Performance of a Surface-Flow Constructed Wetland Treating Landfill Surface-Water Runoff

by

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I hereby declare that I am the sole author of this thesis. This is a true copy of the thesis, including any required final revisions, as accepted by my examiners.

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Abstract

Landfills are a major potential source of groundwater and surface-water contamination. The compounds that can leach from landfilled materials include dissolved organic matter, inorganic macrocomponents, heavy metals, and xebobiotic organic compounds. Landfill surface-water runoff poses a threat to the environment due to high mobility, but has not been rigorously characterized with regards to common pollutants found in landfills. It is well documented that constructed wetlands can serve as an effective treatment option for many pollutants found in landfills. The Napanee Landfill has constructed a wetland in order to treat surface-water runoff coming off the landfill. The objectives of this study were to: 1) characterize the water chemistry of surface-water runoff for an inactive landfill; 2) evaluate the treatment potential for the constructed wetland system at the Napanee Landfill; and, 3) recommend design, maintenance, and operative improvements to enhance effluent water quality. The analysis of the landfill surface-water runoff entering the Napanee Landfill constructed wetland included the pollutants nitrate, ammonia, sulphate, phosphorus, and chloride. The median inflow and outflow concentrations for all of the observed pollutants did not exceed Canadian federal or provincial water quality guidelines. There were sampling days where ammonia, phosphorus, and chloride exceeded guidelines at the inflow and days where ammonia and chloride exceeded guidelines at the outflow. The only pollutant that saw a statistically significant decrease in concentrations was sulphate, with a change of 38% from the inflow to the outflow. Other changes of note were nitrate and phosphorus concentrations increasing by 50% and 23% respectively from the inflow to the outflow. There are a variety of improvements that can be made to the Napanee Landfill constructed wetland that would increase the treatment efficiency of ammonia. Incorporating a vertical-flow wetland would increase available surface area for nitrifying bacteria growth and would provide more oxygen for nitrification processes; both would increase the potential for significant ammonia treatment. Overall, the concentrations of the pollutants found in the surface-water runoff coming off of the Napanee Landfill constructed wetland did not pose a significant threat to the environment at the time of sampling and treatment processes were only successful in reducing sulphate pollutant concentrations.

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1.0 Introduction

Managing municipal waste is a problem that every urban area must address. Municipal waste comes in many forms and is usually characterized as wastewater (sewage) and solid waste. Wastewater consists of sanitary sewage and stormwater. Sanitary sewage comes from homes, business, and industries while stormwater comes from rain or melting snow draining off of various surfaces. Most sanitary sewage is treated before it is discharged into the environment, but in Canada, 150 billion litres of untreated sewage is still discharged to the environment every year (Environment Canada, 2001b). Municipal solid waste consists of commonly used and discarded items; such as, packaging, food scraps, furniture, and electronics (EPA, 2011). In Canada, over 12 million tonnes of municipal solid waste is generated per year (Cameron et al., 2005). Municipal solid waste is managed through recycling, incineration, composting, and disposal in landfills. Approximately 21% of municipal solid waste generated in Canada is recycled; the rest is disposed of through incineration or landfilling.

Landfilling is the more common method for disposing of solid waste. While considered one of the more economical methods for waste disposal, chemicals produced as a result of landfill use have been identified as a dangerous source of environmental pollution (Baedecker and Back, 1979; Arneth *et al*., 1989; Sharma and Lewis, 1994; Bulc, 1997). Of particular concern are the effects landfills have on surrounding ground and surface-water. The two ways landfill pollutants can directly enter the surrounding ground and surface-water are leachate and surface-water runoff.

Landfill leachate is liquid that percolates downward through a landfill extracting components of the landfill material; as a result, leachate characteristics vary with differences in waste composition, waste age, and climate (Bulc, 2006). There are four main groups of pollutants that are found in leachate: dissolved organic matter, inorganic macrocomponents, heavy metals, and xebobiotic organic compounds (Kjeldsen *et al*., 2002). Dissolved organic matter components are qualified as chemical oxygen demand (COD), total organic carbon (TOC), and volatile fatty acids (Kjeldsen *et al*., 2002). The inorganic macrocomponents consist of calcium, magnesium, sodium, potassium, ammonia, iron, manganese, chloride, sulphate, phosphorus, and hydrogen carbonate (Kjeldsen *et al*., 2002). Common heavy metals that are found in leachate are cadmium, chromium, copper, lead, nickel, and zinc (Snow et al., 2008).

Xenobiotic organic compounds enter leachate as a result of household and industrial chemicals and consist of a variety of hydrocarbons, phenols, chlorinated aliphatics, pesticides, and plastizers (Kjeldsen *et al*., 2002). Landfill surface-water runoff has comparable chemistry to that of landfill leachate. While leachate tends to have higher concentrations of dissolved organic matter and xenobiotic organic compounds, landfill surface-water runoff tends to have higher concentrations of inorganic macro components and heavy metals (Marques and Hogland, 2001).

Mitigating the effects of landfill chemicals on the surrounding environment is an essential consideration for landfill management. While leachate is considered the greater threat to the environment; for this reason, it has also received the most attention by researchers and practitioners with approaches to mitigation. Currently, there are numerous ways in which leachate is successfully collected and treated. Surface-water runoff is more difficult to capture and specific research on treatment options specific to landfills for the particular chemicals found in surface-water runoff are limited. Snow et al. (2008) showed that a constructed wetland is effective at treating high concentrations of manganese and iron found in landfill surface-water runoff. Research has been conducted on using constructed wetlands for treating other inorganic macrocomponent pollutants like phosphorus, sulphate, chloride, ammonia, nitrate, and nitrite, but not in relation to these pollutants being found in landfill surface-water runoff.

1.1 Landfill Surface-Water Runoff Water Quality Parameters

Aquatic plants and animals are often exposed to various concentrations of pollutants; this exposure can lead to mortality, physiological deformities, reduced reproductive success, and growth trend impacts. The types of impacts observed depend on a number of variables, including: pollutant concentrations, the type of species making contact with the pollutant, and the age of individuals. Common water parameters observed when analysing water quality are temperature, pH, conductivity, and dissolved oxygen (EPA, 1997). Inorganic macrocomponents are pollutants commonly found in landfill surface-water runoff and include such compounds as phosphorus, sulphate, chloride, ammonia, nitrate, and nitrite. Through a variety of processes, exposure to the previously listed pollutants can lead to negative impacts on exposed plants and animals.

1.1.1 Temperature and pH

Temperature and pH are important water quality parameters. Many species require specific temperature and pH ranges to survive and thrive. The majority of species require a pH between 6.5 and 9.0, which is reflected in national and provincial Canadian water quality guidelines (Table 1-1). In most natural and constructed wetlands, pH follows a consistent mean and does not fluctuate significantly through the year (Kadlec and Wallace, 2009). Aquatic plants and animals can survive in a range of temperatures, but ecosystems contain communities that are adapted to different temperatures, usually categorized into warm and cold water communities. In addition to the inherent characteristics of temperature and pH that affect aquatic plants and animals, temperature and pH also affect the concentrations and interactions of other water quality parameters.

Table 1-1. Landfill leachate pollutant ranges and associated Canadian Environmental Quality Guidelines, Ontario Provincial Water Quality Objectives, and British Columbia Environment Guidelines.

