

**EVALUATION OF LANDFILL LEACHATE AND WASTE DUMPING  
SITE SOIL TREATMENT BY USING CHEMICAL AND BIOLOGICAL  
METHODS**

By

Qian Xu

A Thesis submitted to the Faculty of Graduate Studies of  
The University of Manitoba  
in partial fulfillment of the requirements of the degree of

MASTER OF SCIENCE

Department of Civil Engineering

University of Manitoba

Winnipeg, MB

Canada

Copyright © 2018 by Qian Xu

## Abstract

Solid waste management and landfill leachate has been investigated widely for waste treatment and environmental safety. This study aimed to 1) remove organic contaminants from landfill leachate by sole or combined biological and chemical methods, 2) investigate the solid waste management situations in Canadian First Nations communities, 3) evaluate the feasibility of landfill leachate for plants irrigation, and 4) remove nutrients and salinity from leachate and soil by means of phytoremediation.

Aerobic sequencing batch reactor (SBR) with activated sludge was used as the biological treatment to remove ammonium. The chemical treatments included ozone ( $O_3$ ), ozone + hydrogen peroxide ( $O_3 + H_2O_2$ ), Fenton's reagents ( $H_2O_2 + Fe^{2+}$ ), and ozone + Fenton's reagents ( $O_3 + H_2O_2 + Fe^{2+}$ ). Results indicated that ozonation of raw leachate achieved 16% COD removal after 240 min of treatment while  $O_3 + H_2O_2$  achieved 33% COD removal of raw leachate with 900 mg/L  $H_2O_2$  in 120 min. Moreover, the COD removal from raw leachate was only 33% by using Fenton's reagents. After biological stabilization, ozonation removed 52% of the COD within 240 min. The Fenton's reagents achieved 67% COD removal using equal doses of  $H_2O_2$  and  $Fe^{2+}$ . However, Fenton's reagents in conjunction with  $O_3$  removed 69% and 72% COD in 120 min and 240 min, respectively. The dosage test also revealed that the 2:1 ratio of  $H_2O_2$  and  $Fe^{2+}$  had the best COD removal.

Two native halophytes *Puccinellia nuttalliana* (Alkaligrass) and *Typha latifolia* (Cattail) were selected in this study to investigate bioaccumulation of Sodium ( $Na^+$ ) and Chloride ( $Cl^-$ ) in greenhouse condition. Different treatments were applied on same soil condition, namely control (C), fertilizer (F) and diluted landfill leachate (LL). The results shown that alkaligrass and cattail accumulated  $6.86 \pm 0.96$  and  $7.00 \pm 0.20$  g  $Na^+$ /Kg biomass with the irrigation of LL, respectively.

Alkaligrass and cattail irrigated with LL accumulated 120.15% and 94.48% more  $\text{Cl}^-$  than control. Another test by using cattail to remediate polluted soil (PS) from waste dumping site shown that electrical conductivity (EC) of PS was decreased from  $245.0 \pm 1.4$  ms/m to  $51.9 \pm 9.3$  ms/m.  $\text{Na}^+$  and  $\text{Cl}^-$  content in cattail grown on PS was  $10.82 \pm 1.85$  and  $64.69 \pm 9.15$  g/Kg biomass, respectively. The hydroponic test showed that  $\text{Na}^+$  and  $\text{Cl}^-$  accumulation was  $55.53 \pm 4.82$  and  $78.22 \pm 28.28$  g/Kg biomass.

Overall, these results concluded that the combination of biological and post-chemical treatment methods can effectively remove organic contaminants from mature landfill leachate and the selected plants can significantly accumulate  $\text{Na}^+$ ,  $\text{Cl}^-$  and nutrient in their biomass.

## **Acknowledgement**

This thesis and related work was carried out with all the kindly help and support from many individuals, and I would like to express my deepest gratitude to all of them.

First of all, I would like to extend tremendous thanks to my supervisor, Dr. Qiuyan Yuan, who gave me this great opportunity to conduct research in the field of Environmental Engineering.

Without her instruction and immense encouragement, it would be impossible for me to finish this work.

I would also like to thank my committee member Dr. Sylvie Renault and Dr. Giulio Munz, for all the help during my research and this thesis.

In addition, I would like to express my thanks to our lab technician Dr. Victor Wei and also my colleagues, Yanan Ren, Cornelia Andreea Badila, Md. Mofizul Islam, Huijun Jia, Mengmeng Tian, Abbass Kang and Giovanna Siracusa, for all the help.

I also own my deepest gratitude to my family, for the understanding and all the support.

Finally, I am grateful to the financial support from CREATE H2O program and University of Manitoba.

## Table of Contents

Chapter 1 Introduction & Objectives .....	1
Chapter 2 Literature Review .....	3
2.1. Landfill leachate characteristic.....	3
2.2. Treatment methods.....	4
2.2.1. On-site treatment.....	4
2.2.2. Combined and treated with domestic wastewater.....	4
2.2.3. Biological methods .....	5
2.2.4. Physicochemical treatment .....	7
2.2.5. Phytoremediation .....	12
Chapter 3 COD Removal from Biologically Stabilized Landfill Leachate Using Advanced Oxidation Processes (AOPs).....	14
3.1. Introduction .....	14
3.2. Methods and materials .....	16
3.2.1. Landfill leachate.....	16
3.2.2. Treatment and experimental setup .....	17
a) SBR .....	17
b) Ozonation setup.....	18
c) <i>Vicia faba</i> ecotoxicological plant assay .....	18
3.2.3. Analysis.....	19

3.3.	Results & discussion .....	20
3.3.1.	Biological stabilization: NH <sub>4</sub> -N and COD removal via SBR .....	20
3.3.2.	<i>Vicia faba</i> ecotoxicological plant assay .....	22
3.3.3.	COD removal by AOPs .....	23
3.4.	Conclusion.....	30
Chapter 4 Phytoremediation of Landfill Leachate by Using <i>Puccinellia Nuttalliana</i> and <i>Typha</i>		
<i>Latifolia</i> .....		
4.1.	Introduction .....	31
4.2.	Methods and materials .....	32
4.2.1.	Plants species and germination .....	32
4.2.2.	Treatments.....	33
4.3.	Results & discussion .....	36
4.3.1.	Soil (pot) test.....	36
a)	Alkaligrass and cattail biomass yield.....	36
b)	Nutrient uptake.....	40
c)	EC deduction from soil .....	42
d)	Na <sup>+</sup> and Cl <sup>-</sup> phytoaccumulation.....	43
4.3.2.	Hydroponic cattail.....	46
4.4.	Conclusion.....	48
Chapter 5 Engineering Significance .....		
49		

Appendix 1. Solid Waste Management in First Nations Communities .....	50
1. Introduction.....	50
2. Solid waste management in First Nations communities .....	52
2.1. Open-site dumping and open-air burning.....	52
2.2. Household hazardous waste.....	55
3. Policy & legislation framework and government funding: from very beginning.....	57
3.1. From history and federal aspect.....	57
3.2. The Canadian Environmental Assessment Act .....	58
3.3. The Canadian Environmental Protection Act.....	58
3.4. Canadian Council of Ministers of the Environment.....	58
3.5. Governmental funding.....	59
4. Conclusion and recommendation.....	62
Appendix 2. Phytoremediation of Polluted Soil from Waste Open Dumping Site- A Case Study for Northern Manitoba First Nation Community.....	64
1. Introduction.....	64
2. Methods and materials .....	65
2.1. Plant species and germination .....	65
2.2. Treatments .....	65
2.3. Analysis .....	66
3. Results and discussion .....	67

4. Conclusion .....	70
References .....	71

## List of Tables

Table 2. 1 Leachate classification.....	3
Table 3. 1 Characterization of the raw leachate and primary effluent.....	16
Table 3. 2 Root elongation (RE), germination index (GI), mitotic index (MI), and micronuclei (MN) of <i>Vicia faba</i> seeds treated with deionized water (Control) and effluent collected from the aerobic SBR with activated sludge .....	23
Table 4. 1 $F_v/F_m$ results from Chlorophyll Fluorescence test. 4 replicates were applied one month after the seventh week. Values are the means $\pm$ standard deviation. ....	39
Table 4. 2 $Na^+$ and $Cl^-$ accumulation by alkaligrass and cattail after the last harvest. Values are mean $\pm$ standard deviation. ....	43
Table 4. 3 Phytoaccumulation by hydroponic cattail. Samples were harvested after 5 weeks' study. Values are mean $\pm$ standard deviation.....	46

## List of Figures

Figure 3. 1 Schematic of an aerobic SBR.....	17
Figure 3. 2 Schematic of ozone-based tests.....	18
Figure 3. 3 Quality parameters (TSS, VSS, and SVI) with varying leachate ratio obtained via aerobic SBR.....	21
Figure 3. 4 NH <sub>4</sub> -N removal via biological treatment.....	21
Figure 3. 5 COD removal by biological stabilization.....	22
Figure 3. 6 COD removal efficiency of raw leachate and bio-stabilized leachate via ozonation.	24
Figure 3. 7 COD reduction by O <sub>3</sub> and O <sub>3</sub> /H <sub>2</sub> O <sub>2</sub> treatments.....	25
Figure 3. 8 COD removal efficiency for leachate with different initial COD concentration via Fenton's reagents.....	26
Figure 3. 9 COD removal of bio-stabilized leachate via Fenton's reagents.....	27
Figure 3. 10 Effect of H <sub>2</sub> O <sub>2</sub> /Fe <sup>2+</sup> on COD removal.....	28
Figure 3. 11 Effect of Fe <sup>2+</sup> /H <sub>2</sub> O <sub>2</sub> /O <sub>3</sub> process on COD removal.....	29
Figure 4. 1 Alkaligrass seedlings.....	33
Figure 4. 2 Cattail seedlings.....	33
Figure 4. 3 Plants mean growth in height under C, F and LL during stage 1 and stage 2 (a) alkaligrass (b) cattail. Height was measured weekly during 12 weeks' study. Values are the means ± standard error (error bars).....	37
Figure 4. 4 Plants mean biomass yield under C, F and LL during stage 1 and stage 2 (a) alkaligrass (b) cattail. Biomass was measured 5 times and 7 times for alkaligrass and cattail respectively, by triplicates. Values are the means ± standard error (error bars).....	38

Figure 4. 5 Alkaligrass and cattail nutrient content (DW) after the last harvest (a) & (b) alkaligrass, (c) & (d) cattail. Values are the means $\pm$ standard deviation.....	41
Figure 4. 6 Nutrient distribution of alkaligrass in roots and aboveground tissues (a) nitrogen (b) phosphorous.....	41
Figure 4. 7 Nutrient distribution of cattail in roots and aboveground tissues (a) nitrogen (b) phosphorous.....	42
Figure 4. 8 EC deduction from soil by alkaligrass (a) and cattail (b) after the final harvest. Values are the means $\pm$ standard deviation.....	43
Figure 4. 9 Distribution (roots and aboveground tissues) of Na <sup>+</sup> in plants (a) alkaligrass (b) cattail.....	45
Figure 4. 10 Distribution (roots and aboveground tissues) of Cl <sup>-</sup> in plants (a) alkaligrass (b) cattail.....	45
Figure 4. 11 Hydroponic cattail mean growth in height during 5 weeks' study. Data was recorded with 3 replicates (6 plants for each system, n=18). Values are mean $\pm$ standard deviation. ....	46
Figure 4. 12 NH <sub>4</sub> -N removal via hydroponic cattail with different feeding load. T1, T2 and T3 were 10%, 20% and 30% leachate by volume. Feed was collected right after treatment and effluent was collected after one week retention time. Values are mean $\pm$ standard deviation for 3 replicates. ....	47

## List of Abbreviations

AC	Activated carbon
AOB	Ammonia-oxidizing bacteria
AOPs	Advanced oxidation processes
BOD	Biological oxygen demand
COD	Chemical oxygen demand
DBPs	Disinfection by-products
DO	Dissolved oxygen
DOM	Dissolved organic matter
EC	Electrical conductivity
GI	Germination index
HA	Humic acid
HOCs	Hydrophobic organic chemicals
HS	Humic-like substances
MLTSS	Mix liquor total suspended solid
MLVSS	Mix liquor volatile suspended solid
MSW	Municipal solid waste
NOB	Nitrite-oxidizing bacteria
SBR	Sequencing batch reactor
SVI	Sludge volumetric index
VFA	Volatile fatty acid
WWTPs	Municipal wastewater treatment plants

## **Chapter 1 Introduction & Objectives**

The production of solid waste is increasing rapidly due to fast urbanization and economic growth. In 2012, the worlds' cities generated 1.3 billion tonnes of solid waste, which is expected to rise to 2.2 billion tonnes by 2025 (The World Bank, 2017). Currently, most of the municipal solid waste (MSW) is disposed in landfills, and inappropriate selection of disposal sites and management can result in some environmental hazardous, especially in developing countries and remote areas. Landfilling is an economically feasible approach for the disposal of MSW, however, the leachate generated contains complexed components such as inorganic salts, organic contaminants and heavy metals which are potentially hazardous for the surrounding surface water, ground water, soil and biota. Bortolotto and et al. (2009) evaluated the toxic and genotoxic potential of landfill leachates by using bioassays and their results revealed that the leachate contains genotoxic compounds and exerted mutagenic effects. Li et al. (2007) found out leachate could have genotoxic and physiological effect on plant cells as certain concentrations, and organisms could be cytogenetically damaged when they are exposed to leachate contaminated aquatic environment.

Commonly, chemical oxygen demand (COD) is used as parameter to monitor leachate quality and concentration of organic contaminants (Ghosh and et al., 2017). Techniques, such as biological, and physicochemical methods, have been developed for treatment of landfill leachate and COD deduction, while the methods applied should consider the leachate characteristics since it varies with landfilling age, waste composition and climatic conditions (Ghosh and et al., 2014). It is suggested that landfill leachate could be utilized as fertilizer for plants irrigation because of its nutrient content (Hernandez et al., 1999; McBride 1989). Not only nutrient in landfill leachate, the salinity of leachate is also of special interest, because the salinity could pose

negative effect on plants growth. Therefore, it is significant to study the relationship between plant stress and nutrient accumulation under saline conditions of leachate (Hernandez et al., 1999).

In this study, landfill leachate and soil from waste dumping site was collected for the following objectives:

- Using aerobic sequencing batch reactor (SBR) and chemical methods to remove COD
- Investigating current solid waste management situation in Canadian First Nations communities
- Applying landfill leachate as irrigation for two selected native plants to study nutrient uptake, biomass yield and desalination performance
- Testing the feasibility of leachate as nutrient solution for hydroponic system

## Chapter 2 Literature Review

### 2.1. Landfill leachate characteristic

Microbial and physicochemical degradation of organic matters from landfill waste in conjunction with rainwater penetration results in liquid effluent which is defined as landfill leachate (Gavrilescu and Schiopu, 2010). Since landfill leachate varies greatly due to different climate, seasons, waste types and landfilling years, it may contain large portion of heavy metals, inorganic matters, organic contaminants which can pose threat to surrounding environment and human health. Therefore, stringent regulations and proper engineering management is vitally important. Although landfill leachate varies greatly, there are four successive stages namely aerobic, acetogenic, methanogenic and stabilization for the generation of leachate during landfilling, and the leachate can be classified into three types (Table 2.1) (Baig et al., 1999; Renou et al., 2008). In the early stage of landfilling, there are large portions of readily degradable organics and VFA (volatile fatty acid) due to the anaerobic fermentation and the BOD<sub>5</sub>/COD ratio is relatively higher which means desirable bio-degradability.

Table 2. 1 Leachate classification

Parameters	Young	Intermediate	Old
Landfill age (years)	<5	5-10	>10
pH	<6.5	7	>7.5
COD (mg/L)	>20,000	3,000-15,000	<2,000
BOD <sub>5</sub> /COD	>0.3	0.1-0.3	<0.1
Organic matter	70-90% VFA	20-30% VFA	HMW

“VFA: volatile fatty acids; HMW: high molecular weight humic and fulvic-like material.”  
Source: Baig et al., 1999.

Over time, the landfills become mature which is also called methanogenic stage and during this period VFA is converted to biogas. As a result, the organics maintained in leachate are mainly non-degradable like humic-like substances (HS) (Renou et al., 2008; Harmsen 1983). The chemical and biological process during landfilling leads to the formation of HS which are non-biodegradable anionic macromolecules from 1 kDa MW-FA to 10 kDa MW-HA and it makes up more than 80% of the total organics (Xu et al., 2006; Kang et al., 2012; Zhang et al., 2016).

## **2.2. Treatment methods**

Leachate collected from engineered landfills is usually managed by the following ways (Schiopu et al., 2010):

- On-site treatment
- Discharge to sewage system
- Transport off-site treatment

### **2.2.1. On-site treatment**

Recycling is one of the cheapest treatment methods for landfill leachate by transferring the leachate back through the waste tips. Since moisture content is critical to the microorganism activity and the leachate recirculation is beneficial for nutrient redistribution among the waste (Bae et al., 1998). Significant COD removal and shorter period of leachate stabilization is also observed by other studies (Reinhart et al., 1996; Renou et al., 2008; Rodriguez et al., 2004).

### **2.2.2. Combined and treated with domestic wastewater**

Another extensively used method for treatment of landfill leachate is transporting it to municipal wastewater treatment plants (WWTPs) and treated together with domestic wastewater. This is adopted because of low operation cost and easy maintenance. However, some studies have reported that the combination can negatively affect effluent quality and the complexed

components also interfere with disinfection process for WWTPs using UV-disinfection technique (Zhao et al., 2012). Brennan et al. (2017) also found out that with intermediate aged leachate loaded with volumetric ratio of 4% or 50% of total WWTP  $\text{NH}_4\text{-N}$  load could result in a noticeable decrease in nitrification and account for a larger portion of WWTPs aeration consumption.

### **2.2.3. Biological methods**

Biological methods have been widely conducted to treat young landfill leachate which has higher biological oxygen demand (BOD) concentration. Microorganisms are able to biodegrade the organic matters into carbon dioxide and biogas under different conditions. However, biological method is limited for the treatment of mature landfill leachate for COD deduction because of the low biodegradability of recalcitrant compounds. Depends on the requirement of oxygen or not, the biological processes include aerobic and anaerobic processes.

#### **a) Aerobic process**

Aerobic processes based on suspended-growth biomass, like aerobic lagoon, activated sludge, and SBRs, have been approved to be effective to remove nutrient and organic matters for immature leachate treatment by nitrification, denitrification and mineralization (Abbas et al., 2009). Aerated lagoons with supply of surface aerator or diffuse bubble aeration are widely used in remote small communities and also some developing countries because of its cost-efficiency and operation-convenience. Govahi et al. (2012) examined the feasibility of using aerobic upflow sludge blanket and subsequent aerated lagoon to treat municipal landfill leachate and achieved 54% ammonium removal and 84% COD removal. Mahlum (1995) conducted a research on treatment of leachate by using lagoon and constructed wetland and achieved overall N, P, Fe, and pathogen removal around 70-90%. Mehmood et al. (2009) assessed the treatment efficiency by

microbial oxidation of on-site aerated lagoon at a landfill site and their results shown that the COD removal was around 75% and 80% of nitrogen in the feed was removed.

Activated sludge process is another widely used method for wastewater and leachate-sewage treatment due to the advantage of low aeration consumption and low C/N ratio requirement for partial nitrification and denitrification (Chen et al., 2016). This is achieved by accumulating excess nitrite via AOB (ammonia-oxidizing bacteria) and NOB (nitrite-oxidizing bacteria) (Ge et al., 2015). Chen et al. (2016) established a pilot-scale continuous activated sludge process to treat old landfill leachate under low DO (dissolved oxygen) (0.1-0.5 mg/L) and stable ammonium removal efficiency (95%) was obtained. Tamrat et al. (2012) used activated sludge to stabilize leachate with high organic load (42,310 mg/L) and their results shown the aerobic process removed 64% COD compared to 40% removal by anaerobic process. Even though this process was shown be efficient for COD, nitrogen and nutrient deduction, some drawbacks were also reported as following (Abbas et al., 2009; Renou et al. 2008).

- Low sludge settleability and longer aeration time
- High energy consumption and excess sludge production
- Microbial inhibition from high strength of ammonium- nitrogen

A new biological technology has drawn a lot of research interest recently and been developed is aerobic granular sludge which is formed from activated sludge with dense structure. Aerobic granular sludge has advantages including high metabolic ability, excellent settleability and simultaneous nitrification and denitrification (SND) (Wei et al., 2012). Iaconi et al. (2007) investigated the effectiveness of municipal wastewater and mature landfill leachate treatment by using aerobic granular biomass. Their results shown that 90% COD, 90% nitrogen and 90%

suspended solid was removed from the municipal wastewater and entire content of biodegradable COD was removed from landfill leachate with lower sludge production than conventional systems.

b) Anaerobic processes

Anaerobic digestion process has also drawn increased attention for wastewater treatment because of low energy demand, less solid yield and valuable biogas production. The anaerobic digestion is driven by a varied group of microbes which convert hydrocarbons into CO<sub>2</sub> or CH<sub>4</sub> (Speece 1996; Kawai et al., 2012). Kawai et al. (2012) conducted a lab-scale research by using UASB (upflow anaerobic sludge blanket) to treat synthetic leachate and raw leachate. Their results shown that the COD removal was higher than 95% and the methane accounted for 60% of the biogas production for synthetic leachate, but the COD removal decreased to 40% after switching to raw leachate. This phenomenon may result from the complexed components in landfill leachate such as ammonium ions, heavy metals and high salinity which can interfere with the functional microbes (Alkalay et al., 1998).

#### **2.2.4. Physicochemical treatment**

Both biological and physicochemical treatments have been employed for treatment of wastewater and landfill leachate to remove organic contaminants and ammonium, while biological methods are more favored for treatment of young leachate with high biodegradability and physical-methods have been more recommended for treatment of old leachate containing large portion of recalcitrant compounds.

a) Adsorption

There are numerous materials have been employed for the treatment of wastewater such as zeolite, illite, vermiculite, activated carbon (AC), and municipal waste incinerator bottom ash (Amokrane et al., 1997). It is reported that AC is the most efficient absorbent for organics removal for gaseous or aqueous phase, and zeolite has been widely used to remove inorganic contaminants from wastewater and leachate (Halim et al., 2010). Song et al. (2009) used 3g/L GAC (granular activated carbon) and PAC (powdered activated carbon) for the treatment of biologically pretreated landfill leachate, and the results have shown that hydrophobic organic chemicals (HOCs) and COD removal were 73.4%, 89.2% and 19.1%, 24.6%, respectively. Even though it has been revealed that AC is able to remove COD for wastewater treatment, the lacking of polar surface limits its capacity for ammonia absorbance. Therefore, some new composite materials, such as zeolite-carbon (Z-C), have drawn increasing research interest. Halim et al. (2010) exhibited that a combination of AC and zeolite can provide both hydrophobic and hydrophilic surface, which achieved 24.39 mg/g ammonia and 22.99 mg/g COD absorbance. Gao et al. (2005) also reported that the Z-C composites may be applied widely as high value-added humidity-controlling building material which can be fabricated from industrial wastes.

