2009a). Reed bed systems in the US report SLRs for both aerobic and anaerobically digested sludges varying widely from 13-106 kg TS/m<sup>2</sup>/y from a survey of 24 reed bed filters (Kim and Smith, 1997), or from 18-82 kg TS/m<sup>2</sup>/y, with an average of 41.6 kg TS/m<sup>2</sup>/y, in another survey of 18 systems (Mellstrom and Jager, 1994). Reed bed filters have been applied to septage treatment at the pilot scale in several tropical countries at much higher loading rates of 250 kg TS/m<sup>2</sup>/y (Koottatip *et al.*, 2005) and 100-300 kg TS/m<sup>2</sup>/y in an ongoing pilot study (Kengne *et al.*, 2008). Two full scale systems in France report loading rates of 46 and 109 kg TS/m<sup>2</sup>/y (Paing and Voisin, 2005), while a pilot system was loaded at 30 kg TS/m<sup>2</sup>/y during the commissioning period and will be operated at a design loading rate of 50 kg TS/m<sup>2</sup>/y (Troesch *et al.*, 2009b).

Reed bed systems have not been designed to explicitly take advantage of freeze-thaw conditioning during the winter, although several US reed bed systems have reported operating during the winter (Messtrom and Jager, 1994, Kim and Smith, 1997). Reed *et al.* (1986) suggests a maximum of 1.8m of sludge can be frozen in the US north east in a winter season, which would be similar to conditions experienced in Eastern Ontario. This equates to 61.2 kg TS/m<sup>2</sup>/y at 3.4% solids (average TS in septage) over a 3 month freezing period (mid Dec-mid March). If we assume a conservative loading rate of 60 kg TS/m<sup>2</sup>/y for the remainder of the year, then an annual loading rate of 106 kg TS/m<sup>2</sup>/y should be achievable as a minimum loading rate for biosolids treatment. If we assume a maximum loading rate of 160 kg TS/m<sup>2</sup>/y for the remaining 9 months of the year based on the maximum loading to a sand dewatering filters (USEPA, 1987), a maximum annual loading rate would be 181 kg TS/m<sup>2</sup>/y. These rates are 2-3 times the recommended dosing rate for reed bed filters dewatering WAS or AD sludges in a European temperate climate (Nielson, 2003). Solid loading rates of between 60-180 kg TS/m<sup>2</sup>/y should be investigated.

In addition to SLR, the rest period is important to allow for sludge dewatering and stabilization between doses. Rest periods vary widely, from a minimum of 7 days (Crites and Tchobanoglous, 1998; Burgoon *et al.*, 1997; Koottatep *et al.*, 2005; Kengne *et al.*, 2008), with several authors suggesting 14-21 days to allow for sludge mineralisation (Mellstrom and Jager, 1994; Barbieri *et al.*, 2003; Troesch *et al.*, 2009b) or even longer at 35 days (Paing and Voisin, 2005), 50-60 days (Nielson, 2003), or 98 days (Troesch *et al.*, 2009a). There is clearly a relationship between DM content of the sludge and rest period, as demonstrated in a systems with a 7 day rest period reporting DM content of only 10-12% (Burgoon *et al.*, 1997), while Nielson reports a DM content of 40% at rest periods of 50-60 days and Barbieri *et al.* (2003) reports a 30% DM in summer with a rest period of 13 days compared with a DM of 20% in winter with a rest period of 25 days. Paing and Voisin (2005) reported a 20% DM with a 24 day rest period and a 38% DM at a 35 day rest period, while Troesch *et al.* (2009b) reported 24-32% DM at 17.5 day rest period. Maximum gravity dewatering is well explained by Kopp and Dichtl (2001), who conducted a study on the free water content and dewaterability of sewage sludges. The authors found that free water in sludges dries linearly with time while interstitial water dries exponentially. A

comparison of 58 anaerobically digested sludges and 15 aerobically digested sludges found the maximum theoretical DM for anaerobically sludges to be  $28\pm 4\%$  and the maximum theoretical DM for aerobically digested sludges to be  $23\pm 3\%$ . The limit to dewaterability is defined by the percentage of free water in the sludge ( $92\pm 2\%$  for AD sludges versus  $89\pm 2\%$  for aerobically digested sludges). Therefore, rest periods which achieve DM content of between 20-30% suggest that all free water has drained from the sludge, while achieving DM of 40% or higher suggests that plant ET is removing capillary water from the sludge. The literature suggests rest periods of between 13-25 days produce sludge DM of between 20-30%, which is consistent with what common dewatering technologies such as filter presses, centrifuges will achieve. A final curing period prior to desludging could be used to increase the final DM to 40% or higher, which would reduce hauling and spreading costs. The impact on sludge DM of loading rate and rest period (7-50 days) should be investigated.

Solids accumulation rate is an important design parameter as it determines how frequently a bed must be desludged. The sludge accumulation rate (cm/y) was plotted against SLR (kg TS/m<sup>2</sup>/y) for 12 reed beds described in the literature (Mellstrom and Jager, 1994; Paing and Voisin, 2005; Troesch *et al.*, 2009b). A very good correlation ( $R^2=0.80$ ) was found when the systems with rest periods of greater than 30 days were excluded (Figure 2-1). Data from 12 reed bed systems suggests a sludge accumulation rate 10 cm for every 29 kg TS/m<sup>2</sup> applied to the reed bed, compared with a 10 cm increase for every 50-60 kg TS/m<sup>2</sup> applied with a 50-60 d rest period. This suggests that beyond the required time to dewater the sludge, the process of sludge stability will also lead to a significant reduction in solids through destruction of VS (conversion to CO<sub>2</sub>, NH<sub>3</sub> and H<sub>2</sub>O). However, as the systems studied in Europe are mostly WAS or WAS mixed with AD sludges, while the systems reported in the US are mostly extended aeration plants with either aerobic or anaerobic stabilization, there is less soluble VS to degrade and thus a higher sludge accumulation rate per kg TS applied to the filter. Sludge accumulation should be correlated to VS destruction and other indices of sludge stability as well as solid loading rate and rest period and by season.

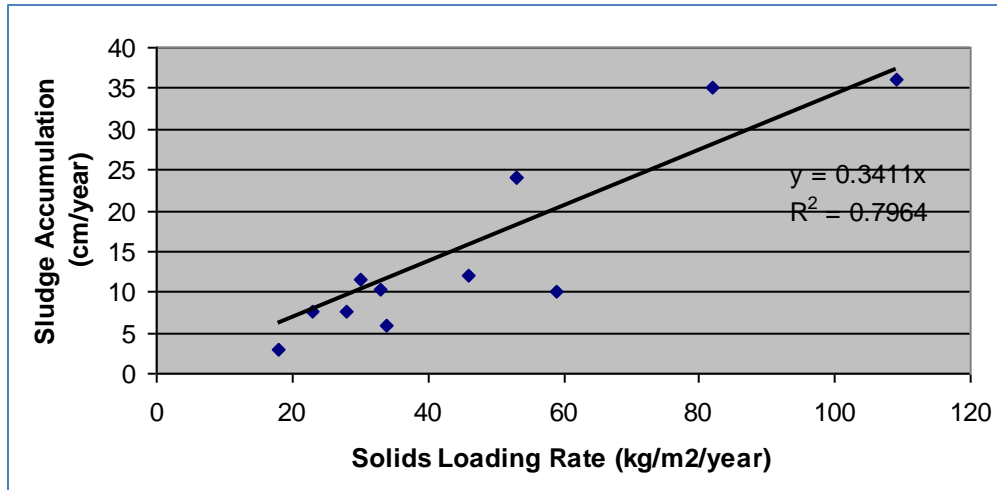


Figure 2-1. Relationship between Sludge Accumulation and Solids Loading Rate

The filter media plays a role in drainage and aeration of the sludge layer. Good capillary connection between the media layers (*i.e.* many layers of gradually smaller media) should increase dewatering. On the other hand, the larger the void spaces, the better oxygen transfer from the passive aeration array up through the filter media layers. The depth and granulometry of the filter firstly protects against short circuiting and secondly acts as a trickling filter to treat the filtrate through processes of filtration and biological degradation. Filter media granulometry vary considerably in the literature, ranging from 45-90 cm in depth. Most filters consist of several layers of coarse, medium and fine gravel and a layer of coarse sand (Table 2-2). Only one system reports using only layers of gravel with no sand layer (Paing and Voisin, 2005). There is no obvious correlation to filter media depth or granulometry on sludge dewatering efficiency; however, there may be an impact on filtrate quality although most authors do not report on filtrate quality. Begg *et al.* (2001) reported exceptionally good filtrate quality (BOD<sub>5</sub>=6 mg/L; TSS=14 mg/L) with a 50-60 cm media layer. Panuvatvanich *et al.* (2009) found that more nitrogen was lost to denitrification when the sand layer was increased from 10 to 20-40 cm. A reed bed system in France treating septage with a 90 cm media depth with the smallest media being fine gravel (2-8 mm) had good filtrate quality (BOD<sub>5</sub>=104 mg/L; TSS=143 mg/L), while another French system treating septage with a 50 cm media layer performed poorly (TSS=952±997 mg/L; COD=2212±2439 mg/L) (Troesch *et al.*, 2009b). It is not clear from the literature what is an optimum filter media design for sludge dewatering. In addition to filter media,

hydraulic loading rate will likely play an important role on liquid residence time in the filter. As rest periods and loading rates increase, hydraulic loading rates will increase and could lead to a significant head of ponding sludge on the filter surface, resulting in effluent short circuiting and poor filtrate water quality. This could be the case with the experiment of Troesch *et al.* (2009b). Reducing the depth of media will reduce construction costs. It seems reasonable that 45cm is the minimum possible media depth as the drainage line is 10cm in diameter and there has to be sufficient depth for root growth (typically 50cm for *phragmites*) (De Maeseneer, 1997). The relationship between bed depth, loading rate and rest period and filtrate quality should be investigated.

#### 2.4 Sludge Stabilization and Quality

Sludge will undergo a non-optimized composting process (aerobic stabilization) in the reed beds. The reed beds should remain aerobic and the dewatered sludge should compost slowly over time. The composting process within the reed bed is an integral component to the treatment system. The advantage of composting the sludge *in-situ* in the filters is that it reduces sludge volume, alleviates downstream processing costs and minimizes labour. The primary end use of the stabilized sludge is the application to agricultural fields as a source of nutrients and organic matter for crop production. Another potential end use is as a feedstock for compost production. The contingency is disposal at a landfill or use as landfill cover.

Composting is described as the mineralization and partial humification of organic matter to a final product, free of phytotoxicity and pathogens or the spontaneous biological decomposition of organic matter in a predominantly aerobic environment (Bernal *et al.*, 2009). The main factors to control the composting process are: bulk density, porosity, particle size, nutrient content, C/N ratio, temperature, pH, moisture and oxygen supply. Optimum C/N ratios for compost are 25-35, based on microbial needs. High C/N ratios slow down the process, while low C/N ratios lead to high N loss. N loss in the reed bed system will reduce the value of the stabilized sludge to farmers; however, it will also decrease the land area required for spreading based on N crop uptake.

Optimum moisture content for the composting process is 50-60%, while optimum temperature is between 40-65°C, although temperatures above 55°C are required to kill pathogenic microorganisms. The composting process generally follows two phases, the biooxidative phase and the maturing phase. Initially mesophilic bacteria degrade simple compounds such as sugars, amino acids, proteins, etc. Temperature increases rapidly and thermophilic microorganisms take over and degrade fats, cellulose, hemicellulose and some lignin. It is during the thermophilic phase that the most OM is destroyed along with the destruction of pathogens. As microbial activity declines, so does the temperature and mesophilic microorganisms once more predominate. During the biooxidative phase, OM is degraded to CO<sub>2</sub>, NH<sub>3</sub> and H<sub>2</sub>O. During the curing or maturation phase, stabilization and humification of the OM occurs and most of the nitrification occurs, resulting in a low NH<sub>4</sub><sup>+</sup>-N/NO<sub>3</sub>-N ratio in mature compost. Quality criteria for compost are defined in terms of: nutrient content, C/N ratio, humified and stabilized OM, pathogen content, heavy metal content and salt content (Bernal *et al.*, 2009).

The Interim Guidelines for the Production and Use of Aerobic Compost in Ontario provides a typical reference for compost quality (MOE, 2004). Temperatures must reach at least 55°C for a minimum of 15 days for pathogen destruction, and oxygen levels within the pile should be maintained between 12-15%. Compost standards are described in Table 2-3.

**Table 2-3. Compost Quality Standard**

<u>Interim Guidelines for the Production and Use of Aerobic Compost in Ontario (MOE, 2004)</u>	
<b>Parameter</b>	<b>Value</b>
Total Nitrogen	0.6 (% DM)
Total Phosphorus	0.25 (% DM)
Total Potassium	0.2 (% DM)
Calcium	3.0 (% DM)
Magnesium	0.3 (% DM)
OM	>30% DM
C/N	22 (typical)
EC	<3.5 mS/cm

SAR	<5
pH	5.5-8.5
Moisture Content	30-55%

Compost and septage metal limits are described in Table 2-4.

**Table 2-4. Metal Limits in Compost and Septage**

<b>Metal</b>	<b>Interim Guidelines for the Production and Use of Aerobic Compost in Ontario (MOE, 2004) (mg/kg DM)</b>	<b>Interim Guideline for the Land Application of Septage (MOE, 2008a) (mg/kg DM)</b>
Arsenic	13	170
Cadmium	3	34
Chromium	210	2800
Cobalt	34	340
Copper	100	1700
Lead	150	1100
Mercury	0.8	11
Molybdenum	5	94
Nickel	62	420
Selenium	2	34
Zinc	500	4200

Note: Identical to the Guidelines for the Utilization of Biosolids and other Wastes on Agricultural Land (MOE/OMAFRA, 1996)

The indicators of compost stability listed in the Interim Guidelines for the Production and Use of Aerobic Compost in Ontario (MOE, 2004) are: volatile solids destruction, spontaneous heating, oxygen uptake rates, toxin production, C/N ratio, seed germination and growth test and redox potential. These indicators are not absolute measures but should be used to compare with the raw feed.

In September of 2008, the MOE posted Draft Guidelines on the Ontario Environmental Registry defining standards for the land application of treated septage for metals and

pathogen as well as application rates based on nitrogen crop needs (MOE, 2008a). The draft guideline stipulates *E. coli* numbers of less than  $2 \times 10^6$  CFU/g DM, nitrogen loading based on crop needs and maximum metals concentration and loading to soil as described in Table 2-4. The metal values are the same as those found in the Guidelines for the Utilization of Biosolids and Other Wastes on Agricultural Land (MOE/ OMAFRA, 1996).

The processes that occur in a reed bed filter are similar to classic compost production in that there is a weak biooxidative phase (where readily degradable organic matter is mineralized) followed by a lengthy maturation phase. Septage is already partially stabilised as a significant proportion of VS has been anaerobically digested in the septic tank. Due to the substrate, bed configuration and loading cycle, it is unlikely that temperatures in a reed bed filter will reach the levels required to produce compost (55°C) so stabilized sludge cannot be classified as a compost; however, measures of compost quality and maturity can be used to characterize the stabilized sludge. Additionally, metal and pathogen limits described in the MOE guideline for the land application of treated septage (MOE, 2008a) will have to be met.

Senesi (1989) provides a good review of various measures of compost maturity. A decline in C/N ratio from >30 in the feedstock to less than 20 in the final product is usually a good indicator of a mature compost; however, substrates rich in N such as sludges may have initial C/N ratios of <10 which actually rise during the compost process. The evaluation of water soluble organic C/N has been found to be a much more reliable index, with stable values of 5-6 found for a variety of well-matured composts. Other common measures of compost stability include: C/N ratio in the water extract, water soluble organic C,  $\text{NH}_4^+\text{-N}/\text{NO}_3^-\text{-N}$ . The presence of VFAs, phenolic acids and  $\text{NH}_4^+\text{-N}$  may indicate immature composts.

As compost matures, OM reaches a high level of humification as humic acids are produced with increasing molecular weight and complexity. Common indices of compost maturity measure the ratio between different forms of organic matter: TOC, humin (very stable), humic acid (stable, complex) and fulvic acid (unstable, simple). Common indices to measure humification level in compost include (Senesi, 1989):

Humification ratio (HR):  $C_{\text{hum}}/C_{\text{org}} \times 100$

Humification index (HI):  $C_{\text{HA}}/C_{\text{org}} \times 100$

Humic acids (HA):  $C_{\text{HA}}/C_{\text{hum}} \times 100$ ;

Polymerization index (PI):  $C_{\text{HA}}/C_{\text{FA}}$

Where  $C_{\text{hum}}$  is alkali-extractable organic-C;  $C_{\text{HA}}$  is humic acid-like organic-C;  $C_{\text{FA}}$  is fulvic acid-like organic-C.  $C_{\text{org}}$  is measured as total organic carbon, while organic matter is fractioned into  $C_{\text{hum}}$ ,  $C_{\text{HA}}$  and  $C_{\text{FA}}$  following a standard method for soil organic matter extraction (Extraction of Soil Organic Matter 30-2 *In Methods of Soil Analysis Part 2 – Chemical and Microbiological Properties*, Page, *et al.*, 1982)

A variety of complex methods based on the identification of the humic like substances in compost and soil have also been applied to attempt to better characterize organic matter maturity (see Peruzzi *et al.*, 2008, below); however, no ideal method has been identified. (Senesi, 1989).

Roletto *et al.* (1985) defined stable compost from a variety of feedstocks as being:  $\text{HR} \geq 7.0$ ;  $\text{HI} \geq 3.5$ ;  $\text{HA} \geq 50$ ; and  $\text{PI} \geq 1.0$ .

Li *et al.* (2000) evaluated the maturity of sewage sludge composting from two compost mixtures: sewage sludge and compost and sewage sludge, swine manure and compost. The most common indices used are: C/N ratio, proportions of humic and fulvic acids, HR and HI. As well, the E4/E6 ratio (absorbance at 465 and 665 nm) of humic fractions were used, where lower E4/E6 ratios are associated with more mature humic and fulvic acid fractions. The C/N ratio reduced from 31.6 and 29.4 to 18.7 and 14.4 after 100 days of composting for the two types of compost. The C/N ratio stabilized after 63 and 35 days for the composts with and without pig manure, respectively. A C/N ratio below 20 is typically an indication of compost maturity; however, the ratio will vary depending upon the compost feedstock. A stable C/N ratio over time is also a good indicator of compost maturity. HR increased from 8.0 and 7.7 to 13.4 and 11.1 for the composts with and without pig manure, respectively. HI increased from 3.5 to 11.1 and 7.0 for the composts with and without pig manure, respectively. Similarly, HA/FA increased from 0.78 and 0.85 to 4.7 and 4.6 for the composts with and without pig manure, respectively. E4/E6 of HA

decreased from 6.0 and 8.2 to 3.7 and 7.1 for the composts with and without pig manure, respectively, while increasing for FA from 8.71 and 7.5 to 26.6 and 17. Li *et al.* (2000) concluded the C/N, HR, HI and E4/E6 ratios all provided a good indication of compost maturity.

Hernandez *et al.* (2005) followed the changes in organic matter during composting of an aerobic and anaerobic sewage sludge mixed with sawdust. Chemical and microbiological parameters as well as pyrolysis-gas chromatography were employed to monitor organic matter humification with time. VS decreased by 8-16% while water soluble organic carbon decreased by 32-49% over the 90 day composting period. Microbial respiration showed a decreasing trend from 1-90 days. The benzene/toluene ratio (B/E<sub>3</sub>) should increase with maturity of the organic matter; however, this was not observed in the study suggesting that this ratio is not useful for determining the degree of humification in poorly evolved materials as sewage sludge/sawdust composts. The acetic acid/toluene (K/E<sub>3</sub>) ratio provided a good indication of mineralization and increased from 18 to 88 percent over the 90 days composting period for the four compost mixtures studied.

Polak *et al.* (2005) conducted a study on the humification of AD sludge applied to a drying bed. Humification processes were studied with EPR, IR and NMR spectroscopic methods. It was found that during the first two weeks after sludge application to the drying beds an intense enrichment of humic acids takes place and that sludge humification occurs in the drying beds.

Peruzzi *et al.* (2008) studied the sludge stabilization in reed beds after one year of operation in Italy. Sewage sludge was applied at a rate of 5-10 cm/week during the summer and 5-10 cm every 2 weeks during the fall/winter period, corresponding to 3.16 and 2.32 m<sup>3</sup>/m<sup>2</sup>/y at the two sites. Sludge sampling was carried out every 3 months and analysed for sludge mineralization: TOC, water-soluble carbon, TN, NH<sub>3</sub>, and dehydrogenase activity (Dhase) and sludge humification: humic carbon, fulvic acids, humic acids, pyrrole/furfural (O/N) and benzene/toluene (B/E<sub>3</sub>). The C/N ratio (TOC/TN) increased from 6.2 and 6.8 to 7.5 in both systems over 9 months. Ammonia decreased substantially, from 929 and 1256 to 202 and 289 mg NH<sub>3</sub>/kg, respectively over 9 months.

Similarly, water soluble carbon decreased from 53,300 and 54,900 to 11,600 and 13,300 mg C/kg, respectively over a nine month period. Humic substances are composed of fulvic acids, which are less stable, and humic acids, which are more stable; however lipids are extracted along with humic acids and may lead to an overestimation of the humic acid fraction of organic matter. More exact methods compare pyrrole/furfural (O/N), with pyrrole derived from nitrogenated and condensed humic structures while furfural represents unstable organic matter. Another index is the ratio of benzene to toluene (B/E3), where benzene is derived from condensed aromatic structures while toluene is derived from aromatic structures with short aliphatic chains. The ratio increases as organic matter matures. The polymerization index (PI) increased from 0.22 and 0.16 to 0.33 and 0.30 for the two systems over a six month period. The O/N ratio decreased by 25 percent, while the B/E3 ratio remained more or less constant. These results suggest that reductions in water soluble carbon and  $\text{NH}_4^+$  are informative, while the C/N ratio increased to 7.5, while a stable compost should be less than 20 and the various indices of humification (O/N) were either contradictory or did not change (B/E3).

Kengne *et al.* (2009) studied the quality of dewatered fecal sludges from pilot reed bed systems in Cameroon. After 6 months acclimatization, 6 months at annual loading rates of 100-300 kgTS/m<sup>2</sup> and 1 month maturation, the biosolids were found to have a C/N ratio of 11.3, a HI of 13.6, HR of 17.8 and PI of 3.7, which is comparable to a mature compost and meets the criteria for a stabilized compost described by Roletto *et al.* (1985).

Ugetti *et al.* (2009) studied three full scale reed beds in Spain. Sludge mineralization was measured through changes in VS, COD and BOD. VS concentrations declined from 52-67% in the influent to 36-45% and 32-39% in the top and bottom layers, respectively of two systems receiving  $\sim 50 \text{ kg TS m}^{-2} \text{ yr}^{-1}$ . However, in a system receiving 125 kg TS/m<sup>2</sup>/y, VS only declined from 58-59% to 46-51% in both layers of sludge. The lower reduction in VS indicates a lower level of mineralization. VS removal of 30% is less than reported for the stabilization of aerobic or anaerobic digestion (50-65%); however, extended aeration sludge is already partly stabilized. BOD<sub>5</sub> values were 100 times lower than COD and BOD<sub>21</sub> values were 50-100 times lower than COD values, which suggest that the organic matter is quite stable.

In a similar study, Nielsen *et al.* (2014), studied cores from 3 systems in Denmark prior to land application after 10-20 years of operation. Levels of total organic carbon and total nitrogen decreased with depth across all systems as did water-soluble carbon, ammonia,  $\beta$ -glucosidase and urease activities and hydrolytic enzymes indicating an increase in mineralisation with depth. Masciandaro *et al.* (2015) studied the sludge in four reed bed systems in Italy over time and concluded that TOC and TN decreased over time due to sludge mineralization with substantial reductions in water-soluble carbon and dehydrogenase activity. The humification index (B/E3) increased over time indicating a maturation of organic matter in all four systems. Heavy metals content remained below Italian reuse standards and was found to be mostly in the fraction bound to organic matter.

Nielsen and Bruun (2015) studied the dewatered sludge residues in reed bed systems in Denmark after 10-20 years of operation. The sludge cake achieved up to 26% DM with heavy metal concentration and hazardous organic compounds below the EU standards for land application. Nitrogen and phosphorus concentrations were 28 and 36 g/kg DM, respectively. Very similar results were reported by Caicedo *et al.* (2015) at two reed bed systems in Germany operating for 6 and 12 years.

#### 2.4.1 Discussion

It is important to measure the degree of stability or maturity of the sludge to determine if the compost processes are progressing within the sludge layer and to determine if the sludge has mineralized and humified into a stable product which can be land applied without further significant degradation, which could create odour issues or negatively impact plant growth. Sludge quality is defined by the MOE Draft Guideline for the Land Application of Treated Septage (MOE 2008a) in terms of metals and pathogen limits, while compost quality is defined by the Interim Guidelines for the Production and Use of Aerobic Compost in Ontario (MOE 2004).

The common measures of compost stability include: volatile solids destruction, spontaneous heating, oxygen uptake rates, toxin production, C/N ratio, seed germination and growth tests and redox potential (MOE, 2004), while most research papers consider reduction in organic matter (VS, TOC, BOD) and a reduction or stabilization in the C/N ratio

as indicators of sludge stability (Li *et al.*, 2000; Roletto, 1985; Hernandez, 2005; Peruzzi *et al.*, 2008), although Sensi (1989) suggests that C/N ratio is not relevant for sludges as the ratio starts at less than 10 and may rise during composting, while compost standards state that C/N should be less than 20. Sensi (1989) suggests that the water soluble C/N ratio of 5-6 is a better indicator of compost stability and has been shown to be constant with a variety of feedstocks. Peruzzi *et al.* (2008) suggests the reduction in NH<sub>3</sub> can also be used as an indicator of sludge stability as NH<sub>3</sub> will either volatilize or nitrify during the composting process. Ugetti *et al.* (2009) used BOD/COD ratio in addition to reduction in VS to characterize sludge stability.

Sludge maturity is a measure of the humification of organic matter within the sludge, or the complexation of organic carbon. The most common indices of sludge maturity are: Humification Ratio (HR), Humification Index (HI), Humic Acids (HA) and Polymersiation Index (PI), which are various ratios between humin (very stable), humic acid (stable), fulvic acid (unstable) and TOC (Sensi, 1989; Peruzzi *et al.*, 2008; Li *et al.*, 2000; Roletto *et al.*, 1985; Kengne *et al.*, 2009). Additionally, there are many complex methods that attempt to characterize the humic substances in organic matter (Peruzzi *et al.*, 2008; Polak *et al.*, 2005; Hernandez *et al.*, 2005); however, the analytical methods are complex and no ideal method has been identified (Senesi, 1989).

The factors which will be used to characterize the maturity and stability of the stabilized sludge are: decrease in VS and water soluble TOC, BOD, COD, BOD/COD, water soluble C/N, decrease in NH<sub>4</sub>, redox, HR, HI, HA and PI. As well, trends in sludge pH, redox, EC, T, bulk density and DM will be monitored and evaluated.

## 2.5 Waste Characterisation

Sludge management is one of the largest challenges in wastewater treatment and represents 30-60% of operating costs of wastewater treatment plants (Bertanza *et al.*, 2016). Sludge is a challenge to treat due to its high solids content (0.25-12% solids) and high levels of organic matter. The primary aims in sludge treatment are to reduce water content (sludge dewatering) and organic matter (sludge stabilization). The processes utilized to reduce moisture content include: thickening (gravity, flotation, centrifugal,

gravity belt, rotary drum), conditioning (ferric chloride, lime, alum, polymers, heat treatment), dewatering (vacuum filtration, centrifugation, belt filter, filter presses, drying beds, vacuum assisted drying beds, lagoons) and heat drying. The most common processes to reduce and stabilize organic matter are digestion (aerobic or anaerobic) and composting. A secondary aim of sludge treatment is often to reduce pathogen counts to a level acceptable by regulation for land application.

The common types of sludge are:

- **Primary Sludge:** Sludge from primary settling tanks. Sludge is young, creates offensive odours and easily degradable.
- **Chemical Sludge:** Sludge from chemical precipitation of phosphorus, usually alum, ferric chloride or lime. Usually primary sludge settled with a coagulant.
- **Activated Sludge:** Activated sludge will digest easily and is often mixed with primary sludge.
- **Digested Sludge (aerobic or anaerobic):** Well digested sludge dewateres easily.
- **Septage:** Accumulated solids from septic tanks. Can be offensive unless well decomposed.

Factors that affect sludge dewatering include surface charge, particle size and percent fixed solids (Shammas and Wang, 2007). Particle size is considered to be one of the most important factors influencing sludge dewaterability. As particle size decreases, surface area and surface to volume ratios increase resulting in more water loosely bound to the sludge particle and higher repulsion between larger areas of negative charge. Biosolids have a negative surface charge and repel each other and attract water molecules either by weak chemical bonding or by capillary action. Chemical conditioning is commonly used to overcome surface charge. Biosolids tend to dewater better as the percentage fixed solids increases. Generally primary sludges are the easiest to thicken followed by fixed film biosolids, with WAS being the most difficult to thicken. Percent solids of thicken sludges range from 5-7% for primary sludge and 2-6% for WAS. Septage is normally characterized by large quantities of grit and grease, a highly offensive odour, poor settling and dewatering characteristics, and high solids and organic content (USEPA, 1984).

Characteristics of different types of biosolids and septage are presented below in Tables 6-8 along with the characterization of raw septage samples collected from the truck of René Goulet Septic Tank Pumping, the full scale experimental site for this research project. The characteristics of the Goulet septage are consistent with values reported by USEPA (1984) in terms of organic matter, solids and nutrients. The Goulet values are higher than USEPA values for a number of metal species including: aluminum, copper, iron, mercury and zinc. *Ecoli* numbers are similar to fecal coliform numbers reported by USEPA (1984).

Table 2-5. Physical and Chemical Characteristics of Septage and Various Sludges

Parameter	Primary Sludge (Metcalf and Eddy, 1991)	WAS (Metcalf and Eddy, 1991)	Digested Primary Sludge (Metcalf and Eddy, 1991)	Septage (USEPA, 1984)	Septage (Goulet Experimental Data) Avg.± SD
	Range (Typical or Average Value)				
TS (mg/L)	20,000-80,000 (50,000)	8,000-12,000	25,000-70,000 (35,000)	1,132-130,475 (34,106)	25,996 ± 22,282
VS (mg/L)	32,500	4,700-10,600	14,000	353-71,402 (23,100)	17,510 ± 15,268
TSS (mg/L)	-	-	-	310-93,378 (12,862)	19,994 ± 19,913
TKN (mg/L)	-	-	-	66-1,060 (588)	791 ± 659
NH <sub>3</sub> -N (mg/L)	-	-	-	3-116 (97)	134 ± 80
TP (mg/L)	-	-	-	20-760 (210)	275 ± 310
BOD <sub>5</sub> (mg/L)	-	-	-	440-78,600 (6,480)	6,235 ± 6,748
COD (mg/L)	-	-	-	1,500-703,000 (31,900)	28,313 ± 30,877
N (N, %of TS)	1.5-4.0 (2.5)	2.4-5.0	1.6-6.0 (3.0)	1.7	3.0
P (P <sub>2</sub> O <sub>5</sub> , % of TS)	0.8-2.8 (1.6)	2.8-11.0	1.5-4.0 (2.5)	1.5	2.5
pH	5.0-8.0 (6.0)	6.5-8.0	6.5-7.5 (7.0)	6.9	7.4 ± 0.4
Alkalinity	500-1,500 (600)	580-1,100	2,500-3,50 (3,000)	522-4,190 (970)	870 ± 395
Sludge Age (d)	-	5-10 days <sup>a</sup>	20-60 days <sup>a</sup>	Years	Years
Grease (FOG) (%TS)	7-35	5-12	5-20 (18)	16	-
Grease (FOG) (mg/L)	-	-	-	208-23,368 (5,600)	-
CST (s)	2550 <sup>b</sup>	100-200 <sup>c</sup>	-	-	-

a Standard Handbook of Environmental Engineering (Corbitt, 1989)

b Baskerville (1971)

c Biosolids Treatment Processes (Shamas and Wang, 2007)

Table 2-6. Metals in Septage and Sludge

Parameter	Typical U.S. Domestic Sludge Mean (Metcalf and Eddy, 1991)	Septage (USEPA, 1984)	Septage (Goulet Experimental Data) Avg.± SD
	(mg/kg DM)		
Ag	-	2.9	4.0 ± 3.5
Al	-	1237	7,597 ± 9,737
As	10	4.1	2.5 ± 5.1
Ba	-	169	280 ± 420
Cd	10	2.8	4.0 ± 2.3
Cr	500	14.4	23.7 ± 44.1
Co	30	11.9	3.6 ± 1.1
Cu	800	142	488 ± 930
Fe	17,000	1,152	5,675 ± 6962
Hg	6	0.15	1.3 ± 1.2
Mn	260	178	110 ± 120
Ni	80	15.4	16.2 ± 18.1
Pb	500	35.5	53.5 ± 121.0
Se	5	2.6	2.8 ± 5.0
Sb	14	2.2	1.5 ± 1.8
Zn	1700	292	817 ± 1053
Cyanide	-	13.8	-

Table 2-7. Pathogen and Pathogen Indicator Organisms in Septage and Sludge

Parameter	Primary Sludge (USEPA, 1984)	Septage (USEPA, 1984)	Septage (Goulet Experimental Data) Geometric mean
	(CFU/100mL)		
Total Coliform	5.6 x 10 <sup>7</sup>	10 <sup>7</sup> -10 <sup>9</sup>	
Fecal Coliform	2.0 x 10 <sup>7</sup>	10 <sup>6</sup> -10 <sup>8</sup>	3.73 x 10 <sup>7</sup> ( <i>E. coli</i> )
Fecal Streptococci	1.1 x 10 <sup>6</sup>	(4.7 x 10 <sup>3</sup> ) 10 <sup>6</sup> -10 <sup>7</sup>	
<i>Pseudomonas aeruginosa</i>		10 <sup>1</sup> -10 <sup>3</sup>	
<i>Salmonella Sp.</i>		1-10 <sup>2</sup>	
<i>Clostridium perfringens</i>	3.4 x 10 <sup>5</sup>	3.3 x 10 <sup>5</sup>	

## 2.6 References

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### **3 Clogging and Freeze-thaw Conditioning of Sand Drying Bed Filters with Biological Sludges**

#### **3.1 Abstract**

Sand drying bed columns filters (7.6 cm dia. with a 5 cm sand layer) were dosed with four biological sludges to the point of clogging before undergoing freeze-thaw (F/T) conditioning. Anaerobic digestion (AD) and primary sludges quickly clogged the filters, while septage and waste activated sludge (WAS) were effectively dewatered through multiple 10 cm doses. Particle size (D10) and particle size distribution (D60/D10) were strongly correlated to clogging dose, while parameters of sludge stability (COD/BOD<sub>5</sub>, %VS) were not. Freeze-thaw (F/T) conditioning was shown to be effective at restoring drainage capacity with the four sludges studied; suggesting that sand drying beds can be operated as freezing beds during the winter without prior desludging. The filters were shown to be very effective at removing suspended solids (>99%), phosphorus (97-99%), organic matter (84-98%) and nitrogen (68-82%) from the four sludges studied. Directly after F/T conditioning, the sludge cakes achieved 25-32% dry matter content which is comparable to mechanical dewatering technologies.