Parameter	Unit	Landfill Leachate Range	Canadian Environmental Quality Guidelines	Ontario Provincial Water Quality Objectives	British Columbia Environment Guidelines
pH	-	4.5 - 9.0[a]	6.5 - 9.0	6.5 - 8.5	-
Conductivity	ms/cm	230-3500[ab]	-	-	-
Dissolved Oxygen	mg/L	-	> 6.5 - 9.5	> 5 - 8	> 5 - 8
Phosphorus	mg/L	0.13 - 4.0[b]	-	0.20	5 - 15
Sulphate	mg/L	22-650[b]	-	-	100
Chloride	mg/L	150 – 4900[ab]	120	-	150
Un-ionized ammonia	mg/L	-	20	19	-
Nitrate	mg/L	-	13	-	3

[a] Kjeldsen et al., (2002)

[b] Oman and Junestedt (2008)

1.1.2 Conductivity

Conductivity is a measurement of the amount of electrical current that can pass through a water sample. A high conductivity indicates the presence of inorganic dissolved solids like chloride, nitrate, sulphate, phosphate, and heavy metals (EPA, 1997). Organic compounds like oil, alcohol, and sugar are not good conductors and lead to low conductivity (EPA, 1997).

Conductivity is heavily influenced by the geology of an area but can also be impacted through anthropogenic sources. Wastewater discharge and landfill surface-water runoff result in increased conductivity due to the presence of inorganic dissolved solid ions (EPA, 1997).

1.1.3 Dissolved Oxygen

Dissolved oxygen is an important component of aquatic ecosystems. It is essential for aerobic aquatic organism survival and influences many chemical reactions. Dissolved oxygen enters water through the atmosphere and photosynthesis by aquatic vegetation (CCME, 1999). The amount of available dissolved oxygen in the water column depends on a variety of factors, including: water depth, inflows, wind, altitude, currents, and water temperature. Cold water can retain more dissolved oxygen than warmer water. Water also hold less dissolved oxygen at high altitudes compared to low altitudes. The presence of aquatic plants results in dissolved oxygen being introduced into the water during the day when photosynthesis occurs and then being removed at night during plant respiration. Macrophytes add more oxygen to water during high growth periods (spring and summer) compared to fall when decomposition of plant organic matter leads to oxygen demand (Stein et al., 2007).

Biological oxygen demand (BOD) and chemical oxygen demand (COD) are two other ways oxygen is removed from water. Biological oxygen demand measures the amount of oxygen removed from water through biological processes. Chemical oxygen demand measures the amount of oxygen removed due to chemically oxidizing reduced minerals and organic matter.

1.1.4 Phosphorus

There are three forms in which phosphorus is found in aquatic environments: inorganic phosphorus, particulate organic phosphorus, and dissolved organic phosphorus. The primary influence of phosphorus on aquatic ecosystems is nutrient loading, with elevated phosphorus levels potentially causing eutrophication (Correll, 1998). Through eutrophication, phosphorus affects an ecosystem by: reducing biodiversity, changing dominant biota, decline in ecologically sensitive species, increases in turbidity, high sedimentation, and anoxic conditions (Mason, 1996). Dissolved inorganic phosphorus is of particular importance since it is considered bioavailable (Reddy et al., 1999).

1.1.5 Sulphate

The sulphate ion is naturally occurring in freshwater environments with ambient concentrations usually ranging between 2 and 50 mg L^{-1} for Canada (Singleton, 2000). While there have been few studies conducted on the toxicity of sulphate, the studies that have been conducted have shown that, with commonly found water chemistry, sulphate toxicity is minimal for most organisms compared to other ions (Mount et al., 1997; Davies, 2007; Elphick et al., 2011). While sulphate toxicity is low under normal water conditions, reduced hardness (concentration of calcium and magnesium ions) can result in greater sulphate toxicity (Elphick et al., 2011). Elphick et al. found that water with hardness of 10-40 mg L^{-1} resulted in sulphate toxicity at concentrations of 100-150 mg L^{-1}.

Although sulphate toxicity is low under most conditions, it can impact the treatment of other compounds. Sulphate has been shown to influence nitrification in treatment wetlands under heavy sulphate and carbon load by reducing available oxygen (Wiessner et al., 2005). Sulphate can also lead to increased phosphorus mobilization due to competition for free iron available for binding (Lamers et al., 2002).

1.1.6 Chloride

The chloride ion is commonly found as a salt but, due to its solubility, often remains in ionic form when in water. Chloride does not readily biodegrade, precipitate, volatilize, bioaccumulate, or absorb onto mineral surfaces (Mayer et al., 1999). Road salt is the primary source of anthropogenically introduced chloride ions to the environment, with an estimated 2,950,728 tonnes of chloride being introduced per year in Canada (CCME, 2011). Chloride is found at high concentrations in municipal solid waste landfill leachate (Kjeldesen et al., 2002), but no studies have observed chloride concentrations for landfill surface-water runoff.

Short-term and long-term chloride toxicity to vertebrates and plants is low, requiring concentrations greater than 500 mg L^{-1} for almost all species (CCME, 2011). Invertebrates are more sensitive to chloride. Some invertebrate species have shown long-term toxicity effects at chloride concentrations as low as 120 mg L^{-1} (Mackie, 1978; Harmon et al., 2003; CCME, 2011). Few studies have been conducted with regards to the toxicity of chloride to plants. The studies that have been conducted have shown plants to be sensitive to chloride concentrations of 1000 mg L^{-1} or greater (Taraldsen et al., 1990; CCME, 2011).

The toxicity of chloride for aquatic animals is a result of osmoregulation disruption. Maintaining ion equilibrium requires energy expenditure, which can be significant enough, if surrounding water ion concentration is high, to cause endocrine imbalance, reduced oxygen consumption, and changes to physiological processes (Varsamos et al., 2005).

1.1.7 Nitrogen and Ammonia

The nitrogen cycle in aquatic systems involves some of the more complicated chemical processes. Total nitrogen concentration is a combined measure of ammonia, nitrate, and nitrite concentrations. Sources of nitrogen include point sources such as wastewaters and mining discharge and non point sources such as agricultural runoff, septic beds, urban runoff, fertilizers, vehicular exhaust, storm sewer overflow, and landfill leachate (NRC, 1978; Constable et al., 2003).

Un-ionized ammonia is the most toxic of the nitrogen compounds, followed by nitrite, and then nitrate (Camargo and Alonso, 2006). Ionized ammonia (ammonium) is considerably less toxic (Constable et al., 2003). Un-ionized ammonia is toxic to two bacteria (*Nitrosomonas* and *Nitrobacter*) involved in the nitrification of ammonia to nitrite to nitrate; so elevated un-ionized ammonia concentrations can result in reduced nitrification and high concentrations of ammonia and nitrite (Anthonisen et al., 1976; Lewis and Morris, 1986).

Ammonia

Organic waste is a common component of landfills, with ammonification transforming the organic waste into ammonia and ammonium. In aqueous solutions, ammonia is found in two states: un-ionized and ionized. The ratio of un-ionized ammonia to ionized ammonia is pH and temperature dependent. At 25 ^{o}C and a pH of 7, un-ionized ammonia is 0.6% of total ammonia, increasing to 72% at 30^{o}C and a pH of 9 (Emerson et al., 1975).