#### b) Air stripping

Air stripping has been reported effectively for ammonia removal from landfill leachate, and also other types of wastewaters, such as those from fertilizer industry, anaerobic digestion, pig slurry or food waste (Ferraz et al., 2013). This technique is based on mass transfer and is favored by alkali conditions. Nurisepehr et al. (2012) studied the sequencing treatment of landfill leachate by combining ammonia stripping and subsequent some other methods, the results indicated that the removal of  $\text{NH}_4\text{-N}$  improved in line with the increasing pH and the highest removal (57.4%) was obtained at pH 10.5. However, a higher pH leads to sludge formation and increases

operation cost arising from extra sludge disposal (Wang et al., 2010). Meanwhile, the  $\text{NH}_3$  from the stripping process can have potential hazardous on air quality, and it may also require large scale stripping facility when lime is employed for pH adjustment (Renou et al., 2008)

#### c) Membrane filtration

Membrane filtration is a separation method using semipermeable membrane to separate influent into two parts: permeate passing thorough the membrane and retentate consisting of components left behind (Mallevalle et al., 1996; Zhou and Smith, 2001). The most common types are based on pressure-driven process including RO (reverse osmosis), NF (nanofiltration), MF (microfiltration), and UF (ultrafiltration) (Zhou and Smith, 2001). Because of its modular designs, automatic operation and small footprints, membrane has been extensively employed for water and wastewater treatment (Koyuncu et al., 2001). However, concerns about the fouling problem, membrane lifetime and high energy input also exist.

#### d) Membrane bioreactor

Membrane bioreactor (MBR) has recently emerged as a promising technology for water and wastewater treatment. MBR has some advantages including better stability, lower sludge production, increased biomass retention (Ahmed and Lan, 2012). MBR consists of two essential parts, one part is bioreactor which is used to biodegrade contaminants, and the other part, membrane module, is responsible for separating treated wastewater from biomass. Studies have noted that MBR can achieve 90-99% of BOD removal from landfill leachate, regardless of the leachate age or operation conditions, but the COD removal efficiency has a wide range from 23% to 90%, and at least 90% ammonia removal was reported also from majority of studies (Ahmed and Lan, 2012).

e) Coagulation-flocculation

Coagulation-flocculation is usually used for pretreatment or a final polishing process to remove recalcitrant matters. It is a simple physicochemical method, but feasible to treat mature landfill leachate. Aluminum sulfate, ferric chloride, polyaluminium chloride and etc. have been commonly used as coagulants, and studies have shown mediate organic removal by using this method (Li et al., 2010). Trebouet et al. (2001) reported a 50-55% COD removal by using ferric chloride for pretreatment, but excessive sludge production is a concern and also it can cause the increase of iron or aluminium concentration.

f) Chemical oxidation

Chemical oxidation has been exhibited to be effective for the treatment of wastewater containing refractory and toxic compounds. Forgie (1988) reviewed that oxidants such as ozone, chloride, hydrogen peroxide, hypochlorite, potassium permanganate were commonly applied for landfill leachate treatment. In recently years, advanced oxidation processes (AOPs) have drawn extensive interest for high strength wastewater treatment, because of the higher oxidative potential compared to conventional chemical methods. AOPs, such as  $O_3$ ,  $O_3/H_2O_2$ ,  $O_3/UV$ ,  $H_2O_2/UV$ ,  $H_2O_2/Fe^{2+}$  and etc., are able to produce highly reactive oxidizing agents which are hydroxyl radicals ( $\cdot OH$ ). The objective and advantage of applying AOPs for old landfill leachate treatment is that AOPs can oxidize refractory compounds to end-products like carbon dioxide and water (i.e., in the case of full mineralization) or break down the high molecular matters to readily degradable substances which could be beneficial for substantial biological treatment (Wang et al., 2003).

g) Ozone and ozone related AOPs

Ozone has two mechanisms in aqueous solution, namely direct reaction and indirect reaction. In the direct reaction, molecular ozone is employed and extremely limited to aromatic compounds and unsaturated aliphatic compounds (Langlais et al., 1991). However, in the indirect reaction,  $\cdot\text{OH}$  is promoted by ozone decomposition in a chain reaction process (Staelin et al., 1984).  $\text{O}_3/\text{UV}$  is also a common practice to generate the hydroxyl radicals when UV radiation has wavelength from 200 nm to 300 nm. Langlais et al. (1991) reported that the maximum UV absorbance can be achieved by ozone at 253.7 nm. Similarly,  $\cdot\text{OH}$  can also be formed by hemolytic splitting of oxygen-oxygen bonds by UV light at 200-300 nm in the process of  $\text{H}_2\text{O}_2/\text{UV}$  (Bolton et al., 1999; Wang et al., 2003). At the present, Fenton's process or so called Fenton's reagents ( $\text{H}_2\text{O}_2/\text{Fe}^{2+}$ ) has been developed significantly for landfill leachate treatment as well as other industrial wastewater. In this process,  $\text{Fe}^{2+}$  improves decomposition of  $\text{H}_2\text{O}_2$  as catalyst to produce  $\cdot\text{OH}$  radicals. Moreover, to increase the concentration of  $\cdot\text{OH}$  radicals, photo-Fenton process by using UV light has also been conducted from studies.

Moro et al. (2018) compared UV/ $\text{H}_2\text{O}_2$  based AOPs as a final treatment or synergized with some biological methods for the treatment of intermediate landfill leachate, and the results shown that  $\text{H}_2\text{O}_2$  alone is not effective, but AOP integrated with the biological method achieved more than 80% of COD and nutrient removal. Ghazi et al. (2014) assessed the  $\cdot\text{OH}$  scavenging effect in young and old landfill leachate, and they concluded that relatively high doses of  $\text{O}_3$  and  $\text{H}_2\text{O}_2$  are required for generation of sufficient  $\cdot\text{OH}$  radicals to target on  $\cdot\text{OH}$ -selective organic micro-pollutants in old leachate and thus applying  $\text{O}_3/\text{H}_2\text{O}_2$  is prohibitively expensive. Studies reviewed that AOPs could effectively increase biodegradability of mature landfill leachate and could be considered as a feasible alternative as pre-treatment. Cortez et al. (2011) examined the performance of Fenton and ozone-based AOPs as a pre-treatment method and it was revealed

that 46% COD could be removed from the old leachate and BOD<sub>5</sub>/COD was increased to 0.15 from 0.01.

### **2.2.5. Phytoremediation**

Phytoremediation by using plants to accumulate, metabolize and stabilize substances from wastes has been conducted widely, because it has been proved to be inexpensive and effective (Ch and Romero, 2016). The development of phytoremediation involving use of the plants to transfer, remove, degrade and stabilize pollutants in sediment, soil and water, has drawn extensive research interest (Hughes et al., 1997; Padmavthiamma et al., 2014; Qadir et al., 2007; Yan et al., 2016). Halophytes are the plants able to thrive on normal habitats and also saline conditions (Hasanuzzaman et al., 2014). A lot of studies have demonstrated the feasibility of halophytes for salt-tolerance and accumulation of salts into plants biomass (Freedman et al., 2014; Glenn et al., 1999; Glenn et al., 2009; Turcios et al., 2016).

The capacity of accumulating large salt quantities also depends on the above-ground biomass, and this ability is important for plants in semi-arid and arid areas where less precipitation leads to saline stress in the rhizosphere of plants (Rabhi et al., 2010; Shiyab et al., 2003). The classification of halophytes is based on the plants mechanism, if they can exclude, accumulate or excrete salts (MsSorley et al., 2016; Yensen and Biel, 2006). Halophytes that exclude salts may increase soil salinity by absorb water and leaving salts to remain in the rhizosphere. However, halophytes that are able to excrete and accumulate salts, can remove salinity from the soil, which is feasible for phytodesalination. The accumulators can uptake salts into the biomass and finally it can approach a threshold of bioaccumulation and the harvest can remove salts from sites (MsSorley et al., 2016). Some halophytes are able to excrete extra salts by liquid and exposure to air makes it visible crystals on the plant tissue surface, and shedding of mature leaves can also

prevent salt toxicity (Mishra and Tanna, 2017). However, if the harvest is not rapid, the salts accumulated on the surface of leaves can return to the soil again.

Since most of landfills are located in rural areas, spray or trickle irrigation of landfill leachate with cultivation of vegetation has drawn study interests (Jones et al., 2006; Haarstad and Maehlum, 1999). Vasseur et al. (1997) conducted a research by irrigating cress and clover with landfill leachate, and they found out that the plants had positive responds compared to other treatments like sewage sludge. However, there are concerns about leachate irrigation, especially for human health. Jones et al. (2016) demonstrated that the harsh environment presented by leachate contains low numbers of pathogens and they are less likely to survive in soil for a long-run.

## **Chapter 3 COD Removal from Biologically Stabilized Landfill Leachate Using Advanced Oxidation Processes (AOPs)**

### **3.1. Introduction**

The disposal of MSW in landfills has always been a common practice because of its cost efficiency and operational convenience (Sfaa et al., 2013; Renou et al., 2008). However, the generation of leachate is a long-standing concern, which could release a wide range of toxic matters such as heavy metals, inorganic compounds, organic pollutants, and xenobiotic matters (Thomas and Xiao-Feng, 1994; Dorthe and Thomas, 2004; Antia et al., 2012). Currently engineering methods and regulatory enforcements are in place to protect surrounding environment and minimize the risk to public health; however, the discharge requirements are not met for every contaminant. Furthermore, improper maintenance of waste dumping sites in developing countries and remote isolated areas is likely to have a more hazardous effect on the surrounding environment (Lalita et al., 2006; Muhammad et al., 2013; Oketola and Akpotu, 2014).

Landfill leachate, generated from percolating excess rain, can be classified as young, medium, and old based on the year of operation. During the early stage of landfilling, the leachate contains a large portion of readily degradable organic content and high ammonium, which can be utilized by microorganisms using conventional, aerobic, and anaerobic biological treatment methods. Over time, there is significant consumption of readily degradable organic matter while refractory portions like humic substances stay in the leachate and cannot be treated solely by the biological method. Therefore, the treatment requires integration of additional techniques, such as adsorption, coagulation, flocculation, nanofiltration, flotation, and reverse osmosis (Majid et al., 2016; Sumona et al., 2014; Xiao-Chun et al., 2013; Zivang et al., 2009).

The development of conventional treatment techniques is limited because of its cost inefficiency and low pollutant removal rate. Therefore, the emergence of AOP has drawn a lot of research interest recently for the treatment of high-strength wastewater and mature landfill leachate due to the characterization of highly reactive radical ( $\cdot\text{OH}$ ). The  $\cdot\text{OH}$  radical ( $E^0=2.8\text{ V}$ ) generated from AOPs is able to non-selectively and rapidly oxidize refractory organic compounds and it also has a higher reaction rate compared to conventional oxidants such as  $\text{O}_3$ ,  $\text{Cl}_2$ ,  $\text{H}_2\text{O}_2$ , and  $\text{KMnO}_4$  (Bautista et al., 2008; Chemlal et al., 2014; Tonni and Wai-Hung, 2006; Wagner et al., 2013). AOPs such as ozone ( $\text{O}_3$ ),  $\text{O}_3/\text{H}_2\text{O}_2$ , Fenton's reagent ( $\text{H}_2\text{O}_2/\text{Fe}^{2+}$ ),  $\text{UV}/\text{H}_2\text{O}_2$ , and  $\text{UV}/\text{H}_2\text{O}_2/\text{Fe}^{2+}$  can react with organics to produce simpler organic compounds, such as  $\text{CO}_2$ ,  $\text{H}_2\text{O}$ , and salts in case of full mineralization (Grzegorz and Andre, 2017).  $\cdot\text{OH}$  generation also degrades refractory organics like aromatic, chlorinated, and phenolic compounds (Tonni and Wai-Hung, 2006). Single-mode process or integrated treatment by AOPs has also been investigated due to its high contaminant removal efficiency. A kinetics study recently reported that the use of  $\text{O}_3$  oxidized contaminants from water and wastewater without forming disinfection by-products (DBPs) and it has widely applied in water treatment plants (Jun et al., 2016). Moreover,  $\text{O}_3$  in conjunction with  $\text{H}_2\text{O}_2$  can enhance decomposition of  $\text{O}_3$  and increase generation of  $\cdot\text{OH}$  radicals (Chedly et al., 2007; Pocostale et al., 2010). Among the aforementioned AOPs, Fenton's reagents offer some advantages as it does not require high energy input to activate  $\text{H}_2\text{O}_2$  (Yongjun et al., 2017). Most aforementioned studies are focused on the treatment of synthetic wastewater or diluted landfill leachate. In this study, raw landfill leachate was treated using different methods. The main objective of this study was to remove COD from landfill leachate by using aerobic SBR with activated sludge and a single or a combination of AOPs. The sludge was acclimatized to achieve stable removal of  $\text{NH}_4\text{-N}$  from 100% leachate by volume for all treatments. The effect of

different parameters such as concentration and ratio of  $\text{H}_2\text{O}_2/\text{Fe}^{2+}$ , treatment period ( $\text{O}_3$ ), and initial COD load were also investigated.

### 3.2. Methods and materials

#### 3.2.1. Landfill leachate

The landfill leachate used in the study was collected from Brady Road Resource Management Facility, Winnipeg, MB, Canada, which has been operating since 1973. The primary effluent was collected from a local WWTP (Winnipeg, MB, Canada). After collection, the leachate and primary effluent was transported immediately to the laboratory, and stored at 4 °C before further processing. Table 3.1 presents characterization of the raw leachate and primary effluent.

Table 3. 1 Characterization of the raw leachate and primary effluent

Parameters	Landfill leachate	Primary effluent
pH	7.8-8.3	7.0
$\text{NH}_4\text{-N}$	890-1,100 mg/L	45 mg/L
Orth-P	3.68 mg/L	3.69 mg/L
$\text{NO}_2\text{-N}$	<0.5 mg/L	<0.1 mg/L
$\text{NO}_3\text{-N}$	<0.5 mg/L	<0.1 mg/L
$\text{BOD}_5$	<100 mg/L	180-210 mg/L
COD	1,050-1,400 mg/L	410 mg/L
$\text{BOD}_5/\text{COD}$	< 0.1	0.42-0.51
EC	371ms/m	N/A
$\text{Na}^+$	1,474mg/L	N/A
$\text{Cl}^-$	183mg/L	N/A

Due to relatively low BOD<sub>5</sub>/COD (<0.1), high ammonium (NH<sub>4</sub>-N) content, and low phosphorous, the leachate for this study can be categorized as old leachate (Jing et al., 2016). In addition, the landfill is located in the northern prairie region where seasonal conditions affect leachate characterization. The feed prepared for aerobic SBR was diluted by using the primary effluent.

### 3.2.2. Treatment and experimental setup

#### a) SBR

The activated sludge was collected from a local WWTP (Winnipeg Pollution Control Centre). As shown in Fig. 3.1, the reactor with a 5-L volume ran 3 cycles per day and was aerated 7.5 hours per cycle with 2 L/min air supply. For each cycle, there was a 5-min settling time before discharging (1.5 L), and then 1.5 L leachate was pumped into the reactor to start another cycle.

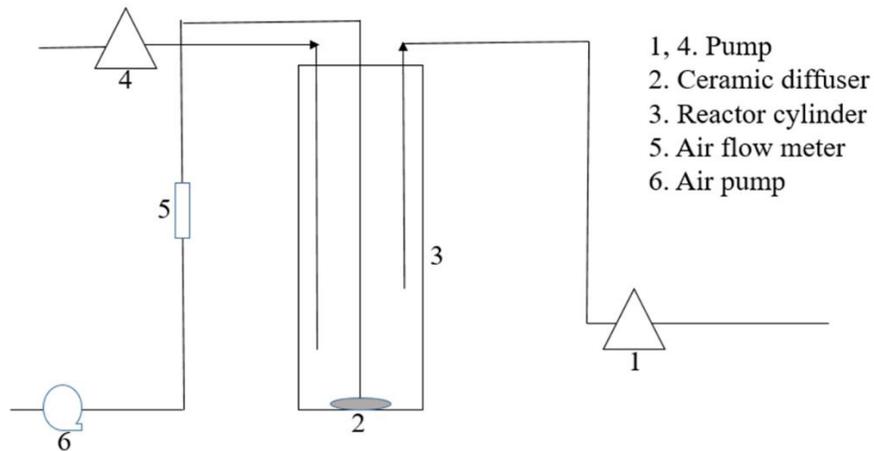


Figure 3. 1 Schematic of an aerobic SBR

### b) Ozonation setup

A lab-scale reactor (3-L volume) was used for the ozone-based experiments (Fig. 3.2). The 1-L volume of leachate to be treated was pumped into the reaction vessel.  $\text{Na}_2\text{S}_2\text{O}_3$  solution was prepared to absorb excess ozone gas. The ozone concentration was titrated and calculated according to the standard method by using KI solution (APHA et al., 1998). The ozone generator (MP8000, A2Z Ozone Inc., Louisville, KY, USA) uses natural air and has a capacity of 1.09 g/h of  $\text{O}_3$  production.  $\text{H}_2\text{O}_2$  (50%) and  $\text{FeSO}_4 \cdot 7\text{H}_2\text{O}$  were used for Fenton's reagents and all chemicals used in this study were of analytical grade.

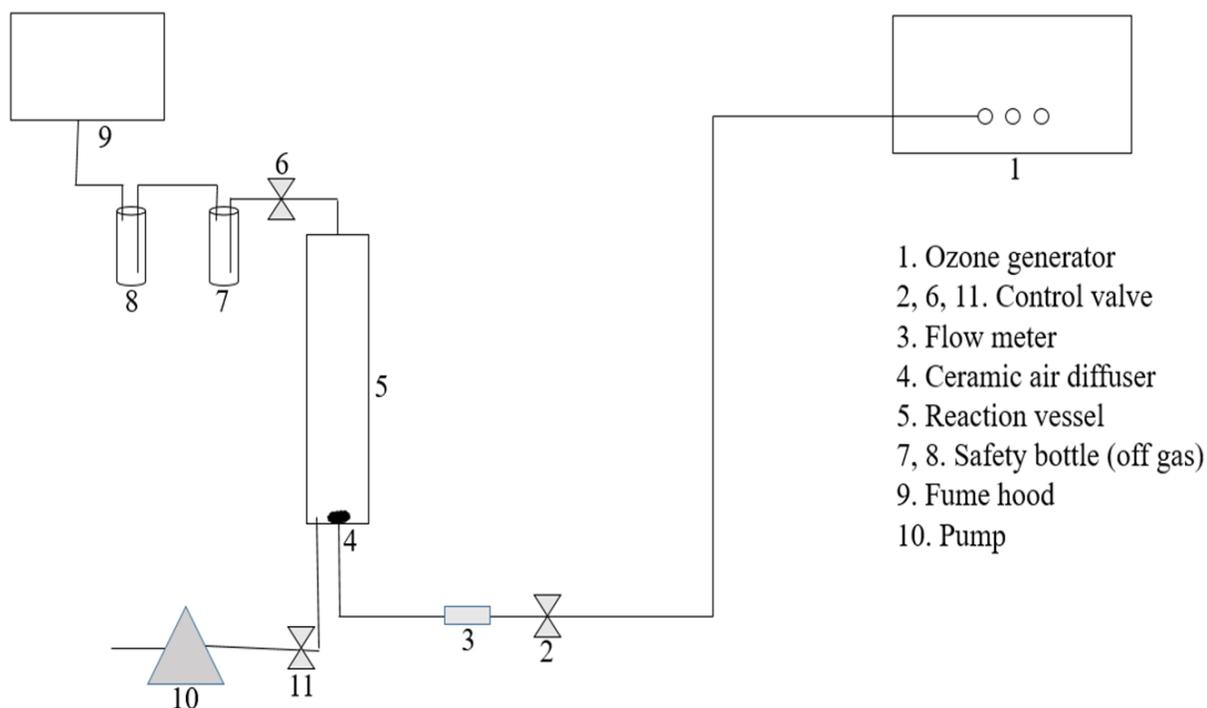


Figure 3. 2 Schematic of ozone-based tests

### c) *Vicia faba* ecotoxicological plant assay

Seeds of *V. faba* L. were analysed for the effect of phytotoxicity, cytotoxicity, and genotoxicity of contaminants according to the ISTISAN protocol (ISTISAN 13/27, 2013). Samples were

collected from the aerobic SBR and directly analysed thereafter. Three petri capsules with 10 seeds per capsule were prepared according to the ISTISAN protocol for each specimen chosen for experimental and control data. After incubation and germination, the primary roots were immersed in Carnoy's fixative solution (EtOH and acetic Acid 3:1) and the vial was incubated for 24 h at 4°C. Subsequently, the root tips were coloured using the Feulgen technique, and then used for DNA sequencing (Venora et al., 2002). For each root tip, at least 1 cell was analysed by counting different regions of the slide that were chosen randomly. Both seed germination rate (% germinated seeds) and root length (cm) with replicates of ten seeds for each sample were calculated. The GI% (germination index) was calculated based on the following equation:

$$GI (\%) = (G_s L_s) / (G_c L_c) \times 100$$

Where  $G_s$  is germination of seeds and  $L_s$  is elongation of roots (mm), respectively;  $G_c$  and  $L_c$  are the counterparts' values of control. The residual apical roots were stored at 4°C. Four different endpoints were analysed: the root elongation, the germination index, the mitotic indices, and the frequency of micronuclei.

### **3.2.3. Analysis**

MLTSS (mix liquor total suspended solid), MLVSS (mix liquor volatile suspended solid), sludge volumetric index (SVI) and soluble COD were measured according to the standard methods (APHA et al., 1998). The concentrations of  $NH_4-N$ , Ortho-P,  $NO_2-N$ , and  $NO_3-N$  were determined by using flow injection analyzer (Lachat Instrument QuikChem 8500, Loveland, CO, USA).

### 3.3. Results & discussion

#### 3.3.1. Biological stabilization: NH<sub>4</sub>-N and COD removal via SBR

The acclimatization process achieved 100% leachate (by volume) in the feed in around 2-3 months. Figure 3.3 illustrates the variation of MLTSS, MLVSS, and SVI with leachate percentage. After the volumetric ratio increased to 40%, the MLTSS kept increasing while MLVSS was relatively stable leading to a decrease in SVI, which indicates good settleability (Ferraz et al., 2016). Figure 3.4 demonstrates NH<sub>4</sub>-N removal, which achieved at least 99.7% reduction by nitrification and volumetric increase did not deter nitrification. Other research studies have also found higher NH<sub>4</sub>-N removal rates during treatment of leachate. For instance, Zheng-Yong et al. (2010) achieved similar NH<sub>4</sub>-N removal rate of 96.7% from raw leachate but their study had a higher effluent NH<sub>4</sub>-N concentration (48 mg/L) and a longer SBR cycle period (12 h). In a different study, Wang et al. (2011) used 30 min intermittent aeration with a longer SBR cycle (24 h) to treat mature landfill leachate with NH<sub>4</sub>-N concentration ranging from 642 to 996 mg/L and found 99.0% NH<sub>4</sub>-N removal.