#### **Key words**

Freeze-thaw conditioning, sand drying bed, sludge dewatering, clogging.

#### **3.2 Introduction and Literature Review**

Sludge drying beds are widely utilized for sludge dewatering, particularly for small community wastewater systems. In the United States fully two thirds of wastewater treatment plants use drying beds and treat 50% of all dewatered sludges (Wang et al., 2007). The advantages to sludge drying beds include low capital costs, low energy and simple operation, while the constraints include lack of efficiency during periods of extended precipitation or freezing temperatures. Recommended annual loading rates can vary between 60-120 kg/m<sup>2</sup>/y for digested sludge, depending on climatic conditions (US EPA, 1987). However, the application of drying beds in cold climates is limited to summer

operation and requires sludge storage capacity for approximately 6 months of the year. To address this issue, sand drying beds can be operated in the winter as freezing beds by extending the side walls to accommodate the accumulating frozen sludge layers (Martel, 1989b; US EPA, 1987). To further reduce operating costs, which relate mostly to bed desludging (Kim and Smith, 1997), it is proposed that sludge loading can be optimized based on sludge characteristics and that drying beds can be loaded to the point of clogging prior to winter with natural F/T conditioning used to restore bed permeability.

The risk of bed clogging depends primarily upon the nature of the sludge, with floc particle size and size distribution considered to be the most important factors influencing sludge dewaterability (Lawler *et al.*, 1986; Karr and Keinath, 1978). Karr and Keinath (1978) fractionated and remixed different sludges with similar particle size proportions and found identical dewaterability, with particle sizes between 1-100  $\mu\text{m}$  having the largest impact on dewaterability. Particle size distribution was shown to vary by sludge type, with Houghton *et al.* (2002) observing particle size distribution peaks of 22-32  $\mu\text{m}$  for digested primary sludge and 110-140  $\mu\text{m}$  for digested WAS while Vincent *et al.* (2012) reported septage particle size ranging from 3.3-71  $\mu\text{m}$  and WAS particle size ranging from 21-163  $\mu\text{m}$ . Several studies have confirmed that small floc particles within a wide particle distribution cause blinding during filtration with small particles filling the pore spaces of the sludge cake (Novak *et al.*, 1988; Sorensen *et al.*, 1995). Mikkelsen and Keiding (2002) compared different sludge types and found that waste activated sludge, with high extra-cellular polymeric substances (EPS), had large floc sizes, low degrees of dispersion and high filterability, while both primary and AD sludges, with low EPS, had small floc size, high degree of dispersion and low filterability. However, several authors have shown a negative relationship between EPS and sludge dewaterability, citing the high interstitial water held by EPS (Jin *et al.*, 2004; Neyens *et al.*, 2004). Mahmoud *et al.* (2006) compared particle size distribution using wet sieve analysis of anaerobically digested primary sludge and found a substantial reduction in floc sizes at all operating conditions with the majority of flocs less than 100  $\mu\text{m}$  and the number of small particles corresponding to a decrease in sludge dewaterability.

F/T conditioning occurs as particulate matter is rejected during ice crystal formation and consolidated into solid particles along the crystal boundary (Reed *et al.*, 1986). F/T conditioning is very effective at agglomerating colloidal particles of less than 10 µm while particles greater than 100 µm tend to become trapped in the advancing ice front (Vesilind and Martel, 1990). Freezing temperature plays an important role in dewaterability, with dewaterability increasing as freezing temperature increases. The most effective freezing temperatures generally ranging between -2 to -10°C and dewaterability decreasing rapidly below -30°C (Hung *et al.*, 1996; Chu and Lee, 1998; Vesilind and Martel, 1990; Wang, 2001). As winter temperatures rarely fall below -30°C in most populated cold climate regions, F/T conditioning should be a widely applicable technique. Many studies have demonstrated the effectiveness of F/T conditioning of sludges including: primary, WAS, AD, chemical coagulant and water plant sludges (Vesilind and Martel, 1990; Vesilind and Hsu, 1997; Chu and Lee, 1998). CST was shown to decrease significantly after F/T conditioning with primary sludge from 2550 to 182 s and WAS from 133 to 16 s (Baskerville, 1971), AD sludge from 212 to 57 s (Martel, 1989a) and WAS from 197 to 40 s (Chu *et al.*, 1999). Freezing beds have been successfully operated at the pilot scale in the N.E. United States to treat WAS, AD sludge and water treatment plant alum sludge (Martel and Diener, 1991; Martel, 1993) and in Ontario, Canada to treat septage (Kinsley *et al.*, 2012). While both laboratory and field studies have demonstrated the effectiveness of F/T conditioning for sludge dewatering, no literature could be found demonstrating the effectiveness of F/T conditioning to restore partially to fully clogged sand drying beds.

The objectives of this laboratory study are to characterize dewatering efficiency and clogging in sand drying bed column filters with four common sources of biological sludge and to evaluate the effect of F/T conditioning to restore bed permeability.

### **3.3 Materials and Methods**

Four types of sludge (Primary sludge, WAS, AD sludge and septage) were selected to represent both undigested and digested biological sludges covering a wide range of sludge age and stability. Septage was obtained from a local septic tank pumper in Eastern Ontario from a truckload containing sludge from three septic tanks where the tanks had not been

pumped for the previous 3-5 years (representative of typical household pump-out frequency). The AD sludge had a solids retention time (SRT) of 10 d and was obtained from a conventional mesophilic anaerobic digester treating primary sludge and thickened waste activated sludge (TWAS) at the Ottawa (ON) wastewater treatment plant (WWTP). WAS and primary sludge samples were obtained from the Hawkesbury (ON) WWTP. The WAS had an SRT of 5-6 days and the wastewater had been treated with alum at 35-40 mL/L. The septage and AD sludge samples were collected at the beginning of the trial, while WAS and primary sludge samples were collected monthly throughout the trial. All sludge samples were stored in 20 L pails at  $4 \pm 2^\circ\text{C}$ . The sludge samples were analysed for the following parameters at the Ontario Rural Wastewater Centre's environmental quality laboratory following Standard Methods (APHA, 2005): COD, sCOD, BOD, TS, VS, TKN,  $\text{NH}_4^+$ ,  $\text{NO}_3^-$ , TP and  $\text{OPO}_4^{3-}$ . sCOD samples were pre-treated by centrifugation at 3800g and filtered at  $0.45 \mu\text{m}$  prior to COD analysis. Samples were sent to an external laboratory for Fats, Oils and Grease (FOG) analysis. Capillary Suction Time (CST) was measured with a CST machine purchased from the Fann Instrument Company of Houston, Texas. The small (1.0 cm) diameter funnel opening was used during the trials.

Mass fraction of sludge particle sizes was determined through serial filtration using diminishing filter pore sizes following a modified TSS Standard Method (APHA, 2005). A 25-50 mL sample of sludge was consecutively filtered through 400, 100, 20-25, 11, 1.5 and  $0.4 \mu\text{m}$  filters. Funnels and glassware were rinsed with distilled water to capture all solids. After the last filter, the remaining filtrate was analysed for TDS. A 10 cm dia. Buchner funnel connected to a 500 mL Erlenmeyer flask was used with the 400, 100 and 20-25  $\mu\text{m}$  filters. Suction was provided using a Barnant Co. Vacuum Pressure Station (21.3" Hg). A standard TSS apparatus was utilized for the remaining 11, 1.5 and  $0.4 \mu\text{m}$  filters. The 400 and 100  $\mu\text{m}$  filters were cut from sheets of long woven nylon obtained from Industrial Netting Inc. of Minneapolis, MN. The rest of the filters were purchased from Fisher Scientific of Ottawa, ON: 40  $\mu\text{m}$  x 9.0 cm dia. VWR-417, 20-25  $\mu\text{m}$  x 9.0 cm dia. Whatman No.4, 11  $\mu\text{m}$  x 5.5 cm dia. Whatman No.1, 1.5  $\mu\text{m}$  x 4.7 cm dia. Whatman Glass microfiber 934-AH and  $0.45 \mu\text{m}$  x 4.7 cm dia. Fisherbrand mixed cellulose esters. The procedure was

carried out in triplicate for each sludge type as well as on freeze-thawed sludge samples which had previously been frozen for 24 hours at  $-12\pm 2^{\circ}\text{C}$ .

The sand filter columns consisted of 7.6 cm (3 inch) dia. ABS piping placed in 320 mL 9.0 cm dia. Buchner funnels. The piping was initially 20 cm in height and later extended to 40 cm to accommodate increasing filter cake levels. The filter media consisted of 2 cm of ceramic aquarium gravel (2-4 mm) and 5 cm of screened silica/granite sand (0.85-2.0 mm) obtained from Bigelow Sands, Grenville, QC. It was decided to use a minimum depth of filter material as the purpose of the experiment was to observe clogging at the sand-sludge interface and to compare filtrate quality between the sludge types while minimizing any effect of sand filtration on treatment. The experiment was carried out on a workbench at room temperature ( $20\pm 4^{\circ}\text{C}$ ). Columns for each sludge type were run in triplicate. Sludge was added to each filter in 10 cm doses. A splash plate (rod with a plastic disk) was utilized to avoid eroding the sand layer while dosing each filter. Columns were dosed weekly with the exception of the WAS columns, where dosing frequency was increased after the first F/T conditioning as it was observed that the filters drained completely within hours of sludge application. Filtrate was collected daily from each filter, volume recorded, and stored at  $4\pm 2^{\circ}\text{C}$ . Filtrate samples from the three replicate filters for each sludge type were collected over each dosing period, mixed together, and analysed for a suite of water quality parameters: COD, BOD, TSS, TKN,  $\text{NH}_4^+$ ,  $\text{NO}_3^-$  and TP. Once clogging was observed in a filter; either by not completely draining within the 7 day period or by observing a flattening of the drainage response curve, the filter was taken out of service and placed in a freezer at  $-12\pm 2^{\circ}\text{C}$  for a 10 day freezing period. The filters were then removed from the freezer and allowed to thaw at room temperature. The filtrate collected during thawing was analysed for the same parameters as listed above. Dosing was then recommenced. A final F/T conditioning was applied at the end of the study with 3 d of freezing time followed by thawing at room temperature in order to dewater the filters prior to sludge cake characterisation. At the end of the experiment the dewatered sludge cake was characterized for: height, mass, TS, VS, *E. coli*, TKN,  $\text{NH}_4^+$ ,  $\text{NO}_3^-$  and TP. All analyses were conducted at the Ontario Rural Wastewater Centre's environmental quality laboratory following Standard Methods (APHA, 2005).

Regression analysis was conducted using data from the four sludge types to correlate clogging dose to measures of sludge stability (COD/BOD, %VS) and sludge particle size (D10, D60/D10) while significant differences in clogging doses between filters were determined using ANOVA. All statistics were carried out using the Data Analysis Toolpack in MSEXcel.

### 3.4 Results and Discussion

The characteristics of the four sludges evaluated are described in Table 3-1. Primary sludge mostly constitutes settled untreated organic matter and is a very young sludge (<1 d) characterized by the highest volatile content (64%), very high soluble organics (CODs=4400 mg/L), high solids (3.9%) and a low COD/BOD<sub>5</sub> ratio (1.8) relative to the other sludges evaluated. WAS mostly constitutes aerobic sludge flocs and is also considered to be a young sludge (5-6 d), with slightly lower volatile content (61%) than primary sludge, but has low soluble organics (CODs=149 mg/L), low solids (0.7%) and a medium COD/BOD<sub>5</sub> ratio (4.1). AD sludge mostly constitutes anaerobic sludge flocs and has a sludge age of 10 d, characterized by a low volatile content (46%), low soluble organics (CODs = 573 mg/L), relatively high solids (3.0%) and a very high COD/BOD<sub>5</sub> ratio (19.2). Finally, septage has an extremely high sludge age (3-5 years), median volatile content (55%), medium solids (1.8%) and a high COD/BOD<sub>5</sub> ratio (10.3). The CST values were lowest for WAS (75 s), as would be expected given the low solids content. For the three remaining sludges, with solids content varying between 1.8 and 3.9%, CST values were similar for primary sludge (465 s) and septage (331 s), while that of AD sludge (2334 s) was considerably higher. The D<sub>10</sub> (diameter of 10% passing) values varied by a factor of two, with AD and primary sludge having the lowest values. The potential for blinding increases with the particle size distribution and is represented by the ratio of D<sub>60</sub>/D<sub>10</sub>, which varies by a factor of three with primary and AD sludge having the highest values. Based on the findings of both Lawler *et al.* (1986) and Mikkelsen and Keiding (2002), which demonstrate the importance of both particle size and particle size distribution in filter clogging, and considering the CST values, it is expected that AD sludge would have the highest clogging potential, followed by primary sludge, then septage and finally WAS.

**Table 3-1. Sludge Characteristics**

Parameter	Primary Sludge	AD Sludge	Septage	WAS
n	5	3	3	5
TS (mg/L)	38500 ± 8000	30000 ± 4400	18300 ± 5000	6800 ± 2600
VS (%)	64 ± 4	46 ± 2	55 ± 3	61 ± 6
COD (mg/L)	37610 ± 18720	28130 ± 1310	20500 ± 3000	9090 ± 1300
CODs (mg/L)	4400 ± 2900	573 ± 191	285 ± 106	149 ± 98
BOD <sub>5</sub> (mg/L)	20500 ± 14800	1468 ± 505	2000 ± 700	2200 ± 800
COD/BOD <sub>5</sub>	1.8	19.2	10.3	4.1
TKN (mg/L)	1088 ± 353	1755 ± 249	697 ± 352	375 ± 292
NH <sub>4</sub> <sup>+</sup> -N (mg/L)	83.3 ± 32.6	651 ± 361	78 ± 23	46.3 ± 31.8
NO <sub>3</sub> -N (mg/L)	0.9 ± 0.4	1.3 ± 0.7	2.4 ± 0.7	0.8 ± 0.3
TP (mg/L)	293 ± 43	422 ± 41	99 ± 24	96 ± 20
FOG (mg/L)	36 ± 18	82	19	5 ± 2
CST (s)	465 ± 71	2234 ± 294	331 ± 37	75 ± 70
D <sub>10</sub> (µm)	34 ± 2	24 ± 2	43 ± 1	43 ± 2
D <sub>60</sub> (µm)	189 ± 111	69 ± 8	92 ± 6	82 ± 7
D <sub>60</sub> /D <sub>10</sub>	5.6 ± 3.5	2.9 ± 0.3	2.1 ± 0.2	1.9 ± 0.1
Sludge Age	<1 d	10 d	3-5 years	5-6 d

The effect of freeze-thaw conditioning on sludge particle size distribution is presented in Fig 3-1. The D<sub>10</sub> of the four sludge samples increased from 24-43 µm in the raw sludge to 73-194 µm after freeze-thaw conditioning. As well, the percentage of solids below 100 µm, the particle size limit shown by Karr and Keinath (1978) to influence dewatering efficiency, decreased from 52-93% in the raw sludges to 7-15% after freeze-thaw conditioning; demonstrating that freeze-thaw conditioning is effective at agglomerating a large fraction of the supra colloidal particles between 1-100 µm in the four types of sludge evaluated.

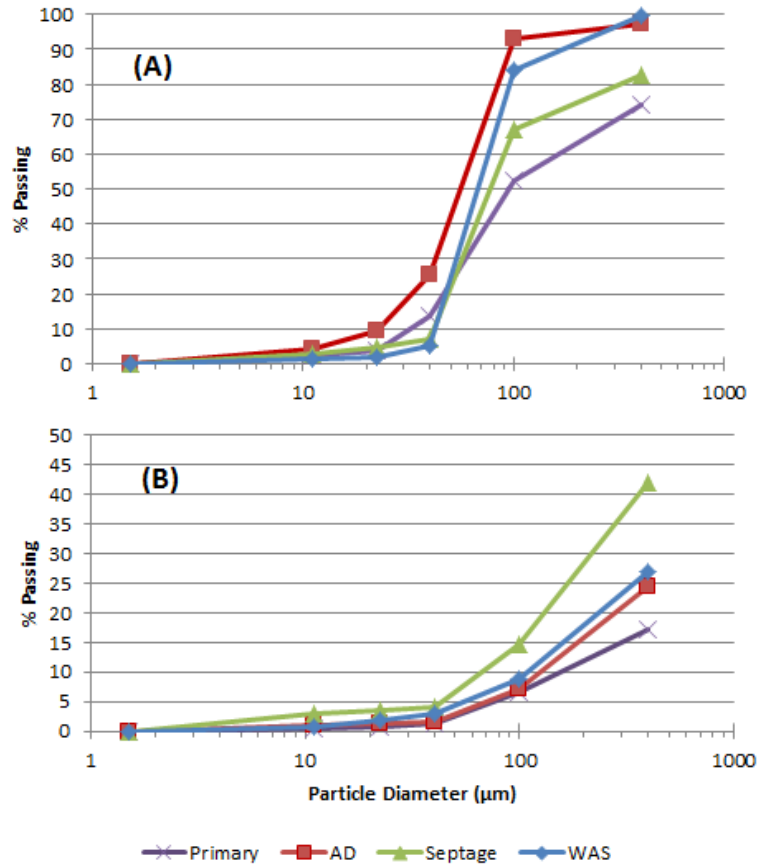


Figure 3-1. Particle Size Distribution by Mass Fraction [(A) Raw sludges and (B) Freeze-thaw conditioned sludges. Average of three replicates.]

The dose response curves for primary and AD sludges are presented in Figure 3-2, while the dose response curves for septage and WAS are presented in Figures 3-3.

The primary sludge columns showed a rapid decline in filterability over all three replicates with daily filtrate drainage reduced to less than 10 mm of the 100 mm dose after only 2 doses. The F/T conditioning was applied after seven dosing events, with a substantial volume of filtrate (65-77 mm) liberated from each filter. The volume liberated could include both interstitial water released from sludge flocs as well as ponding free water due to clogging. The drainage response improved dramatically after F/T conditioning, with full drainage achieved within 1 d for an additional 6 doses, at which time clogging was again observed. However, after conditioning only 50-65 mm of filtrate drained after each dose, suggesting that significant water is held within the sludge flocs. This observation is supported by Shamma and Wang (2007), who suggest that raw sludge does not dewater

as well as digested sludge. A final F/T conditioning was conducted after a further 7 doses which liberated 83-91 mm of filtrate from each of the filters, indicating a high degree of clogging.

The AD sludge columns responded in a similar manner to the primary sludge columns, with two of the three AD sludge columns showing signs of early clogging with daily drainage reduced to less than 10 mm after only 1 dose, while the first column continued to exhibit a reasonable drainage pattern throughout 7 doses. F/T conditioning was applied between 5-7 doses and liberated between 68-97 mm of filtrate, demonstrating a high degree of clogging. The first dose after conditioning exhibited complete drainage within 1 d; however, the drainage rate quickly declined and clogging was again observed after 3-5 doses, with a final F/T conditioning liberating between 46-70 mm of filtrate. Similar to the primary sludge columns, the AD columns only drained between 45-63 mm in the dose after F/T conditioning, suggesting that the AD sludge flocs are also holding significant interstitial water.

The three septage columns performed very well in comparison with both the primary and AD sludge columns. The first and third septage columns showed little signs of clogging with F/T conditioning applied after 22 and 16 doses, respectively and with only 32 and 36 mm of filtrate recovered. The second column showed signs of clogging after 7 doses and underwent F/T conditioning after 10 doses; however, only 30 mm of filtrate was liberated, suggesting that only a limited level of clogging had developed in the filter. Filter drainage was clearly restored in all three columns after conditioning with close to 90 mm of filtrate draining within 1d; suggesting that the septage flocs are holding much less interstitial water than either the primary or AD sludges. A final F/T conditioning was applied to the second and third columns liberating 36 and 13 mm of filtrate, respectively. The septage columns clearly exhibited better drainage properties and a lower propensity for clogging than both the primary and AD sludges both before and after conditioning.

The WAS columns also performed very well with the columns draining consistently within 1 day. The dosing frequency was increased after the first F/T conditioning to increase solids loading to the columns. The columns underwent F/T conditioning after 13-16 doses

with 44-80 mm of filtrate recovered, indicating that a level of clogging had developed. Subsequent doses drained very well with 80-90 mm of filtrate recovered. A final F/T conditioning was applied after a further 4-12 doses, with 25-42 mm of filtrate recovered. The WAS columns exhibited good drainage properties and a low propensity for clogging compared with the primary and AD columns.

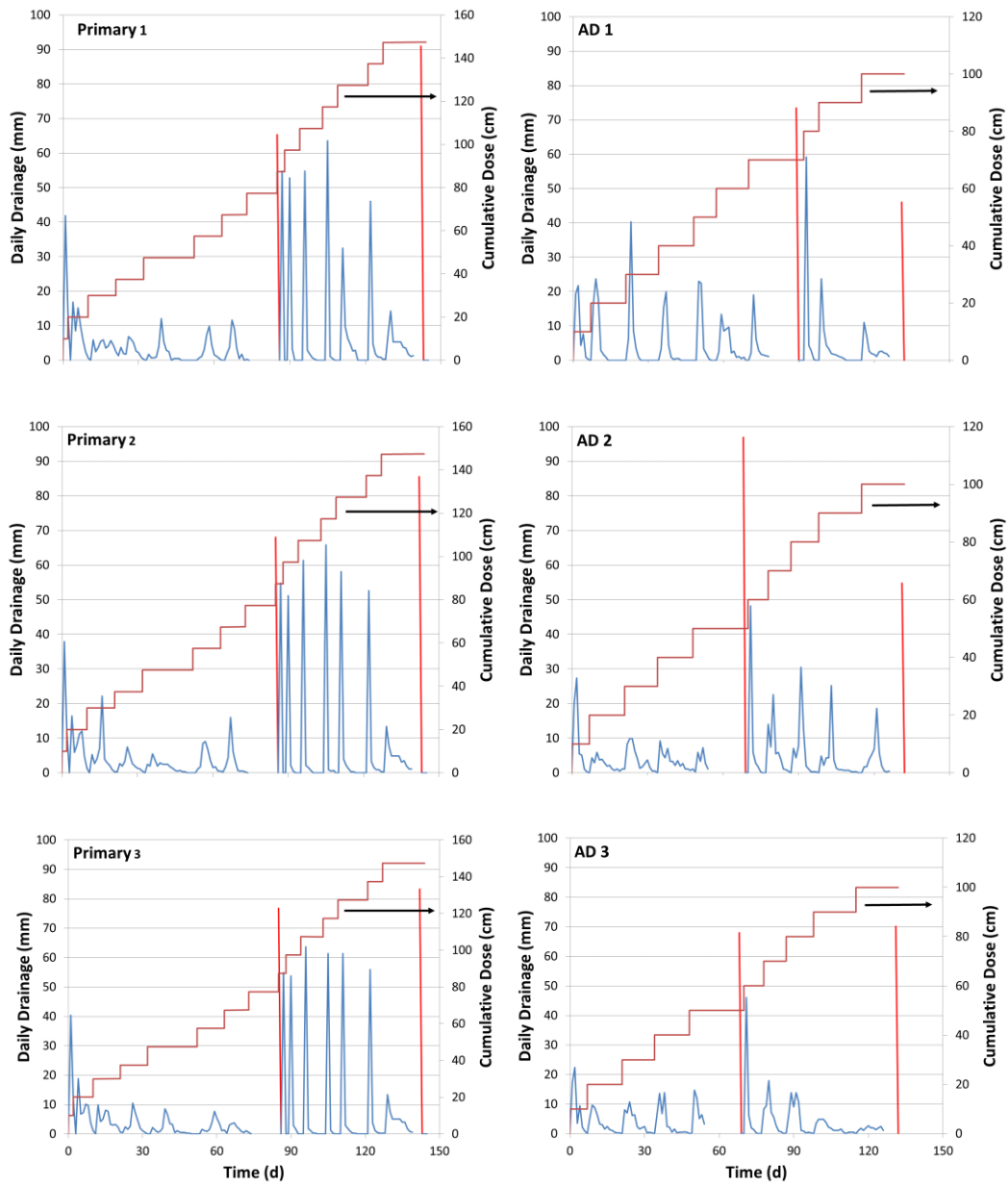
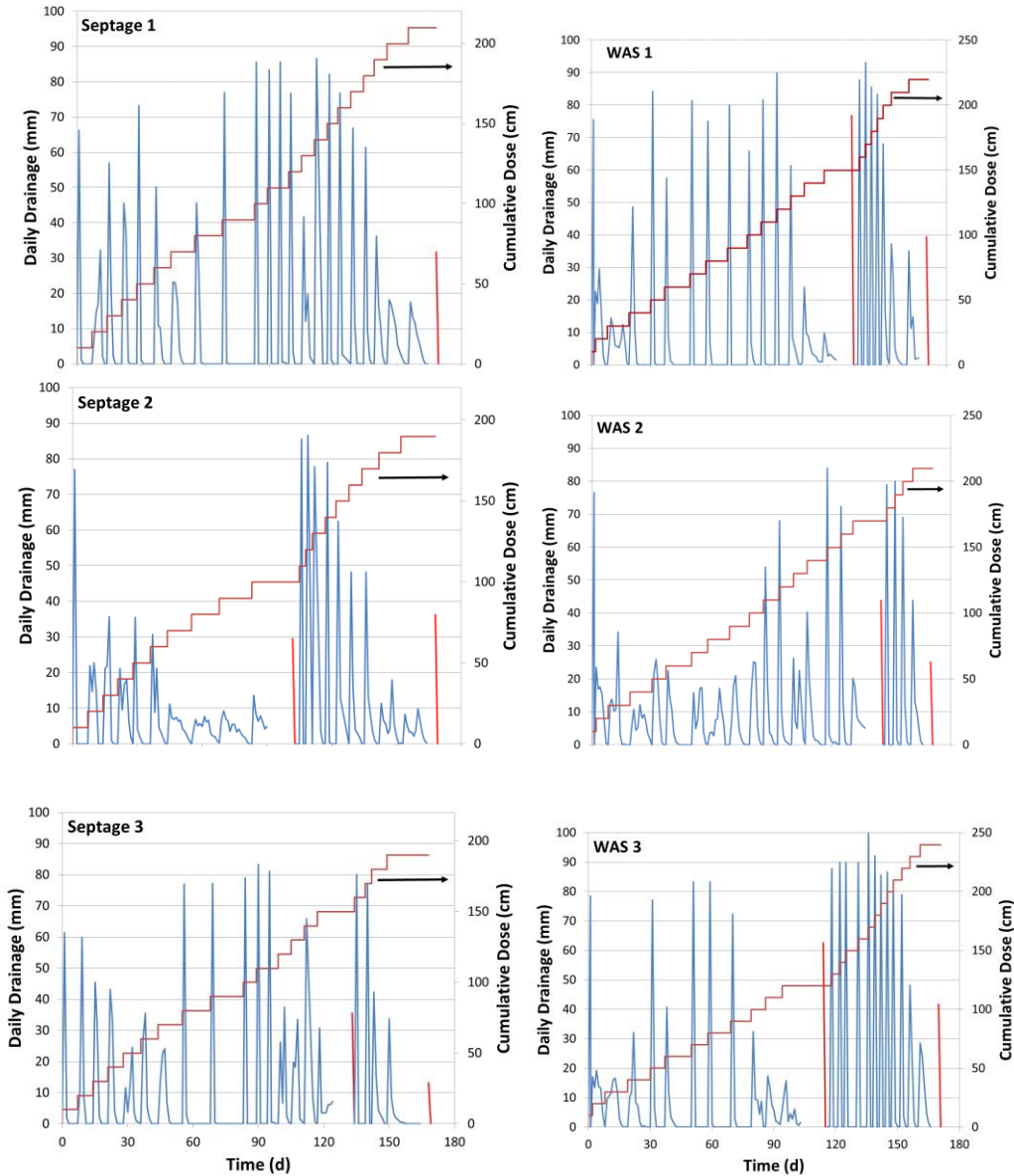


Figure 3-2. Dose Response with Consecutive 10 cm Doses of Primary and AD Sludge Applied to Sand Filters. Red bars represent the filtrate recovered after freeze-thaw conditioning.



**Figure 3-3. Dose Response with Consecutive 10 cm Doses of Septage and WAS Applied to Sand Filters. Red bars represent the filtrate recovered after freeze-thaw conditioning.**

The clogging doses were significantly lower for both primary and AD sludges as compared to WAS and septage ( $P < 0.05$ ). A very strong correlation was observed between clogging dose and the proportion of small particles ( $D_{10}$ ) ( $R = 0.91$ ) as well as strong negative correlation with particle size distribution ( $D_{60}/D_{10}$ ) ( $R = -0.71$ ), which supports the findings of Lawler *et al.* (1986) and Karr and Keinath (1978). However, indicators of organic matter stability (%VS, COD/BOD) showed relatively weak correlation with clogging dose of  $R = 0.25$  and  $R = -0.28$ , respectively.

Freeze-thaw conditioning was shown to be effective at recovering permeability in clogged filters regardless of the type of sludge; however, the lasting impact of the treatment is dependent upon the nature of the sludge. The primary and AD sludges became clogged within 3-7 doses (30-70 cm), which would not significantly reduce the desludging frequency of a sand drying bed. On the other hand, the septage columns were dosed between 10-22 times prior to exhibiting signs of clogging, while the WAS columns were dosed between 13-16 times, suggesting that loading of 100-220 cm can be applied prior to desludging or winter freeze-up. This represents between 2.5 to 5 months of operation at 10 cm/week dosing without requiring sludge removal. As well, both the septage and WAS columns showed very good drainage characteristics through multiple doses after F/T conditioning, indicating that desludging would not be required prior to switching filter operation from a drying bed to a freezing bed for the winter months.

In order to further reduce dewatering costs, the drying beds could be planted with *Phragmites* and operated as reed bed filters during the summer months; reducing the need for desludging to only once every 7-10 years as the plant rhizomes and stems maintain drainage pathways in the accumulating sludge cake (Nielson, 2003; De Maeseneer, J.L., 1997). If operated as a reed bed filter, freeze-thaw conditioning in the early winter would help to maintain bed permeability and could address any clogging occurring in the late fall, when the plants are dormant and evapotranspiration is at a minimum.

### **Filtrate Quality**

Filtrate from sand drying beds must be properly managed, either by pumping the filtrate to a treatment plant headworks, or in the case of a stand-alone septage dewatering facility, to a post-treatment system. Table 3-2 compares filtrate quality and percent separation from the four sludge treatments. Raw sludge quality is presented in Table 3-1.