Organisms have different ways of dealing with ammonia. Freshwater fish excrete the more toxic un-ionized ammonia molecule by diffusion through the gills (Wilkie, 1997). Biological membranes are permeable to un-ionized ammonia but not ammonium (Environment Canada, 2001). When the amount of un-ionized ammonia in the water is high, un-ionized ammonia excretion by freshwater fish is reduced and plasma un-ionized ammonia levels increase (Yesaki and Iwama, 1992; Wilson et al., 1994). Freshwater amphibians excrete a lesser amount of un-ionized ammonia through gill and skin diffusion compared to fish, excreting most of their un-ionized ammonia in the form of urea (Munro, 1953). Urea excretion is not impacted by the ammonia concentrations in the surrounding water, allowing amphibians to maintain healthy internal levels of un-ionized ammonia even if the surrounding water is high in ammonia (Wright and Wright, 1996). Freshwater invertebrates are similar to fish in that the majority of un-ionized ammonia excreted is molecularly unchanged and occurs through diffusion (Wright, 1995; Weihrauch et al., 2012). Due to their un-ionized ammonia excretion method, invertebrates can be very sensitive to high ammonia concentrations (Hickey and Vickers, 1994; Alonso and Camargo, 2006). There is little data on the impacts un-ionized ammonia has on freshwater plant species and the studies available are contradictory and incomplete (Environment Canada, 2001). Studies conducted show that various plant species are sensitive to elevated ammonia concentrations, but at much higher concentrations than fish and invertebrates (Vines and Wedding, 1960; van der Eerden, 1982).

When ammonia concentrations overwhelm aquatic organism coping methods, negative physiological impacts result. The direct physiological impacts of ammonia include: damage to gills due to asphyxiation; suppression of Krebs cycle, resulting in reduced blood oxygen-carrying capacity; inhibition and depletion of ATP production in the brain due to increases in brain extracellular glutamate levels; disruption of osmoregulatory activity affecting the liver and kidneys; and, suppression of the immune system (Randall and Tsui, 2002; Camargo and Alonso, 2006).

Nitrite

Nitrite is the intermediate anion between ammonium and nitrate. Nitrite affects aquatic animals primarily by converting hemoglobin to methemoglobin, methemoglobin being unable to release oxygen into body tissues (Eddy and Williams, 1987; Camargo and Alonso, 2006; Tilak et al., 2007). Other physiological impacts that affect some aquatic animals include: electrolyte imbalance; formation of mutagenic and carcinogenic compounds; damage to mitochondria; and, repression of the immune system (Camargo and Alonso, 2006). High concentrations of chloride can reduce the negative effects of nitrite on aquatic animals. Chloride and nitrite share the same uptake mechanism, so high levels of chloride will create competitive inhibition of nitrite uptake (Tomasso et al., 1979; Harris and Coley, 1991; Bartlett and Neumann, 1998). Similar to un-ionized ammonia, fish and invertebrates are more sensitive to nitrite concentrations than amphibians (Camargo and Alonso, 2006). Amphibian larvae are significantly more sensitive to nitrite than adults (Marco et al., 1999). Aquatic animals cope with nitrite toxicity through the internal conversion of nitrite to the non-toxic nitrate ion.

Nitrate

Nitrate is the product of ammonium ion and nitrite ion nitrification. Nitrate uptake in aquatic animals is limited by the low permeability of diffusion surfaces to nitrate (Jensen, 1996). Due to the lower uptake of nitrate compared to ammonia and nitrite, it is often considered a less toxic substance. The toxicity of nitrate is a result of it being converted into nitrite post-uptake and the physiological impacts associated with nitrite (Cheng and Chen, 2002). Over an extended period of time, high concentrations of nitrate in the water can lead to elevated internal levels of nitrate in aquatic organisms, which leads to reduced conversion of nitrite to nitrate and, in some cases, leads to the internal conversion of nitrate to nitrite (Jensen, 1996; Cheng and Chen, 2002; Camargo and Alonso, 2006). Due to the mechanism for nitrate toxicity, sensitivities of various aquatic animals are similar to nitrite, with fish and invertebrates more sensitive than amphibians (Marco et al., 1999; Camargo et al., 2005)

1.2 Classifying Natural and Constructed Wetlands

Natural wetlands act as a buffer between terrestrial and aquatic systems with various functions mitigating impacts of terrestrial systems on aquatic systems (Kennedy and Mayer, 2002). Wetlands are highly productive due to the efficiency of wetland plants to fix carbon and create biomass (Kennedy and Mayer, 2002). Constructed wetlands are designed to take advantage of the natural processes that result from the unique plant and microbial assemblages found in natural wetlands (Vymazal et al, 2006). There are several different classifications of wetlands with specific classifications or aspects of specific classifications being more desirable for treatment purposes.

1.2.1 Classifying Natural Wetlands

It is generally agreed that wetlands are distinguishable by a permanent or periodic covering or saturation of water (NWWG, 1988; Tiner, 1999; Mitsch and Gosselink, 2007). This inundation of water leads to saturated soils and vegetation that is adapted to the amount of water present in the environment (Tiner, 1999; Mitsch and Gosselink, 2007; Keddy, 2010). Wetlands have been classified into five types, based on the hydrodynamics of the site: bogs, fens, swamps, marshes, and shallow open water (NWWG, 1988).

Bogs are peatlands (peat layer greater than 40cm) with a high water table, a lack of nutrients, and an anaerobic environment (Schwintzer, 1981). Bogs are acidic with pH values that are usually below 4.6 (NWWG, 1988). Bogs are recharged by precipitation and lack an inflow and outflow (Mitsch and Gosselink, 2007). Bogs develop in areas with low temperature and abundant precipitation.

Fens are similar to bogs in that they are peatlands, have a high water table, lack nutrients, are anaerobic, and are acidic (NWWG, 1988). Compared to bogs, fens often have a slightly less acidic pH and higher nutrient and oxygen content (while still being low compared to other wetlands) (NWWG, 1988). Fens also have an outflow and are sometimes recharged through seeps, springs, or surface-water (NWWG, 1988; Rydin and Jeglum, 2006).

Swamps are wetlands that have high nutrient content, neutral to acidic water and soil, and are not oxygen deficient (NWWG, 1988). Swamps are characterized by woody trees and shrubs being the dominant vegetation types (NWWG, 1988).

Marshes are similar to swamps in that they have high nutrient content (NWWG, 1988). Marshes have an alkaline pH and high oxygen content (NWWG, 1988). The dominant vegetation type in marshes is emergent vegetation, like *Typha angustifolia* (cattail) and *Phragmites australis* (common reed) (Keddy, 2010).

Shallow open water are the fifth type of wetland and are small bodies of standing water that are free of emergent vegetation and act as a transition zone from marshes and swamps to large bodies of water like lakes and rivers (NWWG, 1988). Unlike in lakes, shallow open water wetlands have a uniform water temperature, with no stratification (NRC, 1995).

1.2.2 Classifying Constructed Wetlands

Constructed wetlands are created by humans through hydrologic manipulation and/or the moving and processing of soil and rock (Fonder and Headley, 2010). The goal for constructed wetlands is often to mimic the functions and processes found in natural wetlands. There are three purposes for wetland construction: restoration, mitigation, and treatment (Fonder and Headley, 2010). Constructed treatment wetlands often aim to increase the effective processes that would be found in a natural wetland. Constructed wetlands for treatment purposes can be classified into three types: horizontal subsurface-flow (HSSF), vertical flow (VF), and free water surface-flow (FWS).