However, sole biological methods are not desirable to treat old landfill leachate for COD removal due to the low biodegradability and phosphorous content and relatively high refractory matter (e.g. humic-like substances) (Gabriel et al., 2015; Majid et al., 2016). Figure 3.5 shows that COD removal efficiency decreased with the increasing concentration of COD in the influent. Around 25% COD removal was achieved for 100% volumetric leachate. In line with other studies, this work also demonstrates that COD removal via biological stabilization was not desirable (Maranon et al., 2010; Xiao-Feng et al., 2016).

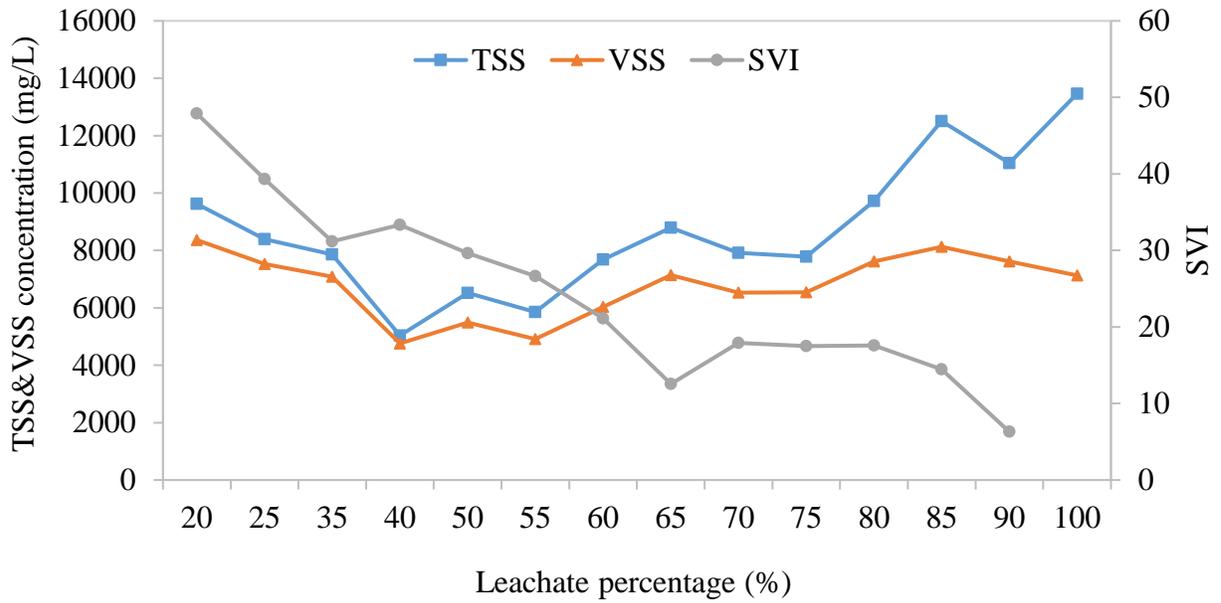


Figure 3. 3 Quality parameters (TSS, VSS, and SVI) with varying leachate ratio obtained via aerobic SBR

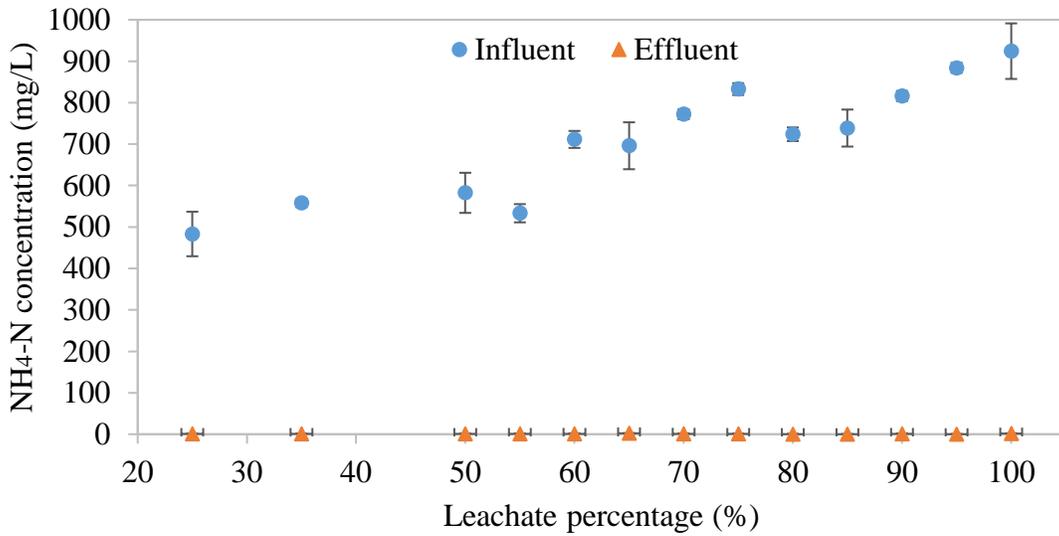


Figure 3. 4 NH<sub>4</sub>-N removal via biological treatment

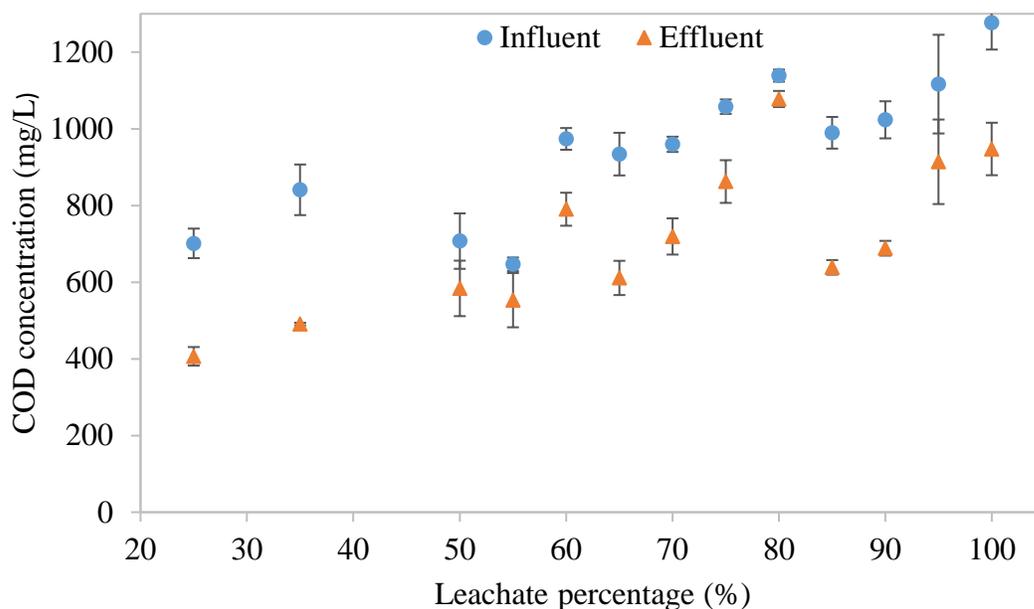


Figure 3. 5 COD removal by biological stabilization

### 3.3.2. *Vicia faba* ecotoxicological plant assay

The biological processes for mature leachate with 100% volumetric ratio only removed 25% of COD. Therefore, to investigate the effect of the biological treatment in terms of toxicity removal, *V. faba* ecotoxicological assay was subsequently implemented on treated old raw leachate. The biological treatment of the old leachate via SBR produced a leachate with phyto-, cyto-, and genotoxic characteristics. GI of biologically treated *V. faba* seeds was 10% lower than the seeds germinated in deionized water (control) and it was characterized by a root length reduction of 56% (Table 3.2). A cytotoxic effect was determined based on the mitotic index. As shown in Table 3.2, the mitotic activity is inhibited by 46% if the seed grown on effluent from SBR. The results show that 25% of COD depletion via biological treatment validated the presence of putative metabolic, intermediate, and recalcitrant compounds in the treated leachate, which damaged the DNA structure and function in the plant.

Table 3. 2 Root elongation (RE), germination index (GI), mitotic index (MI), and micronuclei (MN) of *Vicia faba* seeds treated with deionized water (Control) and effluent collected from the aerobic SBR with activated sludge

Parameters	Control	SBR	Statistical analysis (t-test)
RE (cm)	2.44±0.28	1.13±0.76	p<0.0001
GI (%)	90	80	p<0.001
MI (‰)	9.89±0.03	5.28±0.03	p<0.002
MN (‰)	0.99±0.51	4.71±0.89	p=0.0156

The applicability of AOPs were tested in this study to improve the depletion of COD over a duration of time. According to literature, the use of Fenton's reagents for leachate treatment can also decrease toxicity of the matrix (Hong et al., 2017). The combined effect of AOPs, such as Fe<sup>2+</sup> and H<sub>2</sub>O<sub>2</sub>/O<sub>3</sub>, could reduce the presence of toxic substances that are not treated solely by biological processes. Moreover, the absence of these compounds reduces the bioactivity of microorganisms.

### 3.3.3. COD removal by AOPs

#### a) Ozonation tests

The ozonation tests were conducted and the ozone consumption was calculated based on the inlet and outlet ozone concentration in the system. The time of ozonation, which is relative to doses, plays an important role to break down organic compounds in landfill leachate. The COD reduction of raw leachate and bio-stabilized leachate was 0.040 mgCOD/mgO<sub>3</sub> and 0.144 mgCOD/mgO<sub>3</sub>, respectively, by the end of the treatment period of 240 minutes (Fig. 3.6). Furthermore, the oxidation efficacy decreased significantly after 120 minutes for the bio-stabilized leachate. This is because organic compounds that are readily oxidized by ozone

became less available (Michael et al., 2015). However, the COD removal efficacy achieved during the first and second hour was 0.339 mgCOD/mgO<sub>3</sub> and 0.223 mgCOD/mgO<sub>3</sub>, respectively. These values of ozonation efficiency were within the range of 0.190 to 0.810 mgCOD/mgO<sub>3</sub> reported from literature (Michael et al., 2015).

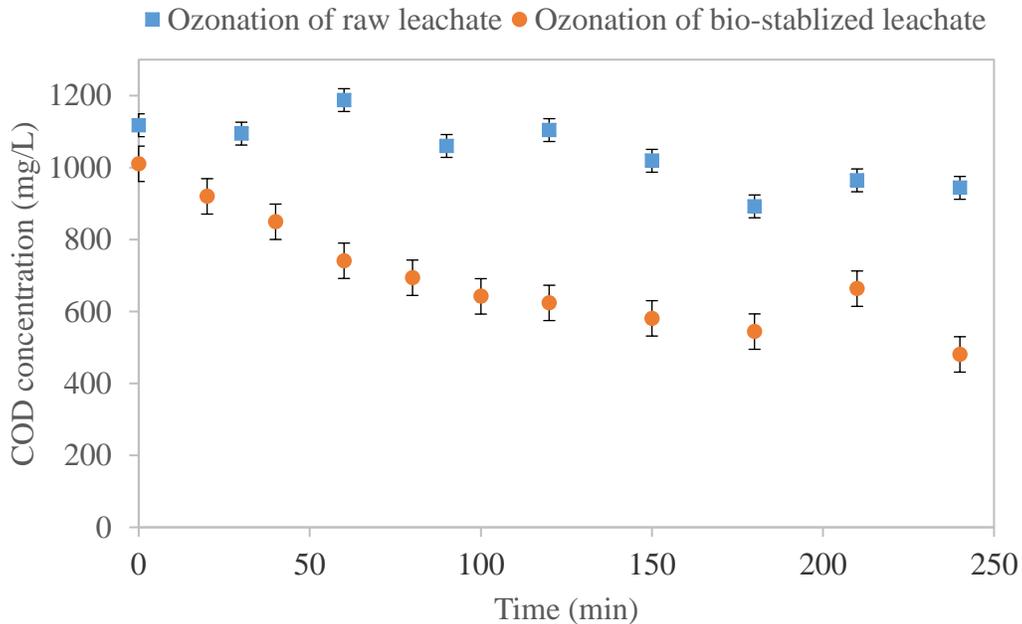


Figure 3. 6 COD removal efficiency of raw leachate and bio-stabilized leachate via ozonation. Due to the high operational cost, the test of O<sub>3</sub> and O<sub>3</sub>/H<sub>2</sub>O<sub>2</sub> was repeated at one time duration of 120 min. Figure 3.7 shows COD reduction under the same condition for both methods. The results indicated that the addition of H<sub>2</sub>O<sub>2</sub> (900 mg/L) did not alter the COD removal rate. This is because there are compounds in old landfill leachate that could compete with H<sub>2</sub>O<sub>2</sub> to react with O<sub>3</sub> to generate •OH (Wang et al., 2010). Another factor is the abundance of dissolved organic matter (DOM) in landfill leachate, which is more likely to act as •OH scavengers (Niloufar et al., 2014). Therefore, to achieve expected oxidation by O<sub>3</sub>/H<sub>2</sub>O<sub>2</sub>, relatively high concentration and

doses are required to produce adequate  $\cdot\text{OH}$ , but this could be prohibitively expensive for the leachate treatment.

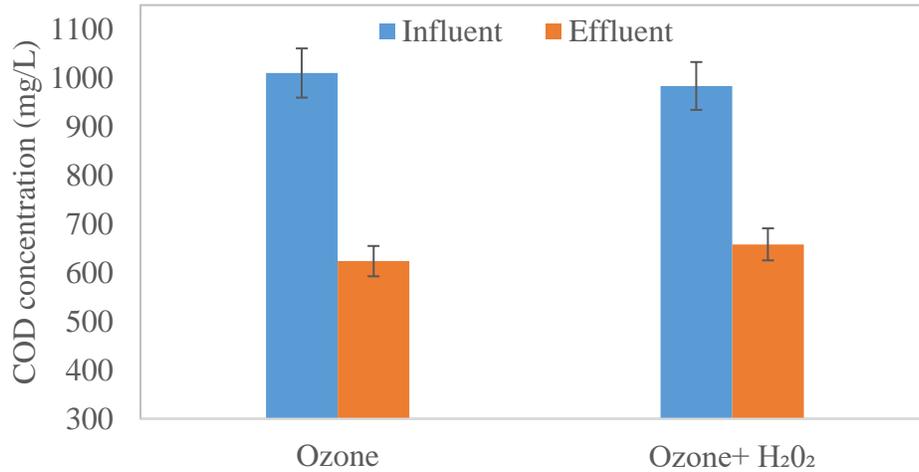


Figure 3. 7 COD reduction by O<sub>3</sub> and O<sub>3</sub>/H<sub>2</sub>O<sub>2</sub> treatments

b) Fenton's reagents test

The tests with Fenton's reagents in this study used 60 mM of 50% H<sub>2</sub>O<sub>2</sub>, 30 mM of Fe<sup>2+</sup>, and leachate at neutral pH (Baharak et al., 2017). As expected, the Fenton's reagents produced the highly reactive radical  $\cdot\text{OH}$  and had better efficacy in removing refractory compounds from the mature leachate. Figures 3.8 shows the COD removal via Fenton's reagents for raw leachate with different initial COD concentrations. When COD was higher than 800 mg/L, disparity of approximately 30% was found between final COD removal efficiencies. However, the same treatment process for bio-stabilized leachate achieved higher removal efficiency (Figure 3.9). Highest COD removal of 67% was achieved for 100% bio-stabilized leachate, as compared to 33% COD removal of 100% raw leachate without bio-stabilization. This revealed that integration of biological treatment and Fenton's reagents is a more efficient and effective approach for the

treatment of old landfill leachate. The wastewater typically treated by Fenton's reagents contains high strength contaminants such as phenol, pharmaceutical, melanoidin, and textile chemicals, and the pre-treatment via conventional methods can reduce the cost of chemicals used in AOPs by removing partial contaminants (Arimi, 2017). Because of the highly oxidative potential, Fenton's reagents can also be adopted as a pre-treatment process to improve the biodegradability of refractory wastewater such as mature landfill leachate. Vilar et al. (2012) used biological oxidation, photo-Fenton reagents, and biological oxidation system to treat pesticide-containing high strength wastewater in sequence, and reported a COD decrease by 25%, 46%, and 36%, respectively, when the initial COD was 830 mg/L. Gupta et al. (2014) studied the UV quenching organics removal from biologically treated landfill leachate, and their results show that Fenton's reagents remove 85% of the humic substances in 30 min, which are the major contributors of UV<sub>254</sub> quenching effect.

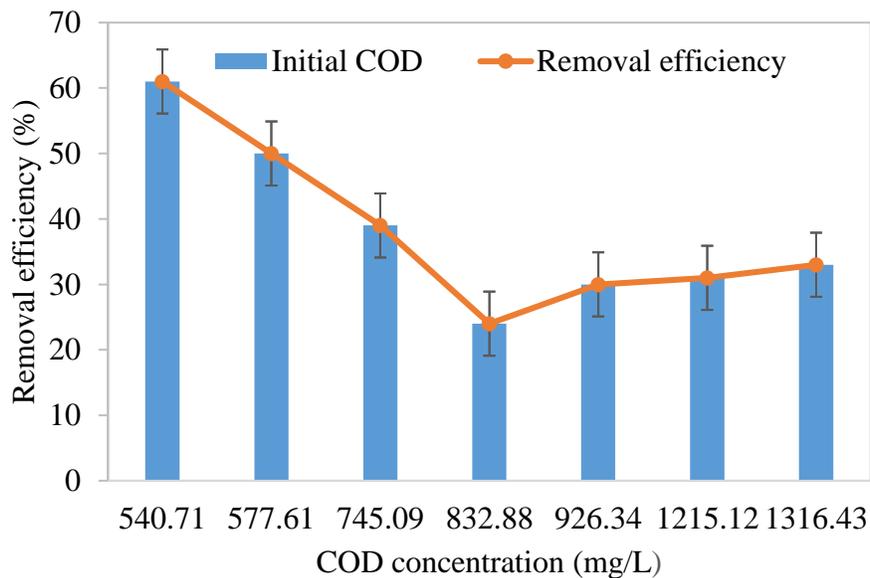


Figure 3. 8 COD removal efficiency for leachate with different initial COD concentration via Fenton's reagents

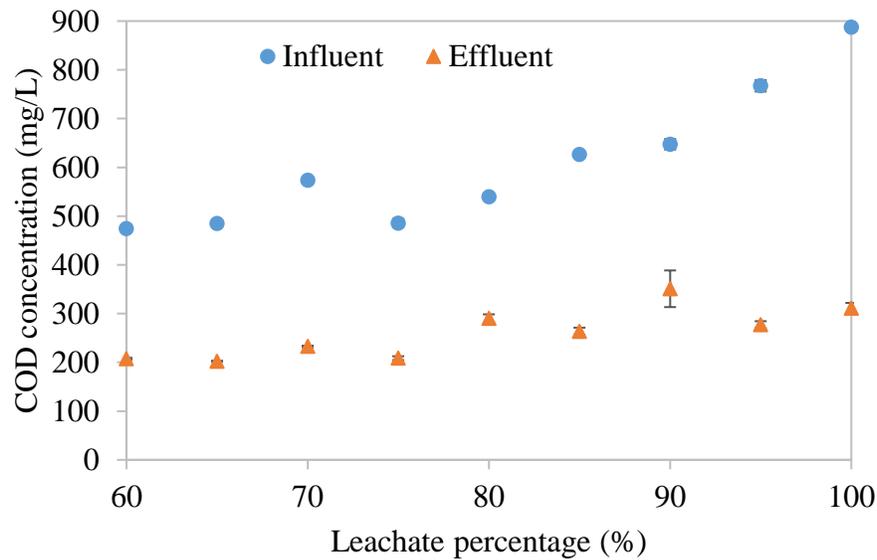
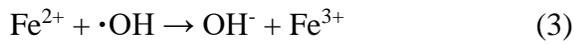
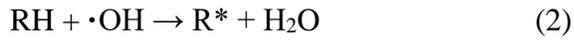
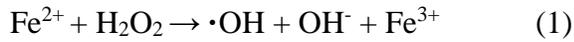


Figure 3. 9 COD removal of bio-stabilized leachate via Fenton's reagents

c) Effect of  $\text{H}_2\text{O}_2/\text{Fe}^{2+}$  ratio

To ensure the maximum production of  $\cdot\text{OH}$  radicals in Fenton's reagents, the ratio of  $\text{H}_2\text{O}_2/\text{Fe}^{2+}$  should be controlled to avoid the scavenging effect of peroxide on  $\cdot\text{OH}$  radicals (Susana et al., 2010; Tang et al., 1996). Therefore, the effect of  $\text{Fe}^{2+}$  concentration was investigated in this study and the results are presented in Figure 3.10. When the  $\text{Fe}^{2+}$  dose was increased from 10 mM to 15 mM, the COD removal slightly increased from 48% to 52%. However, a further increase to 20 and 30 mM achieved the highest removal of 63% and 67%, respectively. This is because an increase in  $\text{Fe}^{2+}$ , which is functionalized as a catalyst corresponds to higher production of hydroxyl radicals. This sequentially leads to higher oxidation rates of organic compounds (R-H). However, the results suggested that excess concentration of  $\text{Fe}^{2+}$  (>30 mM) did not increase the removal rate because it can react with the  $\cdot\text{OH}$  to produce  $\text{Fe}^{3+}$  as shown in equations 1-3 (Brink et al., 2017). Another explanation is the dissociation of  $\text{H}_2\text{O}_2$  into

hydroperoxide radicals ( $\cdot\text{HO}_2$ ), which leads to a decrease in oxidative capacity (Lobna, 2015). Therefore, the optimal ratio found for  $\text{H}_2\text{O}_2/\text{Fe}^{2+}$  was 2:1. Hugh (1964) also reported that 2:1 ratio of  $\text{H}_2\text{O}_2/\text{Fe}^{2+}$  achieved the best oxidation efficiency regarding phenolic compounds removal.



Where RH represents an organic compound.