**Table 3-2. Filtrate Quality from Sand Drying Bed Columns (Dosed 10 cm weekly with four biological sludges over a 6 month period).**

Parameter	Filtrate Concentration [Avg. ± Std. Dev.] (% Separation)			
	Primary Sludge	AD Sludge	Septage	WAS
TSS (mg/L)	250 ± 173 (99.3)	153 ± 100 (99.5)	141 ± 98 (99.0)	26 ± 21 (99.6)
COD (mg/L)	5,417 ± 4,711 (85.6)	1,919 ± 550 (91.0)	529 ± 175 (97.4)	185 ± 81 (97.9)
BOD <sub>5</sub> (mg/L)	3,391 ± 3,265 (83.5)	257 ± 275 (82.3)	153 ± 125 (92.5)	73 ± 51 (96.7)
TP (mg/L)	5.7 ± 8.4 (97.6)	9.5 ± 3.3 (97.7)	2.4 ± 1.8 (96.6)	0.5 ± 0.2 (99.1)
TKN (mg/L)	280 ± 214 (74.3)	455 ± 175 (74.1)	95 ± 47 (86.4)	58 ± 30 (83.7)
NH <sub>4</sub> <sup>+</sup> -N (mg/L)	187 ± 125 -	368 ± 178 (43.5)	63 ± 43 (19.2)	27 ± 18 (41.3)
NO <sub>3</sub> <sup>-</sup> -N (mg/L)	4.2 ± 8.3 -	62.3 ± 108.6 -	32.5 ± 25.3 -	58.4 ± 67.9 -
TN-N (mg/L)	284 ± 222 (73.9)	517 ± 284 (70.6)	128 ± 82 (81.7)	116 ± 98 (67.7)

Filtration through a 5 cm sand layer is shown to be very effective at separating TSS from all four sludge treatments with 99.0-99.6% separation observed and with average filtrate TSS concentrations ranging from 26 to 250 mg/L. Very good separation rates for both COD and BOD<sub>5</sub> were also observed, ranging from 85.6 to 97.9% for COD and 82.3 to 96.7% for BOD<sub>5</sub>. Filtrate BOD<sub>5</sub> concentrations for WAS, septage and AD sludge remained constant over time and were typical of domestic strength wastewater, with average concentrations ranging from 73 to 257 mg/L. The low soluble COD from these sludges (149-573 mg/L) suggests that most of the organic matter is related to the sludge particles and is removed through filtration along with the TSS. However, primary sludge had very high soluble COD (4400 mg/L) and filtrate COD and BOD<sub>5</sub> concentrations declined dramatically over the course of the study (Figure 3-4). This suggests that a biofilm had developed within the filter capable of metabolizing most of the soluble organics in the filtrate; which may have contributed to filter clogging.

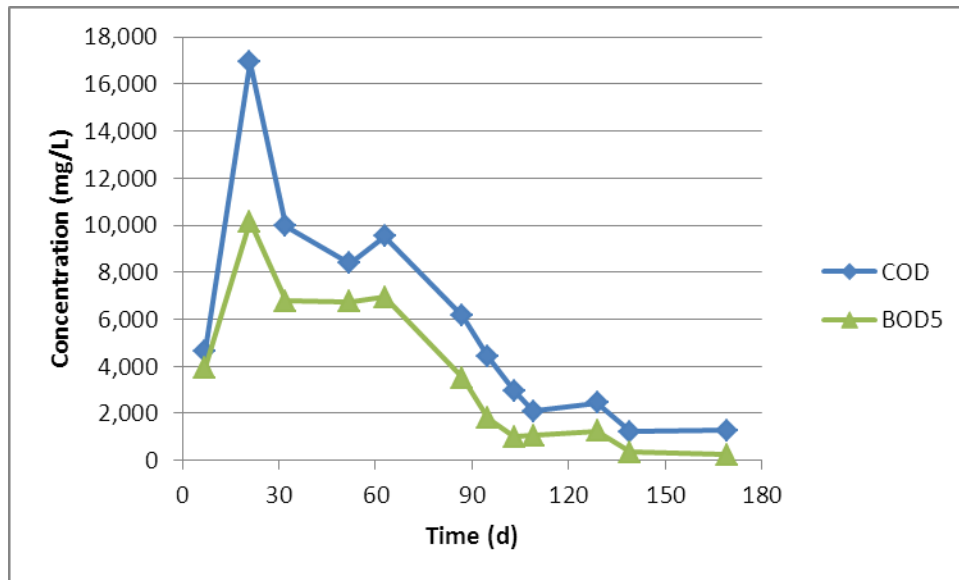


Figure 3-4. Organic Matter in Primary Sludge Filtrate with Time

Excellent separation of TP was observed with all four treatments, ranging from 96.9 to 99.1 % removal; indicating that almost all of the phosphorus is bound to organic solids within the sludge flocs and is effectively removed through filtration.

Nitrogen removal was fairly high for all sludge types with TN separation ranging from 68 to 82%. The N removed is largely related to the filtration of the organic N fraction, with separation rates of 90-95%. There was low  $\text{NO}_3^-$  in all raw sludges (1-3 mg/L); however, nitrification was observed with all trials, with average filtrate  $\text{NO}_3^-$  ranging from 4 to 62 mg/L. The average filtrate TN was similar in septage and WAS (128 and 116 mg/L, respectively), higher in primary sludge filtrate at 284 mg/L, due to observed ammonification, and higher yet in AD sludge filtrate at 517 mg/L due to high ammonium in the raw sludge. These values are all higher than typical domestic wastewater TN of 40-85 mg/L (Metcalf and Eddy, 2003) and should be taken into account when evaluating treatment options. There was evidence of denitrification within the AD sludge columns, as the reduction in  $\text{NH}_4^+$  was not accounted for by an increase in  $\text{NO}_3^-$  concentrations. This suggests the presence of anoxic zones within the filter cake, likely resulting from ponding at the sand-sludge interface.

## Sludge Cake Characteristics

The dewatered sludge cake characteristics are presented in Table 3-3. Solids varied from 25 to 32%, which is typical of gravity separated sludge with limited evapotranspiration (Kopp and Dichtl, 2001). Volatile matter decreased significantly in primary sludge from 64 to 55% and in WAS from 61 to 56%, suggesting the development of active biofilms within the filters. However, no significant decrease in volatile matter was observed in either AD sludge or septage, likely as these two sludges were already largely stabilized. *E. coli* varied between 3.0 and 4.0 log CFU/g DM; which is well below the limit for land application (MOE, 1996). Most inorganic N leached out of the columns, leaving 95-98% organic N in the sludge cake. All four biological sludges can provide a good source of nutrients and organic matter for land application to agricultural soils. As expected, sludge accumulation rates per volume of sludge applied increased with sludge solids content from 3.3 cm/m with WAS at 0.7% solids to 12.4 cm/m with primary sludge at 3.8% solids. Solids accumulation rates were higher for the young sludges (primary sludge and WAS) at 0.17 and 0.14 cm/kg/m<sup>2</sup>, respectively, compared with 0.11 and 0.10 cm/kg/m<sup>2</sup> for AD sludge and septage, respectively, suggesting that the more stabilized sludges form a more dense sludge cake.

**Table 3-3. Sludge Loading and Dewatered Sludge Cake Characteristics (6 month study)**

Parameter	Type of Sludge			
	Primary	AD	Septage	WAS
Hydraulic Loading (cm/m <sup>2</sup> )	177	120	190-210	210-240
Mass Loading (kg/m <sup>2</sup> )	67.0	35.9	34.8-38.5	13.4-15.7
Accumulation Rate (cm/m)	12.4±0.3	7.9±0.4	5.3±0.5	3.3±0.6
Accumulation Rate (cm/kg DM/m <sup>2</sup> )	0.17±0.01	0.11±0.01	0.10±0.01	0.14±0.02
TS (mg/g)	298 ± 11	316 ± 4	298 ± 11	248 ± 23
VS (%)	55 ± 2	42 ± 1	57 ± 0.4	56 ± 4
<i>E. coli</i> (log CFU/g DM)	3.1 ± 0.7	3.4 ± 0.6	3.0 ± 0.8	4.0 ± 0.9
TN (% dry)	2.43	2.33	3.01	3.22
TP (% dry)	1.30	3.00	0.55	1.90

Sludge loading to the filters varied between 13-67 kg/m<sup>2</sup> over the 6 month study without necessitating sludge removal due to the effect of F/T conditioning. With the exception of

WAS, all of the sludges were within the recommended annual loading rates for sand drying beds of 60-120 kg/m<sup>2</sup>/y (USEPA, 1987). The WAS columns received a lower mass loading due to the low solids content of the sludge; however, they received the highest hydraulic loading at 210-240 cm/m<sup>2</sup> over the 6 month study period.

### 3.5 Conclusions

Freeze-thaw conditioning was shown to be effective at restoring drainage capacity in clogged sand drying bed filters regardless of the type of sludge applied; however, the lasting impact of the treatment is dependent upon the nature of the sludge. Particle size and particle size distribution were shown to be good indicators of filter clogging potential, while parameters of sludge stability were not. Primary and AD sludge showed signs of clogging within several 10 cm doses both before and after F/T conditioning, while septage and WAS showed good drainage characteristics over multiple doses both before and after conditioning. This research suggests that it is possible to continuously dose sand drying beds with WAS or septage at 10 cm/week for between 2.5 to 5 months without desludging and that it is not necessary to desludge prior to operating the system as a freezing bed during the winter months.

The sludge cake forming at the sand surface was shown to be very effective at separating practically all solids (99%), phosphorus (97-99%), most organic matter (82-97%) and most nitrogen (68-82%) from the four types of sludge studied. The dewatered sludge can be reused as a source of organic matter and nutrients for agricultural production.

Further research is needed to validate these results at the pilot to field scales with various sources of both aerobic and anaerobic sludges for both sand drying beds and reed bed filters. The application of a combined drying bed – freezing bed or reed bed – freezing bed technology has the potential to significantly reduce sludge management costs for small communities in cold climate regions.

### 3.6 References

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## 4 Development and Modelling of a Sludge Freeze-Thaw Dewatering Bed

[C. Kinsley, K. Kennedy and A. Crolla (2012). Modelling and application of an uncovered freezing bed technology for septage treatment. Canadian Journal of Civil Engineering 39:10, 1136-1144.]. Roles: C. Kinsley - project PI, data collection and analysis, Kennedy - grad student supervisor and revision of manuscript, Crolla – project researcher, lab manager and revision of manuscript.

### 4.1 Introduction

In many jurisdictions in North America the sludge from septic tanks (septage) is routinely land applied to agricultural fields. However, land application is strictly forbidden during winter months when the ground is frozen (USEPA, 1994). Many septic tank pumpers do not possess storage lagoons and either cease to pump tanks during the winter months or haul to often distant wastewater treatment plants. For these haulers a winter treatment system would be beneficial, particularly in regions where there are significant numbers of holding tanks or the distance to a treatment plant is considerable. As well, there is a growing concern over the land application of untreated septage, with various jurisdictions either banning outright the practice or are in the process of doing so (Ontario Ministry of the Environment, 2008). Sludge freezing beds, potentially operated as drying beds during the summer months, could provide an effective treatment option for septic tank haulers. As well, freezing beds can provide an appropriate sludge dewatering solution for small community wastewater treatment plants, where space is often available, and the capital cost of traditional belt press or centrifuge technologies is often prohibitive.

The objectives of the study are to:

1. Develop a pilot sludge freezing bed technology and characterize the treatment of septage in terms of sludge dewaterability, sludge quality and filtrate quality.
2. Model the sludge freezing process with and without snow-cover.
3. Develop a multivariate model to describe sludge thawing considering initial frozen sludge depth, precipitation and degree-days of thawing.
4. Apply the sludge freezing and thawing models to determine the technology's operating limits across Canada and the northern United States.

The freezing process has long been known to improve sludge dewaterability (Vesilind and Martel, 1990). The dewatering process occurs as particulate matter is rejected during ice crystal formation and consolidated into solid particles along the crystal boundary (Reed *et al.*, 1986). Several wastewater treatment plants have incorporated sludge freezing into their operating practices, either by leaving sludge to freeze in a drying bed over winter, or dedicating a sludge lagoon to winter freeze-thaw conditioning (Martel, 1999). For example, the City of Winnipeg successfully utilised sludge freezing during the 1970s to dewater thickened anaerobic digestion (AD) sludge (Penman and Van Es, 1973). The frozen sludge was scraped from the beds and spread on near-by agricultural fields during the winter and incorporated into the soil in the spring when farm machinery had access to the fields. In a survey of eighteen reed bed filters in the US, three operated during the winter with 0.3-0.6 m accumulation of frozen sludge observed (Mellstrom and Jager, 1994). The sludge was observed to dewater very well in the spring.

A pilot freezing bed technology was developed at the US Army Cold Regions Research and Engineering Laboratory in Hanover, New Hampshire (Martel and Diener, 1991; Martel, 1993). The technology consists of a concrete basin with underdrains, a sand filter media and a corrugated fibreglass roof to keep out precipitation. Freeze-thaw treatment of three types of sludge were evaluated over four winters: AD sludge, waste activated sludge (WAS) and alum sludge. A range from 0.58-1.14 m of sludge was frozen and all types of sludge were successfully dewatered with dry matter (DM) ranging from 25% for WAS, to 39% for AD sludge and 82% for alum sludge. Effluent quality was of similar strength to domestic wastewater with average BOD<sub>5</sub> and TSS concentrations of 310 and 100 mg/L, respectively. The stabilized solids had a consistency similar to coffee grounds with an average size of 0.5 mm, a uniformity coefficient of 4.6 and a hydraulic conductivity of 50.4 cm/h, which is similar to a highly permeable soil. A full scale system was put into operation in 1990 at Fort McCoy, Wisconsin. During the first year of operation 1.0 m of AD sludge was applied and dry matter content in the sludge increased from 4.5% in the raw sludge to 78% in the reed bed after dewatering (Martel, 1993).

A full scale freezing bed was established to treat sludge from a WWTP in Sweden (Hellstrom and Kvarnstrom, 1997). The freezing bed design consisted of 30-50 cm of filter

sand over a gravel layer with underdrains and 1.5 m of freeboard to accommodate the sludge during winter. Sludge was applied in 10 cm layers, allowing each layer to freeze before adding the next layer. The pilot project continued over two winter seasons with average DM of the dewatered sludge varying from 53% after the first season to 25% after the second season; which was a mild winter with incomplete freezing of several layers observed (Hellstrom, 1997).

A comprehensive investigation of freeze-thaw conditioning of aerated facultative lagoon sludges was conducted in Quebec at École polytechnique de Montréal (Desjardins and Brière, 1996). The column experiments evaluated successive applications of sludge to a conventional 40 cm sand and gravel drying bed. The columns were loaded between 52 - 204 kg TSS/m<sup>2</sup>. Once thawed, all systems drained very quickly at 5-13 minutes for chemical sludge and 10-37 minutes for biological sludge, with dry matter ranging from 33-37%. When a layer of sludge was only partially frozen, the drainage rate decreased dramatically from 54 to 0.06 cm/h for biological sludge and from 821 to 0.52 cm/h for chemical sludge. The authors recommended either removing any snow cover or melting it with spray irrigated water.

A number of pilot studies have focused on optimizing the design and operation of sludge freezing beds and this technology has been proven to effectively dewater a variety of sludges including: WAS, AD sludge, alum sludge and aerated lagoon sludge (Martel, 1999; Penman and Van Es, 1973; Farrell *et al.*, 1970; Reed *et al.*, 1986; Martel, 1989; Martel and Diener, 1991; Martel, 1993; Hellstrom and Kvarnstrom, 1997; Hellstrom, 1997; Desjardins and Brière, 1996). It is suggested that sludge be applied in 8-10 cm layers and allowed to fully freeze before the next layer is applied (Farrell *et al.*, 1970; Reed *et al.*, 1986, Martel, 1989). Dry matter after freeze-thaw was found to be 39% for an AD sludge and 25% for WAS in pilot studies (Martel and Diener, 1991). According to Reed (1987), dewatered sludge will range between 17-35% DM right after freeze-thaw and can increase to greater than 50% with several weeks of drying.

Several authors have commented that snow should not be allowed to accumulate on the filter, as it will insulate the bed and hinder freezing (Martel, 1996; Reed *et al.*, 1986;

Desjardins and Brière, 1996). Covering the beds will add a significant capital cost and will reduce air flow. Mechanically removing the snow is time consuming and risks machinery becoming stuck in a partially frozen sludge bed. Melting the snow cover with irrigation water is feasible, but involves working with water outdoors during winter with the potential of pipes freezing. One possible operational solution would be to use the layers of sludge applied to the beds to melt any accumulated snow. This approach could work in regions where the annual snowfall is equal or less than the total depth of sludge applied over the winter months. However, as Desjardins and Brière (1996) showed, it is important that each layer of sludge fully freezes before the next layer is applied as an unfrozen layer will impede dewatering in the spring. Therefore, a safety factor in the total depth of sludge frozen in a season would be wise to accommodate situations of heavy snowfall or above average temperatures.

### **Sludge Freeze/Thaw Model Development**

Martel (1989) and Reed *et al.* (1986) proposed models to determine the time to freeze a layer of sludge. Both models are based upon the differential equation describing steady heat flux through a composite slab, which is commonly used to determine ice formation on lakes and streams (USACE, 2002):

$$\frac{\partial h}{\partial t} = \frac{1}{\rho\lambda} \frac{(T_m - T_a)}{\left(\frac{h}{k_i} + \frac{1}{H_{ia}}\right)} \quad \text{(Equation 4.1)}$$

Where:

h = ice thickness

$T_m = 0^\circ\text{C}$

$T_a$  = air temperature,  $^\circ\text{C}$

t = time

$\rho$  = ice density

$k_i$  = thermal conductivity of ice

$H_{ia}$  = heat transfer coefficient from the ice surface to the atmosphere

$\lambda$  = latent heat of ice

If the heat conduction through the ice is the controlling rate of energy flux, then the  $H_{ia}$  term can be ignored and the equation solved as:

$$h = m(\Delta T \cdot t)^{1/2} \text{ (Equation 4.2)}$$

Where

$$m = \sqrt{\frac{2k_i}{\rho\lambda}}$$

$h$  = depth of freezing, cm

$m$  = proportionality coefficient, which depends on thermal conductivity, density and latent heat of material being frozen, cm  $(^\circ\text{C}\cdot\text{d})^{1/2}$

$\Delta T$  = average negative daily temperature,  $^\circ\text{C}$

$t$  = time period, d

Typical values for  $m$  are described in Table 4-1 (USACE, 2002).

**Table 4-1. Proportionality Constant  $m$  for Various Snow Covered Conditions**

<b>Ice Cover Condition</b>	<b><math>m</math></b>
Windy lake with no snow	2.7
Average lake with snow	1.7-2.4
Average river with snow	1.4-1.7
Sheltered small river	0.7-1.4

Reed *et al.* (1986) applied Equation 4.2 to a sludge freezing bed. He found that  $m = 2.01$ - $2.14$  cm  $(^\circ\text{C}\cdot\text{d})^{1/2}$  for sludges of less than 8% solids and suggests a design value of  $m=2.04$

$(^{\circ}\text{C}\cdot\text{d})^{1/2}$ . The model was successfully validated in a field trial in Duluth, Michigan, where successive 20 cm layers of sludge were applied to a lagoon during the winter of 1981, with field observations indicating similar total frozen sludge depth to design calculations based upon Equation 4.2. Ideal freezing layer depth was determined by both Martel (1989) and Reed *et al.* (1986) to be 8 cm for the northern half of continental United States. As average temperatures decline, the freezing layer depth can be increased. Layers of 23 cm were successfully frozen in Duluth, Minn. and a 46 cm layer was frozen in Fairbanks, Alaska (Reed *et al.*, 1986).

While Equation 4.2 is accepted as an expression of ice formation on lakes and streams (USACE, 2002), which can be applied to a sludge freezing bed, there is no accepted model for ice thawing. Hellstrom (1997) presented thawing data from three uncovered sludge freezing ditches where he applied Equation 4.2 to the total accumulated layers of frozen sludge and found  $m$  varied between 3.1-3.7  $(^{\circ}\text{C}\cdot\text{d})^{1/2}$ . The model described in Equation 4.2 assumes that the rate of thawing is dependent upon the depth of the frozen layer, which would not hold true if pathways are present to drain the melt waters. Martel (1989) proposed a differential equation for sludge thawing in a covered bed with the assumptions that melt waters will drain immediately and the thawed sludge will insulate the remaining frozen layer. Martel's model requires parameters for thermal conductivity and solar absorptance of the sludge and transmittance of the roof material. A series of three pilot studies were conducted to validate the model and find differences between the predicted and actual thawed depths of 0, 10 and 27% (Martel and Diener, 1991).

## **4.2 Experimental Design and Methodology**

A series of 12 pilot scale freezing beds were constructed to evaluate and model the freeze-thaw conditioning of sludge. Domestic septic tank sludge (septage) was selected for use in this study as septage has not been evaluated in previous freezing bed studies and freezing beds can provide an interesting treatment option for the significant number of septic tank pumpers who directly land apply septage during the summer months and have no storage capacity for the winter; requiring hauling to often distant WWTPs.

The experimental setup consisted of a 9,000 L holding tank to store the raw septage and 12 pilot freezing bed filters draining to a pump chamber (Figures 4-1 and 4-2). Each filter consisted of a 2.0 m x 0.91 m dia. HDPE plastic pipe installed 1.6 m in the ground on its end with a cap. A 7.5 cm (3”) perforated PVC pipe installed in the bottom of each filter collects the filtrate and flows through a 3.8 cm (1.5”) PVC pipe to a concrete pump chamber (1.6 x 1.6 x 2.2 m). The filters consist of a 15 cm layer of 20-40 mm coarse gravel, a 30 cm layer of 13-20 mm medium gravel and a 15 cm layer of coarse sand (effective size ( $D_{10}$ ) = 0.18 mm; uniformity coefficient ( $D_{60}/D_{10}$ ) = 4.3). The sand used is somewhat finer than the recommended criteria for sand drying beds which is that the sand should have an effective size of 0.3-0.75 mm and a uniformity coefficient of less than 3.5 (Crites and Tchobanoglous, 1998); however, the sand used was that which was locally available.

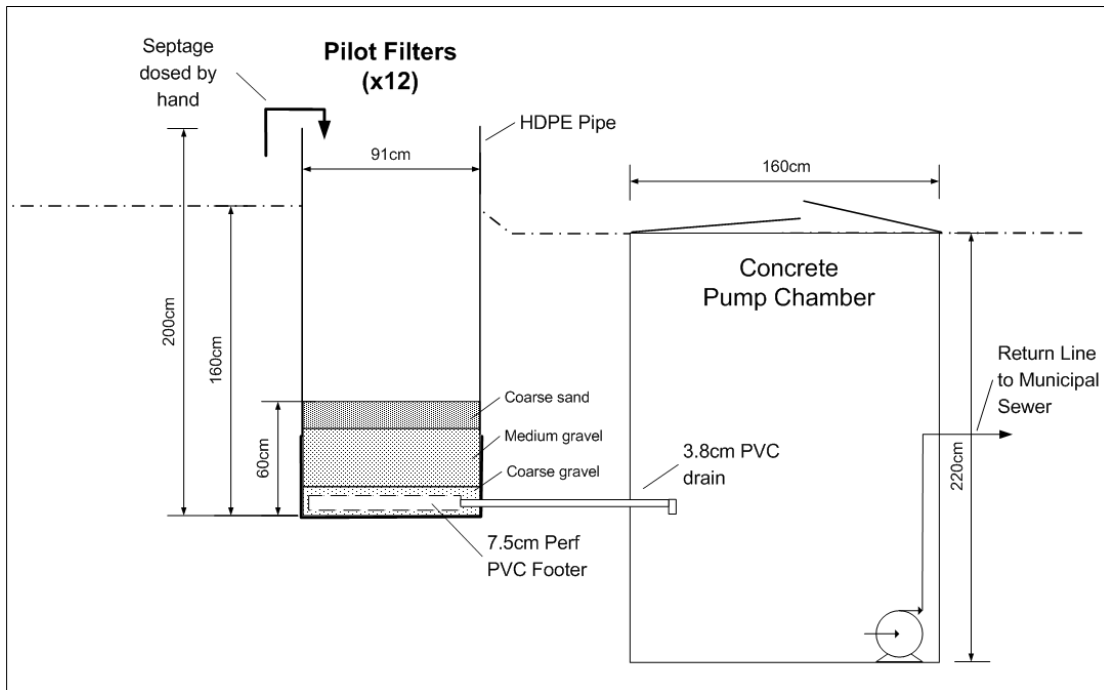


Figure 4-1. Side View Schematic of Pilot Freezing Bed Filters



Pilot Filters before Backfilling



Dosing a Pilot Filter

Figure 4-2. Photos of Pilot Filters

Septage from residential septic tanks is delivered periodically by a local septic tank pumper. Each load of fresh septage is comprised of sludge from three septic tanks which have not been pumped for between 2-10 years. A  $\frac{1}{4}$  HP sewage pump is used to mix the septage prior to dosing the filters. As described in Figures 4-3 and 4-4, three dose depths (8, 16, 24 cm) were tested over the first winter, while only two dose depths (8, 16 cm) were tested over the second winter as the 24 cm depth took a significant time to freeze and would likely not be used in practice. At the same time 3 of 12 filters had the snow removed during the 1<sup>st</sup> winter while 5 of 12 filters had the snow removed during the 2<sup>nd</sup> winter. Each dose of septage (8, 16 and 24 cm corresponding to 50, 100, 150 L) was pumped into 20 L plastic pails to ensure precise volumes applied to each filter. The septage was then poured into the filters onto a removable plastic plate to avoid eroding the filter surface.

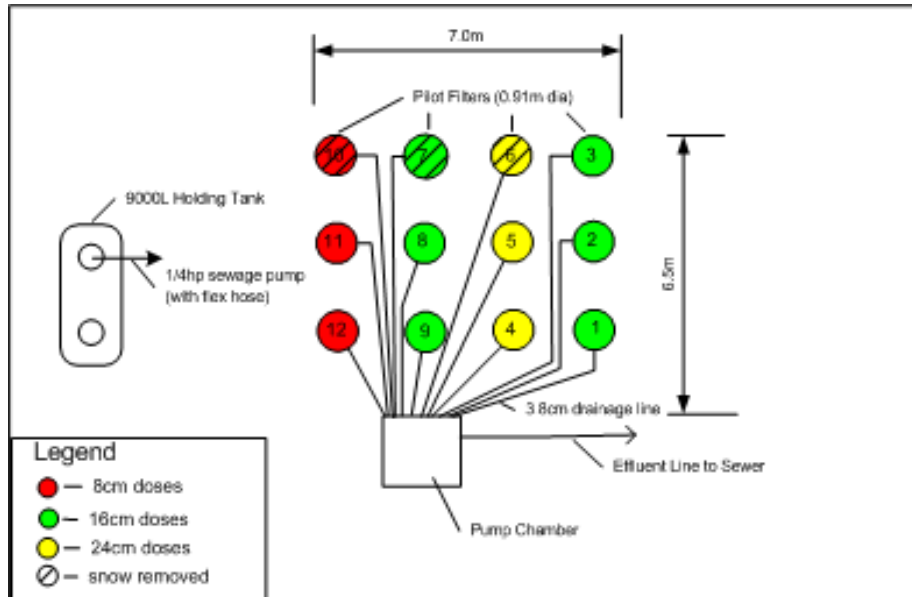


Figure 4-3. Plan View of Freezing Bed Pilot Filters with Dosing Plan (Winter 2010)

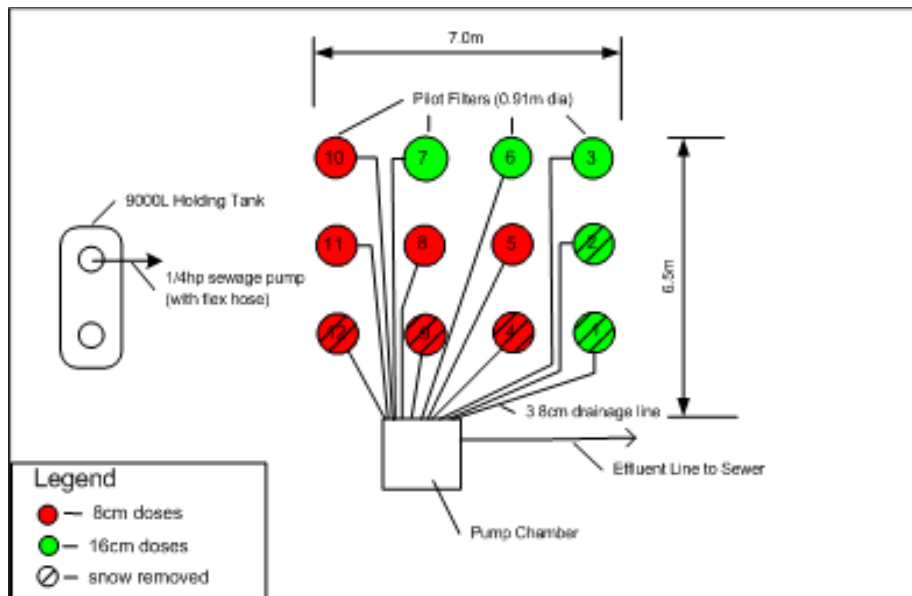


Figure 4-4. Plan View of Freezing Bed Pilot Filters with Dosing Plan (Winter 2011)

Sludge levels were measured before and after each dose during the winter months to quantify the depth of accumulating frozen sludge and every 1-2 days during the spring to quantify the depth of the melting sludge. After a dose of sludge was added to a filter, the ice thickness was measured daily either by creating a slot in the ice with a drill and using a caliper to measure the ice thickness or by drilling a core and measuring the core depth

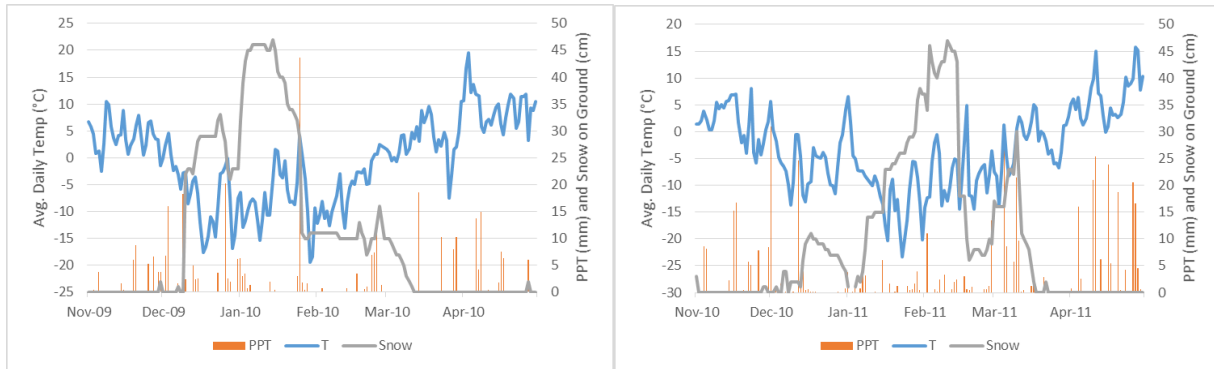
using a 1 cm (3/8") wood auger bit (see Figure 4-5). Once a layer has completely frozen, another dose of sludge is applied.



Figure 4-5. Photos of Measuring Frozen Sludge Layer Thickness

Raw septage samples were collected each time a new load of septage was delivered. The septage was thoroughly mixed before collecting a sample. During the 2010 operating season, both filtrate and dewatered sludge quality was evaluated. Filtrate grab samples were collected once in January before a frozen layer had formed and three times at bi-weekly intervals during the spring melt period, with the samples analysed for: COD, BOD<sub>5</sub>, TSS, TKN-N and *E. coli*. Dewatered sludge samples were collected bi-weekly from the end of the melting period until the end of June using a 2.5 cm soil core sampler with samples analysed for DM and *E. coli*. All analyses were carried out at the Environmental Quality Laboratory of the Université de Guelph-Campus d'Alfred and follow Standard Methods for the Analyses of Waters and Wastewaters (APHA, 2005). Differences between experimental treatments were analysed for significance using ANOVA.

Climatic conditions during the two study seasons are presented in Figure 4-6.



**Figure 4-6. Average Daily Temperature, Precipitation and Snow Cover during the Study Period (Ottawa International Airport Environment Canada Climate Data)**

### 4.3 Results and Discussion

#### Freezing Model

The proportionality constant ( $m$ ) was calculated for each experimental run using Equation 4-2 and plotted versus the frozen depth ( $h$ ) (Figure 4-7). A bias is observed in the data as the proportionality constant ( $m$ ) increases with the depth of frozen sludge. This is due to the latent heat of fusion required to cool the water in the sludge to the freezing point before ice starts to form. Equation 4.2 was modified by subtracting an initial number of degree-days of cooling proportional to the depth of sludge applied. The constant  $k$  was obtained by varying  $k$  until the linear correlation of the  $m^*$  vs  $h$  curve approached zero (Figure 4-8).