Horizontal subsurface-flow wetlands consist of water flowing through soil or gravel beds under the surface of the wetland in a horizontal path (Kadlec and Wallace, 2009; Vymazal, 2010). Wetland plants are planted in the media and water flows around the root systems (Figure 1-1). Plants are important in HSSF wetlands for oxygenation of the media and providing a substrate for bacteria growth (Brix, 1994). HSSF wetlands do not require a lot of space compared to other treatment wetland types; as a result, they have been widely used for secondary treatment for systems with small flow rates (Kadlec and Wallace, 2009). Costs associated with maintaining a HSSF wetland are moderate. The most common maintenance problem associated with HSSF wetlands is clogging due to sediment build-up in the soil/gravel bed. HSSF wetlands operate well in colder environments since the water level is below the surface, making water insulated from freezing (Kadlec and Wallace, 2009). Another benefit of HSSF wetlands is the reduced exposure of humans and animals to the usually contaminated waters flowing through such systems.

Vertical flow wetlands consist of water being introduced to the surface of a constructed wetland followed by the water percolating through a soil/gravel bed (Figure 1-2). VF wetlands are most commonly operated using batch loading, where water is introduced and allowed to fully progress through the system before more water is introduced (Vymazal, 2009). Operation and maintenance costs for VF wetlands are the highest out of the three types of constructed wetland systems. In addition to the potential for clogging to occur, VF wetlands require pumping of water in order to introduce water to the surface of the system. VF wetlands provide superior oxygen transfer compared to HSSF wetlands, resulting in a system that is proficient at oxidizing organics and at nitrification (Vymazal, 2009).

Free water surface wetlands contain open water and appear similar to natural marshes (Figure 1-3). FWS wetlands often consist of a series of ponds separated by berms. The ponds have water depths varying from 0.15 to 2 metres with shallow shelves along the edges for emergent macrophyte growth. Submergent and free-floating macrophytes are found throughout FWS wetlands. Water flows from an inflow and experiences sedimentation, filtration, oxidation, reduction, sorption, and precipitation as it flows to the outflow (Kadlec and Wallace, 2009). FWS wetlands are land-intensive, requiring more space compared to HSSF and VF wetlands. FWS wetlands contain a variety of treatment processes but do not perform any one treatment process exceptionally well; for this reason, FWS wetlands are optimal for treating low to

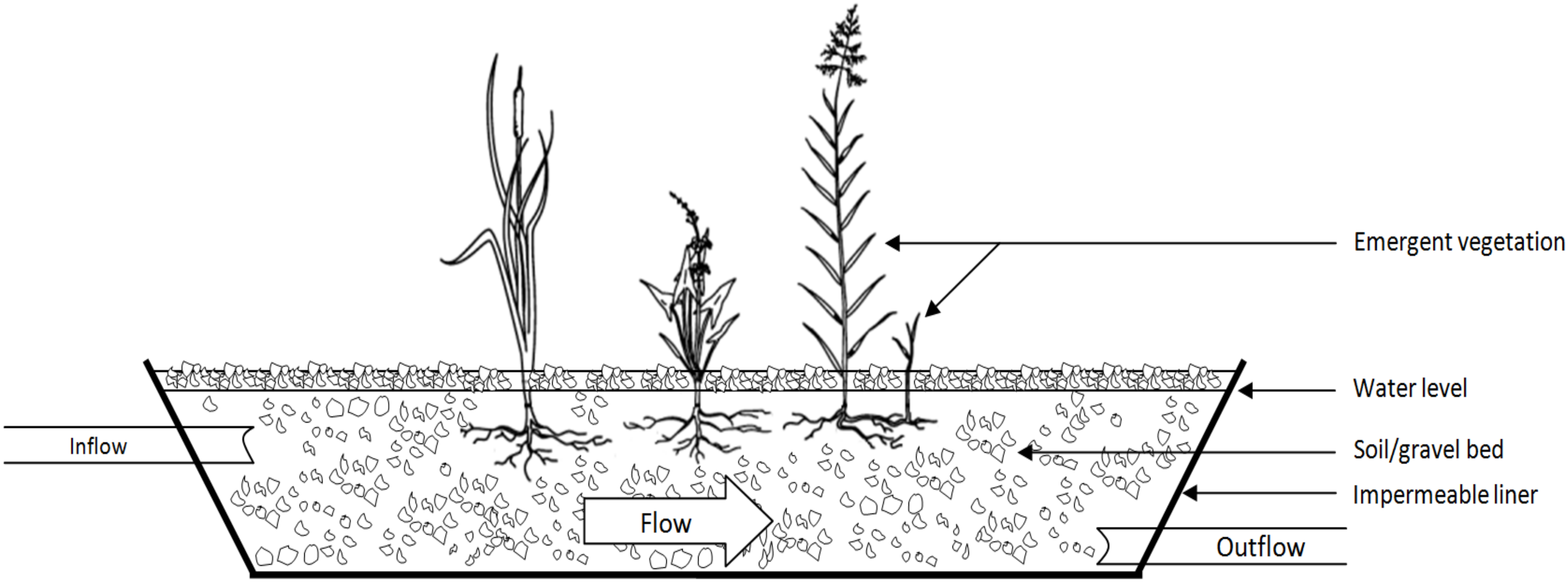

Figure 1-1. Horizontal subsurface-flow wetland schematic. Water enters the system at the inflow pipe, moves horizontally through the wetland, and exits at the outflow pipe. A water level below the surface is maintained.