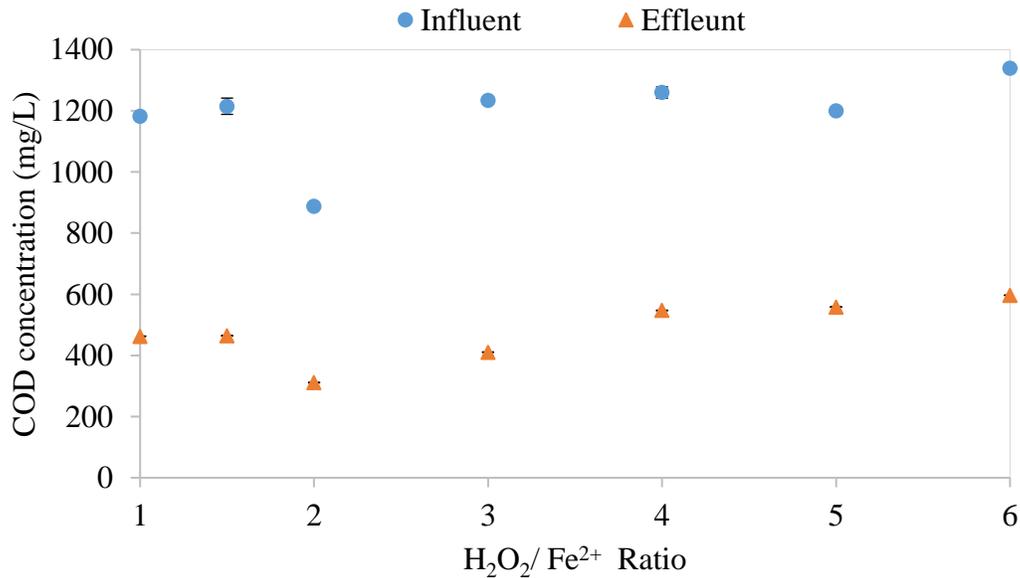


Figure 3. 10 Effect of  $\text{H}_2\text{O}_2/\text{Fe}^{2+}$  on COD removal

d)  $\text{Fe}^{2+}/\text{H}_2\text{O}_2/\text{O}_3$  process

$\text{Fe}^{2+}/\text{H}_2\text{O}_2/\text{O}_3$  process achieved 69% and 72% COD removal after 120 and 180 min of treatment, respectively (Figure 3.11). The benefits of using this process is that Fenton's reagents in

conjunction with  $O_3$  can improve  $O_3$  decomposition, and thus produce more  $\cdot OH$  radicals to oxidize organic matters. There are different pathways for aqueous ozone to react with solutes. It can either react directly or decompose to generate  $\cdot OH$  radicals. Johannes et al. (1985) studied the decomposition kinetics of aqueous ozone in the presence of certain organic compounds. Their results confirmed that the decomposition rate of ozone is generally affected by a radical-type chain reaction and it could be promoted due to the conversion of  $\cdot OH$  radicals to  $\cdot O^{2-}$ , which is a more efficient chain carrier.

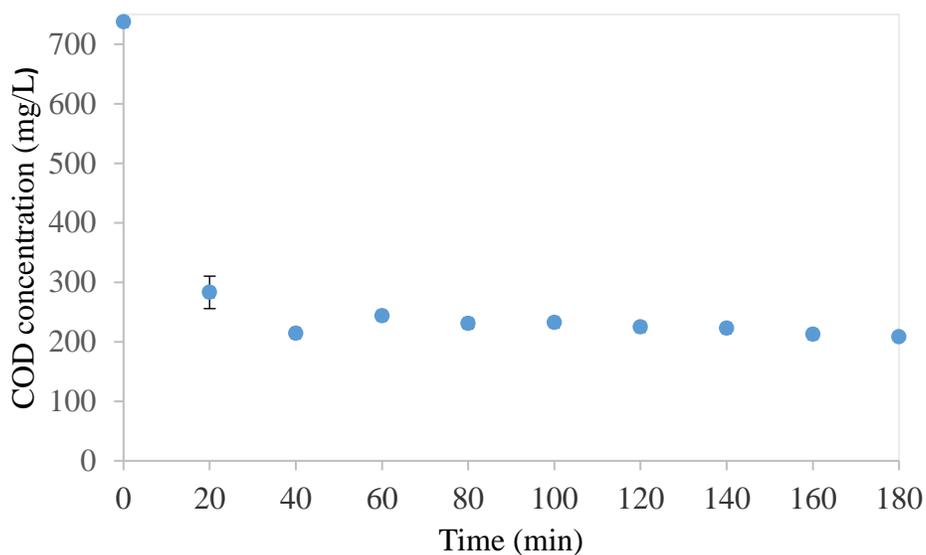


Figure 3. 11 Effect of  $Fe^{2+}/H_2O_2/O_3$  process on COD removal

Lobna et al. (2015) studied the oxidation of diethyl phthalate (DEP) by using  $O_3$  and  $Fe^{2+}/H_2O_2/O_3$ , and their results shown a higher yield of hydroxyl radicals and a higher removal efficiency for  $Fe^{2+}/H_2O_2/O_3$  system than  $O_3$ . However, the chemical dose of  $Fe^{2+}/H_2O_2/O_3$  process is crucial pertaining to the oxidative potential as ozone readily oxidizes ferrous ion into ferric ion ( $Fe^{3+}$ ), which can reduce the oxidative power (Beata et al., 2008).

Arimi (2017) also recommended the combination of Fenton's reagents with another AOP to improve the treatment efficiency. Amr and Aziz (2012) compared several AOPs, namely O<sub>3</sub>, Fenton's reagents, Fenton's reagents/O<sub>3</sub>, for the treatment of mature landfill leachate, and the COD removal efficiency was found out to be 15%, 55% and 65%, respectively, with retention time varying from 60 min to 120 min, which was slightly lower than the results obtained from this study by using different Fenton molar ratio. However, the highest NH<sub>3</sub>-N removal from their study was around 12%, which proved that only treatment via AOPs alone is not sufficient for the treatment of landfill leachate.

### **3.4. Conclusion**

In this study, combined biological methods and AOPs were examined for the treatment of mature landfill leachate to observe the maximum ammonium and COD removal efficiencies. The results indicated that treatment of raw, old leachate without dilution is feasible and effective by biological treatment to remove NH<sub>4</sub>-N. However, significant removal of COD could not be achieved solely by biological method.

In terms of removal of organic pollutants by AOPs, Fe<sup>2+</sup>/H<sub>2</sub>O<sub>2</sub>/O<sub>3</sub> achieved higher COD removal than O<sub>3</sub> and O<sub>3</sub>/H<sub>2</sub>O<sub>2</sub> processes because of its higher oxidation potential to break down refractory compounds in the leachate. Therefore, the synergy of biological method and single AOPs (Fe<sup>2+</sup>/H<sub>2</sub>O<sub>2</sub>/O<sub>3</sub>) is recommended to improve the treatment efficiency and reduce the treatment period. Furthermore, process like electro-Fenton and photo-Fenton can be implemented to improve the performance of Fenton's reagents, for the treatment of high strength wastewater to improve treatment efficiency or the biodegradability of the refractory compounds. This set-up could be a feasible alternative with or without the combination of biological treatment.

## Chapter 4 Phytoremediation of Landfill Leachate by Using *Puccinellia*

### *Nuttalliana* and *Typha Latifolia*

#### 4.1. Introduction

“It was postulated that salt-affected soils cover about 6% (more than 800 million ha) of the world lands, and in 2008 it affected about 2% (32 million ha) of the dryland- farmed areas and 20% (45 million ha) of the irrigated lands in the world” (Zorrigo et al., 2012). Soil salinity is one of the concerns in irrigated agriculture, which can develop both naturally and anthropogenically. Arid and semi-arid environments, like northern prairies, with extreme temperatures and irradiance contribute to higher soil salt concentration (Verheye, 2009). The accumulation of salts in the soil inhibits seeds germination, plants establishment and other physiological processes (including photosynthesis, respiration, transpiration), which could lead in case of severe stress to plant death (Hasanuzzaman et al., 2014).

Municipal landfills are currently the most widely applied method for the disposal of solid waste due to its cost-effectiveness and operation-convenience. However, the leachate generated, especially during acetogenic biodegradation phase, is characterized as high electrical conductivity (EC) caused by increased  $\text{Na}^+$ ,  $\text{Cl}^-$  and  $\text{NH}_4^+$ -N concentrations and high COD (Jones, et al., 2006). With the increasing accumulation of organic (such as xenobiotic compounds) and inorganic pollutants in soil and water which can lead to severe environmental problems, a variety of approaches have been developed for the desalination such as engineering techniques, physical and chemical methods (Padmavathiamma et al., 2014). However, those approaches are relatively expensive and could also make the soil less valuable for plants and microbial growth after remediation because of leaching of nutrient and addition of chemicals (Padmavathiamma et al., 2014). Therefore, it is obligatory to develop a method which can be

cost-affective and environmental-friendly. Even though it is considered as wastewater, the nutrient from landfill leachate such as nitrogen and phosphorous is valuable for plants growth. A lot of studied have been conducted to examine the feasibility of using wastewater for plants irrigation (Zupanc and Justin, 2010; Zalesny et al., 2008). Another commonly studied plant-based method is called hydroponic system which can effectively absorb dissolved matters from wastewater as nutrient for plants establishment (Pan et al., 2007). The hydroponic system plays a role in wastewater purification also by transpiration, which reduces the water volume discharged into the receiving stream. It can also improve the microclimate through plants transpiration, reduce odour nuisance and the aesthetic quality is another benefit (Bawiec, 2018).

In this study, the phytoremediation performance of alkaligrass and cattail was evaluated from soil irrigated with landfill leachate. It also compared the nutrient removal and salts bioaccumulation with soil irrigated with fertilizer and landfill leachate, to demonstrate the feasibility of landfill leachate for plants irrigation. In addition, the hydroponic cattail was applied to treat landfill leachate for nutrient removal and biomass yield was examined.

## **4.2. Methods and materials**

### **4.2.1. Plants species and germination**

Two native halophytes alkaligrass (*Puccinellia nuttalliana*) and cattail (*Typha latifolia*) were selected for the study. Alkaligrass is a caespitose slender-growing perennial salt-tolerant cool-season grass, varying from 20 to 80 cm tall (Liu and Coulman, 2014). Alkaligrass seedlings (Figure 4.1) were collected from Prairie Habitats Inc., Argyle, Manitoba, Canada in June 2017.



Figure 4. 1 Alkaligrass seedlings

Figure 4. 2 Cattail seedlings

Cattail (Figure 4.2) is a perennial aquatic plant species with large biomass yield, which has also been approved feasible for bio-accumulation of pollutants (Jeke et al., 2017). The cattail heads were collected from ditches in autumn 2016, Winnipeg, MB, Canada. The seeds extraction was carried out by the following procedure: the cattail heads were first blended for 30 seconds with Contrad detergent solution using a household blender (M54227C, Hamilton, CA, USA) (McNaughton, 1968). This was subsequently washed with reverse osmosis (RO) water (Jeke et al., 2017; McNaughton, 1968). Afterwards, the seeds were germinated in a plastic tray with potting mix (Miracle-Gro, Scotts, Marysville, OH, USA) in a controlled growth room with the following conditions: day/night temperatures of 22/15°C, 16-h photoperiod, 65% relative humidity and 270  $\mu\text{mol photons m}^{-2}\text{s}^{-1}$  daytime light intensity.

#### 4.2.2. Treatments

The soil used in this part of study was a mixture of peat/sand/clay (1:1:1 by volume) (Department of Plant Science, University of Manitoba, Winnipeg, MB, Canada). The treatments include control (C) only irrigated with tap water, fertilizer (F) (FloraNova, GH, Sebastopol, CA, USA), and landfill leachate (LL) (Brady Landfill, Winnipeg, MB, CA). There were 2 dilutions of leachate applied, namely 20% and 30% leachate which was diluted with tap water by volume

(stages 1 and 2 respectively). All the seedlings (cattail and alkaligrass) were transplanted into pots and grew around 12 weeks in the greenhouse. The first stage started from the second week after transplantation and the second stage started from the seventh week. Equivalent amount of fertilizer (F) was employed for the two stages tests by calculating nitrogen concentration which was based on the value of 20% (200 mg/L  $\text{NH}_4\text{-N}$ ) and 30% (300 mg/L  $\text{NH}_4\text{-N}$ ) LL, respectively. The LL and F were employed at the beginning of second week (20%) and the seventh week (30%). Tap water (100 mL) was used daily for all the treatments in greenhouse environment. The plants samples and soil were collected weekly for analysis of nutrient,  $\text{Na}^+$ ,  $\text{Cl}^-$  content and electrical conductivity.

#### **4.2.3. Analysis**

- a) After each sampling, the plants were washed, and freeze-dried by using a freeze dryer (Labconco, Kansas City, USA) to measure biomass dry weight. Soil samples were collected the same time after each harvest. After air dried, the soil samples were ground to pass through 10-mesh screen. The dried plants and soil samples were digested by using wet oxidation method as following. The digestion mixture was firstly prepared in a beaker with 350 mL 50%  $\text{H}_2\text{O}_2$ , 0.42 g Se powder and 14 g  $\text{LiSO}_4\cdot\text{H}_2\text{O}$ , then it was put in cold-water bath, and then 420 mL 98%  $\text{H}_2\text{SO}_4$  was carefully added with swirling (Rowland and Grimshaw, 1985). The sample digestion was then conducted as following procedure: 0.2 g soil or 0.4 g plants tissue was transferred into kjeldahl digestion tubes and 4.4 mL digestion mixture was carefully added in fume hood, then the tubes were left standing at room temperature. After 1 hour, the tubes were put into the digestion blocks and samples were digested for 3 hours (Akinremi et al., 2003). For the measurement of electrical conductivity ( $\text{EC}_5$ ), 35 mL DI water and 7 g dried and ground soil was put into

a 50 mL centrifuge tube to form soil paste, and then the filtrate was used for EC<sub>5</sub> measurement.

b) Na<sup>+</sup> was analyzed by using Atomic Absorption Spectrometer (AAAnalyst 400, PerkinElmer Inc., MA, USA) with C<sub>2</sub>H<sub>2</sub> and Air flow rate of 2.50 L/min and 10.0 L/min, respectively. The wavelength was set to be 589 nm to detect emission signal based on standards prepared (Robinson, 1996). Cl<sup>-</sup> was detected by using chloride test kit (Model 8-P, Hach company, Loveland, Colorado, USA) (Urgun-Demirrtas et al., 2006). Digested plants samples were neutralized for NH<sub>4</sub>-N and PO<sub>4</sub>-P analysis by using flow injection analysis (FIA, Lachat Instrument QuikChem 8500, Loveland, Colorado, USA) (Ren et al., 2017). EC<sub>5</sub> was examined by using a conductivity meter (Fisherbrand Traceable Conductivity, Resistivity, and TDS Meter, Fisher Scientific, USA) (He et al., 2012). In this study, the photosynthetic efficiency was estimated by measuring chlorophyll fluorescence using chlorophyll fluorometer (Model Opti-Sciences OS-30P, USA). Minimal fluorescence (F<sub>0</sub>), maximal fluorescence (F<sub>m</sub>) and variable fluorescence (F<sub>v</sub>) of mature leaves were measured and calculated to obtain maximum quantum yield of photosystem II (PSII)  $F_v/F_m$  which is an indicator of leaf photosynthetic performance (Maxwell and Johnson, 2000; Murchie and Lawson, 2013; Ritchie, 2006). For data analysis, one-way ANOVA with LSD was used to calculate significance ( $p < 0.05$ ). The statistic was performed by using IBM SPSS 24 with 3 replicates.

### **4.3. Results & discussion**

#### **4.3.1. Soil (pot) test**

##### **a) Alkaligrass and cattail biomass yield**

Figure 4.3 illustrates the height of alkaligrass and cattail under all the treatments (C, F and LL). During the first stage, alkaligrass with addition of fertilizer was observed to be the tallest, starting from  $14.9 \pm 0.4$  cm and ending at  $38.5 \pm 1.9$  cm, compared to C and LL. However, no significant height change (F) was found during the second stage after the ninth week. The alkaligrass irrigated only with tap water (C) kept gaining height until the tenth week, which was similar to F, but about  $8.5 \pm 1.7$  cm taller than that of F in the end. Different trend of alkaligrass growth was observed under the treatment of LL, because the plants were growing relatively slowly compared with C and F, and it took 12 weeks until the establishment levelled off, which was  $2.5 \pm 0.6$  cm taller than that of F, but  $6.0 \pm 1.9$  cm shorter than that of C. Tarasoff et al. (2007) studied the growth of alkaligrass under sodic and normal soil types and obtained the similar grass height during boot stage, but their results revealed that sodic soil was more beneficial for alkaligrass compared to normal soil. However, the LL component, such as toxicity and heavy metals, is exceptionally complex and it could hinder plants growth even if it could contribute to sodic conditions. The control cattail stopped gaining height at approximately the sixth week, and the counterparts under F and LL did not change significantly during the second stage. However, cattail under LL established the highest length at the end of the experiment which was  $101.0 \pm 1.1$  cm.

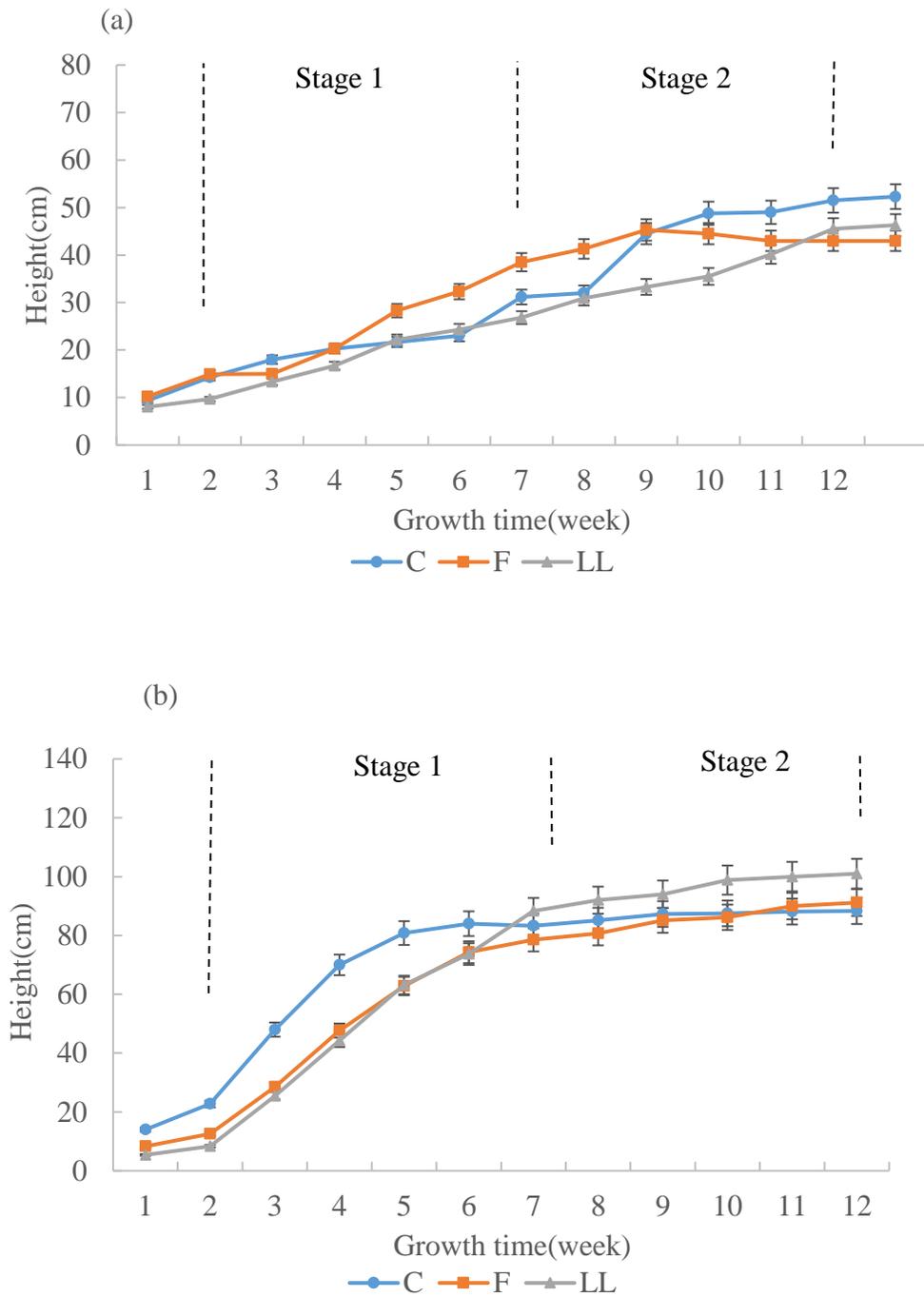


Figure 4. 3 Plants mean growth in height under C, F and LL during stage 1 and stage 2 (a) alkaligrass (b) cattail. Height was measured weekly during 12 weeks' study. Values are the means  $\pm$  standard error (error bars).

As shown in Figure 4.4, the alkaligrass under the treatment of fertilizer gained more biomass compared with C and LL, which was  $9.58 \pm 0.69$  g ( $p < 0.05$ ). The treatment of C and LL did not show significant difference for biomass gaining. Cattail yielded similar biomass under F and LL, they were  $17.30 \pm 0.58$  g and  $16.78 \pm 2.34$  g, respectively.

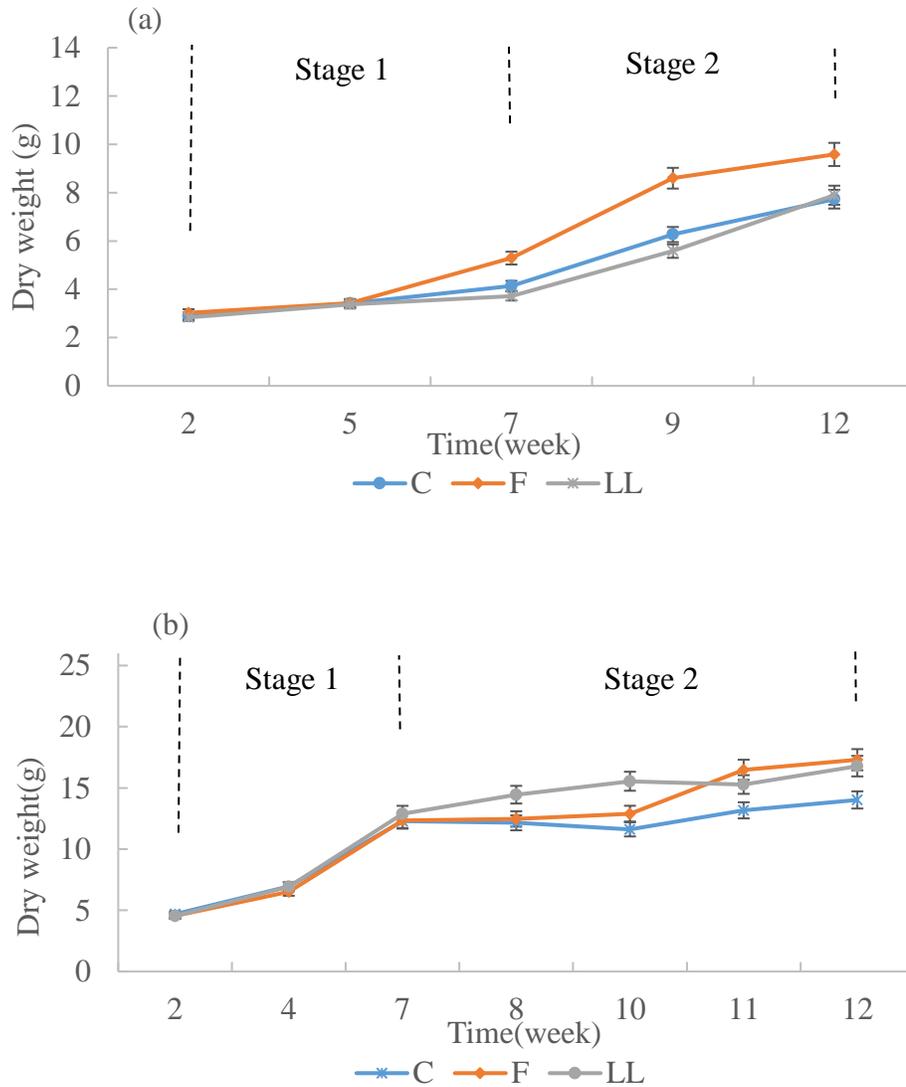


Figure 4. 4 Plants mean biomass yield under C, F and LL during stage 1 and stage 2 (a) alkaligrass (b) cattail. Biomass was measured 5 times and 7 times for alkaligrass and cattail respectively, by triplicates. Values are the means  $\pm$  standard error (error bars).