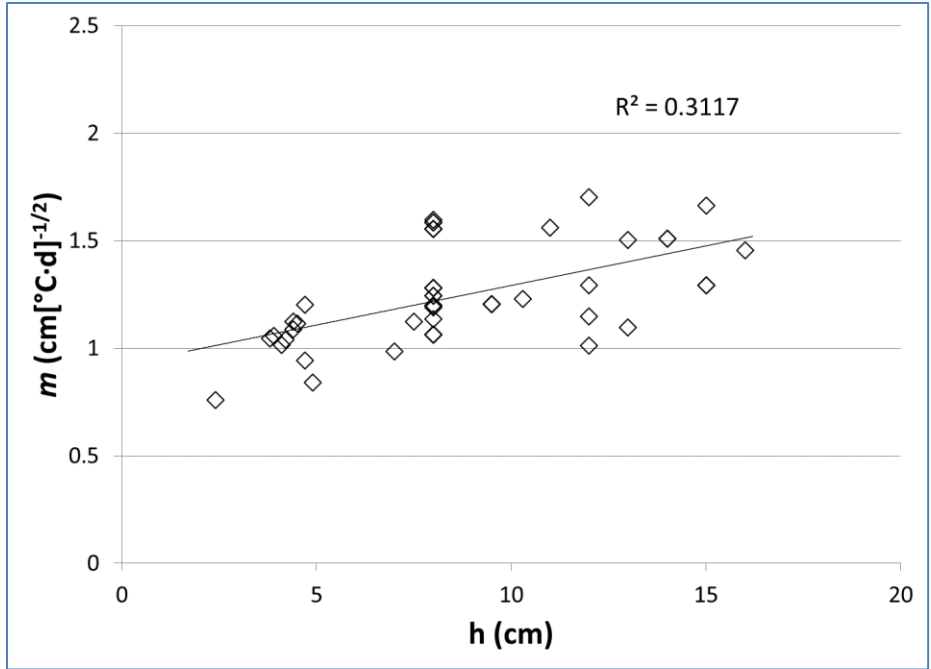


Figure 4-7. Proportionality Constant ( $m$ ) versus Frozen Depth ( $h$ )

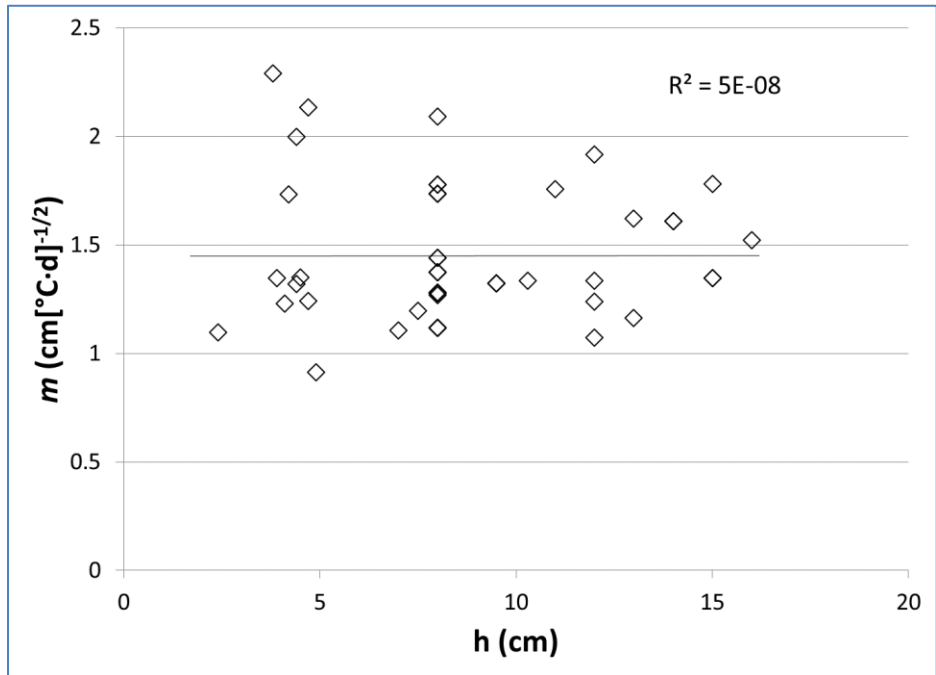


Figure 4-8. Proportionality Constant Corrected for Initial Cooling ( $m^*$ ) versus Frozen Depth ( $h$ )

The proportionality constant ( $m^*$ ) was compared between experimental runs with and without snow removal (see Table 4-2). No significant difference was observed between the

two groups ( $P > 0.05$ ), therefore the data can be treated as a single group with  $m^* = 1.45 \pm 0.09$  ( $^{\circ}\text{C}\cdot\text{d}$ )<sup>1/2</sup> at the 95% C.I.

**Table 4-2. ANOVA Comparison of Freezing Layer Experiments W/WO Snow Removal**

<i>Groups</i>	<i>Count</i>	<i>Mean Proportionality Constant (m) (<math>^{\circ}\text{C}\cdot\text{d}</math>)<sup>1/2</sup></i>	<i>Variance</i>	<i>P-value</i>
W/Snow Removal	19	1.36	0.06	0.11
WO/Snow Removal	30	1.51	0.11	

Therefore, the expression for sludge freezing is defined by Equation 4.3:

$$h = m * (\Delta T \cdot t - k \cdot T_0 \cdot h_0)^{1/2} \text{ (Equation 4.3)}$$

$$\text{where } k = 0.119 \frac{^{\circ}\text{C}\cdot\text{d}}{\text{cm}},$$

$T_0$  = initial sludge temperature ( $^{\circ}\text{C}$ ), and

$h_0$  = thickness of sludge dose (cm)

These results suggest that modest seasonal snowfalls (1.3 and 1.6 m over the two year study period) did not significantly impact the rate of freezing and that snow accumulation on the beds was controlled by melting the snow with subsequent sludge doses.

Plotting  $h$  versus  $t\Delta T$  in Figure 4-9 exhibits the model in the form of Equation (4.3), which fits the data well with  $R^2 = 0.80$ . Replicate trials are displayed with yellow (2010) and orange (2011) markers and are used to calculate experimental pure error. Two statistical tests were carried out on the model: a significance test and a lack of fit test (see Table 4-3 below) (McLean and Burns, 2010). The Significance Test compares the ratio of MS regression and MS residuals to an F distribution. Under the null hypothesis the fitted model explains no variation in the data. Since  $F > F_{1, 48, 0.05}$  we reject the null hypothesis and conclude that the fitted model is significant. The Lack of Fit Test compares the ratio of Lack

of fit MS and Pure Error MS to an F distribution. Under the null hypothesis there is no lack of fit. Since  $F > F_{41, 7, 0.05}$  we reject the null hypothesis and conclude that there is a significant lack of fit to the model. This test indicates that the pure error does not significantly explain the variation in the model output. This stands to reason as changes in climatic conditions, primarily wind and relative humidity, will strongly influence the rate of freezing, while replicate tests carried out under the same climatic conditions resulted in very similar frozen depths (see Figure 4-9). However, the model fits the data reasonably well and can be applied with information readily available to a site operator; that being the thickness of the sludge doses and the degree days of freezing.

**Table 4-3. ANOVA Table for Sludge Freezing Model**

	<i>df</i>	<i>SS</i>	<i>MS</i>	<i>F</i>
Regression	1	3590.45	3590.45	196.83
Residual	48	875.58	18.24	
Pure Error	7	1.29	0.19	115.02
Lack of Fit	41	874.28	21.32	
Total	49	4466.03		

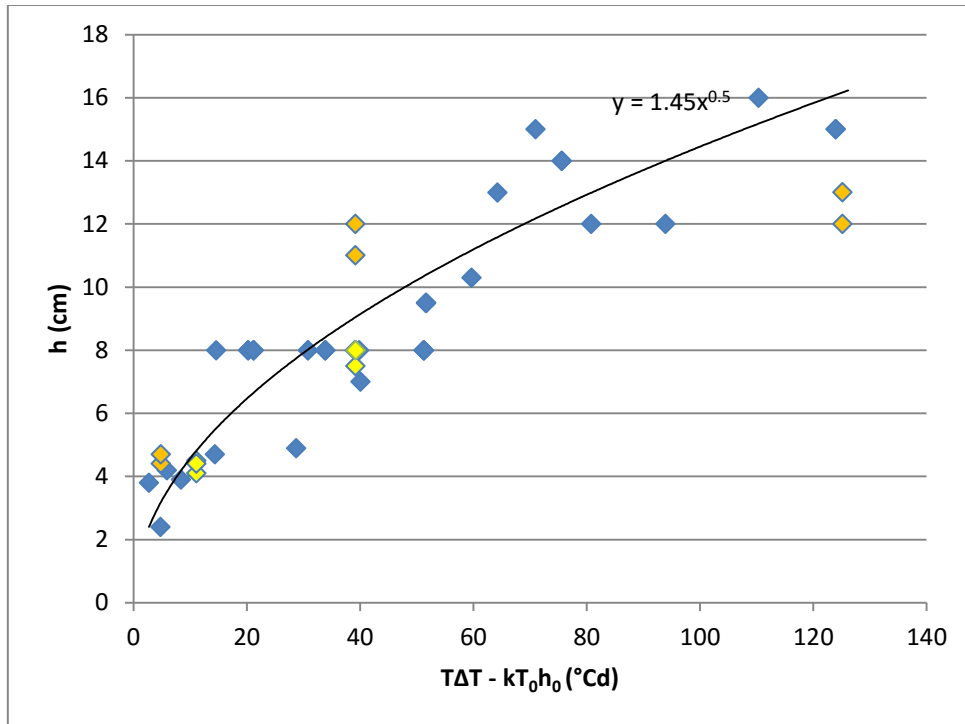


Figure 4-9. Frozen Depth (cm) versus Degree Days of Freezing (°Cd). Replicate trials are identified by yellow (2010) and orange (2011) markers.

The value of  $m^*=1.45\pm 0.09$  ( $^{\circ}\text{C}\cdot\text{d}$ )<sup>1/2</sup> is comparable to but somewhat lower than the range of 2.01-2.14 ( $^{\circ}\text{C}\cdot\text{d}$ )<sup>1/2</sup> reported by Reed (1985) and falls within the range of values reported by USACE (2002) for a river with snow cover. The pilot filters were below the ground surface and thus are subject to heat transfer from unfrozen soil as well as being partially protected from the wind. As well, the experimental site is sheltered from the prevailing winds by a stand of trees and a hill (Figure 4-2). This would suggest that the  $m^*$  value calculated can be considered as a conservative design value and values measured in full scale applications would likely be somewhat greater.

### Thawing Model

As there is no accepted model for sludge thawing, a regression analysis was conducted to develop a relationship of best fit between the independent variables of initial frozen depth ( $h_0$ ), degree days of thawing ( $T\Delta t$ ) and precipitation (PPT) and the dependent variable of thawing measured as the reduction in ice layer thickness. Of the 12 filters used in the study, 3 filters in 2010 and 1 filter in 2011 were removed as significant void spaces had formed

between ice layers due to a sludge layer freezing from the top while still draining from the bottom. An initial regression conducted on the entire data set (n=314) found that both precipitation and initial frozen depth were not significant ( $P>0.05$ ) and were thus removed from the model. In order to avoid the model being skewed toward filters with more data points, only the final depth and degree-days for each filter thawing experiment (n=20) was used to develop the final model. Initially, the 2010 and 2011 data sets were considered separately and the parameter coefficients determined in order to use the two the years of data separately to be able to validate the model. Using the 2010 data set (n=9),  $x_1=0.1430\pm0.0220$  cm/°C·d, while using the 2011 data set (n=11),  $x_2=0.1632\pm0.0142$  cm/°C·d. Since the 95% C.I of the parameter coefficients describe the same value, we can conclude that both data sets describe the same relationship and that the model is validated for the region of the pilot experiment. Ideally the experiment should be replicated in different regions and at different latitudes to rule out any significant effect of variables such as solar incidence or relative cloud cover.

The model was then calibrated using both years of data (n=20) and is described in Figure 4-10 and Equation 4.4. The model fits the data well ( $R^2=0.87$ ) with residuals evenly spaced about zero and provides a parameter coefficient of  $a=0.1579\pm0.0115$  cm/°C·d at the 95% C.I.

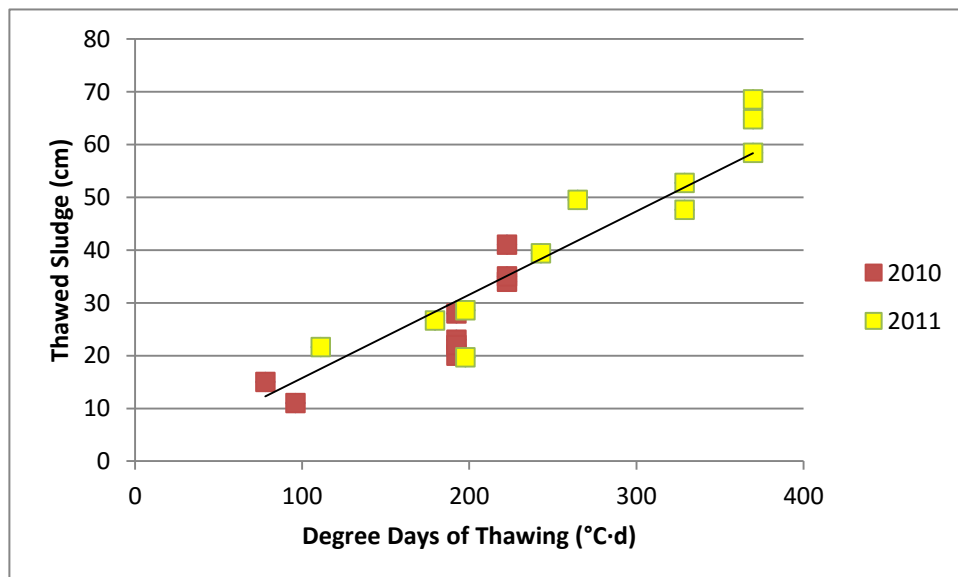


Figure 4-10. Degree Days of Thawing versus Thawed Sludge

$$\text{Thawed Sludge (cm)} = a \cdot T \Delta t \quad - \text{Equation 4.4}$$

As with the freezing model, a significance test and a model adequacy test were carried out on the fitted model (McLean and Burns, 2010) with the ANOVA described in Table 4-4. The Significance Test compares the ratio of MS regression and MS residuals to an F distribution. Under the null hypothesis the fitted model explains no variation in the data. Since  $F > F_{1, 19, 0.05}$  we reject the null hypothesis and conclude that the fitted model is significant. For the Lack of Fit test, under the null hypothesis there is no lack of fit. Since  $F < F_{10, 9, 0.05}$  we accept the null hypothesis and conclude that there is not a significant lack of fit to the model. This test indicates that the pure error significantly explains the variation in the model output. This stands to reason as the degree days of warming should be the predominant variable influencing the rate of melting of the frozen sludge.

**Table 4-4. ANOVA Table for Sludge Thawing Model**

	<i>df</i>	<i>SS</i>	<i>MS</i>	<i>F</i>
Regression	1	29667.41	29667.41	829.58
Residual	19	679.48	35.76	
Pure Error	9	168.70		
Lack of Fit	10			
Total	20	30346.89		

Since the rate of thawing is directly related to degree days of warming, Equation 4.4 can be easily used to predict how long it will take to thaw a known depth of sludge at a given location using temperature normals.

### Model Application

The amount of sludge frozen in a winter period can be easily calculated from Equation 6.3 using average monthly temperature normals for months with negative degree days for a given location, an assumed constant depth for each sludge dose and the calculated constant value of  $m^*=1.45 (\text{°C}\cdot\text{d})^{1/2}$ . At the same time, the maximum depth of frozen sludge that can be thawed can be calculated from Equation 4.4 using average monthly temperature normals for months of positive degree days. Using the limiting sludge depth either from freezing or thawing for locations throughout N. America, a map of iso-depth sludge freezing curves can be created and is depicted in Figure 6-10. Data points in black indicate regions where sludge freezing is controlling while data points in red indicate regions where sludge thawing is controlling. There would likely be little interest in applying the technology in regions where the seasonal frozen sludge depth is less than 1.0 m. As can be seen from Figure 4-11, this would exclude the coastal regions and Southern Ontario.

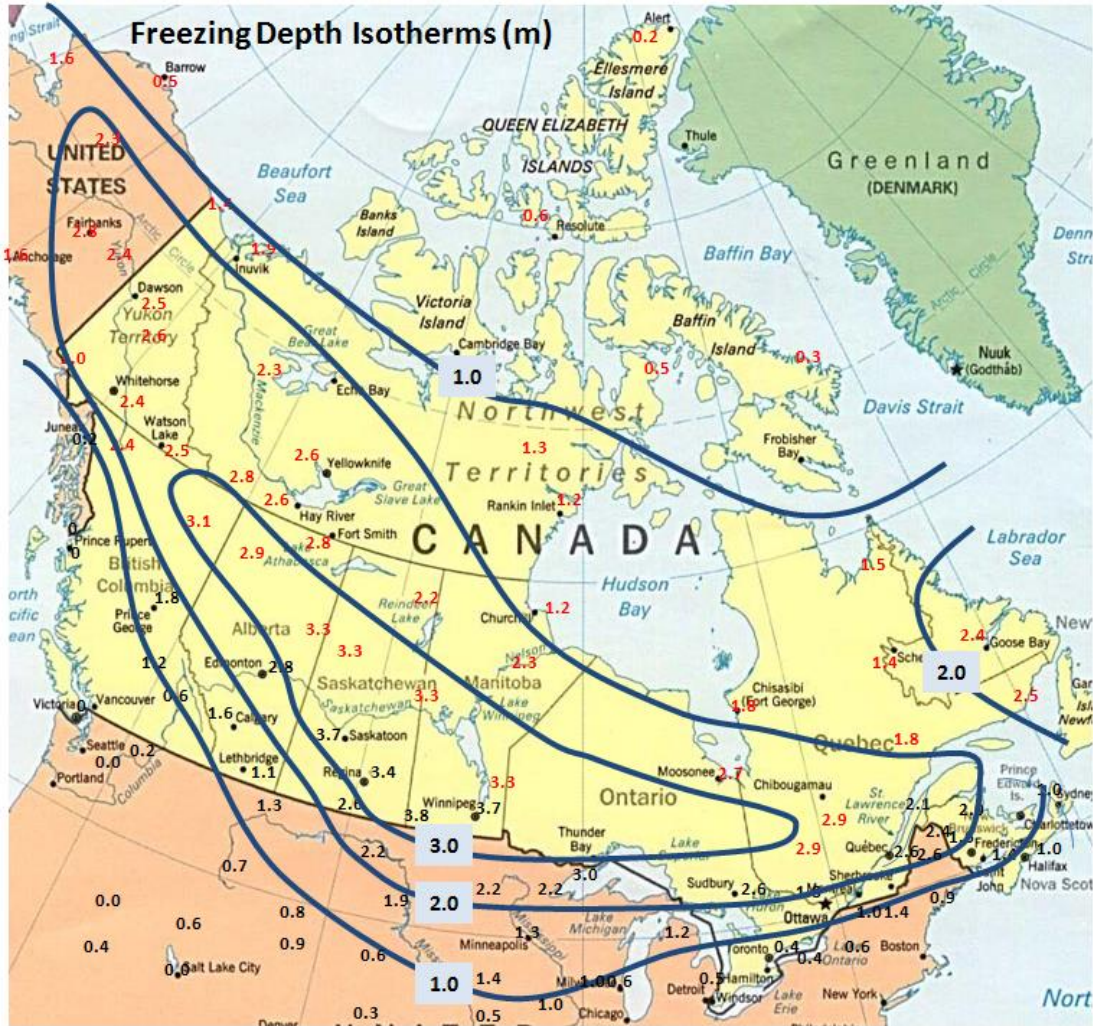


Figure 4-11. Iso-depth Sludge Freezing Curves for N. America

The thawing time dominates most of the North, where it will take the entire summer season to melt the frozen sludge and will thus necessitate parallel sludge drying beds to dewater any sludge produced during the spring to fall period. In regions where freezing dominates it will take a period of time in the spring to melt the frozen sludge: after which the freezing bed can be operated as a sludge drying bed for the remainder of the summer and fall. Either a storage lagoon or a dedicated drying bed will be required for the shoulder season. For example in the Ottawa region, where the pilot site is located, there is a potential to freeze 1.8 m of sludge in an average winter. Based on Equation 4.4, it would take 3 months to thaw the 1.8 m of sludge, requiring either 3 months of storage or a dedicated

drying bed for the April-June period, making the beds available for 5 months from July to November to operate as drying beds.

### Sludge Cake and Filtrate Quality

Filtrate and sludge quality were evaluated over the 2010 operating season. Average filtrate quality and removal rates are presented in Table 4-5. The sand filters are very effective at reducing pollution load in the septage with COD, BOD<sub>5</sub> and TSS levels reduced by 99% to produce a filtrate which is comparable to a low-strength domestic wastewater (Metcalf and Eddy, 2003) and easily treatable in any municipal or onsite wastewater treatment system. TKN-N levels are reduced by 88%; however, 85% of the filtrate TKN is in the form of ammonia. These results strongly suggest that the organic matter in septage is almost exclusively tied to the sludge solids and is readily removed through solid-liquid separation. As well, *E. coli* levels are reduced by 2 logs within the filters. This is also likely due to the reduction in solids which would harbour most of the bacteria.

**Table 4-5. Septage Treatment in Pilot Filters**

Parameter	Units	Raw Septage (Avg. ± STD)	Filtrate (Avg. ± STD)	Removal
COD	mg/L	28,031 ± 23,748	284 ± 39	99.0 %
BOD <sub>5</sub>	mg/L	8,565 ± 6,280	126 ± 108	98.5 %
TSS	mg/L	18,031 ± 21,744	82 ± 21	99.5 %
TKN-N	mg/L	500 ± 506	59 ± 19*	88.2 %
<i>E. coli</i>	CFU/100mL	3.9x10 <sup>7</sup>	4.0x10 <sup>5</sup>	2 log

\*NH<sub>3</sub>-N = 50 ± 23 mg/L

Sludge dry matter was measured bi-weekly from April 20 - June 27, 2010 to observe any trend in DM after the sludge had completely thawed. As can be seen in Figure 4-12, sludge DM appeared to increase from the beginning of May to the beginning of June, from 23 ± 3% to 28 ± 1%; however, the DM then returned to an average of 23-25% for the remainder of the June, likely due to the consistent rainfall encountered during the month. Overall, samples remained fairly constant with time with an average of 25 ± 1% DM across the 12 filters. These results are consistent with the literature (Reed *et al.*, 1986; Martel, 199) and

suggest that while it is possible to increase the sludge DM slightly with two to four weeks of drying, there is little to be gained if springtime rainfall is the norm.

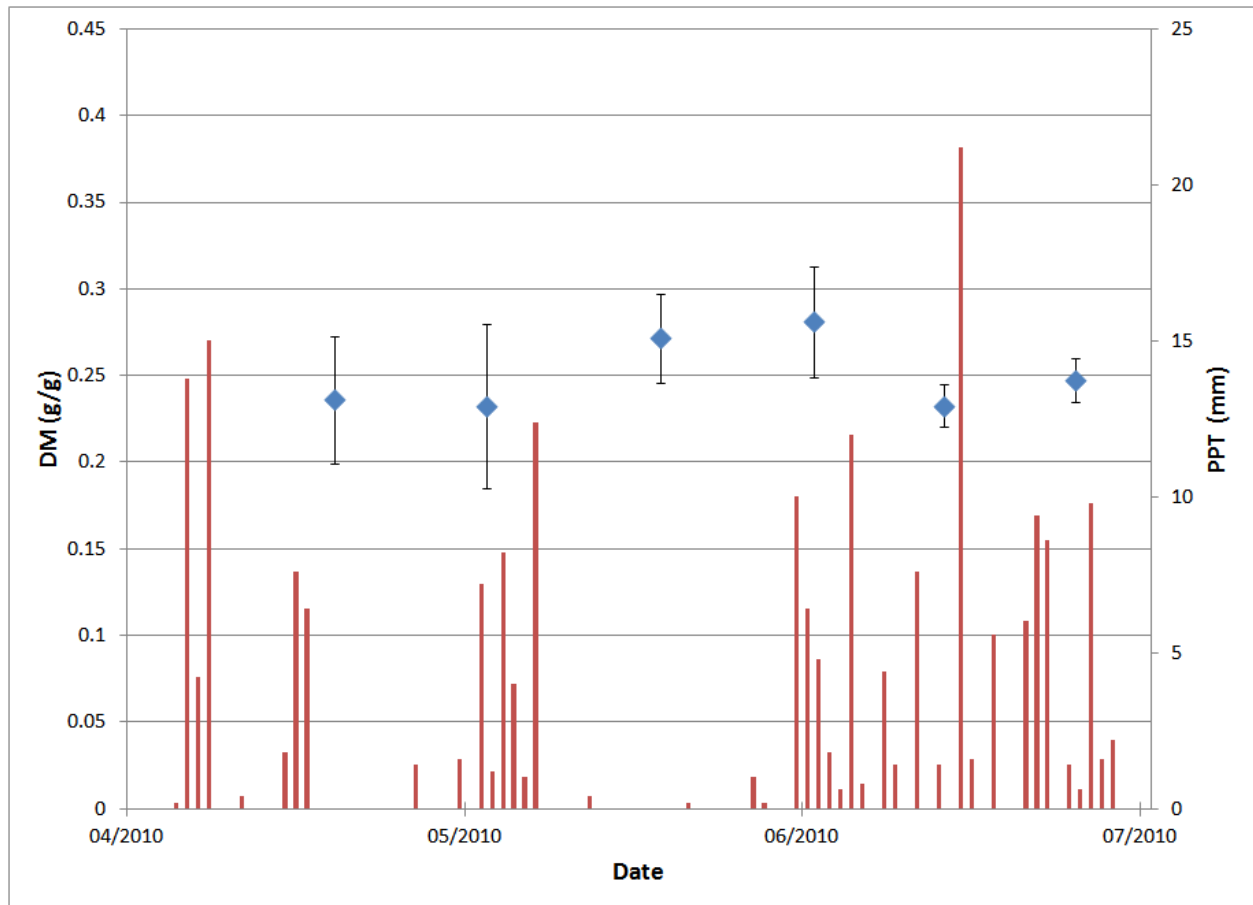


Figure 4-12. Dry Matter with Time (Average of 12 filters  $\pm$  95% C.I.)

DM between individual filters ranged considerably from  $20.5 \pm 2.2\%$  to  $33.1 \pm 3.8\%$ . The lower DM values in some filters are likely due to one or more layers of sludge not freezing completely before the next layer was applied; which would maintain more bound water within the sludge flocs and impede dewatering in the spring. These values also suggest that a DM content of 30% can be achieved if each layer of sludge is allowed to fully freeze before the next layer is applied. Importantly, even with imperfect freezing, DM of 23% is sufficient for sludge removal and transport and comparable to dewatering achievable by belt press or centrifugation (Wang *et al.*, 2007; Shammass and Wang, 2007).

Sludge *E. coli* numbers with time are described in Figure 4-13. The *E. coli* numbers declined by 1.6 logs from an initial value of  $2.0 \times 10^6$  CFU/g DM in April 2010 to an average

value of  $5.2 \times 10^4$  CFU/g DM in samples collected during May and June, 2010. The standard in most jurisdictions for the land application of biosolids is  $2.0 \times 10^6$  *E. coli* CFU/g DM (USEPA, 1994); therefore, the dewatered septage would be suitable for land application one month after the sludge has thawed.

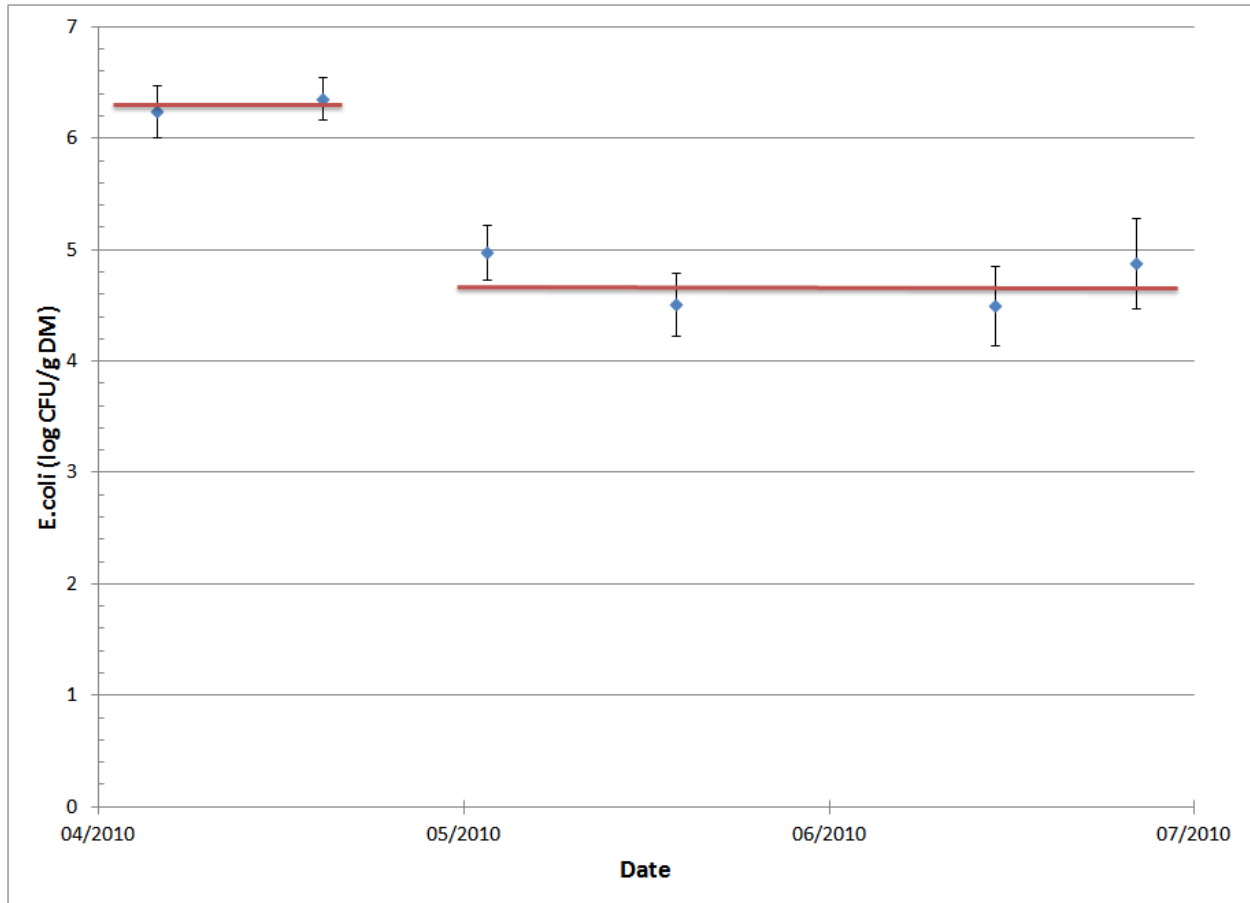


Figure 4-13. Sludge *E. coli* with Time (Average ± 95% C.I.). Red lines represent geometric means.

#### 4.4 Conclusions

This study attempts to model the freeze-thaw processes in uncovered freezing beds as it is hypothesised that the regular addition of fresh sludge will melt any accumulating snow in regions of moderate snowfall. A series of twelve pilot filters were studied over two winters with doses varying between 8 and 24 cm with seasonal snowfalls of 1.3 and 1.6 m observed. The freezing rate was successfully modeled following an accepted model for ice formation on water bodies:  $h = m * (\Delta T \cdot t - k \cdot T_0 \cdot h_0)^{1/2}$ , where  $h$  is the ice thickness in cm,  $\Delta T \cdot t$  are the degree days of freezing,  $k = 0.119 \frac{^{\circ}\text{C}\cdot\text{d}}{\text{cm}}$  is a constant to account for the

cooling of sludge to 0°C and  $m$  is the proportionality constant. No significant difference in the freezing rate was observed between filters with snow accumulation and filters with snow removed ( $P>0.05$ ). The value of  $m$  was found to be  $1.45\pm 0.09$  ( $^{\circ}\text{C}\cdot\text{d}$ )<sup>1/2</sup> at the 95% C.I. The fitted model is significant ( $R^2=0.80$ ); however, a lack of fit test determined that the variability in pure error replicates did not explain the model variability indicating that external variables such as wind speed and relative humidity play a significant role in the rate of freezing (USACE, 2002).

Bed thawing was modelled using a regression analysis. The independent variables of initial frozen depth and precipitation were found to be insignificant ( $P>0.05$ ) with degree days of warming controlling the rate of thawing. The linear model was validated by comparing the parameter coefficients derived from the 2010 and 2011 season data independently. Since the 95% C.I. of the parameter coefficients describe the same value, the model is validated for the region of the pilot experiment. The linear model, Thawed Sludge (cm) =  $a\cdot T\Delta t$ , fits the data well ( $R^2=0.87$ ) and provided a parameter coefficient of  $a=0.1579\pm 0.0115$  cm/ $^{\circ}\text{C}\cdot\text{d}$  at the 95% C.I. The fitted model is significant and there is not a significant lack of fit to the model, indicating that the pure error significantly explains the variation in the model output.

The sludge freezing and thawing models were applied to temperature normals throughout Canada and the United States at a sludge loading depth of 8 cm and a map of iso- sludge freezing curves was developed. Assuming a 1.0 m lower limit of freezing capacity, the freezing bed technology is largely applicable across Canada and parts of the Northern United States with the exception of Southern Ontario and the coastal regions. Annual loading rates will vary between 1.0-3.0 m depending on the location with the condition that annual snowfall must be equal to or below the sludge loading rate.