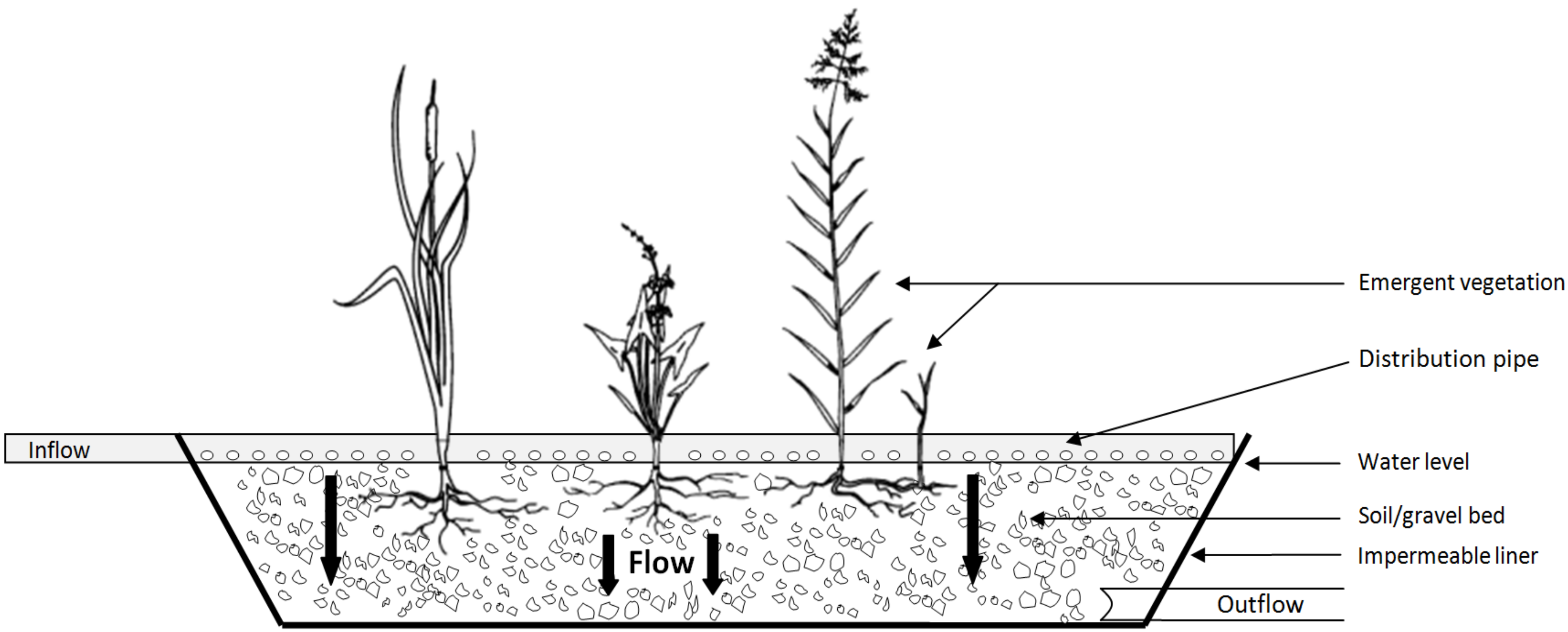

Figure 1-2. Vertical flow wetland schematic. Water travels through the distribution pipes that are laid on the surface of the system. Water exits the distribution pipe, percolates vertically through the wetland, and exits through the outflow pipe. Water is released through the distribution pipe in batches.

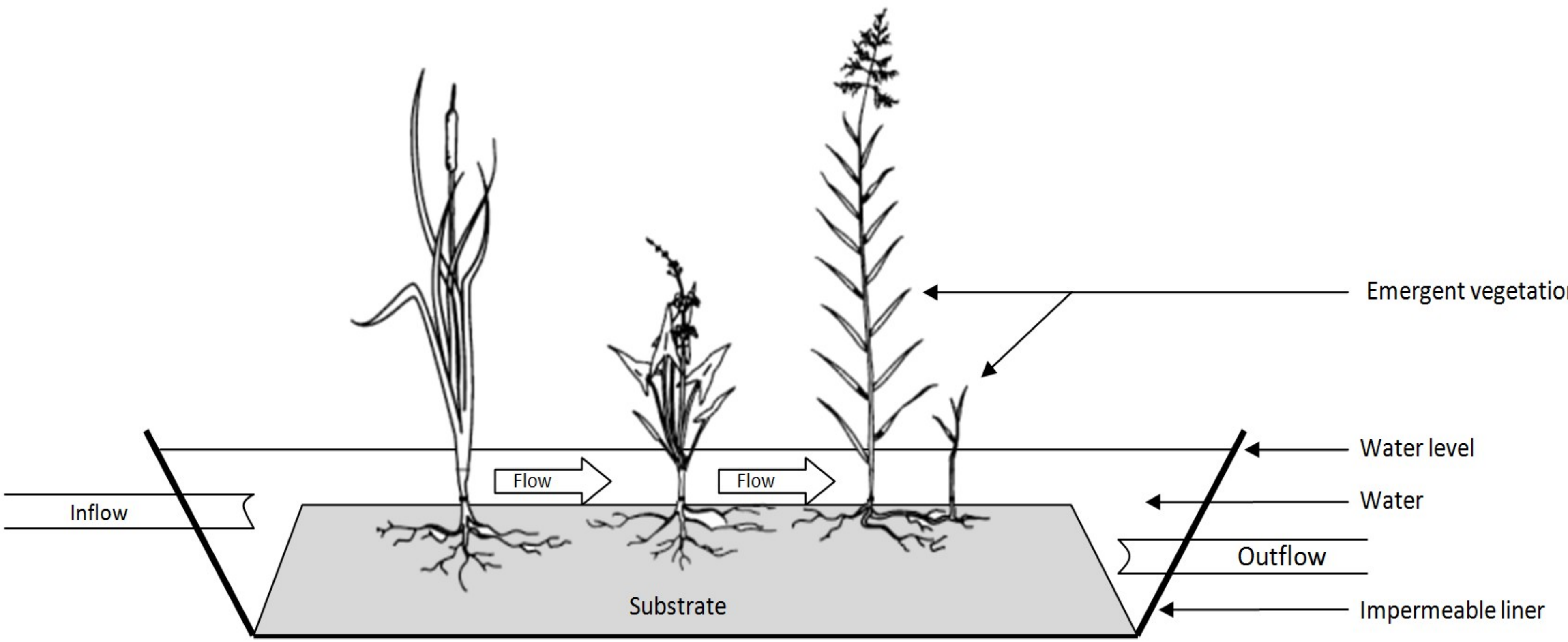

Figure 1-3. Free water surface wetland schematic. Water enters the system at the inflow, travels through a series of ponds and exits at the outflow.

moderately polluted water that contains many different pollutants. Operation costs for FWS wetlands are the lowest out of the three treatment wetland types; they require minimal maintenance and no special operation considerations. Due to the semblance of FWS wetlands to naturally occurring marshes, they provide habitat for a variety of wildlife. FWS are limited by temperature since water flowing through a FWS is exposed to environmental conditions. Freezing during winter can inhibit constructed wetland processes and prevent treatment from occurring (Kadlec, 2001).

Horizontal subsurface-flow, vertical flow, and free water surface wetland systems are all useful tools for treating contaminated water. Each system has advantages and disadvantages and will vary in their ability to treat different pollutants (Table 1-2). The differences in abilities for treating different pollutants are due to the differences in treatment processes found in each of the constructed wetland types.

Table 1-2. Evaluation of pollutant removal efficiencies for horizontal subsurface-flow (HSSF), vertical flow (VF), and free water surface-flow (FWS) constructed wetlands (adapted from Vymazal, 2007).

Parameter	**HSSF**	**VF**	**FWS**
Phosphorus	Medium	High	High
Sulphate	Medium	High	Medium
Chloride	Low	Low	Low
Ammonia	High	High	Medium
Nitrate	High	Medium	Medium

1.3 Treatment Processes in Constructed Wetlands

There are four pathways that pollutants can follow when they enter a wetland, they may be: stored, altered by chemical or biological action, discharged via water, or discharged into the atmosphere (Johnston, 1991). Storage and chemical or biological actions are the mechanisms for pollutant mitigation in constructed wetlands. Through storage and chemical/biological action processes, constructed wetlands can treat the inorganic macrocomponents commonly found in landfill surface-water runoff. Constructed wetlands also have the ability to improve water quality by buffering pH, lowering conductivity, and increasing dissolved oxygen.

1.3.1 Treatment of pH

Constructed wetlands have been shown to be effective buffering agents for water (Mayes et al., 2009). Alkaline waters are buffered through anaerobic microbial respiration and the production of organic acids through cation exchange (Schot and Wassen, 1993; Mayes et al., 2009). The processes involved in a constructed wetland buffering acidic water include: the dissolution of carbonate substrate material, the reduction of iron hydroxides, and the production of carbonate alkalinity due to reduction processes involving sulphate reducing bacteria (Mayes et al., 2009).

1.3.2 Treatment of Conductivity

Conductivity in itself does not require treatment, but can aid in designing a treatment wetland since it indicates the type of compounds entering the wetland. Specifically, water conductivity increases as the number of ions in solution increases (Moore et al., 2008). Temperature and suspended solids can impact water conductivity. Water viscosity decreases as temperature increases, resulting in increased conductivity (Smart, 1992). Suspended solids decrease conductivity due to the desorption of ions on sediment surfaces (Smart, 1992). Due to the ease with which conductivity sampling can be undertaken, conductivity can be an effective means of determining hotspots or discharge zones for high ion concentrations.