Jeke et al. (2017) studied phytoremediation of biosolid by using the same cattail and their results shown 19.5 g biomass yield after 90 days cultivation, which is slightly higher than the biomass yield from this study. Dickerman et al. (1986) estimated the biomass production of cattail from study plot and their records of two years study shown summed shoot maximum was 1505 g/m<sup>2</sup> which was similar to the cattail irrigated with fertilizer (1531 g/m<sup>2</sup>) based on calculation of the pot dimension. Zupanc and Justin (2010) estimated biomass change of *Populus deltoids* with the irrigation of landfill leachate or nutrient solution, and the results shown that plants with irrigation of leachate reduced 21.7% biomass compared with nutrient solution, but more than twice amount than that of control. Zalesny et al. (2008) conducted a 2-year study and concluded that landfill leachate can maximum biomass yield for *Populus* during the early stage of plants establishment compared with control, because the leachate can supply water and plants nutritional benefits.

Table 4. 1  $F_v/F_m$  results from Chlorophyll Fluorescence test. 4 replicates were applied one month after the seventh week. Values are the means  $\pm$  standard deviation.

Treatment	$F_0$	$F_v$	$F_m$	$F_v/F_m$
Alkaligrass				
C	151.6 $\pm$ 22.3	459.3 $\pm$ 67.9	611.0 $\pm$ 90.1	0.751 $\pm$ 0.003
F	149.5 $\pm$ 46.1	365.3 $\pm$ 101.7	515.7 $\pm$ 144.5	0.711 $\pm$ 0.031
LL	105.3 $\pm$ 36.7	319.0 $\pm$ 173.6	420.4 $\pm$ 208.6	0.741 $\pm$ 0.033
Cattail				
C	255.0 $\pm$ 124.1	693.3 $\pm$ 376.9	948.3 $\pm$ 492.1	0.725 $\pm$ 0.039
F	198.5 $\pm$ 48.7	693.0 $\pm$ 233.1	891.5 $\pm$ 281.7	0.773 $\pm$ 0.017
LL	213.3 $\pm$ 67.9	738.5 $\pm$ 190.3	966.3 $\pm$ 268.1	0.781 $\pm$ 0.012

Table 4.1 shows the results of potential quantum yield ( $F_v/F_m$ ) one month after the treatment on stage 2 and it indicated the photosynthetic performance under environmental stress, and there was no significant variations over the treatment, and all the  $F_v/F_m$  values of plants were in the

recommended range of 0.7-0.83, which means no photoinhibition was involved in the plants leaves (Ritchie, 2006; Broetto et al., 2007).

#### **b) Nutrient uptake**

NH<sub>4</sub>-N and PO<sub>4</sub>-P were analyzed in the plants samples and results shown in Figure 4.5.

Significant difference in nitrogen accumulation was observed in alkaligrass biomass under the three treatments ( $p < 0.05$ ). The plants irrigated with LL gained the highest nitrogen amount ( $49.83 \pm 0.27$  g), which was 36.07% higher than that of C and 3.36% more than that of F ( $p = 0.006$ ). The distribution results (Figure 4.6) also revealed that aboveground biomass accumulated more nitrogen with the irrigation of landfill leachate while no significant difference was obtained with F compared to C ( $p = 0.017$ ). However, the phosphorous accumulation between F and LL was not significant ( $p = 0.898$ ) and both of them had higher content than C. Considering that only  $35.91\% \pm 5.78\%$  phosphorous was accumulated in roots by alkaligrass (C), F and LL had higher portion which were  $57.94\% \pm 7.35\%$  and  $59.50\% \pm 1.34\%$  in the roots. The cattail irrigated with fertilizer had the highest nitrogen content compared with C and LL. F and LL phytoaccumulated  $97.11\% \pm 5.35\%$  and  $77.62\% \pm 2.69\%$  more nitrogen than C, respectively ( $p = 0.005$ ) (Figure 4.5). The same trend was also found for phosphorous accumulation, and the cattail under F and LL had significant more P content than that of C ( $p = 0.023$ ), but the difference between F and LL was negligible ( $p = 0.132$ ). The nutrient distribution in roots and aboveground tissues for cattail varied greatly, the nitrogen content increased in the aboveground part in the following order:  $C < F < LL$ , but the phosphorous results between F and LL was not significant (Figure 4.7).

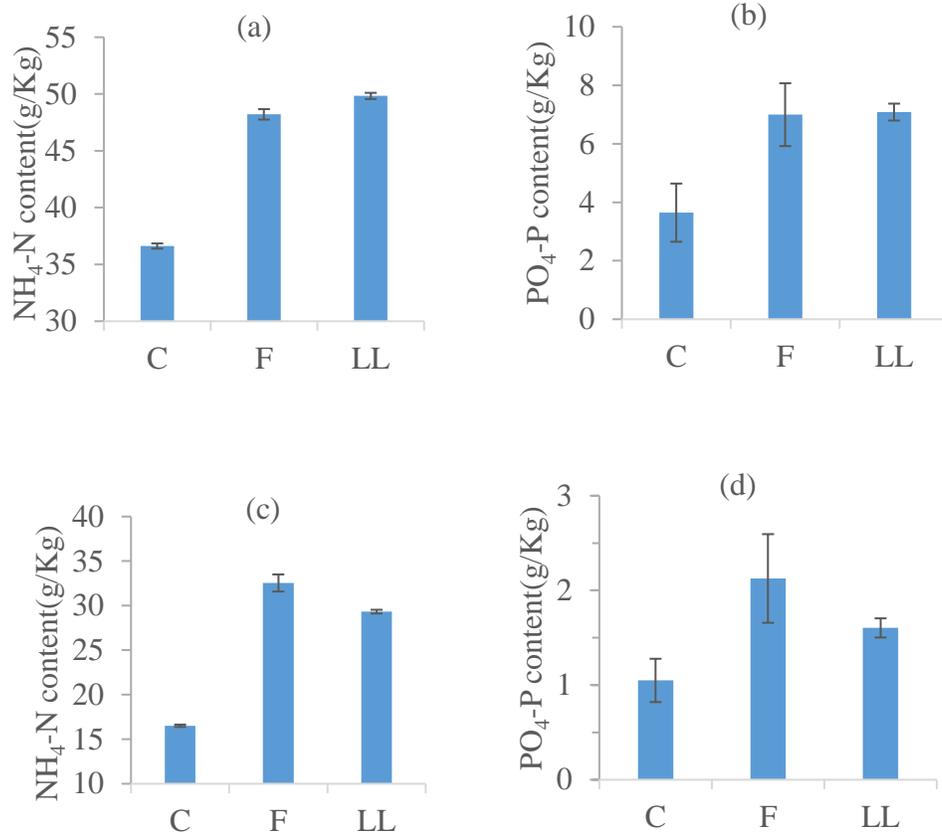


Figure 4. 5 Alkaligrass and cattail nutrient content (DW) after the last harvest (a) & (b) alkaligrass, (c) & (d) cattail. Values are the means  $\pm$  standard deviation.

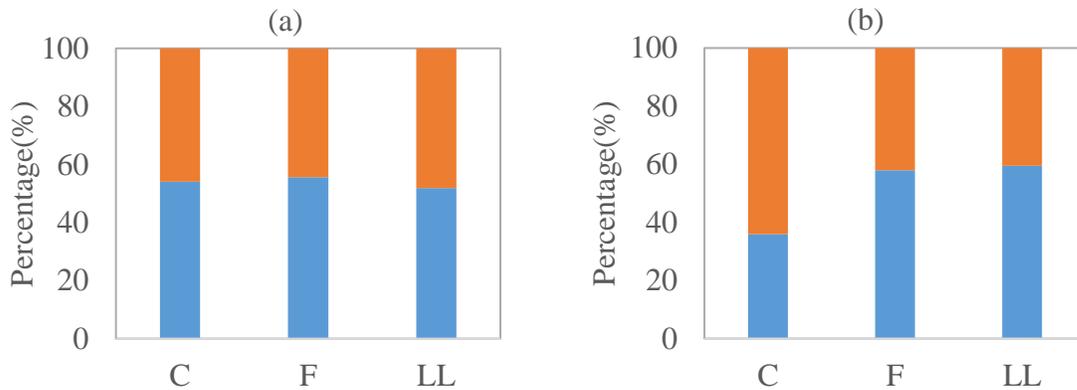


Figure 4. 6 Nutrient distribution of alkaligrass in roots and aboveground tissues (a) nitrogen (b) phosphorous.

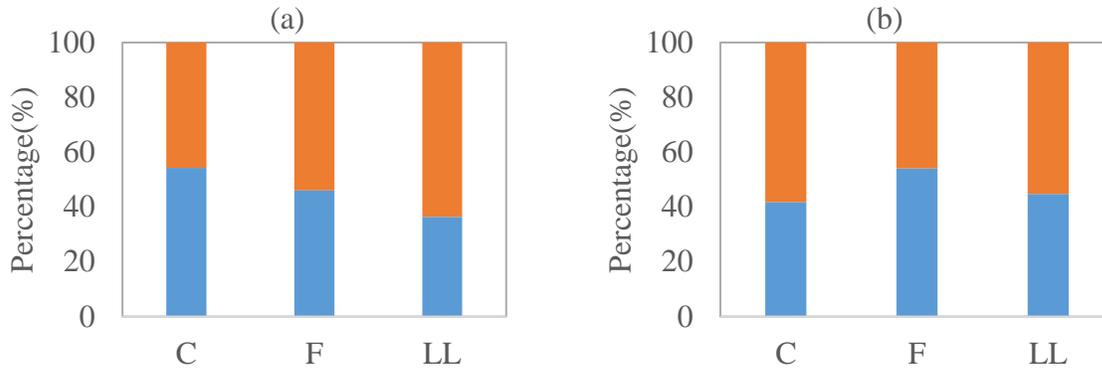


Figure 4. 7 Nutrient distribution of cattail in roots and aboveground tissues (a) nitrogen (b) phosphorous.

### c) EC deduction from soil

After harvest of the plants, EC<sub>5</sub> of the soil under different treatments with alkaligrass and cattail was measured. For alkaligrass (Figure 4.8), EC from leachate was decreased by 71.70%, starting from 124.4 ms/m and ending at 35.2 ms/m, which was lower than that of control and fertilized soil after the harvest (42.0 ms/m and 40.3 respectively). After supplying fertilizer to alkaligrass, the EC<sub>5</sub> of soil was found to be 62.8 ms/m while it dropped to 40.3 ms/m after the harvest. The EC<sub>5</sub> decreased by cattail was also significant concerning leachate and EC<sub>5</sub> was decreased by 45.36% during this period of study. Considering the biomass yield of alkaligrass and cattail with the irrigation of leachate, the reduction in EC<sub>5</sub> by alkaligrass was around 3-fold greater than that of cattail, namely 11.3 ms/m/g DW and 3.8 ms/m/g DW respectively. Ungar (1970) found out that alkaligrass established up to a 1.2-20 ms/cm range in sulphate soils of North Dakota, while Millar (1976) found out that this halophyte developed normally over a range of 15-45 ms/cm in Saskatchewan wetlands (Hammer and Heseltine, 1988), this means the alklaigrass is able to grow with a higher salinity compared to this study, but the plants respond was not mentioned in the above two investigations.

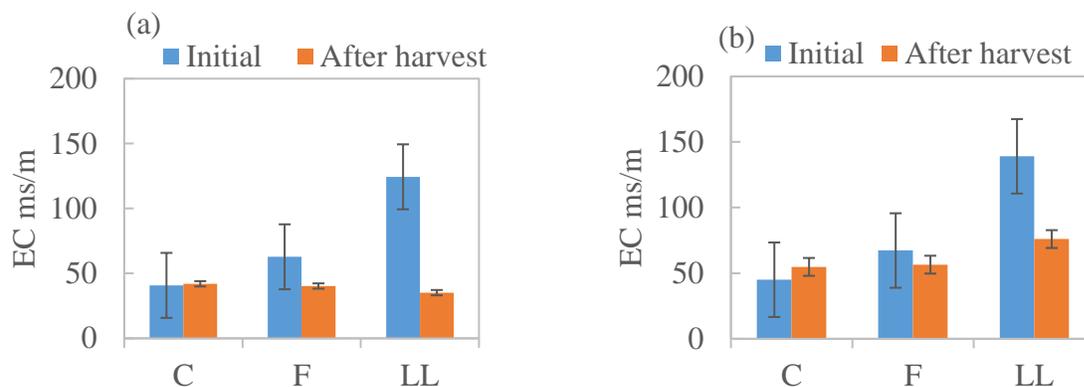


Figure 4. 8 EC deduction from soil by alkaligrass (a) and cattail (b) after the final harvest. Values are the means  $\pm$  standard deviation.

Stewart and Kantrud (1972) reported that cattail was considered a secondary species in North Dakota ponds over a range up to 5 ms/cm which is 3.59-folds compared to the treatment under leachate, however the salinity accumulation was not mentioned in that study.

#### d) Na<sup>+</sup> and Cl<sup>-</sup> phytoaccumulation

Table 4.2 represents Na<sup>+</sup> and Cl<sup>-</sup> phytoextraction by alkaligrass and cattail under C, F and LL. Alkaligrass under C and F shown similar results, and they were 1.24 $\pm$ 0.23 and 1.43 $\pm$ 0.26 g/Kg biomass respectively (p=0.403), however, the plants exposed to LL achieved 4.53 $\pm$ 0.36 and

Table 4. 2 Na<sup>+</sup> and Cl<sup>-</sup> accumulation by alkaligrass and cattail after the last harvest. Values are mean  $\pm$  standard deviation.

		C	F	LL
		g Kg <sup>-1</sup>		
Alkaligrass	Na <sup>+</sup>	1.24 $\pm$ 0.23	1.43 $\pm$ 0.26	6.85 $\pm$ 0.96
	Cl <sup>-</sup>	5.41 $\pm$ 0.70	7.11 $\pm$ 1.22	11.91 $\pm$ 2.78
Cattail	Na <sup>+</sup>	4.31 $\pm$ 1.02	4.93 $\pm$ 0.40	7.00 $\pm$ 0.20
	Cl <sup>-</sup>	15.21 $\pm$ 1.00	22.75 $\pm$ 4.68	29.58 $\pm$ 5.22

3.79±1.21 folds higher Na<sup>+</sup> than C and F, respectively (p=0.001). The highest accumulation of Cl<sup>-</sup> was also achieved by cattail irrigated with LL, but the significance between F and LL was not obvious (p=0.167). In terms of the treatment by cattail, the Na<sup>+</sup> under LL was highest which was also similar to alkaligrass. However, cattail had more efficient performance for Cl<sup>-</sup> extraction.

Figure 4.9 represents the distribution of Na<sup>+</sup> accumulated in roots and aboveground tissues, and the results shown that higher amount of Na<sup>+</sup> was accumulated in the aboveground tissues of alkaligrass under all the treatments. With the irrigation of LL, around 61.50%±2.5% of Na<sup>+</sup> was found in the aboveground tissues, while it was only 57.61%±2.95% for C. However, different trend was observed for cattail, more Na<sup>+</sup> was found in the roots under C and F, whereas no significant difference between roots and aboveground tissues under the treatment of LL.

Unlike the Na<sup>+</sup> distribution in alkaligrass, the Cl<sup>-</sup> under F and LL had a higher amount in the roots, they were 51.74%±2.47% and 54.96%±4.68% respectively, compared to 47.22%±2.27% under C (Figure 4.10). However, similar trend for the Cl<sup>-</sup> distribution was found in cattail compared to Na<sup>+</sup>. The results shown 54.92%±2.34% and 59.97%±8.65% Cl<sup>-</sup> was accumulated in the roots for C and F, however, this figure decreased to 50.88%±1.52% with the treatment of LL.

The results revealed that as expected, the higher concentration of Na<sup>+</sup> and Cl<sup>-</sup> in LL contributes to the accumulation of the elements in biomass tissues compared with C and F. There are also some studies have been conducted for phytodesalination of landfill leachate but using different plants species. Zalesny et al. (2008) studied sodium and chloride accumulation and distribution by using *Populus* and their results shown similar Na<sup>+</sup> accumulation compared with this study. Zhao et al. (2005) used two chenopodiaceae hyperaccumulator to study the accumulation of Na<sup>+</sup> and Cl<sup>-</sup>, and results revealed that the aboveground tissues usually accumulated more elements than

the roots. MsSorley et al. (2006) used three halophytes, namely *Phragmites australis*, *Puccinellia nuttalliana* (salt accumulators), and *Spartina pectinata* (salt excretor), to study the phytoextraction mechanism and their results revealed that alkaligrass shoots accumulated the highest  $\text{Cl}^-$  than the other two species, and the  $\text{Na}^+$  and  $\text{Cl}^-$  concentration was 2.80 g/Kg shoots and 32.00 g/Kg shoots, respectively.

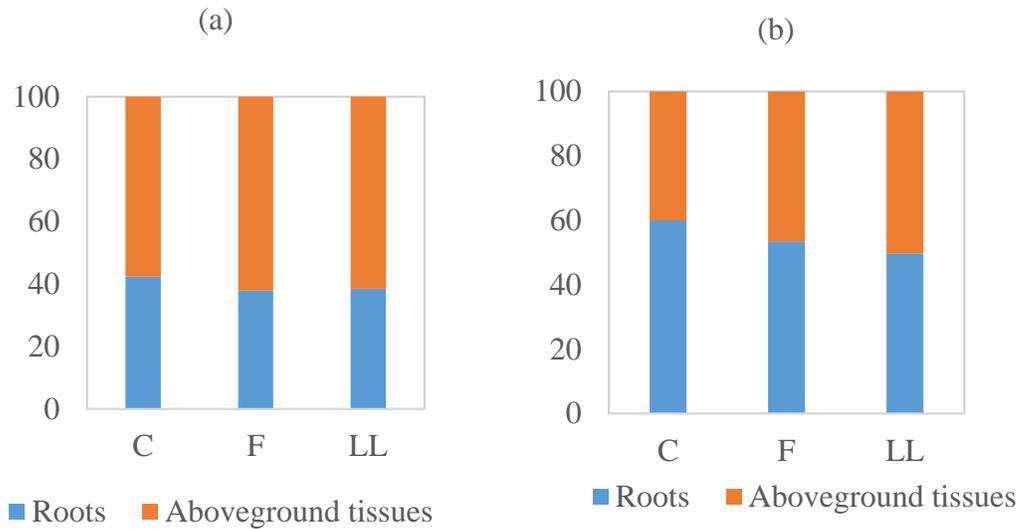


Figure 4. 9 Distribution (roots and aboveground tissues) of  $\text{Na}^+$  in plants (a) alkaligrass (b) cattail.

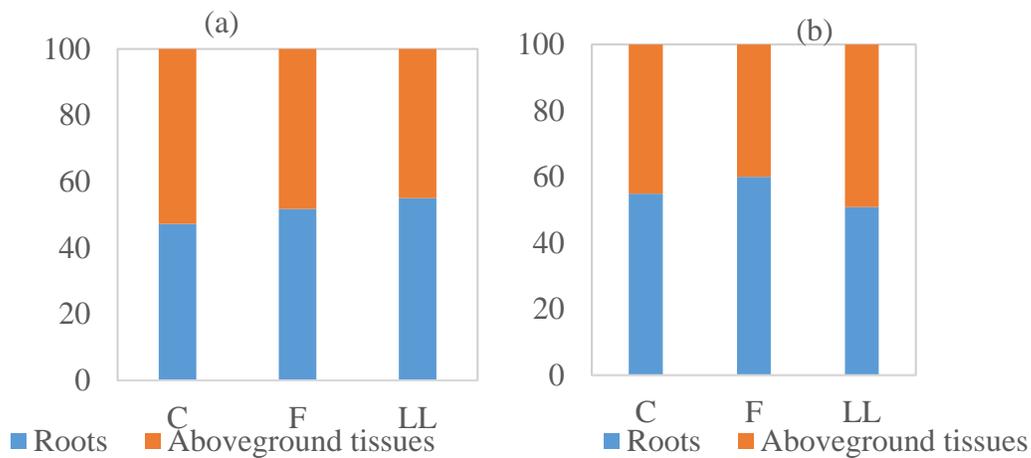


Figure 4. 10 Distribution (roots and aboveground tissues) of  $\text{Cl}^-$  in plants (a) alkaligrass (b) cattail.

### 4.3.2. Hydroponic cattail

The growth rate of hydroponic cattail (Figure 4.11) was found to be approximately  $10.08 \pm 4.87$  cm/week, which was lower than the cattail grown in soils, and also less biomass yield (Table 4.3) considering duration of the hydroponic tests relatively shorter.

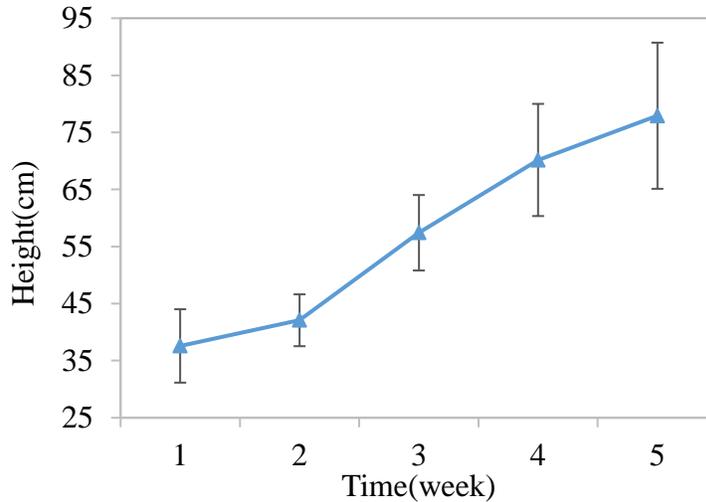


Figure 4. 11 Hydroponic cattail mean growth in height during 5 weeks' study. Data was recorded with 3 replicates (6 plants for each system, n=18). Values are mean  $\pm$  standard deviation.

The  $\text{NH}_4\text{-N}$  removal by hydroponic cattail increased with the increasing  $\text{NH}_4\text{-N}$  load under T1 T2 and T3 (Figure 4.12). When the  $\text{NH}_4\text{-N}$  increased from  $101.6 \pm 5.8$  mg/L in the feed to  $243.3 \pm 9.3$  mg/L, the effluent had the concentration of  $56.9 \pm 6.4$  mg/L and  $72.7 \pm 6.3$  mg/L, respectively. The removal efficiency increased from  $43.96\% \pm 6.68\%$  to  $70.12\% \pm 1.26\%$  ( $p=0.02$ ).

Table 4. 3 Phytoaccumulation by hydroponic cattail. Samples were harvested after 5 weeks' study. Values are mean  $\pm$  standard deviation.