The filters were very effective at reducing pollution load in the septage with COD, BOD<sub>5</sub> and TSS levels reduced by 99% to produce a filtrate which is comparable to a low-strength domestic wastewater and easily treatable in any wastewater treatment plant or onsite wastewater system. *E. coli* numbers were also reduced by 2 logs within the filters. Sludge dry matter varied from  $20.5\pm 2.2$  to  $33.1\pm 3.8\%$  between filters, likely due to some filters

having layers of sludge which did not completely freeze. This range of dry matter is acceptable from a dewatering facility. After the melting period, dry matter did not increase significantly over time. However, sludge *E. coli* numbers did decline by 1.6 logs from an initial value of  $2.0 \times 10^6$  CFU/g DM within 1 month of the sludge having thawed; meeting the *E. coli* limit for land application.

Uncovered freezing beds appear to be an interesting and cost effective sludge dewatering and septage treatment alternative for cold climate regions with moderate snowfall. Depending on the application and region, a parallel sludge drying bed or reed bed filter could permit year-round sludge dewatering capability.

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## 5 Hydraulic Performance of a Combined Reed Bed and Freezing Bed Technology for Septage Dewatering in a Cold Climate

### 5.1 Abstract

The combined application of reed bed and freezing bed technology has been demonstrated to effectively dewater septage year-round under cold climate conditions in a 5-year field scale trial. Solid and hydraulic loading rates were varied from 43 to 147 kg TS/m<sup>2</sup>/y and 1.9 to 5.9 m/y to two 187 m<sup>2</sup> planted and one 187 m<sup>2</sup> unplanted system. Winter freeze-thaw conditioning was shown to consistently double filter drainage rates in spring compared with summer operating conditions at equivalent hydraulic head, indicating that freeze-thaw conditioning can restore bed hydraulic conductivity and mitigate risk of clogging. No significant effect on system drainage was observed between planted versus unplanted systems, between 7 versus 21 d dosing cycles or with solid loading rates between 49-144 kg TS/m<sup>2</sup>/y. However, drainage rates were shown to vary significantly with the hydraulic loading rate. A design loading rate of 2.9 m/y is recommended for septage treatment in reed bed systems operating in cold climates.

**Key Words:** sludge dewatering, reed bed, freezing bed, septage, cold climate

### 5.2 Introduction

The management of septage (accumulated solids in septic tanks) is a challenge facing many rural communities. Traditional approaches include land application or land application following lime stabilization to agricultural soils. However, land application is constrained by plant nutrient demand, plant growth stages and crop harvesting restrictions and is forbidden on saturated soils or during winter months (CCME, 2010; USEPA, 1994). Discharge to local wastewater treatment plants is often not an option due to capacity constraints as small town systems, often wastewater lagoons, are not designed to treat sludge. Therefore, alternative technologies are required. One promising technology is the reed bed system. Reed bed technology shares similar design features with sand drying beds with the key difference being that dewatered sludge is removed from drying beds

after each sludge application, whereas dewatered sludge is only removed from reed beds after 5-10 years of continuous operation, resulting in significantly lower operating costs (Kim and Smith, 1997). The plants are thought to provide a number of important functions including increased evapotranspiration, creation of drainage pathways in the sludge layer through rhizome development and wind induced swaying of plant stems as well as enhanced mineralisation (Nielson, 2003; Gagnon et al., 2013). Another technology based on sand drying bed design is the sludge freezing bed, where layers of sludge are progressively applied to the bed surface during winter and allowed to freeze. During the freezing process, colloidal solids are extruded along the ice crystal boundary and compressed into granules which easily separate as the sludge thaws in the spring (Martel, 1993). It is hypothesised that a reed bed system can be operated year-round by taking advantage of freeze-thaw conditioning during the winter and spring seasons.

Reed bed technology has been widely applied throughout Europe to dewater municipal waste activated sludge (WAS) as well as anaerobically digested (AD) sludge (Nielson, 2003; Barbieri *et al.*, 2003; Troesch *et al.*, 2009a) and to a lesser degree in the United States (Mellstrom and Jager, 1994; Kim and Smith, 1997), while several studies have evaluated septage dewatering in France (Paing and Voisin, 2005; Troesch *et al.*, 2009b; Vincent *et al.*, 2011) as well as in tropical climates (Koottatep *et al.*, 2005; Kengne *et al.*, 2008).

Areal solids loading rate (SLR) is generally used as the primary design parameter in system sizing and is based on empirical experience. Most systems in Europe are designed to process 60 or 50 kg total solids (TS) /m<sup>2</sup>/y treating WAS or digested sludge, respectively (Nielson, 2003). By means of comparison, design loading rates for sand drying beds range from 100 to 160 kg TS m<sup>-2</sup> yr<sup>-1</sup> for AD sludge (USEPA, 1987). Reed bed systems in the US report SLR for both aerobic and anaerobically digested sludges from 13-106 kg TS/m<sup>2</sup>/y (Kim and Smith, 1997), or from 18-82 kg TS/m<sup>2</sup>/y with an average of 41.6 kg TS/m<sup>2</sup>/y (Mellstrom and Jager, 1994). Two full scale reed bed systems in France dewatering septage report loading rates of 46 and 109 kg total suspended solids (TSS) m<sup>2</sup>/y as there are considerable dissolved solids in septage (Paing and Voisin, 2005). Septage reed bed systems have also been applied at the pilot scale in several tropical countries at much higher loading rates of 250 kg TS/m<sup>2</sup>/y (Koottatep *et al.*, 2005) and 100-300 kg TS/m<sup>2</sup>/y

(Kengne *et al.*, 2008). To our knowledge, the effect of hydraulic loading rate (HLR) on system performance has not been investigated.

The rest period between reed bed dosing is important to allow sufficient time for sludge dewatering and mineralization. Rest periods reported in the literature vary widely, from a minimum of 7 days (Burgoon *et al.*, 1997; Koottatep *et al.*, 2005; Kengne *et al.*, 2008), with several studies suggesting 14-21 days be used to allow for sludge mineralisation (Mellstrom and Jager, 1994; Barbieri *et al.*, 2003; Troesch *et al.*, 2009b) or even longer at 35 days (Paing and Voisin, 2005) and 50-60 days (Nielson, 2003).

Reed bed systems have not been designed to explicitly take advantage of freeze-thaw conditioning during the winter/spring, although several US reed bed systems have reported to be operating during the winter (Messtrom and Jager, 1994; Kim and Smith, 1997). However, sludge freezing beds (operating solely in winter) have been demonstrated to effectively dewater a variety of sludges including WAS, AD sludge, alum sludge and aerated lagoon sludge (Reed *et al.*, 1986; Martel, 1989; Martel and Diener, 1991; Martel, 1993; Desjardins and Brière, 1996; Hellstrom and Kvarnstrom, 1997; Martel, 1999) as well as septage (Kinsley *et al.*, 2012). It is suggested that sludge be applied in 8-10 cm layers and allowed to fully freeze before the next layer is applied (Reed *et al.*, 1986; Martel, 1989). The operating season and loading frequency varies with climatic zone with winter loading ranging from 1.0 to 3.0 m from the mid-western U.S. to northern Ontario, Canada (Kinsley *et al.*, 2012).

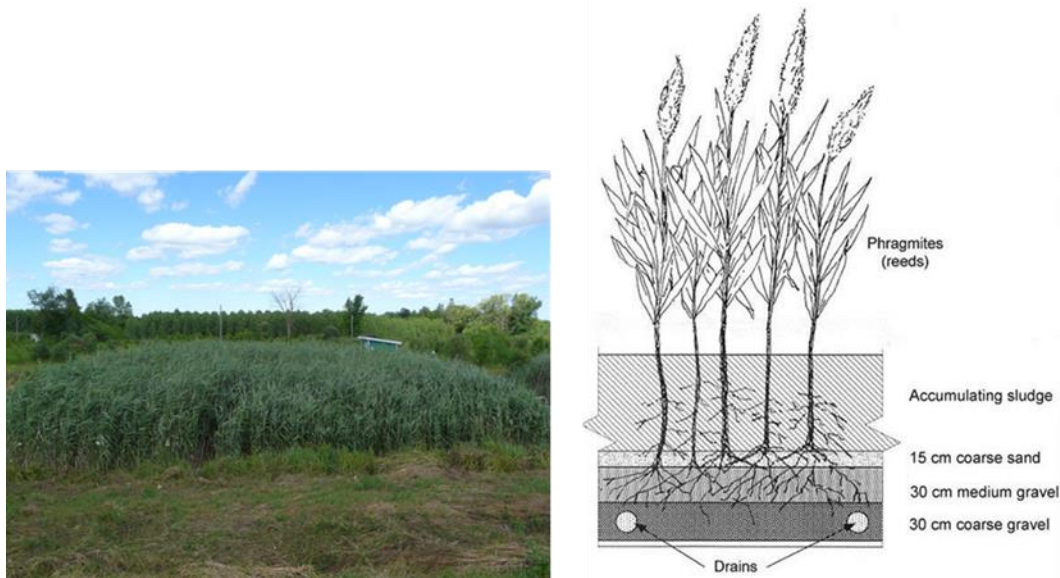
The objectives of this study are to test the hypotheses that integrating reed bed and freezing bed technology can effectively treat septage year-round in cold climates and that *Phragmites* play an essential role in maintaining system drainage during the growing season. Additionally, this study aims to propose system design criteria for application in cold climate regions by determining the impact of areal solid and hydraulic loading rates as well as dose / rest period cycle on system drainage.

### **5.3 Materials and Methods**

Two 187 m<sup>2</sup> reed bed systems (RB1 and RB2) and one 187 m<sup>2</sup> non-planted sand drying bed (SF) were constructed at the septage lagoon of René Goulet Septic Tank Pumping, Green

Valley, ON, Canada (45.32°N 74.64°W). The study site is located between Ottawa, ON and Montreal, QC. Average monthly 25-year temperature climate normals vary from -10.8°C in January to 20.9°C in July, with average temperatures remaining below the freezing point from December through March.

The system design was based upon recommended specifications for sand drying beds (Wang *et al.*, 2007). A cross sectional schematic and photo of the system is presented in Figure 5-1.



**Figure 5-1. Photo of RB1 in Year 5 and Schematic of Reed Bed Filter**

The cross section of each system, from bottom to top consists of: a 6.4 mm non-woven geotextile to protect the geomembrane, a 30 mil geomembrane (Layfield Tantalum 5-30 mil), a 0.30 m layer of washed coarse gravel (20-40 mm dia.), a 0.30 m layer of washed fine gravel (5-10 mm dia.), and a 0.15 m layer of locally available concrete sand ( $D_{10} = 0.18$  mm; Coef. of Uniformity = 3.4; 2.1% fines). Berms were constructed around each system to achieve 2.0 m of freeboard above the system surface to contain the increasing sludge layer over time as well as frozen sludge accumulation during the winter months. The effluent drainage system consists of 9 lines of 10 cm perforated PVC pipe at 1.5 m spacing laid in the coarse gravel layer with a 1% slope to a collector pipe at the toe of each bed. Aeration standpipes were connected to the drainage network at the ends and middle of each

drainage line. The reed beds were planted with native *Phragmites* harvested from nearby ditches. Initially, rhizomes were planted in RB1 and RB2 at 4 rhizomes m<sup>-2</sup>; however, few survived Year 1 and clumps of reeds were excavated and placed directly in the systems in Year 2, which survived and propagated over the following years. Filtrate from each system was collected in a 1.3 m<sup>3</sup> pump chamber and was pumped into an existing lagoon with Myers WR10H-21 1HP pumps. The systems were dosed directly from the vacuum truck onto a splash plate after passing through a 1.0 cm bar screen to remove large non-biodegradable objects.

Information on each truckload of septage was recorded by the vacuum truck operator including: date of last septic tank cleaning, volume, source (holding tank, septic tank) and which system was dosed. Septage was collected mostly from residential houses; however, other sources of waste included schools, retirement homes and restaurants. No industrial waste, food processing waste or portable toilet waste was dosed into the systems. Septage characteristics are described in Table 5-1. As can be observed in the table, average septage values are comparable to typical literature values. Septage will have much higher solids content than waste activated sludge (WAS) but similar solid content to digested sludge (Metcalf and Eddy, 1991). It is expected that there will be much greater heterogeneity in septage quality as compared to both WAS and digested sludge due to multiple sources, varying inputs and pump-out frequencies.

**Table 5-1. Septage Characteristics**

<b>Parameter</b>	<b>Units</b>	<b>Avg. <math>\pm</math> 95% C.I.<sup>a</sup></b>	<b>n</b>	<b>Typical Septage (USEPA, 1984)</b>
TS	g/L	25.8 $\pm$ 3.4	191	34.1
VS	g/L	17.0 $\pm$ 2.2	191	23.1
TSS	g/L	19.5 $\pm$ 2.7	179	12.9
COD	g/L	27.0 $\pm$ 4.1	187	31.9
BOD <sub>5</sub>	g/L	6.4 $\pm$ 1.0	187	6.5
TKN	mg/L	750 $\pm$ 89	195	588
NH <sub>4</sub> <sup>+</sup> -N	mg/L	127 $\pm$ 11	115	97
NO <sub>3</sub> <sup>-</sup> -N	mg/L	0.9 $\pm$ 0.1	115	-
TP	mg/L	265 $\pm$ 43	187	210
PO <sub>4</sub> <sup>3-</sup>	mg/L	29.9 $\pm$ 3.5	117	-
Conductivity	$\mu$ s/cm <sup>1</sup>	4,242 $\pm$ 754	111	-
pH	-	7.4 $\pm$ 0.1	111	6.9
Alkalinity	mg/L as CaCO <sub>3</sub>	895 $\pm$ 62	111	970

<sup>a</sup> C.I. = confidence interval

Climatic data was obtained from AgriCorp rain gauges in the vicinity of the pilot site (four within a 30 km radius) and from the Environment Canada weather station at the Ottawa International airport.

The drainage flow from each bed was measured through pump run-time loggers (OMEGA OM-CP series). Pump run time and flow rates were calibrated periodically.

Representative 2 L septage samples were collected from each truck load and stored in a sample fridge located on site. All analyses were conducted at the Ontario Rural Wastewater Centre water quality lab or at the Ontario Ministry of Environment Laboratory following Standard Methods (APHA, 2005) or EPA methods (SCC, 2013). Sludge depth was monitored over time and percent plant coverage in each system was recorded annually.

### 5.3.1 Experimental Design

The systems were operated as reed beds from May to December with scheduled dosing and as freezing beds during the winter months, where a new dose of sludge was applied once the previous dose had frozen. During Years 1 and 2, HLR and dosing cycle (7-d) were maintained constant, while SLR was varied. During Year 3, the dosing cycle was changed from 7-d to 21-d while HLR remained constant and the effect of plant development compared. During Years 4 and 5, HLR was varied to the systems. Differences in weekly average drainage flow were compared between systems within the same year using a Paired-T test to account for seasonal variability and between years using a Single Factor Anova. Annual loading rates applied to each system are presented in Table 5-2.

**Table 5-2. Annual Solid and Hydraulic Loading Rates by Calendar Year**

Year	SF		RB1		RB2	
	SLR	HLR	SLR	HLR	SLR	HLR
	(kg/m <sup>2</sup> /y)	(m/y)	(kg/m <sup>2</sup> /y)	(m/y)	(kg/m <sup>2</sup> /y)	(m/y)
2007	142	3.2	144	3.3	113	3.4
2008	91	2.4	75	2.8	49	2.8
2009	88	3.6	91	3.1	74	3.4
2010	147	5.9	43	1.9	81	3.5
2011	54	2.4	58	2.6	110	4.6
Avg.	104	3.5	82	2.7	85	3.5

## 5.4 Results and Discussion

### Water Balance

Annual water balances were conducted on each system to determine losses due to evapotranspiration (ET) as the plants developed and the systems matured as well as to determine ponding water in the systems. The water balance equation is presented below:

$$ET = \text{Septage} + PPT - \text{Drainage} - \text{Bound Water} - \text{Ponding} \quad (\text{Equation 5.1}).$$

Where:

ET = Evapotranspiration (mm)

Septage = Volume of septage dosed to each system (mm)

PPT = Precipitation measured from the nearest weather station(s) x 1.18 to account for berm slopes (mm)

Drainage = Filtrate flow calculated from calibrated pump runtime loggers (mm)

Bound Water = Long term storage of bound water in the dewatered sludge. Calculated from the 5-year sludge accumulation rate and average dry matter (DM) content of 23% in dewatered sludge (mm)

Ponding = Excess free water in system (mm)

On an annual basis Equation 5-1 was solved for ET with annual ponding determined by measuring the total depth of sludge at the end of each year and subtracting Bound Water. Yearly ET was then apportioned monthly based on proportional values from a wetland ET study conducted in close proximity to the study site (Lafleur *et al.*, 2005). Weekly ponding was then calculated using Equation 5.1.

Annual ET was fairly constant for each of the 3 filters at  $660 \pm 42$ ,  $510 \pm 42$  and  $680 \pm 45$  mm/y for SF, RB1 and RB2, respectively. The ET calculation will incorporate any errors in measurements from any of the factors in the equation including: septage loading, precipitation, estimation of bound water and pump runtime flow calibration. These errors could explain the difference observed between RB1 and the other two filters. No trends over time were observed even though plant development increased over the course of the study (see Figure 5-2); suggesting that the plant cover does not significantly contribute to net evapo-transpiration. This is reasonable, as plants will increase bed transpiration but also provide shade, increase humidity and reduce wind velocity which will reduce bed evaporation (Kadlec and Wallace, 2009).

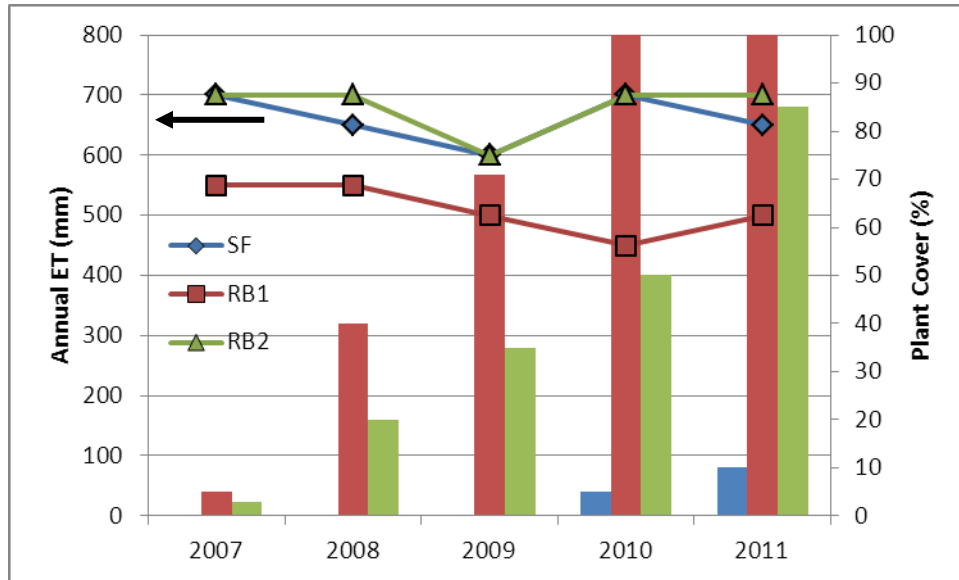


Figure 5-2. Annual ET and Percent Plant Cover. ET was calculated by conducting an annual water balance on each filter using Equation 5.1. Percent plant cover was visually recorded in October of each year.

In this study average ET varied between 510 and 680 mm/y. These values are compared in Table 3 with literature values from multiyear ET wetland studies conducted in similar climatic zones. The results from this study were similar to values measured in a natural wetland in England, UK (Fermor et al., 2001) and in a Southern Ontario lake (Yao *et al.*, 2009) and somewhat higher than values reported in a natural peatland bog in Eastern Ontario (Lafleur et al., 2005), in close proximity to the study site. Several authors report rough equality between lake and wetland ET as reported by Kadlec and Wallace (2009). However, two studies of natural wetlands in Northern Germany (Herbst and Kappen, 1999) and Hungary (Anda et al, 2015) reported ET values between 1.2 – 1.9 times the annual ET measured in this study. The ET values measured in this study are largely consistent with literature values, although no comparable study of sludge dewatering reed beds could be found. The study ET values are lower than the 30-year precipitation normal of 943 mm/y for the study area (Environment Canada, 2016), which would need to be taken into account when designing a filtrate treatment or storage system.

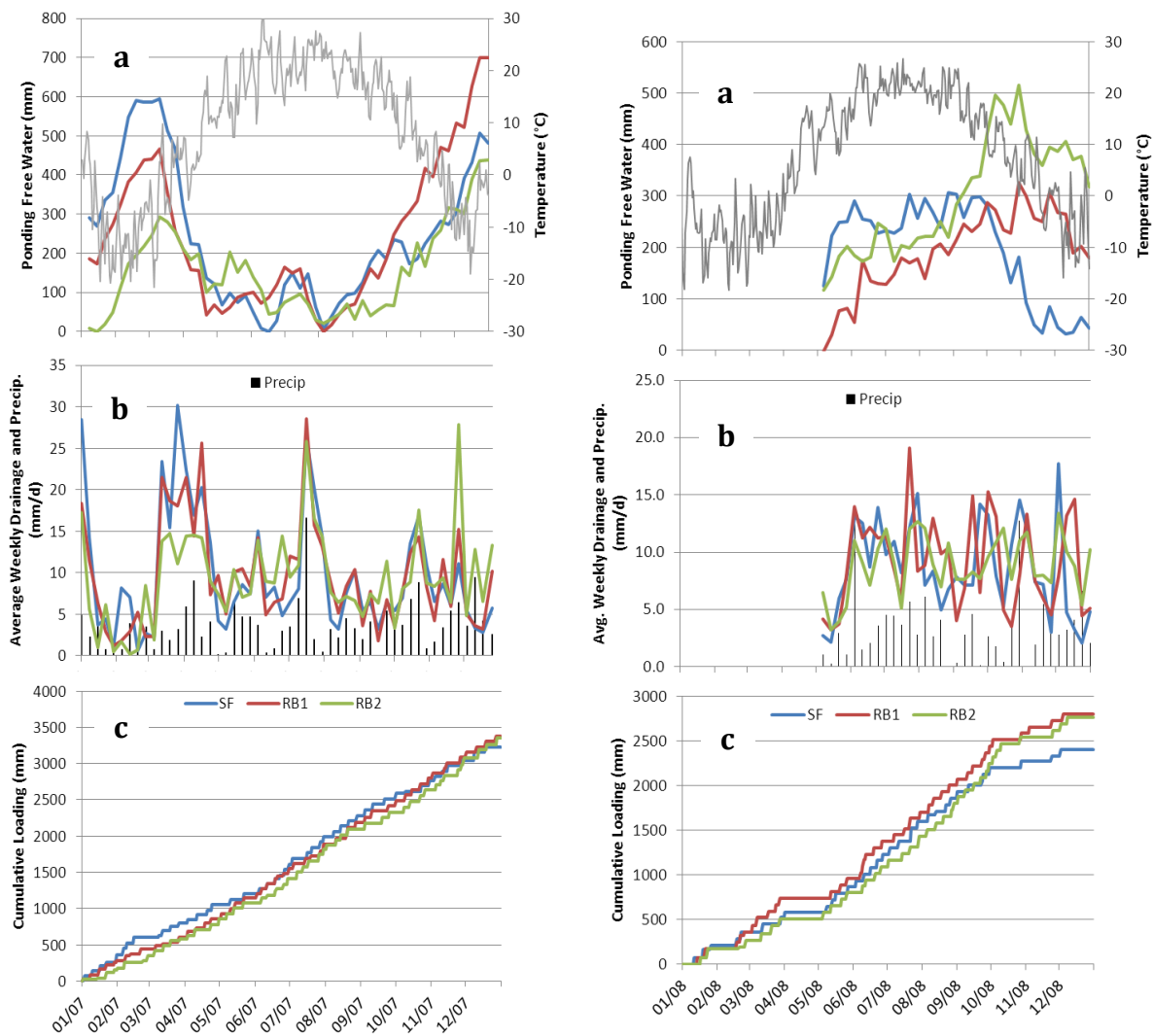
**Table 5-3. Annual Evapo-transpiration Rates of Wetlands in Temperate Climates**

Location	Latitude	Wetland type	Method	Study Duration (y)	Annual ET (mm/y)	Reference
Poland	46°39'N	Natural wetland (Phragmites)	Penman-Monteith	16	877 ± 55	Anda et al. (2015)
Northern Germany	54°06'N	natural wetland ( <i>Phragmites</i> )	S-W model	4	840-1314	Herbst and Kappen (1999)
England, UK	-	natural wetland ( <i>Phragmites</i> )	Phytometer	4	548-840	Fermor et al. (2001)
Ottawa, ON, Canada	45°24'N	natural shrub bog peatland	Eddy covariance	5	392-523	Lafleur et al. (2005)
Blue Chalk L., ON, Canada	45°11'N	lake evaporation	Energy budget	25	426-663	Yao et al. (2009)
Alexandria, ON, Canada	45°19'N	sludge reed and sand drying beds ( <i>Phragmites</i> )	Water balance	5	510-680	this study

#### 5.4.1 Solid Loading Rate (Years 1 and 2)

The water balances for the three systems in Years 1 and 2 are presented in Figure 5-3. In Year 1 the hydraulic loading to the 3 systems was maintained constant at 3.2, 3.4 and 3.4 m/y, while high to medium solid loading rates were applied at 142, 144 and 113 kg TS/m<sup>2</sup>/y to SF, RB1 and RB2, respectively. To vary solids loading, septage loads were selected with higher percent solids; typically from households which had not had their tanks pumped within the previous 5 years. Very little plant establishment had occurred during 2007; therefore, the three filters effectively represented sand drying beds during the first year of operation. Ponding water increased in the filters during the winter months as frozen layers of sludge accumulated in the filters with correspondingly low drainage. As average daily temperatures increased above zero by mid-March, drainage increased with the thawing sludge and the ponding water level correspondingly dropped, draining the accumulated sludge layer. During the growing season (May-November), drainage averaged 8.8±5.4, 9.4±5.1 and 9.8±4.6 mm/d, for SF, RB1 and RB2, respectively. No significant differences between the three drainage rates were observed (P>0.1), suggesting that varying solids loading rates between 113 and 144 kg TS/m<sup>2</sup>/y had no significant impact on filter drainage

during the growing season; however, ponding water increased in all three filters during the fall, indicating that the hydraulic loading rate exceeded the filters' drainage capacity.



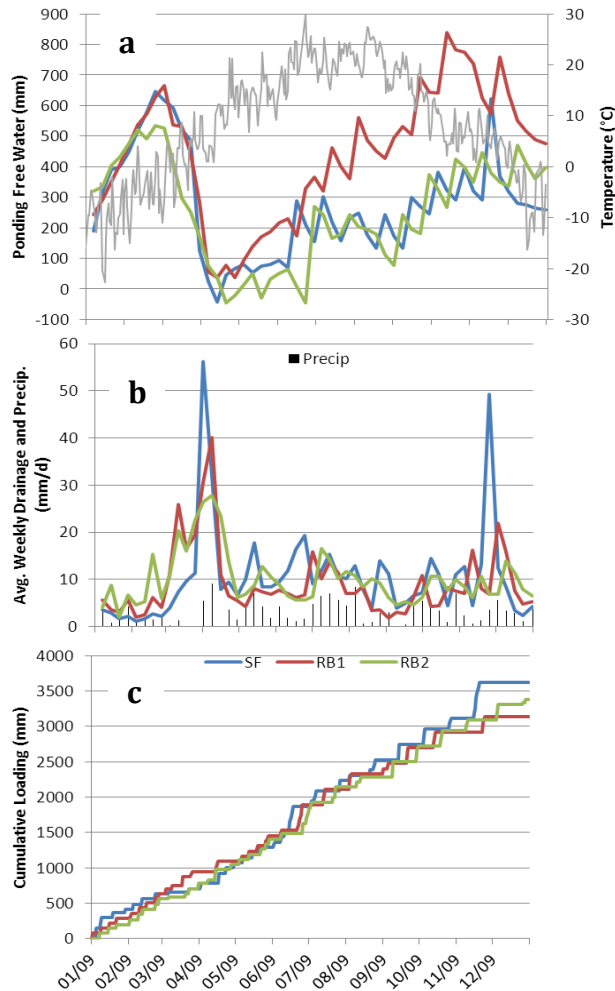
**Figure 5-3. Water Balance for Sand and Reed Bed Filters during Years 1 and 2 [a) ponding water and temperature; b) drainage and precipitation c) cumulative hydraulic loading]. Data is presented on a weekly basis. Outflow data was lost from Jan-May in 2008 due to an error with the data logger.**

The experiment was repeated in Year 2 with lower hydraulic loading rates of 2.4, 2.8 and 2.8 m/y and solid loading rates of 91, 75 and 49 kg TS/m<sup>2</sup>/y for SF, RB1 and RB2, respectively. During the growing season, drainage rates averaged 9.2±3.9, 9.0±4.0 and 8.9±2.6 mm/d, for SF, RB1 and RB2, respectively, with no significant differences observed between the three filters (P>0.1). This again suggests that varying SLR does not impact the drainage rate. Ponding increased somewhat in RB1 and RB2, while ponding declined for

SF, reflecting the lower hydraulic loading rate to this filter during the fall months. Lower hydraulic loading rates in 2008 compared with 2007 resulted in less ponding during the fall months. Plant coverage in RB1 and RB2 had reached 40 and 20 percent, respectively, with no obvious effect on filter drainage observed.

#### **5.4.2 Plant Development (Year 3)**

In Year 3 the hydraulic and solids loading rates to the 3 systems were maintained fairly constant at 3.6, 3.1 and 3.4 m/y and 88, 91 and 74 kg TS/m<sup>2</sup>/y for SF, RB1 and RB2, respectively. Plant coverage in RB1 and RB2 had reached 70 and 35%, respectively, while SF remained without plant cover. As well, in mid-June the loading frequency was changed from one load per week to three loads every third week. The water balance for the 3 filters during Year 3 is presented in Figure 5-4. The increased loading to SF observed in November resulted from 9 loads being dosed to the filter in error and resulted in the spike in the drainage observed.



**Figure 5-4. Water Balance for Sand and Reed Bed Filters during Year 3 [a) ponding water and temperature; b) drainage and precipitation c) cumulative hydraulic loading]. Data is presented on a weekly basis.**

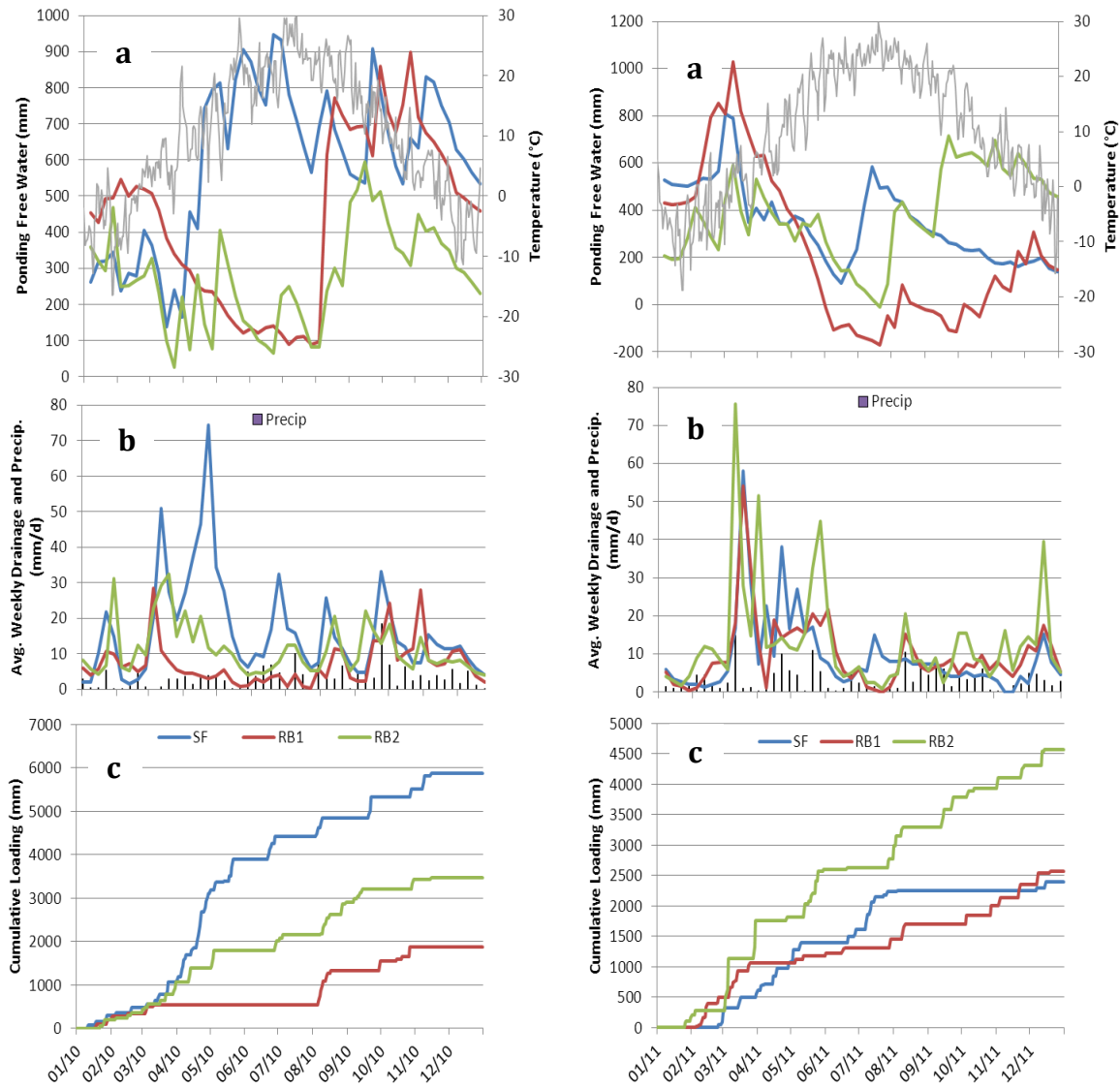
Ponding water increased during the winter months as frozen layers of sludge accumulated along with corresponding low drainage. As average daily temperature increased above zero by early March, drainage increased as the frozen sludge thawed and the ponding water levels correspondingly dropped. During the growing season drainage averaged  $10.4 \pm 4.1$ ,  $7.4 \pm 3.6$  and  $8.8 \pm 3.0$  mm/d for SF, RB1 and RB2, respectively. Ponding increased for all three filters, with RB1 exhibiting the highest increase in ponded water even though SF received the most septage. Drainage from SF was significantly higher than RB1 ( $P < 0.05$ ), corresponding to the higher hydraulic loading to SF; however, no effect of plant development on drainage rates was observed. These results indicate that plants played no significant role in maintaining filter drainage under these experimental conditions.