1.3.3 Treatment of Dissolved oxygen

Dissolved oxygen is important for the treatment functions of a constructed wetland. Many treatment processes require oxygen in order to proceed. Dissolved oxygen concentrations above 1.5 mg L^{-1} are required for most treatment processes (Ding et al., 2012). Constructed wetlands used for treating high organic and nutrient loaded waters are limited by dissolved oxygen availability (Casselles-Osorio and Garcia, 2006). There are many methods used in constructed wetlands to increase dissolved oxygen levels, including: oxygenators, water flow considerations, and promoting emergent macrophyte growth.

Oxygenators consist of oxygen being released through outlets of piping run along the bed of a wetland. Using oxygenators is effective but expensive and maintenance-intensive (Bedessem et al., 2007).

Oxygen can be introduced through the implementation of cascades and water depth management. Cascades are effective, inexpensive, and require little maintenance. Cascades are a pre-treatment technique and do not result in continual oxygenation as water flows through the wetland system. Constructed wetlands can be operated with batch flow, where water depth varies as water fills and is drained. The exposure of soils to the atmosphere increases the available oxygen with batch flow systems (Tanner et al., 1999).

Wetland emergent plant species are effective at introducing oxygen into the soil and sediment. Many wetland plants are hollow and contain channels for oxygen to move to the root system (Vymazal et al., 2006). Plants move oxygen through the channels to the roots for respiration. Once in the root system, some oxygen is lost to the rhizospehere where it becomes available for treatment processes (Brix, 1990).

1.3.4 Treatment of Phosphorus

There are four mechanisms for phosphorus removal/retention in wetland systems: plant uptake, sorption on substrates, precipitation, and the formation and accretion of new sediments (Kadlec, 1997). It is the dissolved inorganic phosphorus form which is subject to removal and retention in wetlands (Richardson, 1985). Organic and particulate phosphorus must undergo transformations to inorganic forms before removal and retention can be achieved (Reddy et al., 1999).

Plant uptake accounts for a low amount of phosphorus removal in constructed wetlands (Vymazal, 2007). Plant uptake is highest during the early growing season, but phosphorus is later released back into the water column during plant senescence (Boyd, 1969; Hill, 1979; Kroger et al., 2007). Floating macrophytes remove phosphorus directly from the water column, but removal is limited and a larger amount of phosphorus is released upon senescence compared to other plant-types (Mitch et al., 1995). Emergent macrophytes do not remove phosphorus directly from the water column but from the soils and sediments through their root systems

(Richardson and Marshall, 1986). While emergent macrophytes do not remove a significant amount of phosphorus from the water column, plant uptake from soils and sediments can lead to a phosphorus concentration gradient between the soils/sediment and water. The resulting gradient can lead to increased soil and sediment phosphorus retention (Reddy et al., 1999).

Phosphorus sorption refers to the adsorption of phosphorus on the surface of a retaining material and the absorption of phosphorus into the retaining material (Reddy et al., 1999). Sorption of phosphorus is a reversible mechanism. The soil chemistry and concentration of phosphorus in the water column will both influence the amount of sorption that occurs and is maintained (Johnston, 1991). When phosphorus concentrations in the water column are high, the amount of phosphorus sorbed is high (Patrick and Khalid, 1974). Low water column phosphorus concentrations result in the mobilization of previously sorbed phosphorus and/or low amounts of phosphorus being sorbed (Barrow, 1983).

Precipitation of phosphorus occurs when phosphate ions react with cations in solution to form amorphous or crystalline solids. Inducing phosphorus precipitation through the introduction of iron, aluminum, or lime is the main process for removing phosphorus from wastewater (Donner and Salecker, 1999; de-Bashan and Bashan, 2004).

The formation of new soils and sediments due to phosphorus accretion is considered one of few major long-term phosphorus sinks (Richardson, 1985). Phosphorus removal due to accretion for freshwater treatment wetlands averages around 0.5 g m^{-2} yr^{-1} (Johnston, 1991). Phosphorus accretion results in up to 20% of phosphorus being permanently stored as new soils and sediments in constructed wetlands (Reddy et al., 1993).

1.3.5 Treatment of Sulphate

Sulphate is relatively non-toxic under most conditions. Treating sulphate may be desirable in order to limit its impact on nitrification processes and if water hardness is low. Sulphate concentrations in the water column can be reduced through plant uptake, emission to the atmosphere, and mineral precipitation.

Sulphate is essential for growth in wetlands plants. While plants uptake sulphate for growth purposes, the removal efficiency of sulphate due to plant uptake is less than 0.3-1% (Winter and Kickuth, 1989; Vymazal and Kropfelova, 2005).

Sulphate can be used as an energy source by sulphate-reducing bacteria. The process of utilizing sulphate for energy use results in sulphide and carbon dioxide bi-products and occurs in the anaerobic root-zone in constructed wetland soils (Wu et al., 2013). Sulphate reduction is greatest during times of active plant growth since more oxygen is being supplied to the soils (Stein et al., 2007). At low pH, sulphide can be emitted to the atmosphere as hydrogen sulphide.

Under anoxic conditions, the resulting sulphide from sulphate reduction can precipitate with heavy metals to form insoluble metal sulphides. The metal sulphides are then immobilized in the wetland soil matrix, resulting in relatively permanent sulphate treatment (Wu et al., 2013).

1.3.6 Treatment of Chloride

Constructed wetlands have been shown to have limited treatment capabilities for chloride. The reason for the limited treatment capabilities is that chloride does not readily biodegrade, precipitate, volatilize, bioaccumulate, or absorb onto mineral surfaces (Mayer et al., 1999). Chloride that enters a wetland systems tends to move through the wetland and exit unaltered (Carlisle and Mulamoottil, 1991; Kadlec and Wallace, 2009; Vidales-Conteras et al., 2010).

1.3.7 Treatment of Nitrogen

The nitrogen cycle is a key component of treating nitrogen pollution in its various forms. Nitrogen can be found as ammonia, nitrate (NO_3), and nitrite (NO_2). Ammonia is found in both the un-ionized ammonia (NH_3) form and as the ammonium (NH_4) ion. Landfill waste will produce ammonia from organic nitrogen sources for many years even after a landfill is closed, resulting in the persistence of ammonia as a pollution problem (Robinson et al., 1992).

The mechanisms for nitrogen transformation are: ammonification, ammonia volatization, nitrification, plant uptake and denitrification (Burgin and Hamilton, 2007; Vymazal, 2007).

Ammonification is the first reaction that occurs to transform organic nitrogen to inorganic forms. Energy released by the ammonification process is used for microbial growth (Vymazal, 2007). Ammonia volatization is the transformation of aqueous ammonia to gaseous ammonia. Ammonia removal due to volatization is only significant at pH 9.3 and above and can result in removals as high as 2.2 g m^{-2} per day (Stowell et al., 1981; Reddy and Patrick, 1984).

Nitrification is a two-step process involving the oxidation of ammonium to nitrite and then nitrate. Nitrification is a chemoautotrophic process involving two types bacteria: one to oxidize the ammonium and one to oxidize the nitrite. The first group of bacteria involved in the nitrification process are chemolithotrophic and entirely rely on the nitrification of ammonia as an energy source for growth (Schmidt et al., 2003). There are two bacteria genera involved in the nitrification of nitrite to nitrate in freshwater environments: *Nitrobacter* and *Nitrospira* (Ehrich et al., 1995). Nitrification of the more toxic nitrite and ammonia compounds into the less toxic nitrate ion is beneficial in the treatment of nitrogen pollution. Nitrification processes are influenced by pH and temperature. The optimal temperature for nitrification to occur is 25 ^{o}C to 35 ^{o}C in water with the optimal pH being 6.6 - 8.0 (Kadlec and Wallace, 2009).