Biomass(g)	$\text{NH}_4\text{-N(g/Kg)}$	$\text{PO}_4\text{-P(g/Kg)}$	$\text{Na}^+(\text{g/Kg})$	$\text{Cl}^-(\text{g/Kg})$
$13.52 \pm 1.29$	$51.14 \pm 5.94$	$9.32 \pm 3.22$	$55.53 \pm 4.82$	$78.22 \pm 28.28$

Maddison et al (2009) obtained similar results by studying cattail biomass and nutrient content in wastewater treatment wetland and their results have shown that the average aboveground

biomass of cattail varied from 0.37 to 1.76 Kg DW/m<sup>2</sup> and the N and P was 52.8-95.3 g/Kg and 8.1-17.3 g/Kg respectively. Another study using cattail for trace metal bioaccumulation has shown that the aboveground biomass yield in the lake was around 0.28-1.99 Kg DW/m<sup>2</sup> (Klink 2017). Tang et al. (2009) studied nutrient removal from constructed wetlands with cattail to

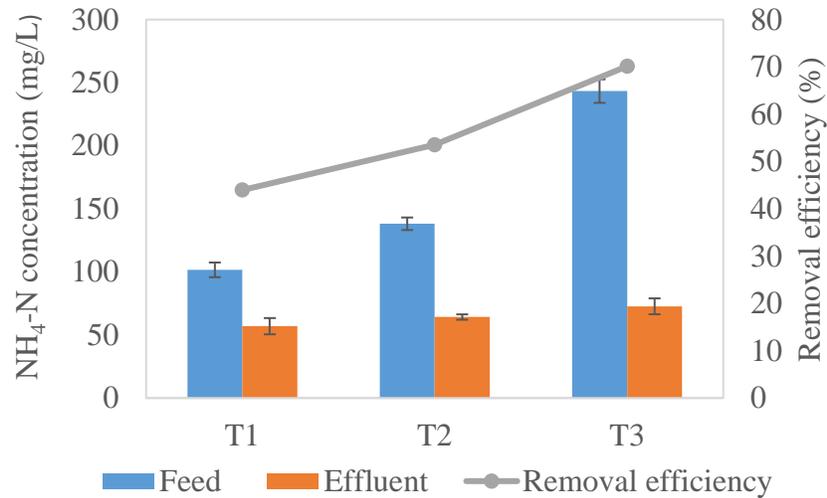


Figure 4. 12 NH<sub>4</sub>-N removal via hydroponic cattail with different feeding load. T1, T2 and T3 were 10%, 20% and 30% leachate by volume. Feed was collected right after treatment and effluent was collected after one week retention time. Values are mean ± standard deviation for 3 replicates.

treat eutrophic river water and their results achieved higher nitrogen removal which was 21.45-79.93 g N/m<sup>2</sup> after harvest, however, the plants had grown for around 8 months compared with 5 weeks in this study. Rozema et al. (2016) used hydroponic system with cattail to study sodium and chloride deduction of recycled nutrient solution, and their results have shown that the Na<sup>+</sup> and Cl<sup>-</sup> content was 5.96 mg/g and 21.8 mg/g respectively after harvest. Even though constructed wetlands are commonly applied for wastewater purification, the construction and maintenance could be relatively expensive than raft floating system or other hydroponic systems. There is also a lack of extensive study from literature regarding the use of hydroponic cattail for wastewater treatment, thus this study provides information for further large-scale investigations.

#### **4.4. Conclusion**

The results from this study revealed that landfill leachate which contains large amount of nutrient could improve plants biomass yield compared to control and can be considered as irrigation alternative, particularly for cattail which yielded similar biomass compared with plants irrigated with fertilizer. Both halophytes could also effectively phytoextract nitrogen and phosphorus from soil irrigated leachate. No photoinhibition was observed in this study under LL, which also proved the selected plants are able to survive in this experiment condition, which gives information for further study with a higher dosage. The EC test indicated that both plants could significantly decrease salts content from the soil and phytoaccumulate sodium and chloride in their biomass. The hydroponic results exhibited that landfill leachate could potentially be applied as nutrient solution, which is durable for plants irrigation. Compared with the constructed wetlands, hydroponic system should be considered as a more cost-effective means for wastewater treatment, regarding the construction and operation cost. Overall, the study proved the possibility of using North American native halophytes for phytoremediation of landfill leachate, which could be a cost-effective and promising approach.

## Chapter 5 Engineering Significance

Combination of biological method and AOPs can significantly save cost compared to only chemical methods. Investigation shown that 72% of schemes used biological methods out of 166 leachate treatment plants surveyed (Vazquez et al., 2004). According to Cortez et al. (2011) and Canizales et al. (2013), Fenton's cost varies based on the influent COD concentration, and the cost is 1.3 €/m<sup>3</sup> and 2.85 €/m<sup>3</sup> for 340 mg/L and 1403 mg/L respectively, which is more cost-effective than other AOPs (Oloibiri et al., 2017).

Phytoremediation by using plants for remediation is considered as an efficient and cost-effective approach for wastewater and contaminated soil treatment. This is particularly important for remote communities and also developing countries where budget is limited (Hasanuzzaman et al., 2014). The usage of wastewater for plants irrigation can also provide certain amount of nutrient for plants growth and this can save addition of fertilizers. The results from this study suggested the feasibility of phytoremediation even for full-scale treatment on-site in the future. Moreover, the hydroponic system from this study shown to be efficient and it can also save cost compared with constructed wetland system, regarding construction and operation.

## **Appendix 1. Solid Waste Management in First Nations Communities**

### **1. Introduction**

First Nations (FN) is a term put forward in the 1970s to replace the word “Indian” that is considered offensive by some aboriginal people (Aboriginal Affairs and Northern Development Canada, AANDC). “First Nations people” refers to status and non-status “Indian” people in Canada. According to the 2011 National Household Survey, more than 1.4 million people in Canada identified themselves as Aboriginal persons, which was 4% of the population, and 50% of these were registered as Indians, 30% as Metis, 15% as non-status Indians, and 4% were Inuit. Currently, there are 618 First Nations communities in Canada, which represent more than 50 nations or cultural groups that speak 50 different Aboriginal languages. In Manitoba, there are 63 First Nations communities and 148,455 registered members as of July 2014. A total number of 88,076 (59.3%) of those members lived on reserves, which is second only to Ontario in terms of total on-reserve population and in total First Nations population (AANDC). The Aboriginal population increased from 312,800 to 1,836,000 during 1901 and 2011, while the increase for the rest of the Canadian population was only 52%. Based on the population size of First Nation reserves, 125 reserves had a population between 250 and 499, and 70% of the reserves had less than 500 inhabitants. The explosion of the population changed the pattern of consumption and also the generation of garbage, which adds up the burden on waste management.

Although more than 50% of the First Nations people live in urban areas, many of them still live in remote and isolated places where the all-weather road is not even available. For example, Manitoba has twenty-three First Nations communities that are not accessible by an all-weather road. This accounts for more than half of all Manitoba First Nations people who live on reserve (AANDC).

In 1701 (Pre-Confederation of Canada), the British Crown entered into solemn treaties to encourage peaceful relations between First Nations and non-Aboriginal people. Subsequently, treaties were signed to define, among other things, the respective rights of Aboriginal people and governments to use and enjoy lands that Aboriginal people traditionally occupied. There is a total of 6 treaties in Manitoba and under these numbered Treaties, the First Nations people who occupied these territories gave up large areas of land to the Crown. As an exchange, the Treaties provided reserve lands and other benefits like farm equipment, animals, annual payments, ammunition, clothing, and certain rights to hunt and fish (Treaty No. 1, 3, 4, 5, 6, and 10). Since historically, first nation communities have gained support and benefits from the treaties in olden times from the government assistance; however, there are still a lot of areas not covered and left on the margins such as poor infrastructure and environmental issues.

Solid waste constitute of discarded material other than liquid. Based on the sources and types, solid waste can be divided into residential, industrial, commercial, institutional, construction, demolition, municipal services, process, and agriculture waste. Municipal solid waste is commonly referred to as trash, garbage or refuse, and rubbish. Although it is deemed waste and thus unwanted; however, some items are valuable for recycling and others are reused for industrial production or energy generation. Solid waste generation, collection, transportation, separation, treatment, and disposal make up an entire solid waste management system. An environmentally sound solid waste management system requires all these processes that collectively meet the appropriate regulations. However, the poor solid waste management in Canada's First Nations communities is a fact and has always been a long-standing concern. Because of the remoteness, non-all-weather roads, and insufficient funding, many First Nations communities do not have access to modern solid waste management facilities as well as the

services, and as a result, open-site dumping and open-air burning are commonly employed practices.

## **2. Solid waste management in First Nations communities**

### **2.1. Open-site dumping and open-air burning**

The Minister of Indian and Northern Affairs Canada (INAC) assumes the responsibility for operation of waste dumps and landfills in First Nations communities and although the INAC has a fiduciary responsibility for waste disposal, most waste sites operated in First Nations communities remain unregulated (Lalita et al., 2006). Waste dumpsites on most FN reserves have been reported as lacking environmental protection measures such as cover materials, engineered liners, or a leachate collection system, and are usually installed without geological considerations (Rebecca et al., 2011). Direct discharge of waste into the environment can pose threat to surrounding soil and groundwater, making it one of the serious concerns.

According to the First Nations on-reserve source water protection plan (guide and template), point source pollution originates from a landfill where leachate contaminates groundwater and subsequently contaminates the downstream source water. Furthermore, according to the on-reserve source water risk assessment results, landfill leachate was one of the most potential sources of water contamination (Table 1).

Ashraf et al. (2013) conducted a research on contaminant transport at an open-tipping waste disposal site and found that contaminants including heavy metals; namely, Fe, Mn, Cu, Cr, Ni, Zn, Pb, and Co were shown to migrate vertically and/or horizontally from the source point (i.e., the disposed garbage) to soil and groundwater. Ajah et al. (2015) systematically established forty sampling nodes around the dumpsite in Ugwuaji, Nigeria and their research found that the status

of soil in the dumpsite and the environs had been heavily compromised due to indiscriminate disposal of untreated waste.

Table 1 First Nations on-reserve source water risk assessment

Contamination Source	Likelihood	Impact	Risk ranking (likelihood × Impact)
Agriculture	5	4	20
Landfill leachate	4	5	20
Private septic systems	4	4	16
Arsenic groundwater	3	5	15
Streambank erosion	2	4	8
Summer recreation	1	3	3
Winter recreation	1	2	2

Source: First Nations on-reserve source water protection plan (2014)

The order of abundance of the monitored heavy metals was

Pb>Fe>As>Zn>Cu>Co>Ni>Cd>Cr>Mn. Another research conducted by Hafeez et al. (2016) revealed the existence of persistent organic pollutants (POPs) including polychlorinated biphenyls (PCBs), polybrominated diphenyl ethers (PBDEs) and dechloran plus (DP) in air, dust, soil, and water samples from a waste dump site in Lahore, Pakistan. It showed baseline data on PBDEs, DP, and PCBs in environmental matrices emitted from waste into the dumping site and health risk assessment of studied POPs in soil and dust through different pathways that presented potential risk due to the carcinogenicity of PCBs. Oketola and Akpotu (2015) also found the concentrations of polyaromatic hydrocarbons (PAHs) and PCBs ranging from 0.85 to 1.47 mg/L and 0.01 to 0.08 mg/L, respectively. However, the values for PCH and PCB in the topsoil were 0.94-2.79 mg/Kg and 10.0-412 ug/Kg, respectively. The authors also revealed that if the

dumpsite is not properly managed, the leachate can seep into the groundwater and surface water via runoff, and thus have adverse effects on human health and the entire ecosystem. Improper solid waste management like uncontrolled dumpsites can also lead to the emission of volatile organic compounds (VOCs), which are hazardous to human health. Majumdar and Srivastava (2012) analyzed the sampled air from the dumpsite in Mumbai, India, using gas chromatography-mass spectrometry (GC-MS) in accordance with U.S. Environmental Protection Agency (EPA) TO-17 compendium method for the toxic compounds, and as many as 64 VOCs were qualitatively identified, among which 13 are listed under Hazardous Air Pollutants (HAPs). Their assessment of total carcinogenic risk for the workers in the dumpsite considering all target HAPs was calculated to be 275 persons in 1 million.

Because of the lack of environmentally sound solid waste management systems, open-air burning of solid waste is also a common practice although it had been banned by regulations and laws. However, based on the principle of “out of sight out of mind”, some people tend to burn the piles of solid waste as to significantly reduce the volume of waste. This open-air burning of waste leads to the toxic emissions, especially with the changing pattern of consumption, and the increasing use and disposal of anthropogenic products, which has quietly emerged as a serious environmental concern. Of particular concern are several families of organochlorine compounds, dioxins, and furans, formed as PICs when plastics are incompletely oxidized in the low-temperature environment of burn barrels (Lighthall and Kopecky, 2000). In Canada, the open burning of garbage produces more dioxins and furans than all industrial activities combined (Environmental and Climate Change Canada, 2015). The biggest source of dioxins and furans in Canada is the large-scale burning of municipal and medical waste. Human’s exposure to dioxins can cause certain health effects including skin disorders, liver problems, impairment of the

immune system, the endocrine system, and reproductive functions, effects on the developing nervous system, other developing events, and certain types of cancers (Health Canada, 2006). Open burning of solid waste is also an important source of particulate matter (PM) emissions, which are associated with various health effects, including respiratory and cardiovascular disease, adverse birth outcomes, and cancer (Ramaswami et al., 2016)

## **2.2. Household hazardous waste**

Household hazardous materials possess any or all of the following characteristics: toxic, corrosive, flammable, and reactive. Several common products used in households possess items: such as solvents, antifreeze, pesticides, oil-based paints, batteries, medication, e-products, and etc. (Vassilis et al., 2015). High disposal costs, lack of disposal facilities, along with increasing stringency of laws and regulations, and declining or limited natural resources have been cited as some of the problems associated with the poor management of household hazardous waste (Ephraim et al., 2014). According to the claim from Environmental and Climate Change Canada (2016), there are three levels of government contribution to environmental protection and the management of hazardous waste in Canada. Municipal governments are responsible for the establishment of the collection, recycling, composting, and disposal programs within their jurisdictions, and the provincial and territorial governments establish measures and criteria for licensing hazardous-waste generators, carriers, and treatment facilities, in addition to controlling movements of waste within their jurisdictions. The federal government regulates transboundary movement of hazardous waste and hazardous recyclable material, in addition to negotiating international agreements related to chemicals and waste (Environmental and Climate Change Canada, 2016).

In Canada, under the Canadian Environmental Protection Act of 1999 (CEPA, 1991), regulations are released toward the management of hazardous waste. Also, there are a lot of organizations in Canada and Province of Manitoba that assume the operations of management, recycling, and disposal of hazardous waste such as Canadian Association of Tire Recycling Agencies (CATRA), Tire Stewardship Manitoba (TSM), Canadian Plastics Industry Association (CPIA), Medications Return Program (MRP), Manitoba Electronic Products Recycling Association (EPRA), and Used Oil Management Association of Canada (UOMA). All of the aforementioned disposal agencies and also other related organizations have improved the recycling and safe handling of household hazardous waste; however, when it comes to the activities in remote and isolated First Nations communities, there is little work been done.

Without proper collection and treatment of household hazardous waste, it is likely that the First Nations people dispose of the waste items such as electronic products, used oil, scrap tires, old batteries, etc. in the open dumpsites. This household waste is being collectively called e-waste. Uncontrolled disposal of e-waste is an emerging issue elevated by the rapidly increasing quantities of complex end-of-life electronic products; informal e-waste recycling has introduced large amounts of toxic substances in which poses health risks to exposed population (Zeng and et al., 2016). Polychlorinated biphenyls, polychlorinated diphenyl ethers, and heavy metals are a major health concern for workers engaged in waste disposal and processing, residents living near these facilities, and are also a detriment to the natural environment (Awasthi et al., 2016). The penetration of e-waste related materials into the subsurface layer of the earth can also contaminate soil and groundwater. Anna et al. (2013) found that the highest average total Polycyclic aromatic hydrocarbon (PAHs) concentration in combusted residues of wires, cables, and other computer components located at two e-waste open dumping and open burning areas

were 195- and 113-fold higher than the PAHs concentration of soil at the control site. The establishment of a comprehensive system for the management of healthcare waste is also essential for environment protection. However, improper waste disposal, insufficient financial resources, lacking awareness of health hazards, and lack of data on healthcare waste generation and disposal are some of the main issues impeding the development of waste management (Issam et al., 2010).

### **3. Policy & legislation framework and government funding: from very beginning**

#### **3.1. From history and federal aspect**

Solid waste management including source separation, collection, transportation, recycle, reduce, treatment, and disposal has developed a lot in the past 150 years, and most of the Western Nations have policies and regulations for the continuous monitoring and evaluation of the social and environmental effects (Lalita et al., 2016). This is also true for many Canadian cities and regions. Although several agencies and policies have been developed to protect people from environmental hazards in Canada, no equivalent mechanisms exist at present within the terms of self-government agreements that enable First Nations people to control environmental impacts on their lands. Historically, First Nations communities have been left at the margins in policy development, and the revenues for their participation in federal, provincial, and territorial review processes are still unclear or unsatisfactory (Bharadwaj et al, 2006). Under the 1978 Indian Reserve Waste Disposal Regulations, no person shall operate a garbage dump or use any land to dispose of, burn, or store waste without a permit. However, based on the information from INAC, there are 365 First Nations communities in British Columbia, Saskatchewan, and Ontario whose land is managed by the Indian Act, but only 14 of those communities have been issued the permit. Therefore, the other communities without a permit are prone to using the site-specific

waste management techniques such as open burning and open-air dumping. While the provision of solid waste management is a municipally led function, the regulations of relative practices are supposed to be set up by the provincial governments under their justifications.

### **3.2. The Canadian Environmental Assessment Act**

In 1992, the Canadian Environmental Assessment Act (CEAA) was enacted by the Government of Canada to achieve sustainable development by evaluating and mitigating adverse environmental effects resulting from projects under the federal jurisdiction. Therefore, this legislation supports planning and decision-making for designated projects at a federal level and it also delineates distinct roles and responsibilities to reduce the potential for overlap between the jurisdictions of the Federal and Provincial governments. However, in 2012, the Government of Canada repealed this legislation and substituted for a new CEAA 2012, which significantly narrows the nature and scope of the federal environmental assessment obligations. Consequently, the Canadian Environmental Law Association (CELA) views this new CEAA 2012 as an unjustified and ill-conceived rollback of the federal environmental law.

### **3.3. The Canadian Environmental Protection Act**

Assented to 14<sup>th</sup> September 1999, the Canadian Environmental Protection Act (CEPA) declared that the protection of the environment is essential to the well-being of Canadians and that is the primary purpose of this Act, which is to contribute to the sustainable development through pollution prevention. Also, CEPA is the primary legislation that gives the Federal government jurisdictional authority for involvement in solid waste related matters.

### **3.4. Canadian Council of Ministers of the Environment**

The Canadian Council of Ministers of the Environment (CCME) serves as a forum for Federal and Provincial Environmental Ministers to collaborate in developing overarching tools that jointly undertake initiatives to address major environmental issues. Through CCME they have developed a variety of Canada-wide policies and a wide range of supporting technical products. For management of solid waste, it has issued the policies and guidelines for bio-solids, compost quality, extended producer responsibility, hazardous waste, packaging, PCBs, along with other waste management. Particularly, the Solid Waste Management Task Group of the CCME commissioned UMA Environmental to study and evaluate small-scale waste management models (SSMs), which are appropriate for implementation in rural, remote, and isolated Canadian communities and regions. In this case, the methods for managing solid waste were identified through an investigation of existing small-scale waste management models in a variety of Northern American and European jurisdictions (EMA Environment, 1995).

### **3.5. Governmental funding**

#### **a) Infrastructure Canada**

In 1999, the Government of Canada outlined a new vision, which provided measures to improve the quality of life for Canadians and make a long-term contribution towards a dynamic economy through the building of infrastructure for the 21st century. Infrastructure Canada was created as a separate organization in 2002 under the Financial Administration Act and it delivers a broad range of infrastructure programs along with providing flexible and effective funding support to provincial, territorial, municipal, the private sector, and not-for-profit infrastructure projects.

In the Budget 2000, the Government of Canada reiterated its commitment to supporting the country's physical infrastructure by allocating \$2.05 billion over six years to improving urban and rural municipal infrastructure across Canada through the Physical Infrastructure Project

(PIP) (Infrastructure Canada, 2010). Of this, a total number of \$31.13 million was allocated towards the First Nations component to improve the quality of life in First Nations communities. The First Nations component of the Infrastructure Canada Program (ICP-FN) operated from 2001-2007 within the ICP. The INAC was solely responsible for the program delivery of ICP-FN, and it was operated within INAC's Capital Facilities Program. The ICP-FN was a multi-year collaborative initiative amongst the Government of Canada, First Nations communities, and their partners like the neighboring municipalities. Through ICP-FN, 37 new green infrastructures along with 9 cultural and recreational facilities, 3 local transportation infrastructure, 3 affordable housing and 3 other projects were funded. However, only 17 of the 37 green infrastructures went to solid waste components, which were mainly new landfill site construction. Moreover, not all the provinces or regions received the funding for the upgrade. Financially, around \$2.5 million was distributed to solid waste management facilities, which merely accounted for 8% of the total funding of ICP-FN. Under the management of infrastructure Canada, \$120 billion funds were planned over next 10 years for public transit, social infrastructure, and green infrastructure (2016).

#### b) Capital Facilities and Maintenance Program

The Capital Facilities and Maintenance (CFM) program within Aboriginal Affairs and Northern Development Canada (AANDC) is the main pillar of the Government of Canada's effort to support community infrastructure for First Nations on reserve (INAC, 2015). The objective of the CFM program is to provide financial support to First Nations and other eligible recipients to invest in physical assets (or services) that mitigate health and safety risks in their communities, establishment of codes and standards for the aforesaid assets, management of those assets in a cost-effective and efficient manner that protects, maintains and maximizes asset lifecycle, and

ensure that the above activities are undertaken in an environmentally sound and sustainable manner. The expected result of the Capital Facilities and Maintenance Program (CFMP) is that First Nations communities have a base of infrastructure that ensures health and safety and enables engagement in the economy

c) First Nations Infrastructure Funds (FNIF) Program

The FNIF program was created as a complementary source of funding to the Capital Facilities and Maintenance Program for six eligible categories of infrastructures projects:

- planning and skills development
- solid waste management
- roads and bridges
- energy systems
- connectivity

The objective of the FNIF program is to improve the quality of life and the environment for First Nations communities by improving and increasing public infrastructure on reserves. Other goals include increased access to the Crown Land, which is land set aside for the use and benefit of a First Nations community, or access to an off-reserve in the case of a cost-shared project with non-First Nations partners such as neighboring municipalities. The expected result of the FNIF includes improving health and safety of the First Nations communities; contributing to a cleaner and healthier environment; enhancing collaboration between the Government of Canada, First Nations communities, municipalities, provinces, and the private sectors; and leveraging other sources of funds for infrastructure projects in First Nation communities.