Possible explanations for these observations are that septage, due to its high sludge age and lack of easily degradable volatile organics, does not exhibit the same clogging potential as other biological sludges or that freeze-thaw conditioning each winter acts to restore and maintain bed permeability.

Average growing season drainage rates were compared between Year 1 and Year 3 to observe the effect of dosing frequency (7 vs 21 days) on drainage with equivalent hydraulic loading rates. No significant differences were observed ( $P>0.1$ ); suggesting that either dosing frequency is appropriate for filter operation.

#### **5.4.3 Hydraulic Loading Rate (Years 4 and 5)**

The effect of hydraulic loading rate on filter drainage was explored in Years 4 and 5 and is presented in Figure 5.5. In Year 4 the hydraulic loading to the 3 systems was varied, with RB1 left to rest for five months. The filters were dosed 5.9, 1.9 and 3.5 m/y while solids loading rates were 148, 43 and 81 kg/m<sup>2</sup>/y for SF, RB1 and RB2, respectively.



**Figure 5-5. Water Balance for Sand and Reed Bed Filters during Years 4 and 5 [a] ponding water and temperature; b) drainage and precipitation c) cumulative hydraulic loading]. Data is presented on a weekly basis.**

The level of ponding in the filters remained stable over the winter as drainage continued from the bottom of the filters, likely due to the mild winter. However, as average daily temperatures increased above zero by early March, drainage increased with the thawing sludge and the ponding water level correspondingly dropped. During the growing season drainage averaged  $14.6 \pm 8.7$ ,  $6.8 \pm 6.7$  and  $9.9 \pm 4.8$  mm/d for SF, RB1 and RB2, respectively, varying significantly with hydraulic loading rate ( $P < 0.05$ ). In particular, SF responded very well to the very high hydraulic loading rates of 5.9 m/y with only a net increase of 300 mm

of ponded water over the year. This indicates that the sand filter technology is capable of processing up to twice the average hydraulic loading rate without signs of clogging.

In Year 5 the hydraulic loading to the 3 systems was also varied. The filters were dosed 2.4, 2.6 and 4.6 m/y while solids loading rates varied between 54, 58 and 110 kg/m<sup>2</sup>/y for SF, RB1 and RB2, respectively. Ponding water increased over the winter months and drained in the spring once daily temperature increased to above zero. During the growing season drainage averaged 7.3±5.4, 8.7±5.7 and 11.5±10.3 mm/d for SF, RB1 and RB2, respectively. Drainage was significantly higher for RB2 than both SF and RB1 (P<0.05) due to a higher hydraulic loading rate. In particular, RB1 responded very well to the very high hydraulic loading rates of 4.6 m/y with only a net increase of 250 mm of ponded water over the year. This would indicate that the reed bed technology is capable of processing up to 1.5 times the average hydraulic loading rate without signs of clogging. The negative ponding values observed in RB1 represents a decline in bound water due to evaporation.

#### 5.4.4 Freeze-thaw Conditioning

Average drainage rates were calculated during the spring melt period (typically from March to April) and were compared with drainage rates from June – November. As can be observed in Figure 5-6, drainage was consistently higher during the spring thaw compared with the growing season. This is in keeping with the physical action of freeze-thaw solid-liquid separation, where small colloidal particles are compressed into granules during ice crystal formation, resulting in a porous media after the sludge has thawed (Martel, 1993). Drainage rates were also shown to increase with increased levels of ponding water, due to increased hydraulic head, with drainage rates on average 2.2 to 2.6 times higher during spring thaw than from June to November at equivalent levels of ponded water. May drainage data varied between the two trend lines, suggesting that the positive effect on filter drainage from freeze-thaw conditioning diminished over time as new septage was added and macropores were filled in with colloidal and supracolloidal sludge particulates. Throughout the five year experiment, it could be observed that freeze-thaw conditioning increased the hydraulic conductivity each spring in all three filters. This annual rejuvenation of filter hydraulic conductivity could mask any positive effects the plants may play in maintaining filter drainage.

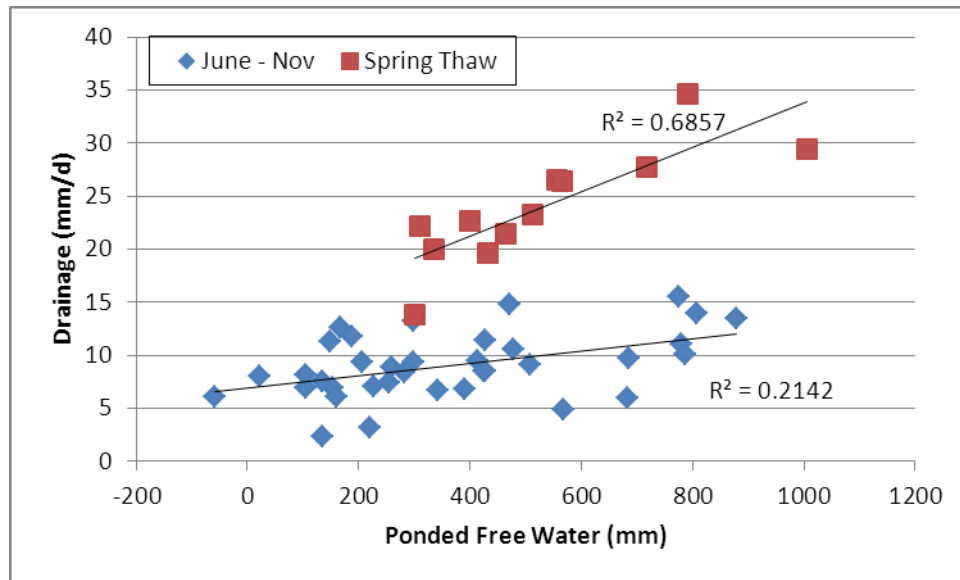


Figure 5-6. Drainage versus Poned Free Water during Spring Thaw (March-April) and Growing Season (June to November). Each data point is a 2 month average from each of 3 filters over the 5 year study period (excluding 2008 were flow data was lost).

#### 5.4.5 Design Hydraulic Loading Rate

Throughout the five year study, ponding water often increased throughout the growing season, which is not ideal as it could impact plant growth by submersing new plants, create odour issues and reduce aerobic stabilization of the sludge. As well, the risk of filter failure increases if the system water balance is reliant on freeze-thaw conditioning to drain excess ponding water from the previous year. Therefore, an optimum design hydraulic loading rate would ensure that no significant ponding occurs during the growing season. This rate was calculated using Equation 5.2 for each growing season (May-November) during the first three years of the study, when consistent dosing was applied, with the assumption that reduced loading will not significantly affect drainage rates. The results are presented in Table 5-4.

Design Hydraulic Loading Rate (mm/d) = [Total Volume Dosed (mm) – Poned Water (May-November) (mm)] / #days – (Equation 5.2)

**Table 5-4. Design Hydraulic Loading Rate by Year and Filter**

Year	Design Hydraulic Loading Rate (mm/d)		
	SF	RB1	RB2
2007	8.4	8.5	9.5
2008	8.4	8.2	8.8
2009	10.8	7.1	8.1
Avg. ± SD	9.2±1.4	7.9±0.7	8.8±0.7

A design hydraulic loading rate was determined to be  $8.6 \pm 0.7$  mm/d (Avg $\pm$ 95% C.I.). It is recommended to use the lower confidence interval value of 7.9 mm/d for design purposes. This converts to an annual hydraulic loading rate 2.9 m/y or an annual solid loading rate of 75 kg TS/m<sup>2</sup>/y at an average solids content of 2.6%. This recommended rate is higher than the 50-60 kg TS/m<sup>2</sup>/y recommended by Neilson (2003) for digested WAS or WAS, but within the ranges reported by other authors (Mellstrom and Jager, 1994; Kim and Smith, 1997; Paing and Voisin, 2005).

#### 5.4.6 Sludge Accumulation

Sludge levels fluctuated considerably during operating periods as frozen sludge accumulated during the winter and drained in the spring or as dosing and draining cycles occur. Sludge depths were measured after 5 years following a rest period to allow for any ponded water to drain. Specific sludge accumulation based on both hydraulic and solid loading are presented in Table 5.5. Sludge accumulation is consistent between the three filters with average rates of  $5.7 \pm 0.4$  cm/m of septage applied or  $0.21 \pm 0.01$  cm/kgTS·m<sup>2</sup> of septage applied. The upper 95% confidence interval (C.I.) should be used when designing freeboard to store the accumulated sludge. Additional freeboard should also be included in the design to accommodate the accumulated layers of frozen sludge over the winter period.

Table 5-5. Specific Sludge Accumulation

Filter	Sludge Accumulation after 5 Years (cm)	Specific Sludge Accumulation	
		Hydraulic basis (cm/m septage applied)	Mass basis (cm/kg·m <sup>2</sup> TS applied)
SF	103	5.9	0.20
RB1	80	5.8	0.20
RB2	94	5.3	0.22
Avg. ± 95% C.I.	-	5.7 ± 0.4	0.21 ± 0.01

## 5.5 Conclusions

Two 187 m<sup>2</sup> reed bed systems and one 187 m<sup>2</sup> unplanted sand filter system were evaluated under varying loading conditions in a controlled 5 year study carried out in a cold climate. The combined application of reed bed and freezing bed technologies has been demonstrated to effectively dewater septage year-round under Canadian climatic conditions. Bed evapotranspiration was found to be similar to natural wetland systems in temperate climates and somewhat higher than lake evaporation for both planted and unplanted systems. Sludge freeze-thaw conditioning consistently doubled drainage rates during spring thaw compared with the growing season at equivalent head, indicating that freeze-thaw conditioning increased filter hydraulic conductivity. No significant differences in drainage rates between the planted and unplanted filters were observed at equivalent hydraulic loading rates, suggesting that planting *Phragmites* in the filter beds did not play a significant role in maintaining filter drainage under the operating conditions of this study. Drainage rates varied significantly with hydraulic loading rate but not with solid loading rate suggesting that it is more appropriate to use the hydraulic loading rate when designing septage reed bed systems. Changing the dosing frequency from 7 to 21 days had no significant effect on drainage rates, suggesting that either dosing frequency is appropriate for filter operation. It is recommended to use a design hydraulic loading rate of 2.9 m/y for the sizing of combined reed bed and freezing bed systems treating septage in cold climate applications.

## 5.6 References

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## 6 A Combined Reed Bed / Freezing Bed Technology for Septage Treatment and Reuse in Cold Climate Regions

### 6.1 Abstract

A combined reed bed-freezing bed technology was effective at treating septage under Canadian climatic conditions over a 5 year period with average loading rates of 82-104 kg TS/m<sup>2</sup>/y. Varying hydraulic and solid loading rates as well as the increasing sludge cake with time had little to no effect on treatment efficiency with almost complete removal of organic matter, solids, heavy metals and nutrients. Filtrate concentrations varied significantly between the freeze-thaw and growing seasons for many parameters, although the differences were not important from a treatment or reuse perspective with filtrate quality similar to a low to medium strength domestic wastewater. The potential to reuse the filtrate as a source of irrigation water will depend upon local regulations; however, filtrate metal concentrations remained well below irrigation guideline limits. The dewatered sludge cake consistently met biosolids land application standards in terms of pathogen and metals content, with *E. coli* numbers declining with time as sludge cake depth increased. A combined reed bed – freezing bed technology can provide a cost-effective solution for septage management in northern rural communities with potential for beneficial reuse of both the filtrate and dewatered sludge cake.

Keywords: septage, reed bed, freezing bed, metals, pathogens, agricultural reuse

### 6.2 Introduction

Septage, the solids accumulated in septic tanks, has traditionally been applied to agricultural land without treatment in Ontario (Canada). However, public policy is moving towards regulating septage as a biosolid with pathogen and metals limits; as is the case in many jurisdictions throughout North America (CCME, 2010; USEPA, 1994a). Disposal of septage at municipal treatment plants is often not feasible, as town wastewater systems, which are often lagoons, are generally not equipped to receive and treat septage. A low-cost technology which can dewater and treat septage year-round and be able to meet biosolids reuse standards is needed for rural communities in cold-climate regions such as

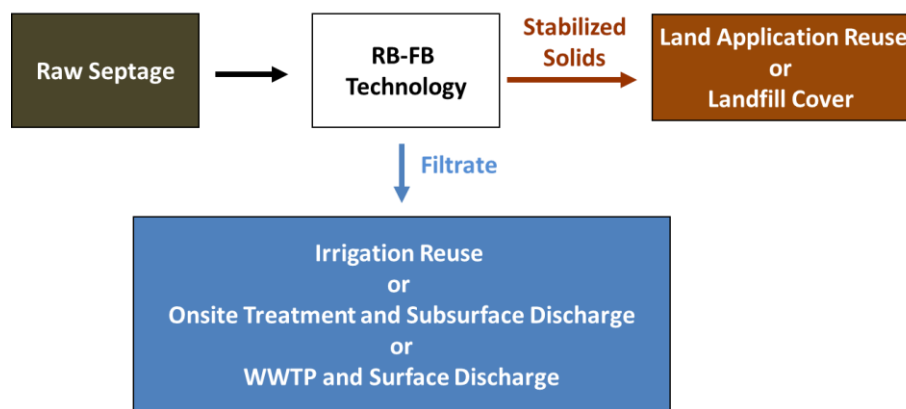
Ontario. It is hypothesized that a combined reed bed and freezing bed technology can meet these requirements.

Reed bed filters are similar in design to conventional sand drying beds only planted with common reeds (*Phragmites*). The main difference between a reed bed and a sand drying bed is that the sludge is left to accumulate in a reed bed over a period of 6-10 years, greatly reducing operating costs. The reeds play two important roles: firstly, the growing rhizomes and movement of the stems in the wind break apart the accumulating sludge layer and permit continuous filter drainage and secondly, plant evapotranspiration increases sludge dewatering (De Maeseneer, 1997). Reed beds have been used extensively in Europe for dewatering municipal waste activated sludges (WAS) and mixed WAS and anaerobic digestion (AD) sludges with recommended loading rates of 50 and 60 kg total solids (TS)/m<sup>2</sup>/y, respectively (Nielsen, 2003). As well, a limited number of studies have shown reed beds to be effective at septage dewatering: two full scale systems in France at loading rates of 46 and 109 kg TS m<sup>2</sup>/y (Paing and Voisin, 2005), and pilot system in France at 50 kg TS/m<sup>2</sup>/y (Vincent *et al.*, 2011) and two pilot systems in tropical countries at much higher loading rates of 250 kg TS m<sup>2</sup>/y (Koottatep *et al.*, 2005) and 100-300 kg TS m<sup>2</sup>/y (Kengne *et al.*, 2009). However, the reed bed technology has not been adapted to operate under freezing conditions.

It is hypothesized that reed bed filters can be operated as freezing bed filters during the winter months. Martel (1993) conducted pioneering work with freezing beds and proposed a design for a freezing bed filter consisting of a sand drying bed with extended side walls to accommodate the accumulating layers of sludge applied and frozen during the winter, with dewatering occurring in the spring. As sludge freezes, particulate matter is rejected during ice crystal formation and consolidated into solid particles along the crystal boundary, greatly increasing dewaterability (Reed *et al.*, 1986). Freezing temperature plays an important role in dewaterability, with the most effective freezing temperatures ranging between -2 to -10°C and dewaterability decreasing rapidly below -30°C (Hung *et al.*, 1996; Vesilind and Martel, 1990; Wang *et al.*, 2001). As winter temperatures rarely fall below -30°C in most populated cold climate regions, freeze-thaw (FT) conditioning should be a widely applicable technique. Many laboratory studies have demonstrated the

effectiveness of FT conditioning on sludge dewaterability including: primary, WAS, AD, chemical coagulant and water plant sludges and RBC sludge (Vesilind and Martel, 1990; Chu and Lee, 1998; Diak *et al.*, 2011). Freezing beds have been successfully operated at the pilot scale in the N.E. United States to treat WAS, AD sludge and water treatment plant alum sludge (Martel and Diener, 1991; Martel, 1993) and in Ontario, Canada to treat septage (Kinsley *et al.*, 2012).

This study explores the application of a combined reed bed / freezing bed (RB-FB) technology to treat and dewater septage and focusses on the effect of operating conditions (loading rate, operating season, accumulating sludge cake with time) on filtrate and sludge cake quality for reuse applications. Figure 6-1 depicts potential reuse and disposal options for RB-FB by-products.



**Figure 6-1: Reuse and Disposal Options for Solid and Liquid Streams from Septage Treated in a Reed Bed - Freezing Bed Technology**

The potential for reuse of the sludge cake will typically be governed by biosolids regulations with limits on toxic metals and pathogens (CCME, 2010). Nutrient (N:P:K) and organic matter (OM) content are the key economic drivers for beneficial reuse as the dewatered sludge cake can substitute manure, compost or organic soil applied to agricultural land, parkland or land reclamation sites (USEPA, 1994b). Most jurisdictions specify one threshold for unrestricted reuse with very low metal concentrations and non-detect pathogen numbers as well as a second threshold for restricted reuse with higher metal concentrations and moderate pathogen numbers (typically  $2.0 \times 10^6$  *E. coli* / g dry

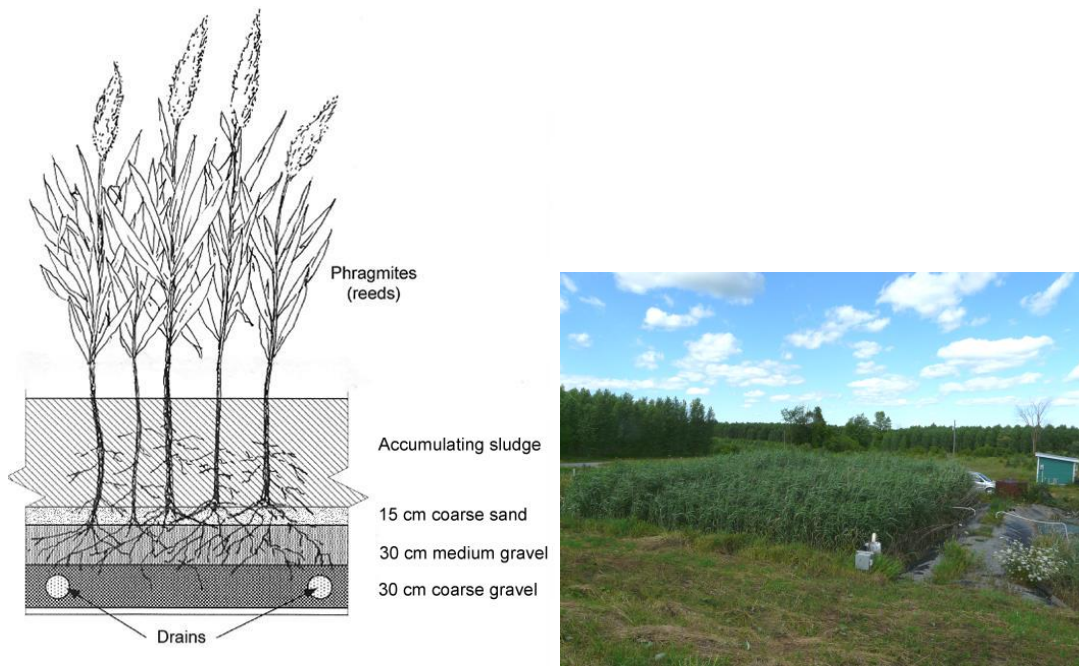
matter (DM)) (Iranpour *et al.*, 2004). The potential for reuse of the filtrate for irrigation purposes will depend upon wastewater reuse guidelines or regulations, and will typically include limits on pathogens, salinity, biochemical oxygen demand (BOD), total suspended solids (TSS) and toxic metals (USEPA, 2012; WHO, 2006; Alberta Environment, 2000).

### 6.3 Materials and Methods

Two reed bed systems (RB1 and RB2) and one non-planted sand filter (SF) were constructed at the septage lagoon of René Goulet Septic Tank Pumping, Green Valley, ON, Canada (45.32°N 74.64°W). The study site is located between Ottawa, ON and Montreal, QC. Average monthly 25-year temperature climate normals vary from -10.8°C in January to 20.9°C in July, with average temperatures remaining below the freezing point from December through March.

Each filter was 187 m<sup>2</sup> and was sized to receive individual loads of 13.6 m<sup>3</sup> from a septage vacuum truck; which represents a 7.3 cm dose, and is slightly lower than the 8.0 cm dose recommended by Martel (1993) for freezing bed operation. The filter design was based upon recommended specifications for sand drying beds (Wang *et al.*, 2007). A cross sectional schematic and photo of the system is presented in Figure 6.2. The cross section of each system from bottom to top consists of: 6.4 mm non-woven geotextile to protect the geomembrane, a 30 mil geomembrane (Layfield Tantalum 5-30 mil), a 0.3 m layer of washed coarse gravel (20-40 mm dia.), a 0.30 m layer of washed fine gravel (5-10 mm dia.), and a 0.15 m layer of locally available concrete sand (D10 = 0.18 mm; Cu = 3.4; 2.1% fines). Berms were constructed around each system to achieve 2.0 m of freeboard to contain the increasing sludge cake layer over time as well as frozen sludge accumulation during the winter months. The effluent drainage system consists of 9 lines of 10 cm perforated PVC pipe at 1.5 m spacing laid in the coarse gravel layer with a 1% slope to a collector pipe at the toe of each system. Aeration standpipes were connected to the drainage network at the ends and middle of each drainage line. The two reed beds were planted with *Phragmites* harvested from nearby ditches. Initially, rhizomes were planted in RB1 and RB2 at 4 rhizomes m<sup>2</sup>; however, few survived Yr 1 and clumps of reeds were excavated and placed directly in the systems in Yr 2, which survived and propagated over the following years.

Filtrate from each system was collected in a 1.3 m<sup>3</sup> pump chamber and was pumped into an existing lagoon with Myers WR10H-21 1HP pumps. The systems were dosed directly from the vacuum truck onto a splash plate after passing through a 1.0 cm bar screen to remove large non-biodegradable objects. Lagoon supernatant was used to irrigate poplar plantations during the summer months.



**Figure 6-2. Pilot Reed Bed - Freezing Bed System Schematic and Photo with Poplar Plantation in Background**

The systems were operated as reed beds from May to November with scheduled dosing and as freezing beds during the winter months, where a new dose of septage was applied once the previous dose had frozen. During Years 1 and 2, hydraulic loading rate (HLR) was maintained fairly constant at 3.2-3.4 m/y in 2007 and 2.4-2.8 m/y in 2008 to the three filters while solid loading rate (SLR) was varied by selecting septage loads with varying solids concentrations. This resulted in SLR varying from 113 to 144 kg TS/m<sup>2</sup>/y in 2007 and from 49 to 91 kg TS/m<sup>2</sup>/y in 2008 (Table 6-1). During Year 3, HLR (3.1-3.6 m/y) and SLR (74-91 kg TS/m<sup>2</sup>/y) remained fairly constant to compare the effect of non-planted SF to the planted RB1 and RB2. During Years 4 and 5, HLR was varied to the systems (1.9-5.9 m/y), which also impacted SLR. Over the course of the study solid loading rates to the filters ranged from 43 to 147 kg TS/m<sup>2</sup>/y, with average loading between 82 – 104 kg

TS/m<sup>2</sup>/y; which is consistent with the loading rates reported by Paing and Voisin (2006) but is considerably higher than the 50-60 kg TS/m<sup>2</sup>/y recommended by Nielson (2003) for WAS and WAS mixed with AD sludge.

**Table 6-1: Annual Solid and Hydraulic Loading Rates to Systems by Calendar Year**

Year	SF		RB1		RB2	
	SLR	HLR	SLR	HLR	SLR	HLR
	(kg/m <sup>2</sup> /y)	(m/y)	(kg/m <sup>2</sup> /y)	(m/y)	(kg/m <sup>2</sup> /y)	(m/y)
2007	142	3.2	144	3.3	113	3.4
2008	91	2.4	75	2.8	49	2.8
2009	88	3.6	91	3.1	74	3.4
2010	147	5.9	43	1.9	81	3.5
2011	54	2.4	58	2.6	110	4.6
Avg.	104	3.5	82	2.7	85	3.5

Representative 2 L septage samples were collected from each truck load and stored in a sample fridge located on site. A peristaltic pump activated by the effluent pump in each pump chamber collected flow-proportional composite filtrate samples from each system into a 20 L carboy. The pump and carboy were housed inside a cooler with a heat trace cable and 40 Watt light bulb to maintain the temperature above freezing during the winter months with filtrate samples collected on a bi-weekly basis. Grab samples were collected for bacteria analysis and stored in sterile sample bottles for transport to the laboratory. Sludge cake samples were collected at the end of the study at varying sampling frequencies using a 5 cm dia. x 90 cm soil corer. Each bed sample consisted of a composite of 4 cores. Composites were created either from the entire cores or the cores were split into 3 x 30 cm segments if depth profiles was studied. Raw septage and filtrate samples were analysed for: COD, BOD<sub>5</sub>, TS, TSS, TKN, NH<sub>3</sub>, NO<sub>3</sub>, TP, *E. coli* and metals, while raw septage and cake samples were analysed for: TS, VS, N, P, K, C, NH<sub>3</sub>, NO<sub>3</sub>, *E. coli*, *C. Perfringens*, *Salmonella*, *Enterococci* and heavy metals. Metals analyses and sludge N, P, K were analysed at the Ontario Ministry of Environment laboratory following EPA methods (SCC, 2013) while the

remaining analyses were conducted at the Ontario Rural Wastewater Centre environmental quality laboratory following Standard Methods (APHA, 2005).

### 6.3.1 Statistical Design

#### Filtrate

Filtrate quality was compared between filters over the entire study period for each parameter using a single factor ANOVA. Where significance was found ( $P < 0.05$ ), a post hoc T-test assuming equal variance with Bonferroni correction was conducted (Dunn, 1961). To evaluate the effect of loading rate on filtrate concentration, a two-way ANOVA without replication was conducted for each year with sample date and filter as variables. If significance was found ( $P < 0.05$ ), a paired T-test with Bonferroni correction was conducted between each of the filter pairs. To compare the effects of operating period (Dec-April freeze-thaw vs May - November growing season) and time, a two-way ANOVA with replication was conducted. Blocked average data for each period was used with the three filters acting as replicates. All pathogen data was log normalized prior to conducting statistical analyses.

#### Sludge cake

The effect of increasing sludge cake with time on *E. coli* numbers was compared using a two-way ANOVA without replication with filter and year as variables. As well, a correlation coefficient was determined between *E. coli* and sludge cake depth. At the end of the study, dosing was stopped to SF and RB1 (RB2 remained in service for the hauler) and the effect of drying on bacteria numbers was investigated with a two-way ANOVA without replication conducted for each indicator species with depth and sample date as variables. Bacteria numbers between filters was compared using a two-way ANOVA with replication (3 sample depths) using filter and sample date as variables. Bacteria numbers were compared between raw, sludge cake during filter operation and sludge cake during filter drying for each indicator species using a single factor ANOVA. Where significant was found ( $P < 0.05$ ), a post hoc T-test assuming equal variance with Bonferroni correction was conducted.

All statistical analyses were conducted using the Data Analysis Toolpack™ in Microsoft Excel.

## 6.4 Results and Discussion

### 6.4.1 Organic Matter, Solids and Nutrients

The filters performed exceptionally well at removing organic matter, solids and nutrients from septage with average removal rates of 99% for COD, BOD<sub>5</sub> and TSS, 98% for TP, 90% for TN and 93% for TS (see Table 6-2). These results are very similar to those reported by Paing and Voisin (2005) at a full scale reed bed system treating septage in France and by Burgoon *et al.* (1997) in a reed bed system treating lagoon biosolids in the Northwestern United States. The very high removal rates strongly suggest that the organic matter and nutrients are mostly related to particulate matter, which is removed through filtration. No significant differences in average filtrate concentration were observed between the three filters for COD, BOD and TSS ( $P > 0.1$ ), while TN filtrate concentrations were significantly higher in SF compared with both RB1 and RB2, and TP filtrate concentrations were significantly lower in RB1 compared with both SF and RB2 ( $P < 0.05$ ). These observed differences could be due to the fact that SF had the highest average SLR and HLR, while RB1 had the lowest in addition to the potential role of plant uptake. Filtrate TS was also significantly higher in SF, reflecting higher SLR to this filter over the course of the study. However, these differences are not relevant from a treatment perspective as the range of filtrate concentrations measured were typical of weak to average domestic wastewater (Metcalf and Eddy, 2003), which can be discharged to the headworks of a municipal wastewater treatment plant or lagoon system, easily treated in any decentralized wastewater treatment system, or potentially used as a source of irrigation water.

**Table 6-2: Septage Treatment in Sand and Reed Bed Filters (Averages over 5 years) (Removal on a volumetric basis)**

Parameter	Raw Septage (Avg. $\pm$ SD) (mg/L)	Filtrate (Avg. $\pm$ SD)			Avg. Removal (%)
		SF (mg/L)	RB1 (mg/L)	RB2 (mg/L)	
COD	27,000 $\pm$ 28,300	274 $\pm$ 258	219 $\pm$ 144	231 $\pm$ 132	99.1
BOD <sub>5</sub>	6,400 $\pm$ 6,800	71 $\pm$ 88	55 $\pm$ 43	59 $\pm$ 47	99.0
TS	25,800 $\pm$ 24,000	*2310 $\pm$ 800	1720 $\pm$ 620	1750 $\pm$ 600	92.5
TSS	19,500 $\pm$ 18,400	114 $\pm$ 155	79 $\pm$ 80	102 $\pm$ 97	99.5
TN (TKN + NO <sub>3</sub> )	750 $\pm$ 630	*91 $\pm$ 43	67 $\pm$ 31	65 $\pm$ 28	90.0
TP	265 $\pm$ 300	5.2 $\pm$ 4.0	*3.8 $\pm$ 2.5	5.4 $\pm$ 2.8	98.2

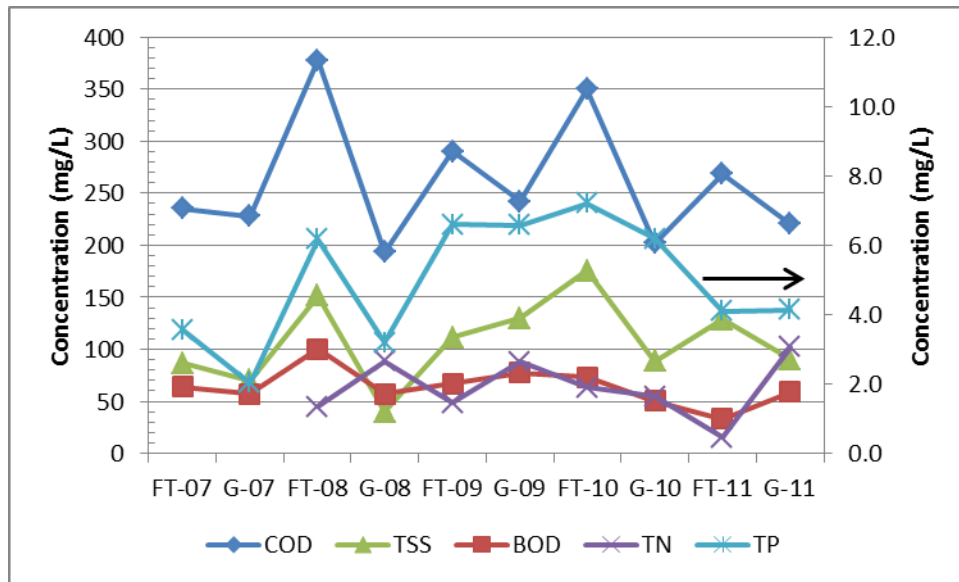
\* Significant difference at 95% confidence level using a single factor ANOVA with post hoc T-Test assuming equal variance with Bonferroni correction.