Plant uptake of nitrogen is achieved through nitrogen assimilation. Nitrogen assimilation is the process of converting inorganic nitrogen to organic nitrogen compounds that can be used to build cells and tissues (Vymazal, 2007). The two forms of nitrogen that are assimilated by plants are ammonia and nitrate with a preference for ammonia (Kadlec and Wallace, 2009). Constructed wetlands that rely on plant uptake for nitrogen removal require plant harvesting in order for permanent large nitrogen reductions from water. If plant harvesting occurs, plant uptake can provide the most nitrogen removal compared to other nitrogen removal pathways (Gumbricht, 1993). If harvesting does not occur, much of the assimilated nitrogen is released during plant decomposition when senescence occurs (Gumbricht, 1993.

Denitrification, in the absence of plant uptake and plant harvesting, is the most prevalent mechanism for permanent nitrogen removal (Sharma and Ahler, 1977; Gersberg et al., 1983; Gumbricht, 1993). Denitrification is a low-oxygen process involving denitrifying bacteria transform nitrate into nitrogen gas by using nitrate as a terminal oxygen acceptor (Kadlec and Wallace, 2009; Lee et al., 2009). Since nitrification needs to occur prior to the availability of nitrate for denitrification, constructed wetland designers and operators attempt to create oxygen gradients where nitrification and denitrification can occur in sequence (Lee et al., 2009). The denitrification process requires organic carbon to proceed and studies have shown that carbon-limited constructed wetlands can have difficulty treating nitrogen through denitrification (Kozub and Liehr, 1999).

1.4 Free Water Surface Wetland Design and Implementation for Pollutant Treatment

There is a variety of design considerations that need to be satisfied when constructing a treatment wetland. The size, layout, vegetation, and operation of the wetland need to be considered and will be impacted by variations in seasonal temperatures and precipitation, concentration and types of inflow chemicals, and treatment goals.

Prior to the construction of a treatment wetland, water quality sampling should be performed in order to identify pollutants of concern and determine treatment efficiencies necessary to mitigate identified pollutants of concern.

1.4.1 Layout and configuration

Models have been created in order to determine the pollutant removal capacity of various sizes of constructed wetlands. While there are many design factors that influence constructed wetland treatment performance, areal loading rate modeling can provide information for sizing estimates when constructing a treatment wetland (EPA, 2000). The issue arises that there are no models for free water surface wetlands based on a robust dataset and the number of pollutants covered by the models that do exist are limited (EPA, 2000). The best available means for sizing a constructed wetland is finding comparable reference sites that have had success treating identified pollutants of concern.

While the overall surface area of a constructed wetland is a major factor influencing treatment performance, water depth is impactful as well. Water depth will impact vegetation by limiting what species can populate different parts of the constructed wetland. For example, *Typha latifolia* L. does not grow in water deeper than 100 cm and *Phragmites* does not grow in water deeper than 15 cm (Grace, 1989; Weisner, 1996; Borst et al., 2002). Similar to *T. Latifolia* and *Phragmites*, most other macrophyte species do not grow well in deep water (Lieffers and

Shay, 1981; Squires and Van der Valk, 1992). Submergent and floating vegetation does grow well in deep water and varying the water depth throughout a constructed wetland can allow for the proliferation of different plant-types and species throughout (Thullen et al., 2005). Ibekwe et al. (2007) found that constructed wetlands with 50% plant cover provided greater treatment performance than constructed wetlands with 100% plant cover or 0% plant cover. The open water sections in a surface-flow constructed wetland are important because they result in increased dissolved oxygen concentrations in the water due to increased exposure of water to the atmosphere and increased mixing due to wind (Kim et al., 2010).

Using liners to seal a constructed wetland from possibly contaminating surrounding groundwater is necessary (Davis, 1995). Liners should be impermeable and robust. Clay is an often used natural liner due to its impermeability but can encounter problems with cracking if it becomes dry (Mariappan et al., 2011). Synthetic liners can also be used and can be made of asphalt, rubber, or plastic (Davis, 1995). Rubber and plastic liners are impermeable and are more durable than clay liners (Mariappan et al., 2011).

The configuration of a wetland system is important for performance. Length, width, and cell configuration all contribute to the treatment efficiency of a constructed wetland. Many constructed wetlands contain cells in sequence or parallel (Kadlec and Wallace, 2009). Constructed wetland cells can vary in size, depth, and plant species in order to target a specific treatment goal; for example, goals involving different pollutants or removal efficiency targets.

1.4.2 Vegetation

The incorporation of plants into constructed wetlands is common practice due to the many benefits they provide. Plants found in constructed wetlands perform a variety of treatment tasks, including: reducing water velocity (Brix, 1997), providing surface area for microbial growth (Ibekwe et al., 2007), nutrient uptake (Iamchaturapatr et al., 2007), and soil oxygenation (Yao et al., 2011).

There are five factors that should be considered when choosing plants for constructed treatment wetland use: (1) ecological acceptability; (2) tolerance to local climate conditions; (3) method of introduction; (4) wetland hydrology; and, (5) pollutant removal capacity (Tanner, 1996; Kadlec and Wallace, 2009). It is important to choose plants that will not cause problems for the local ecology. The introduction of invasive exotic species could cause harm to the biodiversity and genetic integrity of the surrounding environment (Levine et al., 2003). The ability for species to thrive in the local climate is necessary to ensure plant performance in the constructed wetland (Kadlec and Wallace, 2009). Plants can be introduced to the constructed wetland through planting or natural colonization. Planting allows for the introduction of specific species and for plant distribution management. Natural colonization requires little effort but is subject to the available species found in the local environment and species establishment takes longer compared to planting (Kadlec and Wallace, 2009). The success and establishment of different wetland plant species is highly dependent on water depth and system hydrology (Ibekwe et al., 2007). In addition to water depth influencing the establishment of species, changes in water depth due to the hydrology and operation of a constructed wetland will influence species survival and treatment success. Short frequent fluctuations in flooding and drying can lead to greater plant biomass, species richness, and pollutant removal (Tanner et al., 1999; Casanova and Brock, 2000). Some wetland species take more time to reduce pollutant concentrations and this can influence the required retention time of water in a constructed wetland (Iamchaturapatr et al., 2007).

There has been limited research conducted on treatment potentials comparing various wetland plants. The studies that have been conducted on surface-flow constructed wetlands test a limited range of species capacity to treat a limited range of pollutants (Gumbricht, 1993; Weisner et al., 1994; Tanner, 1996; Iamchaturapatr et al., 2007; Brisson and Chazarenc, 2009; Jiang et al., 2011). Studies that have compared species capacity to treat different pollutants have been laboratory-controlled experiments conducted using small-scale simulated wetland systems (Brisson and Chazarenc, 2009). The experiments fail to account for the fact that many wetland plants show increased treatment performance during periods of active plant growth (Stein and Hook, 2005).

1.4.3 Operation

Constructed wetlands are often advertised as being low-maintenance treatment options (Vymazal, 2010). While it is true that constructed wetlands require less maintenance compared to other treatment option, it is important to consider and implement a wetland operation plan for a constructed wetland system to perform optimally. There are three stages that occur in the construction and operation of a constructed wetland for pollutant treatment: start-up phase, stabilization phase, and routine operation phase.