In Budget 2007, the FNIF was announced as part of the Canada's Infrastructure Plan and \$127 million funding was pooled from three existing federal sources, namely Infrastructure Canada's Municipal Rural Infrastructure Fund (MRIF), Gas Tax Fund (GTF), and the Capital Facilities and Maintenance Program (CFMP). In 2009, AANDC accessed an additional \$107.6 million to increase the total FNIF funding envelope to \$234.9 million. However, the funding contributed to solid waste management only account for 11 % of the total funding among other projects, namely connectivity, energy system, planning and skill, roads and bridge development (FNIF activity report, 2007-2012).

In 2014-2015, the Government of Canada announced \$155 million over ten years from the New Building Canada Fund and \$139 million over five years from the Gas Tax Fund for the FNIF. Starting in 2016-2017, Budget 2016 proposed an additional \$255 million over two years to the First Nation Infrastructure Fund.

#### **4. Conclusion and recommendation**

The environmentally sound management of waste and used materials through a hierarchy of actions or the 5Rs regarding waste, namely reduce, reuse, recycle, recover, and retain. The goal of the 5Rs is to divert solid waste materials out of the waste disposal stream. However, because of the remoteness, insufficient funding, unclear jurisdiction, and lacking enforcement, the situation in First Nations communities is a long-standing concern. Historically, First Nations also had limited involvement in research concerning environmental contamination and policies with respect to solid waste management, which definitely needs to be improved.

For remote and isolated First Nations communities, the unique or innovative components for a long run should include 1) waste and household hazardous waste reduction including reuse, recycle and proper disposal supported through public education program; 2) segregated waste

management programs, and 3) reuse of waste within the community that is supported and encouraged through public. A feasible and suitable scenario to improvement should get better involvement of the government, tribal council, band office, and also other stakeholders. The involvement of stakeholders such as retailers, taking the electronic products retailers as an example, could be a suitable solution for short-term plans. Properly designed and mandatory take-back programs through retailers can significantly increase users' involvement and convenience in waste recycling and management compared to voluntary collection programs.

## **Appendix 2. Phytoremediation of Polluted Soil from Waste Open Dumping**

### **Site- A Case Study for Northern Manitoba First Nation Community**

#### **1. Introduction**

Solid waste management has been widely studied for contaminants deduction. As a cost-effective approach, landfilling of waste is adopted the most commonly. Despite some benefits of landfilling, the generation of leachate from waste decomposition and rainfall permeation is a long-standing concern, which could pollute the surrounding environment (El-Fadel et al., 1997; Paoli et al., 2012; Zhang et al., 2013). Moreover, in some remote and small communities or developing countries, waste open dumping is still a common practice because of the lacking of engineering technique and funding resources (Zhang et al., 2011). The composition of solid waste may contain electronics, paper, plastic, metal, glass and textile which are source of organic and inorganic contaminants (Hafeez et al., 2016). Open waste dumping sites can cause environmental hazardous through waste burning in the open, leaching and run-off from the waste decomposition, and it also provides the breeding of disease vectors (Oketola and Akpotu, 2015). In recent decades, a lot of research has been focused on the treatment of landfill leachate and also remediation of the polluted soil such as biological and physicochemical methods (Schiopu and Gavrilescu, 2010). In order to reduce the treatment expenses, phytoremediation by means of plants' accumulation of contaminants has drawn extensive research interest. The plants used for remediation can transfer, remove and degrade pollutants in sediments, soil and water (Hughes et al., 1997; Yan et al., 2016). The advantage of phytoremediation is that it saves cost by reducing chemicals and energy input, and it does not cause secondary pollution issues (Maneesuwannarat et al., 2013). In this study, we aimed to evaluate bioremediation performance of cattail (*Typha*

*latifolia*) regarding salinity removal from polluted soil of a waste open dumping site in Northern Manitoba.

After eight weeks' cultivation, all the plants from soil test were harvested and analyzed the nitrogen, phosphorous, sodium and chloride accumulation in biomass. This study extended the technique for wasted soil bio-treatment in small and remote communities of Northern Prairies.

## **2. Methods and materials**

### **2.1. Plant species and germination**

The cattail used in this study was germinated based on the same methods described in Chapter 4.

### **2.2. Treatments**

#### a) Polluted soil test

In the soil test, cattail was used to treat polluted soil (PS) from a waste open dumping site of Northern Manitoba, Canada. Two cattail seedlings were transplanted into each pot (38.7\*24.8\*29.2 cm) and grown 8 weeks in greenhouse condition. Two types of soil was used, one is mixture of clay/sand/peat with 1:1:1 ratio by volume (Department of Plant Science, University of Manitoba, Winnipeg, MB, Canada) which is also considered as control (C), and the other type of soil was the PS. Three replicates were employed for both C and PS. The plants and soil were sampled after 8-week experiment.

#### b) Hydroponic test

3 months old seedlings were used for the hydroponic test in the floating raft system (40.6\*27.3\*17.8 cm) (Figure 1), and the landfill leachate was diluted with tap water by volume. Rock wool cubes were pre-treated with acid-solution overnight. The acid-solution was prepared with 0.1N phosphoric acid by adjusting pH to be around 6.4 with distilled water. The rock wool cubes were then rinsed with distilled water. 6 seedlings

were transplanted into each container and held by the rock wool cubes. 1.5 gallons of solution was changed weekly and pH was checked daily and adjusted to be around 6.4. Aeration was distributed from the bottom by using aquarium air pump (Airpod, Penn Plax, NY, USA). The percentage of leachate was increased from 10% (T1) to 15% (T2) and finally 20% (T3), by weekly. The water sample (feed and effluent) was collected weekly. After 5-week experiment, the plants were harvested for analysis.

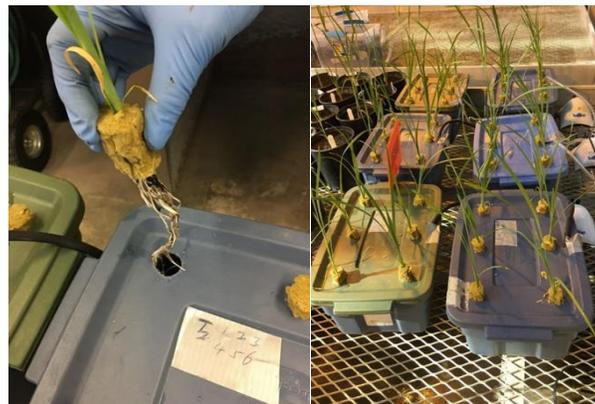


Figure 1 Hydroponic test with cattail seedlings

### 2.3. Analysis

The plants and soil samples were prepared as the same methods described in Chapter 4. The same digestion method was also used for the plants and soil samples. For the measurement of electrical conductivity ( $EC_5$ ), 35 mL DI water and 7 g dried and ground soil was put into a 50 mL centrifuge tube to form soil paste, and then the filtrate was used for  $EC_5$  measurement.

$Na^+$  was analyzed by using Atomic Absorption Spectrometer (AAAnalyst 400, PerkinElmer Inc., MA, USA) with  $C_2H_2$  and Air flow rate of 2.50 L/min and 10.0 L/min, respectively. The wavelength was set to be 589 nm to detect emission signal based on standards prepared (Robinson, 1996).  $Cl^-$  was detected by using chloride test kit (Model 8-P, Hach company,

Loveland, Colorado, USA) (Urgun-Demirrtas et al., 2006). Digested plants samples were neutralized for NH<sub>4</sub>-N and PO<sub>4</sub>-P analysis by using flow injection analysis (FIA, Lachat Instrument QuikChem 8500, Loveland, Colorado, USA) (Ren et al., 2017). EC<sub>5</sub> was examined by using a conductivity meter (Fisherbrand Traceable Conductivity, Resistivity, and TDS Meter, Fisher Scientific, USA).

For data analysis, one-way ANOVA with LSD was used to calculate significance ( $p < 0.05$ ). The statistic was performed by using IBM SPSS 24 with 3 replicates.

### **3. Results and discussion**

Figure 2 represents the cattail grown on soil condition (C and PS) during this period of study. The cattail under PS showed 15.25 cm/week growth rate over time, while the plants under C were slightly slower with 14.17 cm/week ( $p < 0.05$ ). The biomass yield of cattail grown in the polluted soil also had  $25.9\% \pm 1.2\%$  more biomass compared with that grown in control soil (Table 1) ( $p < 0.05$ ). The difference between C and PS in terms of biomass yield could be related to the nutrient levels of two types of soil, the relatively lower N content in C may have adverse effect on the biomass yield (Smith et al., 2015). Alternatively, the soil from waste dumping site is regarded as polluted soil, because of the discarded waste may contain hazardous wastes. On the other hand it could also be rich in nutrient and minerals which are essential for plants establishment. This is especially the case when leachate generated from waste piles penetrates into surrounding soil, and a lot of studies have also revealed that the leachate can be a potential supply of N for plants growth due to high concentration of ammoniac substances. Cheng and Chu (2011) irrigated *Hibiscus tiliaceus* and *Litsea glutinosa* with leachate and the results demonstrated that the trees doubled shoot biomass when they were irrigated with leachate compared with control irrigated with only tap water.

Table 1 Biomass yield and salinity change. Samples were from the harvest after 8 weeks' experiment. Values are mean  $\pm$  standard deviation.

	Biomass (g)	Plants (g/Kg biomass)				Soil (EC) (ms/m)	
		Dry weight	NH <sub>4</sub> -N	PO <sub>4</sub> -P	Na <sup>+</sup>	Cl <sup>-</sup>	Initial
C	35.28 $\pm$ 4.28	5.23 $\pm$ 0.03	0.92 $\pm$ 0.01	5.81 $\pm$ 2.05	29.45 $\pm$ 0.77	44.6 $\pm$ 0.6	45.1 $\pm$ 6.9
PS	44.43 $\pm$ 3.29	7.90 $\pm$ 0.37	1.59 $\pm$ 0.17	10.82 $\pm$ 1.85	64.69 $\pm$ 9.15	245.0 $\pm$ 1.4	51.9 $\pm$ 9.3

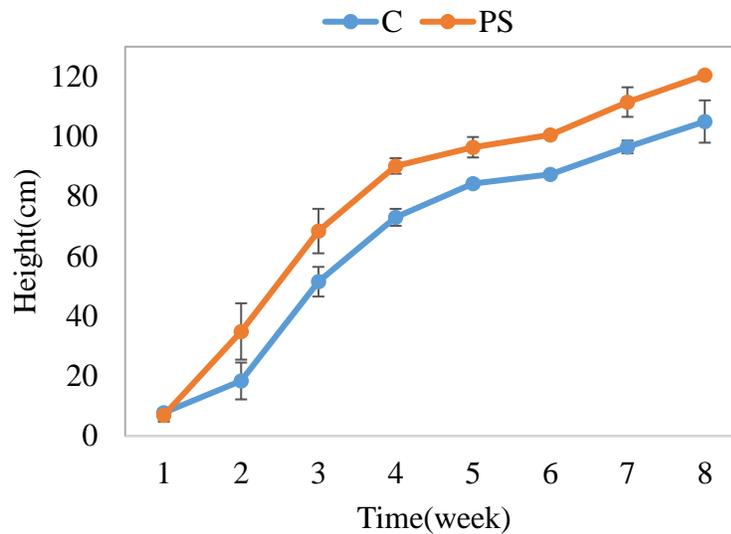


Figure 2 Mean plants growth (height) in control soil (C) and polluted soil (PS). Height was measured weekly with 3 replicates. Values are mean  $\pm$  standard deviation.

Figure 3 shows nutrient content deduction in soils for both C and PS. The NH<sub>4</sub>-N decreased from 2.87 $\pm$ 0.45 g/kg soil to 2.49 $\pm$ 0.16 g/kg soil (p=0.002), whereas 69.55% $\pm$ 10.55% PO<sub>4</sub>-P deduction was observed from C (p=0.001). Considering the nutrient removal from PS, 19.64% $\pm$ 4.49% of N and 68.22% $\pm$ 15.81% of P were removed under this experiment, respectively (p=0.0018). The

cattail grown in PS also accumulated more  $\text{NH}_4\text{-N}$  and  $\text{PO}_4\text{-P}$  in the biomass as shown in Table 1.

A field study of *Typha angustifolia* in Tingitan Peninsula has shown that the biomass yield was  $5,653 \text{ g/m}^2$  which was higher than that from this study, and the N content was 1.44% dry weight compared to 0.52% (C) and 0.79% (PS) in this study (Ennabilia et al., 1998). This deduction could be related to several environmental stress for the cattail growth in controlled environment, such as the effect of plants density which can change the plant physiology and morphology (Poorter et al., 2016).

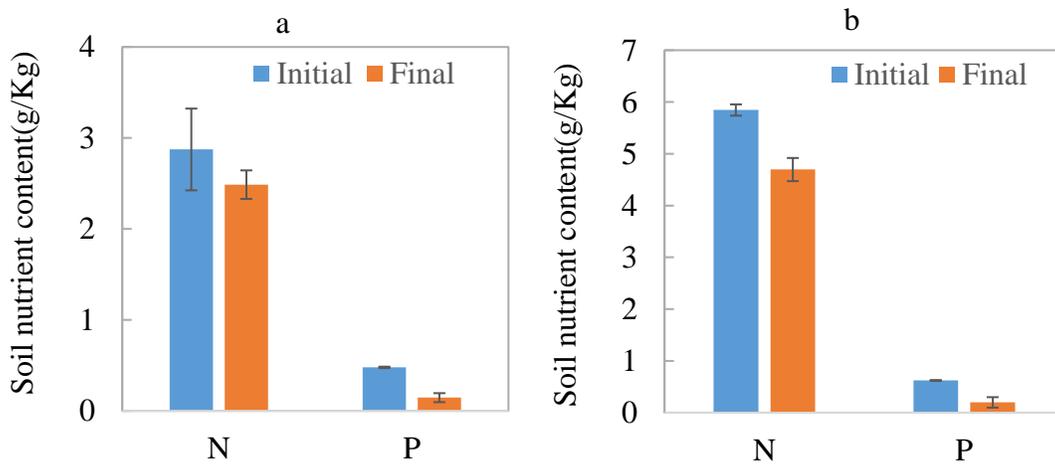


Figure 3 Nutrient (nitrogen and phosphorous) content deduction in control soil (a) and polluted soil (b) by cattail. Soil samples were collected after 8 weeks' experiment when the plants were harvested. Values are mean  $\pm$  standard deviation.

Another factor is the complex components in the PS compared to the soil from field test. A lot of studies have proved that heavy metals and persistent organic compounds can be formed and leached into the surrounding soil in dumping site (Kulikowska and Klimiuk, 2008; Lou et al., 2009). Lyubenova and Schroder (2011) demonstrated that the combination of heavy metal and

organic xenobiotic from waste could inhibit plants function. The results also showed that  $\text{Na}^+$  was significantly higher in plants grown in PS than C, they were  $64.69 \pm 9.15$  g/Kg biomass and  $29.45 \pm 0.77$  g/Kg biomass, respectively. Morteau et al. (2009) compared the performance of *Typha latifolia*, *Atriplex patula* and *Spergularia Canadensis* for treatment of salted road runoffs and the results demonstrated that *Typha* had the highest efficiency for salt accumulation and the  $\text{Cl}^-$  was found to be 63 mg/g dry biomass, which is slightly lower than the results from PS in this study. McSorley et al. (2016) investigated chloride extraction by an accumulator (*Puccinellia nuttalliana*) and they observed that the chloride content was about 31 g/Kg biomass. The decrease of EC from soil was also found significant ( $p=0.04$ ), starting from  $245.0 \pm 1.4$  ms/m and ending at  $51.9 \pm 9.3$  ms/m. Sargeant et al. (2008) evaluated the desalination performance by *Distichlis spicata* on a saline discharge zoon and they found out that the surface soil salinity ( $\text{EC}_5$ ) was reduced by 50% over a 8 years' monitoring.

#### **4. Conclusion**

Results from this study shown higher biomass yield for cattail growing in the waste dumping site soil compared with the control soil, which suggests that the contaminated soil may have rich content of nutrient that is beneficial for plants growth. The cattail is also proved to be efficient for  $\text{Na}^+$  and  $\text{Cl}^-$  accumulation. Therefore, phytoremediation of contaminated sites in remote and small communities by using native plants should be considered as a feasible approach and further real site study.

## References

- Akinremi O.O., Armisen N., Kashem M.A., Janzen H.H., 2003. Evaluation of Analytical Methods for Total Phosphorus in Organic Amendements. *Communications in Soil Science and Plant Analysis*. 34(19&20), 2981-2991.
- Amr, S.S., Aziz, H.A., 2012. New treatment of stabilized leachate by ozone/Fenton in the advanced oxidation process. *Waste Management*. 32(2012), 1693-1698.
- Anna, O., Leung, W., Cheung, K.C., Wong, M.H., 2013. Spatial distribution of polycyclic aromatic hydrocarbons on soil, sediment and combusted residue at an e-waste processing site in southeast China. *Environmental Science and Pollution Research*. 22 (12), 8786-8801.
- Anita, J., Tatjana, T, Andreja, Z.G., 2012. Assessment of landfill leachate toxicity reduction after biological treatment. *Arch Environ Contam Toxicol*. 2012 (62), 210-221.
- Anu, R., Navneet, K.B., Ajay, S.N., 2016. Exploring social and infrastructural factors affecting open burning of municipal solid waste in Indian cities: A comparative case study of three neighborhoods of Delhi. *Waste Management & Research*. 34 (11), 1164-1172.
- Awasthi, A.K., Zeng, X.L., Li, J.H., 2016. Relationship between e-waste recycling and human health risk in India: a critical review. *Environmental Science and Pollution Research*. 23 (12), 111509-11532.
- Baharak, P.A., Abdul, S., Wan, R.A.A., 2017. Ozone (O<sub>3</sub>) and sono (US) based advanced oxidation processes for the removal of color, COD and determination of electrical energy from landfill leachate. *Separation and Purification Technology*. 172 (2017), 442-449.

Bautista, P., Mohedano, A.F., Casas, J.A., Zazo, J.A., Rodriguez, J.J., 2008. An overview of the application of Fenton oxidation to industrial wastewater treatment. *Chemical Technology and Biotechnology*. 83(10), 1323-1338.

Bawiec, A., 2018. Efficiency of nitrogen and phosphorus compounds removal in hydroponic wastewater treatment plant. *Environmental Technology*. 10(1080).

Beata, B., Krzysztof, P., Zbigniew, L., Jan, B., 2008. Degradation of microcystin- LR by ozone in the presence of Fenton reagent. *Journal of Environmental Science and Health*. 43(2), 186-190.

Brink, A., Sheridan, C.M., Harding, K.G., 2017. The Fenton oxidation of biologically treated paper and pulp mill effluents: A performance and kinetic study. *Process Safety and Environmental Protection*. 107 (2017), 206-215.

Broetto, F., Duarte, H.M., Luttge, U., 2007. Responses of chlorophyll fluorescence parameters of the facultative halophyte and C<sub>3</sub>-CAM intermediate species *Mesembryanthemum crystallinum* to salinity and high irradiance stress. *Journal of Plant Physiology*. 164 (2007), 904—912.

Ch, J.A.J., Romero, R.M., 2016. Evaluation of *Cajanus cajan* (pigeon pea) for phytoremediation of landfill leachate containing chromium and lead. *International Journal of Phytoremediation*. 18 (11), 1122-1127.

Chedly, T., Latifa, B., Loubna, M., Ahmed, G., 2007. Landfill leachate treatment with ozone and ozone/hydrogen peroxide systems. *Journal of Hazardous Materials*. 140 (1-2), 316-324.

Chemlal, R., Azzouz, L., Kernani, R., Abdi, N., Lounici, H., Grib, H., Mameri, N., 2014. Combination of advanced oxidation and biological processes for the landfill leachate treatment. *Ecological Engineering*. 72 (2014), 281-289.

Cheng, C.Y., Chu, L.M., 2011. Fate and distribution of nitrogen in soil and plants irrigated with landfill leachate. *Waste Management*. 31(2011), 1239–1249.

David, R.L., Steven, K., 2000. Confronting the Problem of Backyard Burning: The Case for a National Ban. *Society & Natural Resources*. 13 (2000), 157-167.

Dickerman, J.A., Stewart, A.J., 1986. Estimates of Net Annual Aboveground Production: Sensitivity to Sampling Frequency. *Ecology*. 37 (3), 650-659.

Dipanjali, M., Anjali, S., 2012. Volatile organic compound emission from municipal solid waste disposal sites: A case study of Mumbai, India. *Journal of the Air & Waste Management Association*. 62 (2012), 398-407.

Dorthe, L.B., Thomas, H.C., 2004. Speciation of heavy metals in landfill leachate: a review. *Waste Management & Research*. 2004 (23), 3-23.

El-Fadel, M., Findikakis, A.N., Leckie, J.O., 1997. Environmental Impacts of Solid Waste Landfilling. *Journal of Environmental Management*. 50(1997), 1–25.

Ennabilia, A., Atera, M., Radoux, M., 1998. Biomass production and NPK retention in macrophytes from wetlands of the Tingitan Peninsula. *Aquatic Botany* 62(1998), 45-56.

Ephraim, M., Tye, L., Laura, V., Kelly, B., 2014. Voluntary Approach to Solid Waste Management in Small Town: A Case Study of Community Involvement in Household Hazardous Waste Recycling. *Journal of Environmental Health*. 76 (10), 26-33.

Fernandez, R.T., Perry, R.L., Flore, J., 1997. Drought Response of Young Apple Trees on Three Rootstocks. II. Gas exchange, Chlorophyll Fluorescence, Water Relations, and Leaf Abscisic Acid. *J. Amer. Soc.Hort. Sci.* 133 (6), 841-848.

Ferraz, F.M., Bruni, A.T., Povinelli, J., Vieira, E.M., 2016. Leachate/domestic wastewater aerobic co-treatment: A pilot-scale study using multivariate analysis. *Journal of Environmental Management*. 166 (2016), 414-419.

Freedman, A., Gross, A., Shelef, O., Rachmilevitch, S., Arnon, S., 2014. Salt uptake and evapotranspiration under arid conditions in horizontal subsurface flow constructed wetland planted with halophytes. *Ecological Engineering*. 70 (2014), 282-286.

Gabrial, T.M., Alexandre, G., Edson, A.C., Marco, A.S.R., Alvaro, M., Andrea, M.B., 2015. The effect of sanitary landfill leachate aging on the biological treatment and assessment of photoelectrooxidation as a pre-treatment process. *Waste Management*. 36 (2015), 177-183.

Glenn E.P., Brown J.J., Blumwald E., 1999. Salt tolerance and crop potential of halophytes. *Critical Reviews in Plant Sciences*. 18(2), 227-255.

Glenn, E.P., Mckeon, C., Gerhart, V., Nagler, P.L., Jordan, F., Artiola, J., 2009. Deficit irrigation of a landscape halophyte for reuse of saline waste water in a desert city. *Landscape and Urban Planning*. 89 (2009), 57-64.

Grzegorz, B., Andre, F., 2017. Wastewater treatment by means of advanced oxidation processes at basic pH conditions: A review. *Chemical Engineering Journal*. 320 (2017), 608-633.