The filtrate was used to irrigate a poplar plantation as part of the Hauler's land application approval. Total dissolved solids (TDS) in the filtrate averaged 1641  $\pm$  540 mg/L, which falls within the upper range of typical irrigation water and could require moderate restrictions for use depending on the risk of increasing soil salinity (WHO, 2006). However, there is no risk in eastern Ontario due to an annual surplus water budget where precipitation exceeds evaporation and any accumulated salts will be leached out of the root zone. Wastewater reuse guidelines vary across jurisdictions; for example, Alberta Environment (2000) recommends CBOD and TSS < 100 mg/L while USEPA (2012) recommends secondary treatment with BOD and TSS < 30 mg/L in addition to pathogen limits. Reuse of the filtrate as a source of irrigation water will depend upon local regulations and may require further treatment depending upon the intended use.

To evaluate the effect of loading rate on filtrate quality, filtrate data was compared between filters for each year of the study. No significant difference for any year was found for COD, BOD and TSS indicating that varying HLR and SLR did not significantly impact filtrate quality for these parameters and suggests that most organic matter was tied to the sludge solids and was effectively removed through filtration. No significant differences in TN and TP were observed in 2007, 2010 and 2011. However, in 2008, filtrate TN was significantly higher in SF than both RB1 and RB2 (99 $\pm$ 33 versus 86 $\pm$ 27 and 74 $\pm$ 27 mg/L, respectively)

and TP was significantly lower in RB1 than in both SF and RB2 ( $2.5 \pm 1.7$  versus  $3.1 \pm 1.2$  and  $4.8 \pm 1.7$  mg/L, respectively), which is consistent with the higher SLR to SF. In 2009, filtrate TN was significantly higher in SF than RB1 ( $95 \pm 50$  versus  $61 \pm 20$  mg/L), with no significant differences observed with RB2 ( $67.7 \pm 26.6$  mg/L). The difference observed in 2009 could relate to plant N uptake in the reed bed filters. It can be concluded from these results that a combined sand bed-freezing bed or reed bed-freezing bed technology can effectively separate organic matter and nutrients from septage at SLRs between 43 and 147 kg/m<sup>2</sup>/y and HLRs between 1.9 and 5.9 m/y, with little to no effect on filtrate quality. Furthermore, differences between the planted RB1 and RB2 compared with the unplanted SF were mostly non-significant and where significance was found, the differences were not important from an effluent quality perspective.

The effect of operating period (freezing-thaw vs reed bed) and accumulating sludge cake with time on filtrate quality is presented in Figure 6-3. Significant differences between Periods were observed for COD, TSS, TP and TN ( $P < 0.05$ ), while no significant differences were observed between Years ( $P > 0.05$ ). A pattern of higher concentrations in the FT period can be observed for all parameters except TN, which displays the opposite behaviour. The higher solids migration observed during thawing events could result from preferential pathways created from FT conditioning of the sludge cake in conjunction with higher levels of soil saturation (Mohanty *et al.*, 2014). The higher concentrations of nitrogen observed from May to November (G) could result from higher dissolved ammonia concentrations in the raw septage, as proportionally more holding tanks, with lower strength wastewater, are pumped during winter. While filtrate was observed to vary by operating period, no trend over time was observed, indicating that the accumulating sludge cake did not affect filtrate quality and that the filters were operating in a steady state condition.



**Figure 6-3. Filtrate Quality with Operating Period and Time. Comparison of Freeze-thaw (FT) period from December-April to the Growing (G) period from May-November. Each period is an average of the three filters. Error bars not shown for clarity. A two-factor ANOVA with average concentration from the three filters as replicates was used. Significant effect of Period for COD, TSS, TN and TP was observed ( $P < 0.05$ ), while no effect for Year ( $P > 0.05$ ) was observed.**

Dewatered septage cake quality is described in Table 6-3 and is compared with raw septage and solid dairy manure. Solids increased from 2.6 to 23.8 percent dry matter (DM), which is comparable to solid dairy manure. Approximately 25% of nitrogen was lost in the filters (on a DM basis), likely from nitrification/denitrification reactions, with dried septage cake containing 78% of solid dairy manure N. P was almost entirely conserved in the sludge cake and was somewhat higher than that of solid dairy manure. Septage, however, was not a significant source of K compared with solid dairy manure as K is water soluble and is not strongly bound to the sludge solids. No change in organic matter (OM) between raw septage and dried septage cake was observed, suggesting that the readily degradable organics in domestic wastewater had already been largely consumed by anaerobic bacteria in the septic tanks prior to application to the filters. OM was lower in the dewatered septage compared with solid dairy manure, reflecting the more stabilized nature of septage, while total carbon concentrations were the same between the two materials at 36% (on a DM basis). The C/N ratio in the dried septage cake was somewhat higher than solid dairy manure, while the  $\text{NH}_4/\text{TKN}$  ratio was lower, reflecting the loss of ammonia in the filter beds. In summary, the dried septage cake can provide a good source of organic

matter and nutrients for agricultural production and has similar agronomic value to solid dairy manure.

**Table 6-3. Nutrient Content in Dewatered Septage. Average of three filters (Avg. ± SD)**

Parameter	Raw Septage	Dewatered Septage Cake	Solid Dairy Manure
Dry Matter (%)	2.6 ± 2.4	23.8 ± 6.9	25.9*
OM (%)	65±13	63±3	72**
C (% DM)	-	36.0±1.8	35.6**
N (% DM)	2.91 ± 2.43	2.09 ± 0.49	2.78*
P (% DM)	1.03 ± 1.15	0.96 ± 0.01	0.77*
K (% DM)	0.36 ± 0.60	0.13 ± 0.05	2.36*
C/N	-	17.2	12.8**
NH <sub>4</sub> <sup>+</sup> /TKN	0.17	0.08	0.21*

\* Brown, C. (2013) Available Nutrients and Value for Manure from Various Livestock Types OMAFRA FactSheet;

\*\*Pettygrove and Heinrich (2009). Dairy Manure Content and Forms, UC Extension.

#### 6.4.2 Metals and Salts

Metal partitioning between the raw septage and filtrate was evaluated over a four year period and is presented in Table 6-4. Overall percent removal of heavy metals was very high with greater than 99% removal observed for Cu, Zn, Al, Fe and Ti. This is consistent with the strong association of most heavy metal species with sludge solids and organic matter (Lake *et al.*, 1984) and corresponds to the > 99% removal of TSS observed in the filters. However, Ni was found to be the most water soluble of the heavy metal species across several studies with Ni solubility varying between 1.9-14.3% (Lake *et al.*, 1984), which is consistent with the 10.9% concentration of Ni observed in the filtrate. Wang *et al.* (2006) showed that metal adsorption to sludge particles generally increases with pH and that Co and Ni are the least absorbable heavy metal species, which is also consistent with the removal rates reported in Table 6-4. Lower removal rates were observed with salts (Ca, K, Mg and Na), which are all water soluble. Significantly higher concentrations of salts (Ca, K, Mg, Na) as well as Mn were observed in the SF filtrate compared with at least one of the

reed bed filters and significantly lower concentrations of Co and Ba in the RB2 filtrate were observed compared with the other two filters; generally reflecting higher average SLR and HLR in SF throughout the study.

**Table 6-4: Raw and Dewatered Septage Metal Quality (Year 1-4) (Avg. ± SD). Note: Filtrate concentrations below detection limit values (Cd, Cr, Hg, Mo, Pb, Se, Ag, Be, Sb, V) were not reported. Avg. percent separation was calculated on a volumetric basis.**

Parameter	Raw Septage (mg/L)	Filtrate			Irrigation Guideline <sup>1</sup> (mg/L)	Avg. Separation (%)
		SF (mg/L)	RB1 (mg/L)	RB2 (mg/L)		
As	0.06 ±0.12	0.0032 ±0.0014	0.0031 ±0.0017	0.0029 ±0.0013	0.1	95.0
Co	<sup>a</sup> 0.094 ±0.025	0.012 ±0.006	0.012 ±0.005	* 0.009 ±0.005	0.05	>88.3
Cu	11.6±22.1	0.077±0.086	0.064±0.057	0.060±0.047	0.2	99.4
Ni	0.38±0.42	0.045±0.029	0.046±0.042	0.036±0.023	0.20	89.1
Zn	19.8±24.4	0.15±0.20	0.10±0.12	0.11±0.11	2.0	99.4
Al	208.6±295.1	0.59±0.72	0.45±0.50	0.54±0.56	5.0	99.7
Ba	6.6±9.9	0.11±0.07	0.12±0.11	* 0.08±0.05	-	98.4
Fe	135.2±165.5	0.97±1.02	0.89±1.16	0.90±0.84	5.0	99.3
Mn	2.9±3.7	* 0.24±0.17	* 0.17±0.13	0.23±0.17	0.20	92.4
Sr	13.2±36.8	3.1±2.0	2.4±1.1	2.6±1.1	-	79.5
Ti	1.5±1.4	0.012±0.012	0.012±0.020	0.014±0.020	-	99.1
Ca	784.4±878.2	* 240.9±92.0	195.5±87.3	193.4±74.2	-	73.5
K	95.1±157.9	* 67.9±29.1	46.7±30.4	46.0±29.1	-	44.4
Mg	83.2±73.3	* 41.5±13.5	34.4±12.7	33.9±10.4	-	56.5
Na	490.8±621.1	* 412.8±207.8	* 316.7±136.9	360.1±155.9	-	26.4

<sup>a</sup> Detection limit values often reported

<sup>1</sup> WHO, 2006

\* Significance at 95% confidence level using a single factor ANOVA followed by a post hoc T-test assuming equal variance with Bonferroni correction.

Metals in the filtrate were lower than the recommended maximum concentrations for irrigation (WHO, 2006), with the exception of Mn, which was just above the recommended

limit value. At these concentrations, manganese can be toxic to a number of crops, but usually only in acidic soils. The sodium adsorption ratio (SAR), is a ratio of  $\text{Na}^+$  to  $\text{Ca}^{++}$  and  $\text{Mg}^{++}$  used to determine the suitability of water for irrigation. As SAR increases, the risk of damage to the soil structure increases, which can reduce soil infiltration (Equation 6.1). Filtrate SAR was calculated to be 2.0, which falls within the typical range for irrigation water and poses no risk of affecting soil infiltration rates. (WHO, 2006)

$$\text{SAR} = [\text{Na}^+]/(0.5([\text{Ca}^{++}] + [\text{Mg}^{++}]))^{0.5} \quad \text{(Equation 6.1)}$$

where concentrations are in meq/L

To evaluate the effect of loading rate on filtrate quality, filtrate data was compared between filters for each year of the study. No significant differences were observed between filters for any metal species that exhibited > 99% removal (Cu, Z, Al, Fe, Ti) for any of the four years, indicating that increasing SLR or HLR did not significantly affect filtration of sludge solids for these species. No significant differences were observed between filters in 2007 for any species even though SLR varied widely. In 2008, SF metal filtrate concentrations were significantly higher in 4 species (Sr, Ca, Mg, Na), while either RB1 or RB2 were significantly lower in 4 species (Ni, Ba, Mn, K), which is consistent with the higher SLR of SF. In 2009, SF metal filtrate concentrations were significantly higher in 4 species (Ni, Sr, Ca, Mg), while RB2 filtrate concentrations were significantly lower in 2 species (As, K), possibly reflecting the small differences in both SLR and HLR between the filters in addition to the potential effect of plant uptake in the two planted filters. In 2010, filtrate K was significantly higher in SF than in RB2, and filtrate Na was significantly lower in RB1 than in RB2. The limited differences in filtrate concentrations observed, which mostly relate to readily dissolved salt species, suggests that varying both HLR or SLR had little to no impact on filtrate heavy metal concentrations.

The effect of operating period (FT vs G) and accumulating sludge cake with time on filtrate concentration are described in Figure 6-4. In Figure 6-4i and 6-4ii metal concentrations in FT periods were significantly lower ( $P < 0.05$ ) than those in the G periods with the exception K, which followed the same trend but showed no significant difference. This could be due to the fact that raw metal concentrations were lower during the FT period when more

holding tanks and fewer septic tanks are pumped. Correlating raw septage to filtrate concentration for these metals found strong correlation coefficients ranging from  $R=0.48$  to  $0.89$ , with a clear trend of increasing correlation with decreasing percent removal ( $R=-0.85$ ). This stands to reason as the more water soluble the metal, the higher the correlation between raw and filtrate concentration. The opposite effect was observed with the metals in Fig 8-4iii, where metal concentrations in the FT periods were significantly higher ( $P<0.05$ ) than during the G periods, with the exception of Zn and Cu, which followed the same trend but showed no significant differences. What is interesting about these metals is that they all represent heavy metals with removal rates of greater than 99%, with the exception of Mn at 92.4%. Correlating raw septage to filtrate concentration for these metals showed negative correlations ranging from  $R= -0.21$  to  $-0.76$ . Mohanty *et al.* (2014) found that freeze-thaw cycles in soils increased mobilization of metals through increased migration of colloids through preferential pathways and increased hydraulic conductivity from saturated soils during thaw events. The same phenomena could be observed in this study, with the effect masked in Figures 6-4i and 6-4ii by a larger impact from differences between seasonal influent concentrations.

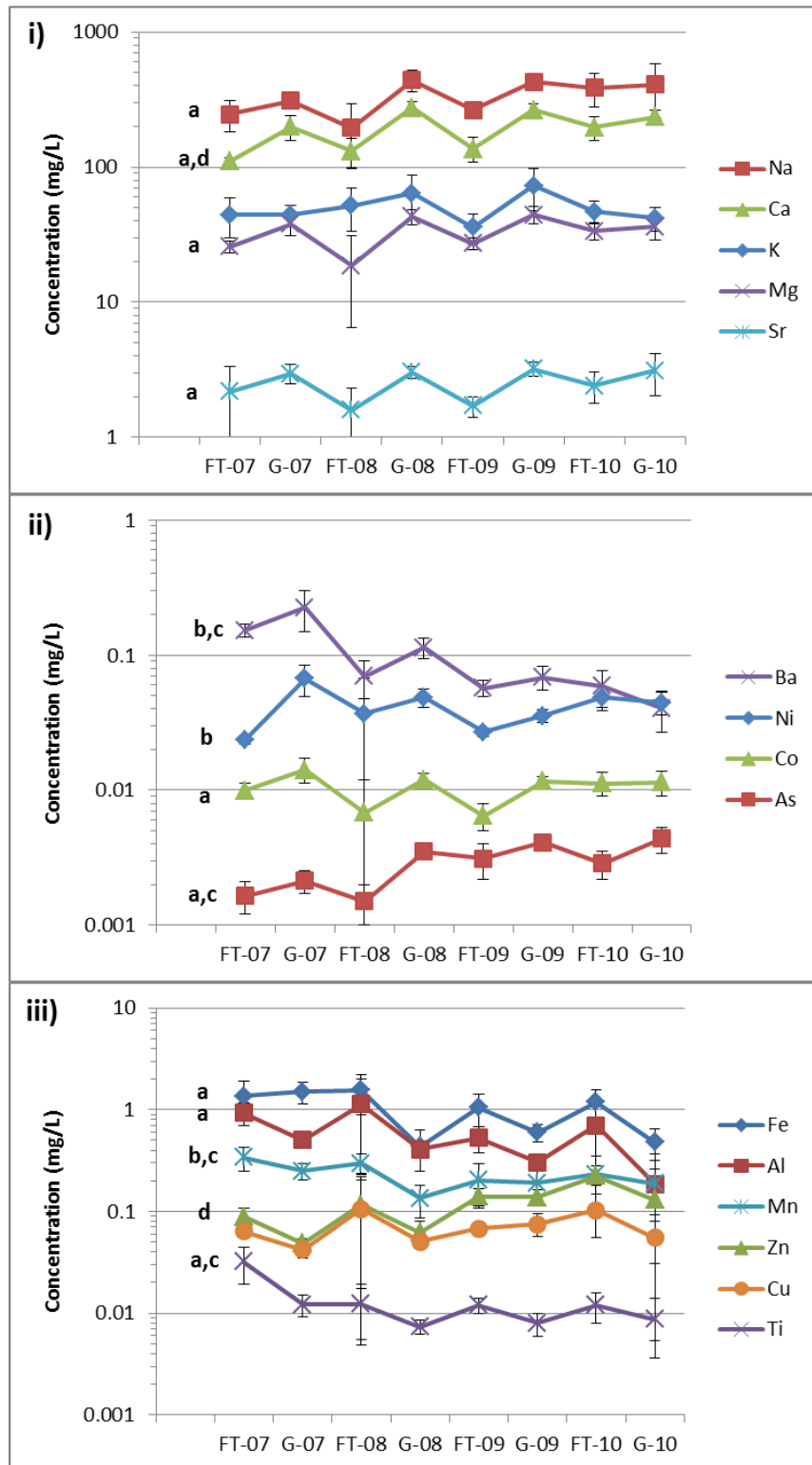


Figure 6-4: Filtrate Metal Concentration with Operating Period and Time. Comparison of Freeze-thaw (FT) period from December-April to the Growing (G) period from May-November. Data grouped by operating period with average of 3 filters  $\pm$  SD. Significance tested using a two way ANOVA with: a: Operating Period  $P < 0.01$ ; b: Operating Period  $P < 0.05$ ; c: Year  $P < 0.01$ ; d: Year  $P < 0.05$ .

The effect of year, which represents the accumulating sludge cake, was less evident, with only 6 of 15 metal species exhibiting a significant difference; with Ba, Mn and Ti decreasing with time as As, Ca and Zn increased with time. Decreases could relate to more effective filtering through the accumulated sludge cake, while increases could relate to increased desorption or solubilisation and release. Even with the slight variations observed, the data indicated that heavy metals were stable within the sludge cake with variations by season and time very small compared with raw sludge concentrations. With the exception of Mn under acidic soil conditions, metals in filtrate should pose no concerns for irrigation reuse.

The regulated metals were evaluated in the dewatered septage cake at the end of year 4 and are presented in Table 6-5 and compared with both domestic sludge and the regulatory limits for Ontario, Canada. The dewatered septage cake had lower concentrations than municipal sludge for most metal species, with similar values observed for Cu, Mo, Se and Zn. This stands to reason, as municipal sludge derives from mixed domestic and industrial sources, which could increase metal content. No significant differences in the sludge metal concentration were observed between the three filters ( $P>0.05$ ), with values substantially below the CM2 regulated limits; however, several species were higher than the CM1 limits. These results indicate that dewatered septage cake meets the CM2 metals limits for restricted land application in Ontario. It should also be noted that the septage cake meets the USEPA exceptional quality (EQ) metals standard for unrestricted land application (Iranpour *et al.*, 2004); which is considerably less stringent than Ontario's CM1 Standard.

**Table 6-5. Dewatered Septage Cake Concentration and Limits for Regulated Metals**

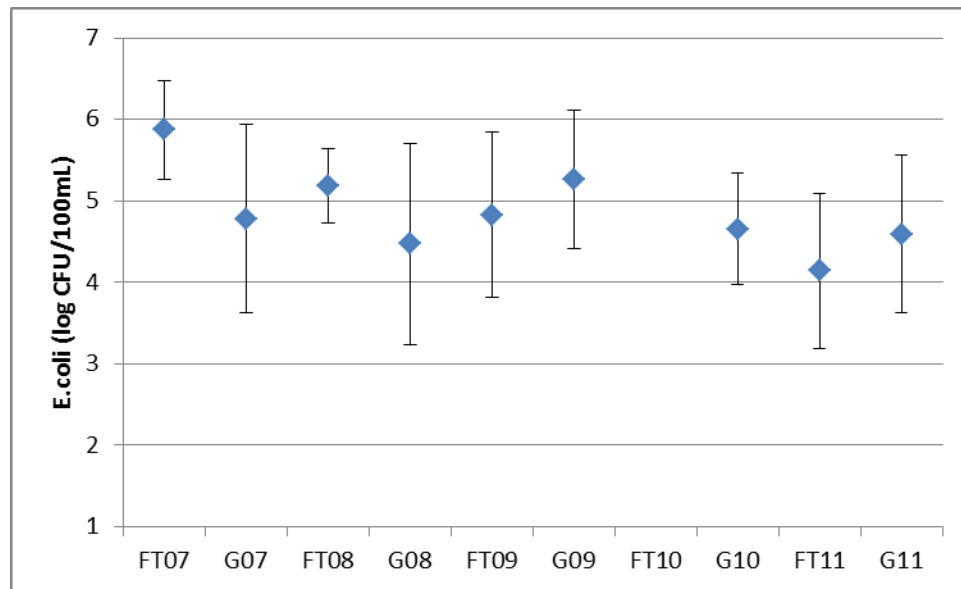
Regulated Metal	Typical Domestic Sludge (USEPA, 1984a)	Dewatered Septage Cake (Avg. ± SD) (mg/kg d.s.)			Land Application Metal Limits (O.Reg.267/03) (mg/kg d.s.)	
		RB1	RB2	SF	CM1	CM2
As	10	*<2.5	*<2.5	*<2.5	13	170
Cd	10	2.8±0.2	2.2±0.3	2.6±0.3	3	34
Co	30	1.8±0.1	1.6±0.3	1.7±0.3	34	340
Cr	500	35±5	25±2	26±5	210	2,800
Cu	800	722±117	518±85	695±54	100	1,700
Pb	500	68±8	94±24	60±20	150	1,100
Hg	6	*<2.2	*<2.2	*<2.2	0.8	11
Mo	4	9.2±0.9	7.1±1.9	9.0±1.2	5	94
Ni	80	23±3	20±2	22±2	62	420
Se	5	8.3±2.6	7.5±2.7	7.5±2.7	2	34
Zn	1700	1267±103	1110±217	1113±82	500	4200

\* less than method detection limit value

### 6.4.3 Pathogens

Pathogen numbers could potentially limit reuse applications of both the filtrate and the dewatered sludge. *E. coli* numbers were compared between filters for each year of the study to compare the effect of varying SLR and HLR on filtrate quality. No significant differences between filters were observed for any year ( $P > 0.1$ ), indicating that varying SLR and HLR had no significant impact on filtrate *E. coli* numbers. *E. coli* numbers were compared and are presented in Figure 6-5 between operating period (FT vs G) and with increasing sludge cake by year, with significant differences observed both with period and year ( $P < 0.05$ ). Declining pathogen numbers with time suggests that as the filters matured and the sludge cake depth increased, there was increased *E. coli* die-off within the filter, likely due to a combination of filtration, predation and retention time. While a significant

difference between periods was observed, no trend was apparent as *E. coli* in the FT period was higher than in the G period during 2007 and 2008 and lower during 2009 and 2011.



**Figure 6-5: Filtrate *E. coli* with Operating Period and Time. Comparison of Freeze-thaw (FT) period from December-April to the Growing (G) period from May-November. No samples were collected during FT10 period. Each data point is the average of 3 filters  $\pm$  SD. Significance was tested using a two way ANOVA with both Period and Year found to be significant ( $P < 0.05$ ).**

*E. coli* numbers were reduced from  $7.2 \pm 0.9$  log CFU/100 mL in raw septage to annual averages ranging from  $5.3 \pm 0.2$  log CFU/100 mL in Year 1 to  $4.4 \pm 0.2$  log CFU/100 mL in Year 5, with log reductions of between 1.9 and 2.8 on a volumetric basis. While there is no pathogen standard for irrigation water in Ontario, the WHO (2006) recommends  $< 5$  log *E. coli* CFU/100 mL for restricted irrigation with treated wastewater and USEPA (2012) recommends  $< 3$  log *E. coli* CFU /100 mL. On average, the filtrate would meet a 5 log limit; however, would not meet a 3 log limit without further treatment.

Average annual *E. coli* numbers in the sludge cake with cumulative sludge cake depth are presented in Figure 6-6. No significant differences between the three filters was observed ( $P > 0.1$ ), while a significant difference between years was observed ( $P < 0.01$ ). *E. coli* numbers are shown to decline from Years 1-4, with a levelling off in Year 5 with a very strong negative correlation with sludge cake depth ( $R = -0.95$ ). This suggests that as sludge cake depth increases, the impact of new sludge dosing diminishes, which is consistent with observations of Nielson (2007), who found a sharp reduction in pathogen numbers in the

first 40 cm of a reed bed shortly after dosing. During years 4 and 5 the *E. coli* numbers were below the limit for restricted land application of biosolids in Ontario and other jurisdiction of  $2 \times 10^6$  *E. coli*/g DM (CCME, 2010); therefore, in principal the sludge cake could be removed and land applied without requiring further treatment such as composting or lime stabilization.

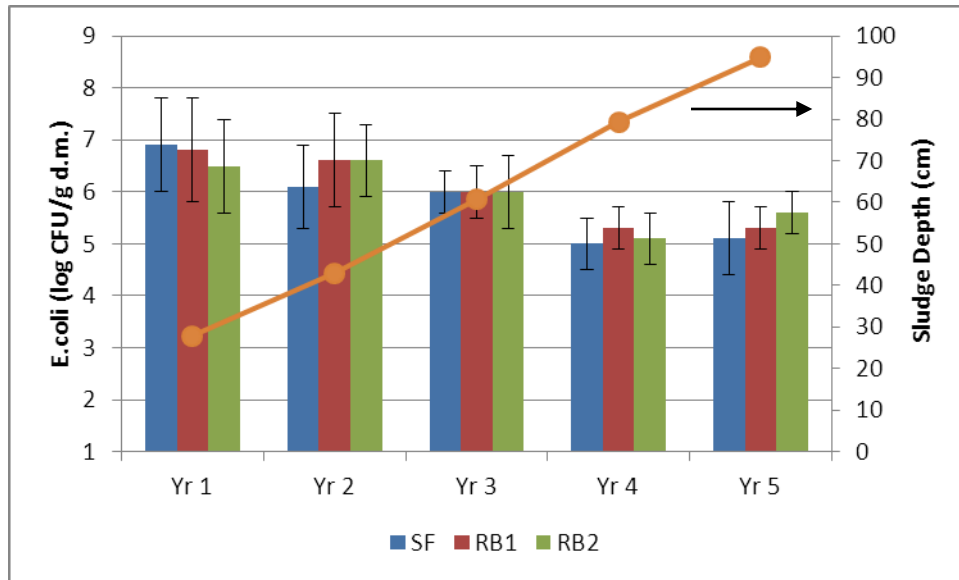


Figure 6-6. Dewatered Sludge Cake *E. coli* with Time and Cake Depth. *E. coli* are yearly geometric mean  $\pm$  SD for each filter. Two-way ANOVA with Filter and Year as variables. No significant difference between Filters ( $P > 0.1$ ) while significant difference between Years was observed ( $P < 0.01$ ).

The effect of freeze-thaw conditioning in addition to sludge cake drying and stabilization was investigated during the spring of Year 6. Dosing was stopped to both SF and RB1 in December of Year 5, with a sludge cake sampling campaign carried out from May - August, 2012. Samples were collected at three depths (0-30, 30-60, 60-90 cm) in the two filters and analysed for four pathogen indicator species as well as for dry matter. Dry matter increased in the 0-30 cm layer of both filters to a maximum of 45% over a nine week period, while the dewatered cake in both the 30-60 cm and 60-90 cm layers remained stable at approximately 20 % DM (Figure 6-7). Average temperature over the sampling period varied from 13.2-23.7°C. Bacteria numbers were compared between depth ranges as well as between the two filters for each indicator species. No significant differences between depth ranges were observed for any of the bacteria indicators ( $P < 0.05$ ). Bacteria numbers were compared between the two filters with significant differences found for all

four indicator species (Figure 6-7). The three non-spore forming bacteria (*E. coli*, *Salmonella*, *Enterococci*) all exhibited higher numbers in RB1 compared with SF ( $P < 0.01$ ) while *C. Perfringens* exhibited the opposite effect ( $P < 0.05$ ). The differences observed between the two filters could be due to relative differences in moisture, solar radiation and heat transfer between the two filters, with SF having less plant cover and higher DM content in the first 30 cm than RB1, although by Year 6, *Typha* and *Phragmites* had largely naturally populated SF. The data suggest that a variability of 1-2 logs in sludge cake bacteria should be expected between similar filter beds at a given time.

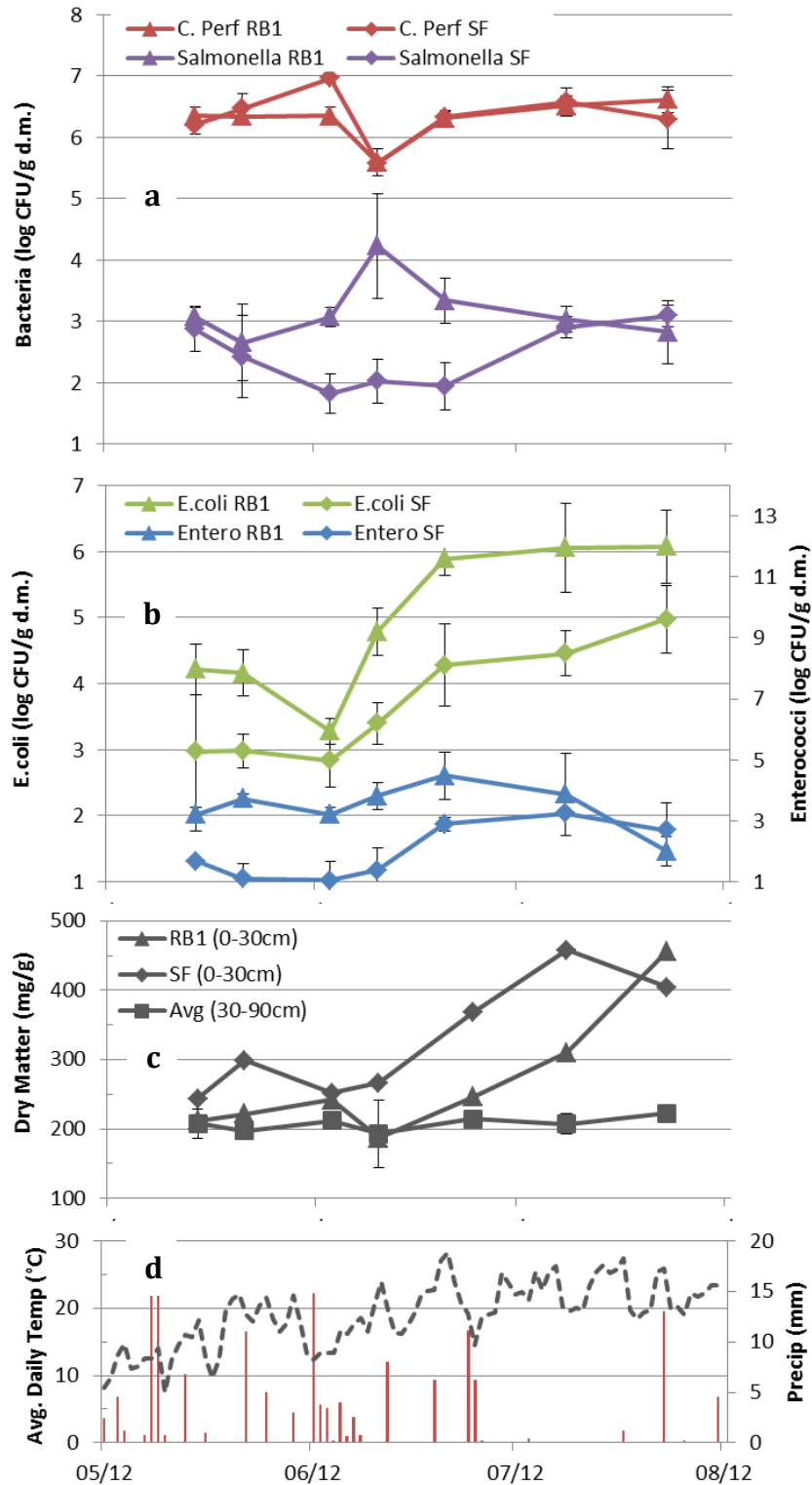


Figure 6-7: Bacteria and Dry Matter in SF and RB1 Sludge Cake with Time (no new dosing) [a] *C. perfringens* and *Salmonella*; b) *E. coli* and *Enterococci*; c) DM; d) Temperature and Precipitation]. Dry matter for 30-90 cm depths is presented as an average of both filters  $\pm$  SD as individual values showed no statistical differences ( $P>0.05$ ). Bacteria numbers are geometric means of 3 depths  $\pm$  SD.

No specific trends were observed with time for *Salmonella*, *Enterococci* and *C. perfringens* suggesting that a 2 month stabilization period after FT conditioning had no significant effect on these indicator species. *E. coli*, however, showed an increasing trend over time, with RB1 *E. coli* numbers increasing from 4.2 to 6.1 log CFU/g DM and SF *E. coli* numbers increasing from 3.0 to 5.0 log CFU/g DM. The observed increase in *E. coli* numbers without increased sludge addition was surprising and contrary to the results presented by Nielson (2007), where pathogen numbers declined to almost non-detect levels within two months after dosing to the filters had ceased. Possible reasons for the increase in pathogen numbers could include more optimal regrowth conditions such as a reduction in moisture content and/or an increase in temperature. Conditions conducive to regrowth of pathogenic bacteria in sludge include moisture levels of greater than 20%, pH from 5.5 to 9.0 and temperatures between 10 and 45°C (Ward *et al.*, 1984; Santamaria *et al.*, 2003), all of which were met in the filters. The *E. coli* numbers were initially up to 2.5 log units lower than the values measured during filter dosing, possibly due to FT conditioning; however, after 9 weeks of drying, the *E. coli* numbers had increased to levels approaching the limit for land application. These results were similar to what was observed by Gibbs *et al.* (1997), who studied stored dewatered anaerobic digestion sludge over a 1 year period and found fecal coliform and *Salmonella* numbers initially declined to non-detect levels but later increased when the environmental conditions changed; and in the case of fecal coliform to even larger numbers than were initially present in the sludge. Ward *et al.* (1999) also observed regrowth of Class A biosolids in a treatment plant prior to shipment. These results suggest that indicator bacteria can survive and potentially regrow in dewatered septage beds several months after dosing has ceased.

Average pathogen reduction from raw septage to the drying sludge cake is presented in Figure 6-8. Similar reductions were observed for *E. coli*, *Enterococci* and *Salmonella* at 1.9, 2.1 and 2.4 log removal, respectively. These log reductions were consistent with conventional sludge treatment processes, such as anaerobic or aerobic digestion, which will typically remove 2 log units of pathogens (Carrington, 2001). *C. perfringens*, however, exhibited a lower reduction of only 0.8 logs, which was expected as *C. perfringens* is a spore producing bacteria which is more resistant to sludge treatment (Chauret *et al.*, 1999).

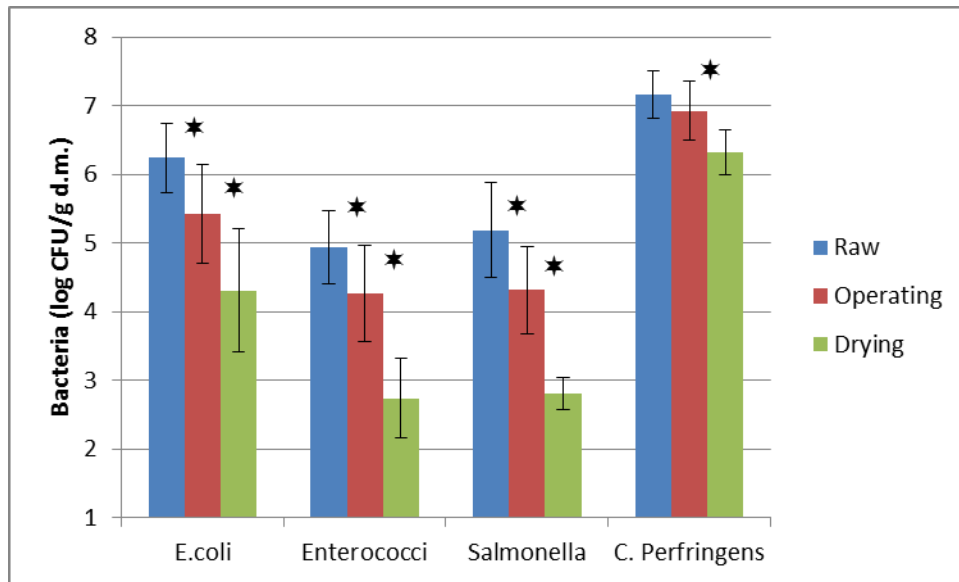


Figure 6-8: Pathogen Reduction in Filters during Operating and Drying Periods. Geometric Mean  $\pm$  SD of three filters (two for drying). Significant differences at 95% confidence level (\*) determined using a single factor ANOVA with post hoc T-test assuming equal variance with a Bonferroni correction.

## 6.5 Conclusions

A combined reed bed-freezing bed technology was effective for treating septage under Canadian climatic conditions over a period of 5 years of continuous dosing with average loading rates of 82-104 kg TS/m<sup>2</sup>/y. Varying hydraulic and solid loading rates as well as the increasing sludge cake with time had little to no effect on treatment efficiency, with 99% removal of BOD and TSS in addition to many heavy metal species (Cu, Zn, Al, Fe, Ti), 98% removal of TP and 90% removal of TN observed over the study period. This strongly suggests that heavy metals, organic matter and nutrients are largely tied to sludge solids and are effectively removed through filtration. Varying solid and hydraulic loading rate also had no significant effect on filtrate *E. coli* numbers; however, *E. coli* numbers declined with time, suggesting that the increasing sludge cake played a positive role in pathogen removal, either through increased filtration, retention or predation. Filtrate concentrations were significantly higher for many parameters during the December to April freeze-thaw period than the May-November growing season period even though influent concentrations were lower. It is possible that freeze-thaw conditioning creates preferential pathways for migration of colloidal particles and in combination with increased hydraulic gradient due to saturated soil conditions can lead to increased filtrate concentrations.

However, the differences were not significant when considering percent removal and not important when considering water quality. Filtrate quality was consistent with a low to medium strength domestic wastewater which is easily treatable in any municipal or decentralized wastewater system. The potential to reuse the filtrate as a source of irrigation water will depend upon local regulations in terms of organic matter, solids and pathogens; however, filtrate metal concentrations remained well below irrigation guideline limits and remained stable with time, suggesting that heavy metals remain strongly bound to the sludge cake solids.

The dewatered sludge cake consistently met biosolids land application standards in terms of pathogen and metals content, with *E. coli* numbers declining with time as sludge cake depth increased. Four pathogen indicators in the sludge cake declined significantly following winter freeze-thaw conditioning with no new dosing but remained stable or increased over the following summer. The sludge cake exhibited similar dry matter, organic matter, carbon, nitrogen and phosphorus content to solid dairy manure and can provide an excellent source of nutrients and organic matter for crop production.

A combined reed bed – freezing bed technology can provide a low capital and very low operating cost solution for septage management in rural communities in cold-climate regions with potential for beneficial reuse of both the filtrate and dewatered sludge cake.

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## 7 Conclusions

A new combined Reed Bed – Freezing Bed technology to dewater and treat septage has been successfully developed and applied under Canadian climatic conditions. The development of the technology progressed through a series of lab, pilot and field scale investigations.

### Lab Scale Study

The role of freeze-thaw conditioning in restoring drainage to clogged sand drying beds was investigated in a laboratory column study with four biological sludges: primary, WAS, septage and AD sludge. This application of freeze-thaw conditioning had not been previously studied. It was hypothesized that the annual freeze-thaw cycle can not only condition sludge applied during winter, but can help rejuvenate permeability in reed bed filters which may experience partially clogging in the late fall period when plants are dormant. The study found:

1. Freeze-thaw conditioning was shown to be effective at restoring drainage capacity in clogged sand drying bed filters regardless of the type of sludge applied; indicating that reed bed filters can be utilized as freezing beds during the winter months without desludging and that any clogging layer developed in the late fall can be successfully remediated through freeze-thaw conditioning.
2. Particle size and particle size distribution were shown to be good indicators of filter clogging potential, while parameters of sludge stability were not. Primary and AD sludge showed signs of clogging within several 10 cm doses both before and after conditioning, while septage and WAS showed good drainage characteristics over multiple doses both before and after conditioning.
3. It is possible to dose WAS and septage continuously at 10 cm/week for between 2.5 to 5 months before filter clogging is observed. This suggests that there is a low risk of clogging in applying a combined reed bed – freezing bed technology to dewater either septage or WAS as the periods between plant senescence and freezing temperatures is within this timeframe.

## Pilot Scale Study

Freezing bed literature recommends that the beds be covered to avoid snow accumulation and its insulating effect. This is impractical in a combined reed bed – freezing bed system and would add significantly to construction costs. It was hypothesized that the regular addition of fresh sludge will melt any accumulating snow. A pilot study was conducted over two winters varying sludge dose and comparing snow covered to non-snow covered beds. Study findings include:

1. No difference in sludge freezing was observed with or without snow removed from the freezing beds. This suggests that it is not necessary to cover the beds or remove snow from the beds in regions with modest snowfall that is less than the total sludge depth applied, as new layers of sludge will melt any accumulated snow.
2. Septage freezing was successfully modelled following an accepted model for ice formation on water bodies corrected for the initial temperature of the sludge with a model coefficient of  $m = 1.45 \pm 0.09 \text{ cm } (^{\circ}\text{C}\cdot\text{d})^{-1/2}$  at the 95% C.I. The model describes the data reasonably well ( $R^2 = 0.80$ ), although factors such as wind speed and relative humidity are not accounted for in the model. The model can be used to design freezing bed filters using average temperature data from the nearest weather station and a design sludge loading rate.
3. Septage thawing was modelled using a regression analysis. Initial frozen depth and precipitation were found to be insignificant with degree days of warming controlling the rate of thawing. The linear model describes the data well ( $R^2 = 0.87$ ).
4. The sludge freezing and thawing models were applied to temperature normals throughout Canada and the United States and a map of iso-sludge freezing curves was developed. The freezing bed technology is shown to be largely applicable across the northern United States and Alaska and most of Canada with the exception of coastal regions and southern Ontario.
5. Freezing beds have been shown to be an effective technology to treat septage and can provide a low cost winter treatment option for septic tank pumpers or for small communities. Filtrate quality is similar to a low strength domestic wastewater and the sludge cake has a dry matter content of 25% with *E. coli* numbers below  $2.0 \times 10^6$  CFU/g dry matter 1 month after thawing.

## Field Scale Study

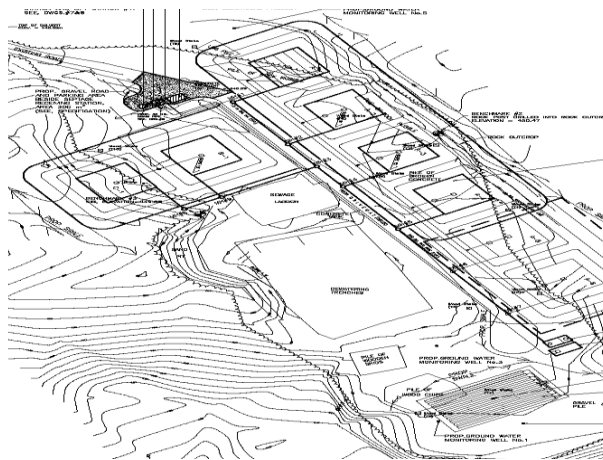
Two 187 m<sup>2</sup> reed bed systems and one 187 m<sup>2</sup> unplanted sand filter system were evaluated under varying hydraulic and solid loading conditions in a controlled 5 year study carried out in a cold climate. The combined application of reed bed and freezing bed technologies has been demonstrated to effectively dewater septage year-round under Canadian climatic conditions. Specific findings include:

1. System evapotranspiration was found to be similar to natural wetland systems in temperate climates and somewhat higher than lake evaporation for both planted and unplanted systems.
2. Freeze-thaw conditioning consistently doubled drainage rates (at equivalent levels of ponding water) compared with during the growing season. This suggests that freeze-thaw conditioning increases *in-situ* filter hydraulic conductivity and likely acts to maintain filter drainage.
3. No significant differences in drainage rates between the planted and unplanted filters were observed at equivalent hydraulic loading rates, suggesting that planting *phragmites* in the filter beds does not play a significant role in maintaining filter drainage. However, the positive effects of freeze-thaw conditioning on filter drainage could be masking a plant effect.
4. Drainage rates varied significantly with hydraulic loading rate but not with solid loading rate suggesting that it is more appropriate to use the hydraulic loading rate when designing septage reed bed systems.
5. Changing the dosing frequency from 7 to 21 days had no significant effect on drainage rates, suggesting that either dosing frequency is appropriate for filter operation.
6. A conservative design hydraulic loading rate of 2.9 m/y or a solids loading rate of 75 kg TS/m<sup>2</sup>/yD is recommended for the sizing of combined reed bed and freezing bed systems treating septage in cold climate applications.
7. Almost complete removal of organic matter, metals and nutrients was observed independent of hydraulic and solid loading rates or with accumulating sludge cake over time. This strongly suggests that contaminants are largely tied to sludge solids and are effectively removed through filtration. Filtrate quality was consistent with a low to medium strength domestic wastewater which is easily treatable in any municipal or decentralized wastewater system.

8. The dewatered sludge cake consistently met biosolids land application standards in terms of pathogen and metals content. The sludge cake exhibited similar dry matter, organic matter, carbon, nitrogen and phosphorus content to solid dairy manure and can provide an excellent source of nutrients and organic matter for crop production.
9. A combined reed bed – freezing bed technology can provide a low capital and very low operating cost solution for septage management in rural communities in cold-climate regions with potential for beneficial reuse of both the filtrate and dewatered sludge cake.

### Full Scale Application

The first full-scale application of the technology has been implemented to treat the septage and holding tank waste from the comfort stations at Algonquin Park, the largest Provincial Park in Ontario. This is the first full-scale approval of the technology by the Ontario Ministry of Environment and will act as a demonstration site and aid in future approvals as the technology moves from the University to acceptance by regulatory authorities and implementation by end user groups including consulting engineers, municipalities and septage haulers.



**Figure 7-1. Plan View and Photo of Algonquin Park Septage Reed Bed System (Credit: Ontario Parks)**

## Future Research

Future research avenues will include studying both reed bed-freezing bed and drying bed-freezing bed systems with different types of sludge, in particular WAS and lagoon sludge, in order to determine solid and hydraulic loading rates for engineering design purposes. The role of plants in maintaining filter dewatering and in aiding with sludge stabilization should be studied in detail as well as looking at various methods of optimizing oxygen transfer to the filters through both passive and active aeration. Much work is required in gaining a better understanding of clogging mechanisms in sludge gravity dewatering systems as related to sludge characteristics. This can lead to better system design and operation. The use of coagulation aids can also be studied to increase system performance. Finally, the desludging cycle and beneficial reuse in agricultural production should be evaluated and a life-cycle and economic assessment of the technology should be conducted to provide valuable information to industry practitioners as well as to government policy makers.

## **Appendix A - Reed and Sand Bed Filter Design and Construction**

Two reed bed filters (each 187 m<sup>2</sup>) and one sand bed filter (187 m<sup>2</sup>) were constructed at the septage lagoon of Goulet Pumping in Green Valley, Ontario. The design sizing was based upon annual solids loading of 100-200 kg TS/m<sup>2</sup>/y (one to two weekly loads of 12,000 L septage to each filter). The average solids content of septage was assumed to be 3 percent for design purposes.

The cross sectional and plan views of the pilot filters are presented below in Figures A-1 and A-2. The cross section of each filter, from bottom to top consists of: a felt liner, a 30 mil PVC geomembrane, a 0.23-0.38 m layer of coarse gravel (20-40 mm dia.), a 0.3 m layer of fine gravel (5-10 mm dia.), and a 0.15 m layer of coarse sand (0.3-0.6 mm dia.). Berms were constructed around each filter to achieve 1.2 m of freeboard above the filter surface to contain the increasing sludge layer over time as well as any frozen sludge accumulation during the winter months (later increased to 2.0 m). The effluent drainage system is comprised of 5 lines of 10 cm (4 inch) perforated PVC pipe laid in the gravel layer with a 2% slope flowing to a collector pipe at the toe of the filter. Aeration standpipes connect to the drainage network. The reed bed filters were planted with native *Phragmites australis* harvested from nearby ditches. Filter percolate from each filter flows into a 1300L pump chamber, where the percolate is pumped into the existing septage lagoon.

The filters were dosed from the pump on the septage truck, with the septage first passing through a 1.0 cm (3/8 inch) bar screen to remove large non-biodegradable objects such as gravel, hair, rags, plastic materials, etc. The filter was connected to the end of the truck pump hose with a quick connect coupling.

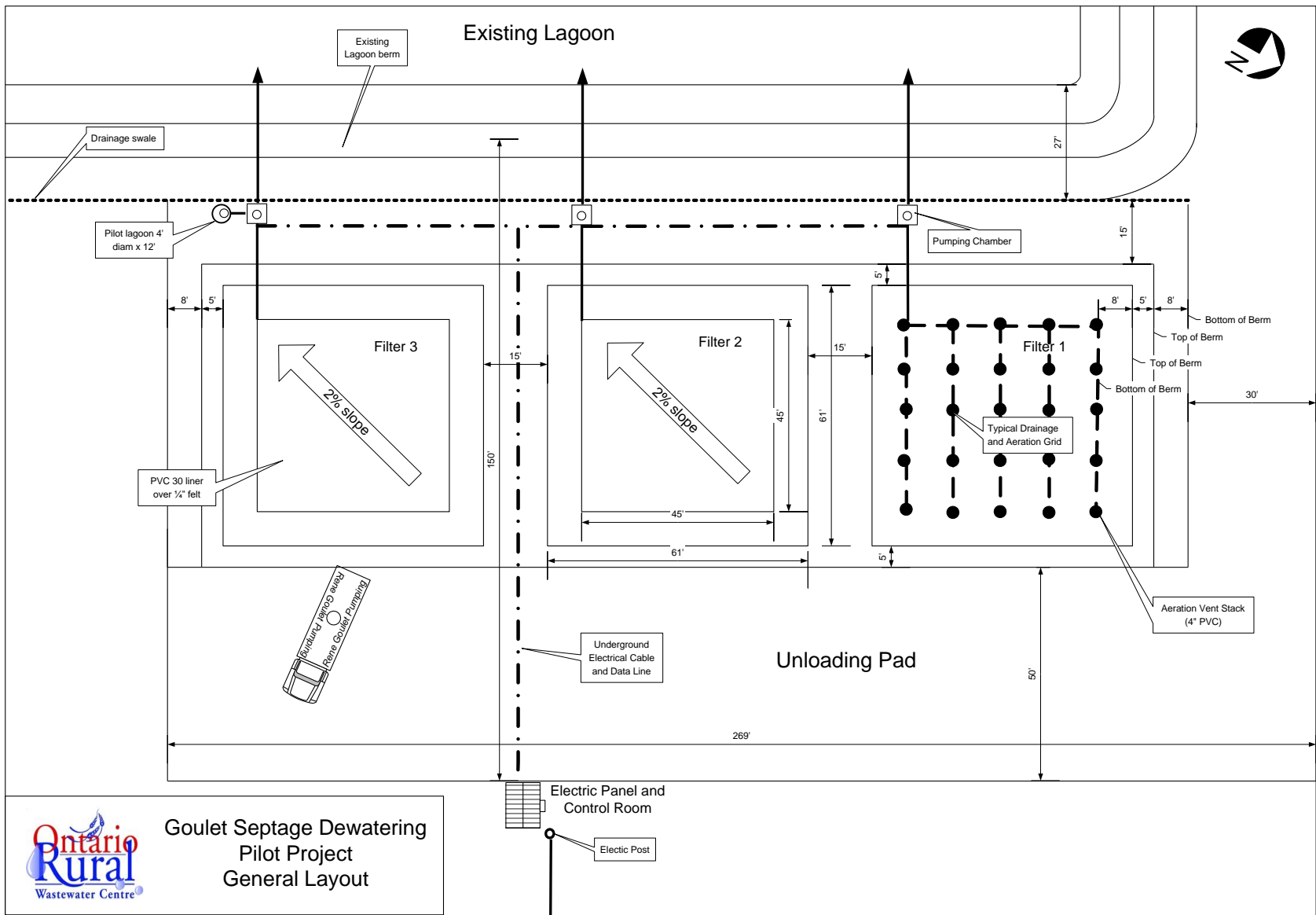


Figure A-1. Goulet Pilot Septage Reed Bed Plan View

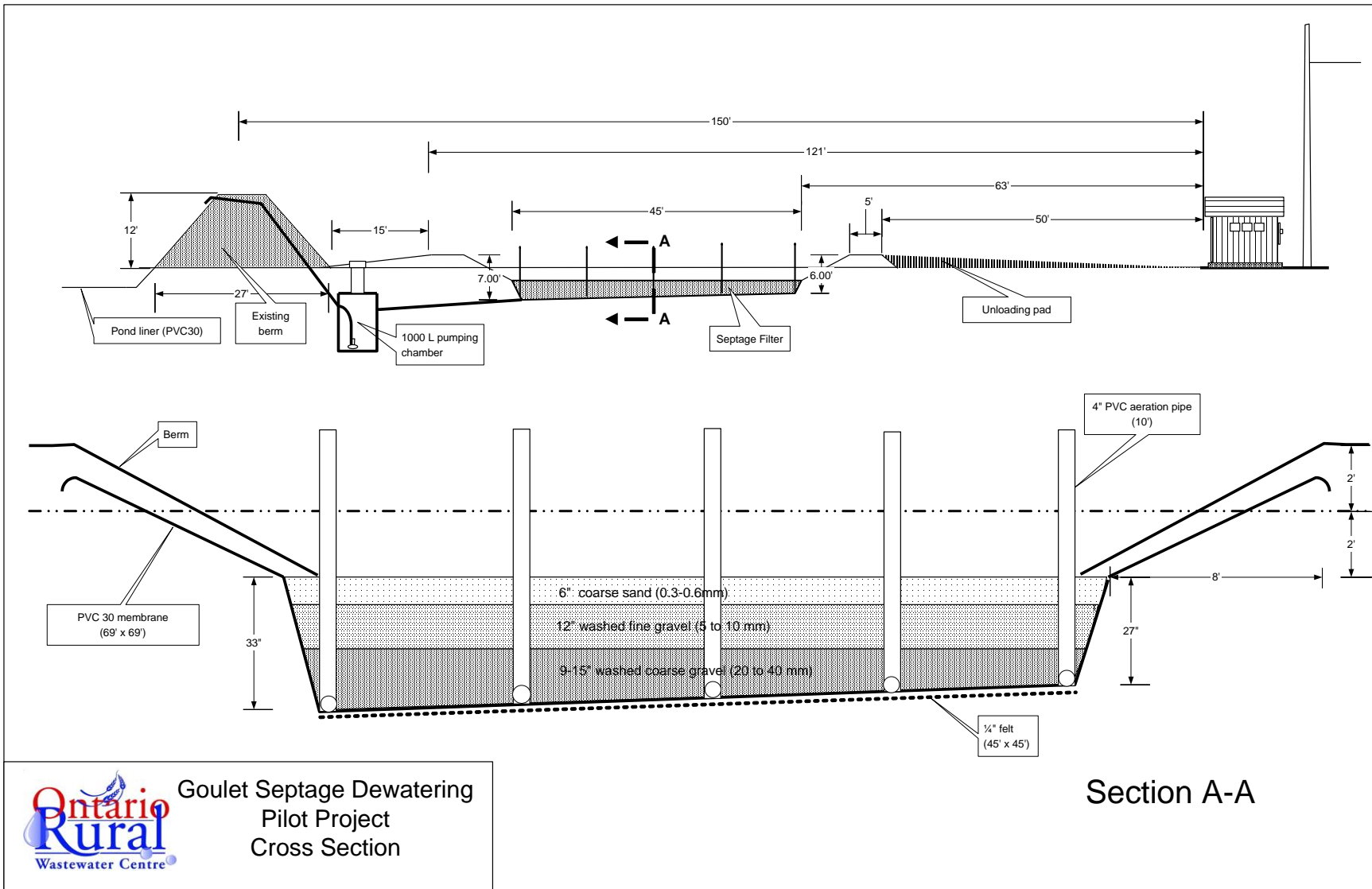


Figure A-2. Goulet Pilot Reed Bed Filter Cross Section View

The beds were excavated using an excavator and bulldozer. Concrete pump chambers were installed at the toe of each bed with a drainage line run from each pump chamber to each bed. The pump chambers were then backfilled. A 6.4 mm (1/4 inch) non-woven geotextile from Layfield was laid on each bed floor to protect the liner from puncture. Next, a 30 mil geomembrane (Layfield Tantalum 5-30 mil) was laid out with the edges rolled around 2x4 boards which were spiked into the berms to fasten the liners. Another layer of geotextile was placed along the berm slopes to protect the geomembrane from sunlight. A series of 10 cm (4 inch) perforated PVC drainage lines at 1.5 m (5 foot) spacing were installed sloping to a footer line which was run through the geomembrane at the bottom corner of the filter and connected to the pump chamber. The geomembrane perforation was sealed with tar. Non-perforated 10 cm (4 inch) PVC aeration stacks were connected to the drainage network at the ends and middle of each line. The gravel and sand layers were then added with the use of a stone slinger. Sections of *Phragmite* rhizomes, harvested from a nearby ditch, were planted in the sand layer at 4 rhizome segment per m<sup>2</sup>. Construction photos are presented below in Figure A-3.



**Installed pump chamber with excavated bed (1)**



**Geotextile laid out (2)**



**Liner unrolled (3)**



**Unfolding the liner (4)**



**Liner laid out (5)**



**Liner fastening (6)**



**Placing gravel in base of filter (7)**



**Placing Sand in Filter (8)**



**Geotextile to protect liner from UV rays (9)**



**Completed filter with planted reeds (10)**



**Reed Bed Filter (2010) (11)**



**Sand Filter operating as a freezing bed (12)**

**Figure A-3. Reed Bed Construction Photos**

The reed beds and sand drying bed filter were dosed directly from the septage truck using the truck's vacuum pump. Each full load of septage was approximately 3000 gallons (13,600 L) and represented a dose of 7.3 cm. The septage first passes through a 1.0 cm (3/8 inch) bar screen to remove large non-biodegradable objects such as gravel, hair, rags, plastic materials, etc. The hose from the truck was connected to the bar screen filter with a quick connect coupling. The screened septage flowed by gravity through a 15 cm (6 inch) PVC pipe to one of the beds. The pipe was placed on an inner tube to float on the accumulating sludge layer. Photos of a bar screen filter and dosing pipe are presented below in Figure A-4.



Figure A-4. Septage Screening and Dosing Pipe Photos

## Appendix B – Analytical Methods

### Ontario Rural Wastewater Centre Environmental Quality Laboratory - Campus d'Alfred-University of Guelph Analytical Methods

**Ammonia (as  $\text{NH}_4^+\text{-N}$  +  $\text{NH}_3\text{-N}$ ):** Wastewater samples analyzed using the Ammonia-Selective Electrode Method outlined in Standard Methods for the Examination of Water and Wastewater (APHA, 2005) (SM No. 4500-NH<sub>3</sub> D.). This methodology has a MDL = 0.02 mg/L.

**Biochemical Oxygen Demand (BOD<sub>5</sub>):** Wastewater samples analyzed using the 5-Day BOD Test outlined in Standard Methods for the Examination of Water and Wastewater (APHA, 2005) (SM No. 5210 B.). This methodology has a MDL = 10 mg/L.

**Chemical Oxygen Demand (COD):** COD was determined using the Standard Methods for the Examination of Water and Wastewater (APHA, 2005) (SM No. 5220 D. Closed Reflux, Colorimetric Method.) sCOD samples were pre-treated by centrifugation at 3800g and filtered at 0.45  $\mu\text{m}$  prior to COD analysis.

**Dissolved Oxygen (DO):** Dissolved oxygen measurements were taken at the Campus d'Alfred wet chemistry laboratory within 1 hour of the samples arriving to the lab. The Membrane Electrode Method outlined in Standard Methods for the Examination of Water and Wastewater (APHA, 2005) (SM No. 4500-O G.) was used to measure dissolved oxygen. The methodology has an MDL = 0.1 mg/L.

**Indicator Pathogens (*E. coli*, *Salmonella*, *C. perfringens*, *Enterococci*):** Bacteria samples processed by adding 10g (or 10mL) of sample to 90 mL of sterilized phosphate buffer and shaken at 40 rpm for 5 minutes. The diluted sample then analyzed using the following methods:

*E. coli*: Difco m-FC Basal Medium with BCIG Method.

*Salmonella*: Difco SS Agar-Salmonella Shigella Agar Method.

*C. perfringens*: Difco SFP Agar Base Egg Yolk Enrichment 50% Antimicrobial Vial K-Antimicrobial Vial P Method.

*Enterococci*: Difco KF Streptococcus Agar TTC Solution 1% Method.

**Nitrate (as N-NO<sub>3</sub><sup>-</sup>):** Wastewater samples analyzed using the Nitrate Electrode Method outlined in Standard Methods for the Examination of Water and Wastewater (APHA, 2005) (SM No. 4500-NO<sub>3</sub><sup>-</sup> D.). This methodology has a MDL = 0.14 mg/L.

**Ortho-Phosphates (as O-PO<sub>4</sub><sup>3-</sup>):** Ortho-phosphate was analyzed at the Campus d'Alfred wet chemistry laboratory. The wastewater samples were analyzed using the Ascorbic Acid Method outlined in Standard Methods for the Examination of Water and Wastewater (APHA, 2005) (SM No. 4500-P E.). This methodology has a MDL= 0.03 mg/L.

**pH:** pH measurements of the wastewater samples taken to the Campus d'Alfred wet chemistry laboratory within 1 hour of the samples arriving at the lab. The Electrometric Method outlined in Standard Methods for the Examination of Water and Wastewater (APHA, 2005) (SM No. 4500-H+ B.) was used to measure pH. This methodology has an MDL = 0.1.

**Temperature:** Temperature readings of wastewater samples taken at the wetland site using a Fisherbrand Traceable metal digital thermometer. The thermometer is accurate to ± 1 °C and has a resolution of 0.1 °C. The thermometer was calibrated by the manufacturer according to the standards provided by the National Institute of Standards and Technology (NIST).

**Total Dissolved Solids (TDS):** Total dissolved solids were analyzed at the Campus d'Alfred wet chemistry laboratory. The wastewater samples were analyzed using the Total Dissolved Solids Dried at 180°C Method outlined in Standard Methods for the Examination of Water and Wastewater (APHA, 2005) (SM No. 2540 C.). This methodology has a MDL = 2mg/L.

**Total Kjeldahl Nitrogen (TKN):** The wastewater samples were analyzed using the Macro Kjeldahl Method outlined in Standard Methods for the Examination of Water and Wastewater (APHA, 2005) (SM No. 4500 N<sub>org</sub> B.).

**Total Phosphorus (TP):** The wastewater samples were analyzed using the Persulfate Digestion Method followed by The Ascorbic Acid Method as outlined in Standard Methods for the Examination of Water and Wastewater (APHA, 2005) (SM No. 4500-P B. 5). This methodology has a MDL = 0.03 mg/L.

**Total Solids (TS) and Volatile Solids (VS):** Total solids and volatile solids were determined using the Standard Methods for the Examination of Water and Wastewater (APHA, 2005) (SM 2540G. Total, Fixed and Volatile Solids in Solid and Semisolid Samples Method).

**Total Suspended Solids (TSS):** The wastewater samples were analyzed using the Total Suspended Solids Dried at 103-105 °C Method outlined in Standard Methods for the Examination of Water and Wastewater (APHA, 2005) (SM No. 2540 D.). This methodology has a MDL = 2 mg/L.

**Ontario Ministry of Environment Laboratory Services Branch Analytical Methods (SCC, 2013)**

**pH:** Method PHSOIL-E3137 – The Determination of pH in Soil and Dried Sludge by Potentiometry.

**Solids:** Method SOLIDS-E3188 – The Determination of Solids in Liquid Matrices by Gravimetry

**Conductivity, pH, Alkalinity:** Method WATS-E3218 – The Determination of Conductivity, pH, and Alkalinity in Water and Effluents by Potentiometry

**TKN & TP:** Method STKNP-E3368 – The Determination of Total Kjeldahl Nitrogen and Total Phosphorus in Water, Sewage, Leachate and Industrial Waste by Colouimetry

**Moisture Content:** Method PHYSOLID-E3139 – The Determination of Moisture Content, RST, TSTA and LOI in Solids by Gravimetry

**BOD:** Method SBBOD-E3182 – The Determination of Biochemical Oxygen Demand in Surface Water and Sewage by Dissolved Oxygen Meter

**Nutrients:** Method SDNP-E3366 – The Determination of Ammonia Nitrogen, Nitrite Nitrogen, Nitrite+Nitrate Nitrogen and Reactive Ortho-Phosphate in Water, Sewage, Leachate and Industrial Effluents by Colourimetry

**Cations:** Method PRAA400-E3146 – The Determination of Cations in Atmospheric Deposition by Atomic Absorption Spectrophotometry (AAS)

**As, Se, Sb:** Method HYDSWG-E3091 – The Determination of Arsenic, Selenium and Antimony in Sewage and Sludges by Hyride-Flameless Atomic Absorption Spectrophotometry (HYD-FAAS)

**Mercury:** Method HGSSV-E3059 – The Determination of Mercury in Soils, Sediments and Vegetation by Cold-Vapour –Atomic Absorption Spectrophotometry (CV-AAS)

**Heavy Metals:** Method HMPNSOIL-E3075 – The Determination of Heavy Metals in Soils and Sediments by Atomic Absorption Spectrophotometry (AAS)

**Metals:** Method HMRAWSWG-3181 – The Determination of Metals in Raw Sewage by Inductively Coupled Plasma-Atomic Emission Spectroscopy (ICP-AES)

**Metals:** Method SCANICP-E3094 – The Determination of Metals in Final Effluent, Industrial Waste and Landfill Leachates by Inductively Coupled Plasma-Atomic Emission Spectroscopy (ICP-AES).

**Metals:** Method E3470 – The Determination of Metals in Solid Matrices and Extracts Using Hot Block Digestion and Inductively Coupled Plasma-Optical Emission Spectrometry (ICP-OES)