Gupta, A., Zhao, R., Novak, J.T., Goldsmith, C.D., 2014. Application of Fenton's reagent as a polishing step for removal of UV quenching organic constituents in biologically treated landfill leachates. *Chemosphere*. 105(2014), 82-86.

Haarstad, K., Maehlum, T., 1999. Important aspects of long-term production and treatment of municipal solid waste leachate. *Waste Management Research*. 17, 470-477.

Hafeez, S., Mahmood, A., Syed J.H., Li, J., Ali, U., Malik, R.N., Zhang, G., 2016. Waste dumping sites as a potential source of POPs and associated health risks in perspective of current waste management practices in Lahore city, Pakistan. *Science of the Total Environment*. 562(2016), 953–961.

Hammer, U.T., Heseltine, J.M., 1988. Aquatic macrophytes in saline lakes of the Canadian prairies. *Hydrobiologia*. 158 (1988), 101-116.

Hasanuzzaman, M., Nahar, K., Alam-Mahabub, Md., Bhowmik-C, P., Hossain-Amzad, Md., Rahman-M., M., 2014. Potential Use of Halophytes to Remediate Saline Soils. *BioMed Research International*. 2014, 1-12.

He, Y., DeSutter, T., Prunty, L., Hopkins, D., Jia, X., Wysochi, D.A., 2012. Evaluation of 1:5 soil to water extract electrical conductivity methods. *Geoderma*. 185–186 (2012) 12–17.

Hong M., Lu G., Hou C., She S., Zhu L. Advanced treatment of landfill leachate membrane concentrates: performance comparison, biosafety and toxic residue analysis. *Water Sci Technol*. 2017 76(11):2949-2958. doi: 10.2166/wst.2017.469.

Hugh, R.E., 1964. Oxidation of Phenolic Wastes. *Water Pollution Control Federation*. 36 (9), 1116-1128.

Hughes, J.B., Shanks, J., Vanderford, M., Lauritzen, J., Bhadra, R., 1997. Transformation of TNT by Aquatic Plants and Plant Tissue Cultures. *Environ. Sci. Technol*. 1997 (31), 266-271.

Issam, A., Maria, M., Salem, A.M., Mohammed, M.A, Despo., k., 2010. Dental solid and hazardous waste management and safety practices in developing countries: Nablus district, Palestine. *Waste management & Research*. 28 (2010), 436-444.

Jeke, N.N., Hassan, A.O., Zvomuya, F., 2017. Phytoremediation of biosolids from an end-of-life municipal lagoon using cattail (*Typha latifolia* L.) and switchgrass (*Panicum virgatum* L.). *International Journal of Phytoremediation*. 19 (3), 270-280.

Jing, Z., Tao., Y., Hongyu, W., Kai, Y., Cheng, F., Bin, L., Xiaojun, Y., 2016. Study on treating old landfill leachate by Ultrasound-Fenton oxidation combined with MAP chemical precipitation. *Chemical Speciation & Bioavailability*. 27 (4), 175-182.

Johannes, S., Jurg, H., 1985. Decomposition of Ozone in Water in the Presence of Organic Solutes Acting as Promoters and Inhibitor of Radical Chain Reactions. *Environ. Sci. Technol.* 1985 (19), 1206-1213.

Jones, D.L., Williamson, K.L., Owen, A.G., 2006. Phytoremediation of landfill leachate. *Waste Management*. 26 (2006), 825-837.

Jun, L., Kunquan, L., Yan, Z., Xuebin, L., Tao, T., 2016. Kinetic analysis of *Legionella* inactivation using ozone in wastewater. *Chemosphere*. 168 (2017), 630-637.

Kanayochukwu, C.A., Joel, A., Chidozie, C.N., 2015. Spatiality, seasonality and ecological risks of heavy metals in the vicinity of a degenerate municipal central dumpsite in Enugu, Nigeria. *Journal of Environmental Health Science & Engineering*. 2015, 13:15.

Klink, A., 2017. A comparison of trace metal bioaccumulation and distribution in *Typha latifolia* and *Phragmites australis*: implication for phytoremediation. *Environ. Sci. Pollut. Res.* 24(2017), 3843–3852.

Kulikowska, D., Klimiuk, E., 2008. The effect of landfill age on municipal leachate composition. *Bioresource Technology*. 99 (2008), 5981–5985.

Lalita, B., Suzie, N., Ian, J.H., Gene, O., Laura, P., Ceal, T., Daryl, W., Darcy, B., Gilbert, L., Austin, B., 2006. Waste disposal in First-Nations Communities: the issues and steps toward the future. *Journal of Environmental Health*. 68 (7), 35-9.

Liu, Y., Coulman, B.E., 2014. Morphological and agronomic variation of *Puccinellia nuttalliana* populations from the Canadian Great Plains. *Can. J. Plant Sci.* 95 (2015), 67-76.

Lobna, M., Marina, S., Sven-Uwe, G., Latifa, B., 2015. Catalysed ozonation for removal of an endocrine-disrupting compound using the O<sub>3</sub>/Fenton reagents system. *Environmental Technology*. 36 (13), 1721-1730.

Lou, Z., Zhao, Y., Yuan, T., Song, Y., Chen, H., Zhu, N., Huan, R., 2009. Natural attenuation and characterization of contaminants composition in landfill leachate under different disposing ages. *Science of the Total Environment*. 407 (2009), 3385–3391.

Maddisona, M., Soosaara, K., Muringa, T., Mandera, U., 2009. The biomass and nutrient and heavy metal content of cattails and reeds in wastewater treatment wetlands for the production of construction material in Estonia. *Desalination*. 246(2009), 120–128.

Majid, P., Mohsen, J., Mahsa, A., Etesam, G., 2016. Old landfill leachate treatment through multistage process: membrane adsorption bioreactor and nanofiltration. 39 (2016), 1803-1816.

Maneesuwannarat, D., Maneesuwannarat, S., Nilratnisakorn, S., Thiravetyan, P., 2013. Effect of Silicon on Calcium, Proline, Growth Rate and Salt Stress of Narrow-Leaved Cattails in Synthetic Reactive Dye Wastewater. *International Journal of Phytoremediation*. 15(1), 24-37.

Maronon, E., Castrillon, L., Fernandez-Nava, Y., Fernandez-Mendez, A., 2010. Colour, turbidity and COD removal from old landfill leachate by coagulation-flocculation treatment. *Waste Management & Research*. 28 (2010), 731-737.

- Maxwell, K., Johnson, G.N., 2000. Chlorophyll fluorescence-a practical guide. *Journal of Experimental Botany*. 51 (345), 659-668.
- McNaughton, S.L., 1968. Autotoxic Feedback In Relation to Germination and Seeding Growth in *Typha Latifolia*. *Ecology*. 49 (2).
- McSorley, K.A., Rutter, A., Cumming, R., Zeeb, B.A., 2016. Chloride accumulation vs chloride excretion: Phytoextraction potential of three halophytic grass species growing in a salinized landfill. *Science of the Total Environment*. 572(2016), 1132–1137.
- Michael, C., Violet, A.O., Wim, T.M.A., Kristof, D., Stijn, W.H.V.H., 2015. Ozonation of biologically treated landfill leachate: efficiency and insights in organic conversions. *Chemical Engineering Journal*. 277 (2015), 104-111.
- Millar, J.B., 1976. Wetland classification in western Canada: a guide to marshes and shallow open water wetlands in the grasslands and parklands of the prairie provinces. *Canadian Wildlife Service*. 37.
- Milton, M.A., 2017. Integrated of Fenton with biological and physical-chemical methods in the treatment of complex effluents: a review. *Environmental Technology Reviews*. 6(1), 156-173.
- Mishra, A., Tanna, B., 2017. Halophytes: Potential Resources for Salt Stress Tolerance Genes and Promoters. *Front. Plant Sci*. 8 (2017), 829.
- Morteau, B., Triffault, B.G., Galvez, R., Martel, L., Leroueil, S., 2009. Treatment of salted road runoffs using *Typha latifolia*, *Spergularia canadensis*, and *Atriplex patula*: a comparison of their salt removal potential. *Journal of ASTM International*. 2009(6), 4.

Muhammad, A.A., Ismail, Y., Mohamad, Y., Yatimah, A., 2013. Study of contaminant transport at an open-tipping waste disposal site. *Environ Sci Pollut Res.* 20 (2013), 4689-4710.

Murchie, E.H., Lawson, T., 2013. Chlorophyll fluorescence analysis: a guide to good practice and understanding some new applications. *Journal of Experimental Botany.* 64 (13), 3983-3998.

Niloufar, M.G., Andres, A.L., Michael, J.W., 2014. Hydroxyl radical ( $\cdot\text{OH}$ ) scavenging in young and mature landfill leachates. *Water Research.* 56 (2014), 148-155.

Oketola, A.A., Akpotu, O.S., 2014. Assessment of solid waste and dumpsite leachate and topsoil. *Chemistry and Ecology.* 31 (2), 134-146.

Padmavathiamma, P. K., Ahmed, M., Rahman-Abdel, H., 2014. Phytoremediation-A sustainable approach for contaminant remediation in arid and semi-arid regions-a review. *Environ. J. Food Agric.* 26 (9), 757-772.

Pan, J., Sun, H., Nduwimana, A., Wang, Y., Zhou, G., Ying, Y., Zhang, R., 2007. Hydroponic plate/fabric/grass system for treatment of aquacultural wastewater. *Aquacultural Engineering.* 37 (2007), 266–273.

Paoli, L., Corsini, A., Bigagli, V., Vannini, J., Bruscoli, C., Loppi, S., 2012. Long-term biological monitoring of environmental quality around a solid waste landfill assessed with lichens. *Environmental Pollution* 161(2012), 70-75.

Pocostales, J.P., Sein, M.M., Konolle, W., Sonntag, C.V., Schmidt, T.C., 2010. Degradation of Ozone-Refractory Organic Phosphates in Wastewater by Ozone and Ozone/Hydrogen Peroxide (Perozone): The Role of Ozone Consumption by Dissolved Organic Matter. *Environ. Sci. Technol.* 44 (21), 8248-8253.

Poorter, H., Fioranni, F., Pieruschka, R., Wojciechowski, T., Putten, W.H., Kleyer, M., Schurr, U., Postma, J., 2016. Pampered inside, pestered outside? Differences and similarities between plants growing in controlled conditions and in the field. *New Phytologist*. 212 (2016), 838–855.

Qadir, M., Oster, J.D., Schubert, S., Noble, A.D., Sahrawat, K.L., 2007. *Advances in Agronomy*. 96, 199-239.

Rabhi, M., Ferchichi, S., Jouini, J., Hamrouni, M.H., Koyro, H.W., Ranieri, A., Abdelly, C., Smaoui, A., 2010. Phytodesalination of a salt-affected soil with the halophyte *Sesuvium portulacastrum* L. to arrange in advance the requirements for the successful growth of a glycophytic crop. *Bioresource Technology*. 101 (2010), 6822-6828.

Rebecca, Z., Ian, J.H., Suzie, N., Lalita, B., 2011. Perspectives on Past and Present Waste Disposal Practices: A Community-Based Participatory Research Project in Three Saskatchewan First Nations Communities. *Environmental Health Insight*. 5 (2011), 9-20.

Ren, Y.N., Ferraz, F., Lashkarizadeh, M., Yuan, Q.Y., 2017. Comparing young landfill leachate treatment efficiency and process stability using aerobic granular sludge and suspended growth activated sludge. *Journal of Water Process Engineering*. 17(2017), 161-167.

Renou, S., Givaudan, J.G., Poulain, S., Dirassouyan, F., Moulin, P., 2008. Landfill leachate treatment: Review and opportunity. *Journal of Hazardous Materials*. 150 (2008), 468-493.

Ritchie, G.A., 2006. Chlorophyll Fluorescence: What Is It and What Do the Numbers Mean. *USDA Forest Service Proceedings RMRS-P-43*. 23(1960), 458-461.

Robinson, J.W., 1960. Determination of sodium by atomic absorption spectroscopy. *Analytica Chimica Acta*.

Rowland, A.P., Grimshaw H.M., 1985. A wet oxidation procedure suitable for total nitrogen and phosphorous in soil. *Communications in Soil Sciences and Plant Analysis*. 16(1985), 551-560.

Rozema, E.R., Gordon, R.J., 2014. Plant Species for the Removal of Na<sup>+</sup> and Cl<sup>-</sup> from Greenhouse Nutrient Solution. *Hortscience*. 49(8), 1071–1075.

Rozema, E.R., Gordon, R.J., Zheng, Y., 2016. Harvesting Plants in Constructed Wetlands to Increase Biomass Production and Na<sup>+</sup> and Cl<sup>-</sup> Removal from Recycled Greenhouse Nutrient Solution. *Water Air Soil Pollut*. 227(2016), 136.

Saba, H., Adeel, M., Jabir, H.S., Li, J., Usman, A., Riffat, N.M., Zhang, G., 2016. Waste dumping sites as a potential source of POPs and associated health risks in perspective of current waste management practices in Lahore city, Pakistan. *Science of the Total Environment*. 562 (2016), 953-961.

Safaa, M.R., Ahmed, M.M., Hala, A.H., 2013. Treatment of leachate from municipal solid waste landfill. *HBRC journal*. 2013 (9), 187-192.

Sargeant, M.R., Tang, C., Sale, P.W.G., 2008. The ability of *Distichlis spicata* to grow sustainably within a saline discharge zone while improving the soil chemical and physical properties. *Australian Journal of Soil Research*. 2008(46), 37-44.

Schiopu, A.M., Gavrilesco, M., 2010. Options for the Treatment and Management of Municipal Landfill Leachate: Common and Specific Issues. *Soil Air Water*. 38(12), 1101-1110.

Shiyab, S.M., Shibli, R.A., Mohammad, M.M., 2006. Influence of Sodium Chloride Salt Stress on Growth and Nutrient Acquisition of Sour Orange In Vitro. *Journal of Plant Nutrition*. 26 (5), 985-996.

- Smith, S.M., Thime, A.R., Zilla, B., Lee, K., 2015. Responses of Narrowleaf Cattail (*Typha angustifolia*) to Combinations of Salinity and Nutrient Additions: Implications for Coastal Marsh Restoration. *Ecological Restoration*. 33 (3), 297-302.
- Steward, R.E., Kantrud, H.A., 1972. Vegetation of Prairie Potholes, North Dakota, in Relation to Quality of Water and Other Environmental Factors.
- Sumona, M., Soumyadeep, M., Mohd, A.H., Bhaskar, S.G., 2014. Contemporary Environmental issues of landfill leachate: assessment and remedies. *Critical Reviews in Environmental Science and Technology*. 45 (5), 472-590.
- Susana, C., Pilar, T., Rosario, O., Manuel, M., 2011. Evaluation of Fenton and ozone-based advanced oxidation processes as mature landfill leachate pre-treatments. *Journal of Environmental Management*. 92(3), 749-755.
- Tang, W.Z., Huang, C.P., 1996. 2,4-Dichlorophenol Oxidation Kinetics by Fenton's Reagent. *Environmental Technology*. 17 (12), 1371-1378.
- Tang, X., Huang, S., Scholz, M., Li, J., 2008. Nutrient Removal in Pilot-Scale Constructed Wetlands Treating Eutrophic River Water: Assessment of Plants, Intermittent Artificial Aeration and Polyhedron Hollow Polypropylene Balls. *Water Air Soil Pollut*. 197(2009), 61–73.
- Tarasoff, C.S., Mallory-Smith, C.A., Ball, D.A., 2007. Comparative plant responses of *Puccinellia distans* and *Puccinellia nuttalliana* to sodic versus normal soil types. *Journal of Arid Environments*. 70 (2007) 403–417.
- Thomas, R.H., Xiao-Feng, Z., 1994. Sorption by kaolinite of Cd<sup>2+</sup>, Pb<sup>2+</sup> and Cu<sup>2+</sup> from landfill leachate-contaminated groundwater. *Journal of Contaminant Hydrology*. 16 (1994), 271-287.

Tonni, A.K., Wai-Hung, L., G.Y.S., C., 2006. Radicals-catalyzed oxidation reactions from degradation of recalcitrant compounds from landfill leachate. *Chemical Engineering Journal*. 125 (2006), 35-57.

Turcio, A., Weichgrebe, D., Papenbrock, J., 2016. Potential use of the facultative halophyte *Chenopodium quinoa Willd.* as substrate for biogas production cultivated with different concentrations of sodium chloride under hydroponic conditions. *Bioresource Technology*. 203 (2016), 272-279.

Ungar, I.A., 1970. Species-Soil Relationships on Sulfate Dominated Soils of South Dakota. *The American Midland Naturalist*. 83 (2), 343-357.

Urgun-Demirtas, M., Stark, B.C., Pagilla, K.R., 2006. Comparison of 2-chlorobenzoic acid biodegradation in a membrane bioreactor by *B. cepacia* and *B. cepacia* bearing the bacterial hemoglobin gene. *Water Research*. 40(2006), 3123-3130.

Vasseur, L., Fortin, M.J., Cyr, J., 1998. Clover and cress as indicator species of impacts from limed sewage sludge and landfill wastewater land application. *The Science of the Total Environment*. 217(1998), 231-239.

Vassilis, J.I., Konstaninos, M., 2015. Household hazardous waste management: A review. *Journal of Environmental Management*. 150 (1), 310-321.

Venora G., Blangiforti S., Ruffini Castiglione M., Pignone D., Losavio F., Cremonini R. Chromatin organization and computer aided karyotyping of *Triticum durum* Desf. Cv. Timilia. 2002 *Caryologia* 55(1), 91-98.

Verheye, W., 2009. Soil of arid and semi-arid areas. *Land use, land cover and soil science-(VII)*.

Vilar, V.J.P., Moreira, F.C., Ferreira, A.C.C., Sousa, M.A., Goncalves, C., Alpendurada, M.F., Boaventura, R.A.R., 2012. Biodegradability enhancement of a pesticide-containing bio-treated wastewater using a solar photo-Fenton treatment step followed by a biological oxidation process. *Water Research*. 46(2012), 4599-4613.

Wang, F., Gamal-EI-Din, M., Smith, D.W., 2010. Oxidation of aged raw landfill leachate with O<sub>3</sub> only and O<sub>3</sub>/H<sub>2</sub>O<sub>2</sub>: Treatment efficiency and molecular size distribution analysis. *Ozone: Science & Engineering*. 26 (2004), 287-298.

Wagner, G.M., Miriam, C.S.A., Lisete, C.L., 2013. Evaluation of landfill leachate treatment by advanced oxidative process by Fenton's reagent combined with membrane separation system. *Waste Management*. 33 (2013), 89-101.

Xiao-Chun, X., Hong-Tao, Z., Zhi-Yong, D., Yu-Feng, F., 2013. Pretreatment of old-age landfill leachate by microwave-assisted catalytic oxidation in the presence of activated carbon. *Environmental Technology*. 34 (20), 2853-2858.

Xiao-Song, H., Bei-Dou, X., Ru-Tai, G., Hui, Z., Qiu-Ling, D., Dan, L., 2016. Insight into the composition and degradation potential of dissolved organic matter with different hydrophobicity in landfill leachates. *Chemosphere*. 144 (2016), 75-80.

Yan, K., Xu, H., Zhao, S., Shan, J., Chen, X., 2016. Saline soil desalination by honeysuckle (*Lonicera japonica Thunb.*) depends on salt resistance mechanism. *Ecological Engineering*. 88 (2016), 226–231.

Yensen, N.P., Biel, K.Y., 2006. Soil remediation via salt-conduction and the hypothesis of halosynthesis and photoprotection. *Ecophysiology of high salinity tolerant plants*. 313–344.

Yongjun, C., Qihuui, X., Dandan, G., Huirong, S., 2017. Degradation of an anthraquinone dye by Ozone/Fenton: Response Surface Approach and degradation pathway. *Ozone: Science & Engineering*. 39 (4), 219-232.

Yu, W., M., Pelkonen, J., Kaila, 2010. Cost-saving biological nitrogen removal from strong ammonia landfill leachate. *Waste Management & Research*. 29 (8), 797-806.

Zalesny, J.A., Zalesny, R.S., Wiese, A.H., Sexton, B., Hall, R.B., 2008. Sodium and chloride accumulation in leaf, woody, and root tissue of *Populus* after irrigation with landfill leachate. *Environmental Pollution*. 155 (2008), 72-80.

Zalesny, J.A., Zalesny, R.S., Wiese, A.H., Sexton, B., Hall, R.B., 2008. Uptake of Macro- and Micro-Nutrients into Leaf, Woody, and Root Tissue of *Populus* after Irrigation with Landfill Leachate. *Journal of Sustainable Forestry*. 27 (3), 303-327.

Zeng, X., Xu, X.J., Boezen, H.M., Huo, X., 2016. Children with health impairments by heavy metals in an e-waste recycling area. *Chemosphere*. 148 (2016), 108-415.

Zhang, H., Hu, H., Zhang, X., Wang, K., Song, T., Zeng, F., 2012. Detecting *Suaeda salsa* L. chlorophyll fluorescence response to salinity stress by using hyperspectral reflectance. *Acta Physiol Plant*. 34 (2012), 581–588.

Zhang, H., Yan, X., Cai, Z., Zhang, Y., 2013. Effect of rainfall on the diurnal variations of CH<sub>4</sub>, CO<sub>2</sub>, and N<sub>2</sub>O fluxes from a municipal solid waste landfill. *Science of the Total Environment*. 442(2013), 73–76.

- Zhang, H., Zhang, D.Q., Jin, T.F., He, P.J., Shao, Z.H., Shao, L.M., 2011. Environmental and economic assessment of combined biostabilization and landfill for municipal solid waste. *Journal of Environmental Management*. 92(2011), 2533-2538.
- Zhao, K.F., Fan, H., Song, J., Sun, M.X., Wang, B.Z., Zhang, S.Q., Ungar, I.A., 2005. Two Na<sup>+</sup> and Cl<sup>-</sup> Hyperaccumulators of the Chenopodiaceae. *Journal of Integrative Plant Biology*. 47 (3), 311-318.
- Zheng-Yong, X., Guang-Ming, Z., Zhao-Hui, Y., Yong, X., Ming, C., Hong-Song, S., Li-Li, J., Ying, C., 2010. Biological treatment of landfill leachate with the integration of partial nitrification, anaerobic ammonium oxidation and heterotrophic denitrification. *Bioresource Technology*. 101 (1), 79-86.
- Ziyang, L., Youcai, Z., Tao, Y., Yu, S., Huili, C., Nanwen, Z., Renhua, Z., 2009. Natural attenuation and characterization of contaminants composition in landfill leachate under different disposing ages. *Science of the Total Environment*. 407 (2009), 3385-3391.
- Zorrig, W., Rabhi, M., Ferchichi, S., Smaoui, A., Abdeily, C., 2012. Phytodesalination: a Solution for Salt-affected Soils in Arid and Semi-arid Regions. *Journal of Arid Land Studies*. 22 (1), 299-302.
- Zupanc, V., Justin, M.Z., 2010. Changes in soil characteristics during landfill leachate irrigation of *Populus deltoids*. *Waste Management*. 30 (11), 2130-2136.