Start-up Phase

The start-up phase involves introducing vegetation to the system, managing water flow and water depth to promote plant growth, and obtaining early monitoring results on treatment performance. Water that initially enters a new constructed wetland should not be released until treatment processes have begun. The time required for treatment processes to stabilize depends on the pollutant being treated.

Stabilization Phase

The stabilization phase is the period when treatment processes and vegetation dynamics stabilize. Lin et al. (2002) found that a surface-flow wetland could take up to three months for treatment of nitrogen and phosphorus to stabilize.

Routine Operation Phase

The routine operation phase of constructed wetland management involves maintaining the desired removal efficiencies for pollutants of concern. Tasks involved in the operation phase include: water flow management, water quality monitoring, vegetation management, pest control, structural maintenance, and seasonal adjustments.

Water flow in a constructed wetland can be managed as batch flow or continuous flow. Batch flow involves controlled release of water into and/or out of the wetland; releases can vary in frequency to achieve different goals. Batch flow systems require less engineering and less management to maintain desired retention times, but are more complicated to manage since mechanisms need to be in place to manage water flow between the inflow, outflow, and treatment cells. The primary benefit of batch flow management is the oxygenation of wetland substrate that results from fluctuations in water levels (Tanner et al., 1999). Short frequent fluctuations in water level can lead to greater plant biomass and species richness and oxygenation of the substrate that results from water level fluctuations can aid in oxidation processes in the substrate (Tanner et al., 1999; Casanova, 2000). Increased plant biomass results in greater pollutant uptake by wetland plants (Tanner, 1996; Greenway and Woolley, 2001). Constructed wetlands managed as continuous flow require more vigorous monitoring compared to batch flow systems. While batch-flow systems can be tested for pollutants of concern prior to water release, continuous flow systems need to be frequently monitored with monitoring frequency being dependent on water retention time (Kadlec and Wallace, 2009). There has not been additional research into the advantages and disadvantages of using batch flow or continuous flow constructed surface-flow wetlands, particularly to treat different pollutants in different treatment scenarios.

Constructed wetland water should be monitored at the inflow and outflow for pollutants of concern and other water quality parameters like pH, temperature, conductivity, and dissolved oxygen. In Ontario, a Certificate of Approval must be issued by the Ontario Ministry of the Environment for landfill operations. The Certificate of Approval is issued based on plans for eliminating environmental impacts, including stated plans for monitoring frequency and monitoring locations for landfill surface-water runoff (Ministry of the Environment, 2010). In addition to remaining in compliance with government regulations, monitoring also allows for modifications to a wetland system in order to increase treatment efficiency. Having internal monitoring sites in addition to the inflow and outflow can aid in identifying treatment hotspots and determine treatment efficiency related to the size of a constructed wetland.

Vegetation management in the routine phase can involve species management and plant harvesting. Managing undesirable plant species that enter a wetland system can be difficult to manage. Species removal can be costly, but some undesirable species can have large negative impacts on treatment performance; for example, plants can form floating mats or leaf canopies that block sunlight and oxygenation, resulting in extremely limited treatment performance for many pollutants (Kadlec and Wallace, 2009). Plant senescence can lead to large nutrient releases, compromising the treatment abilities of a constructed wetland (Alvarez and Becares, 2006; Kroger et al., 2007). Plants have the highest removal efficiencies during their growth period; this combined with the senescence phenomenon present a convincing argument for instituting plant harvesting in a constructed wetland. While studies have shown that plant harvesting can increase treatment performance of a constructed wetland (Karathanasis et al., 2003; Toet et al., 2005; Jinadasa et al., 2008), the costs associated with harvesting plants in surface-flow constructed wetlands may outweigh the benefits. No economical method for harvesting plants in surface-flow constructed wetlands has been developed; the most common method currently used is hand cutting plants or burning (Alvarez and Becares, 2008; Jinadasa et al., 2008; Kadlec and Wallace, 2009). More evidence is needed showing the benefits of harvesting in surface-flow constructed wetlands and an economical harvesting method needs to be developed in order for harvesting to become common practice.

Routine maintenance is necessary when operating a constructed wetland. Erosion of wetland banks needs to be managed. If too much erosion takes place, it can have negative impacts on the system, including: changes to wetland water depth; vegetation changes; and, clogging of parts of the wetland (Davis, 1995; Kadlec and Wallace, 2009). Sediment build-up can impact a constructed surface-flow wetland by changing flow dynamics and plant species composition due to changes in water depth. Dredging may be required to manage sediment build-up.

Temperature impacts most biochemical processes, including those involved in the treatment of pollutants in surface-flow constructed wetlands (Kadlec, 1999). Temperature fluctuations are greatest in temperate climates due to seasonal temperature differences. Temperature affects water flow, bacteria activity, and plant activity. Depending on how a surface-flow wetland is constructed, if temperatures are below zero degrees Celsius, surface-water flow may be non-existent. If water flow is present, treatment processes that rely on bacteria (e.g. denitrification and sulphate reduction) will be impacted in the short-term by low temperatures (Bachand and Horne, 2000; Kadlec and Reddy, 2001; Allen et al., 2002; Werker et al., 2002; Stein and Hook, 2005; Stein et al., 2007). In the long-term, the bacteria involved in constructed wetland treatment processes can adapt to cold temperatures (Werker et al., 2002). The primary influence of temperature on treatment wetlands is the impacts low temperatures have on wetland plants. Low temperatures put wetland plants into a dormant state where respiration does not occur and oxygen is not released in the root-zone (Stein and Hook, 2005; Stein et al., 2007). While cold temperature limits the treatment performance of constructed wetlands for nitrogen-based and sulphate-based pollutants, cold temperatures (if water-flow persists) do not impact other treatment processes like sorption, accretion, and precipitation (Kadlec and Reddy, 2001).

1.5 Research Question and Objectives

The research question I address is whether a constructed wetland system is a viable/necessary treatment option for surface-water runoff from an inactive landfill. The objectives of the study were to:

1. Characterize the water chemistry of surface-water runoff for an inactive landfill.

2. Evaluate the treatment potential for the constructed wetland system at the Napanee Landfill.

3. Recommend design, maintenance, and operative improvements to enhance effluent water quality.

1.6 Project Background

The research project was funded by Waste Management of Canada Corporation (WMCC) (Appendix B). Research was conducted independently of WMCC operations with no direct influence by WMCC on the project planning, execution, or results interpretation. Where possible, WMCC provided background information about the research site and site operation.

The operation of the Napanee Landfill and the ponds by WMCC is contingent on compliance with the certificate of approval (No. A 371203) issued by the Ministry of Environment of Ontario and all associated amendments. Key conditions for the operation of the ponds at the Napanee Landfill include: down-flow water quality testing of surface and groundwater throughout the year and ensuring that water quality is below Ontario Provincial Water Quality Objectives before water is released from the ponds at the outlet. Fines can be issued and WMCC made to make reparations to affected parties if contaminated water was emitted from the ponds.

2.0 Material and Methods

2.1 Site Description

The Napanee Landfill is located on a 16.2 ha footprint at 1271 Beechwood Rd RR 6, Greater Napanee, Ontario (Figure 2-1).

There are five surface-water impoundment ponds located south of the landfill. The two western most ponds were constructed in 1991 and expanded to their current state in 2009. There is a ditch encompassing the landfill mound that channels surface-water runoff to the ponds. The ponds are linked in series with rock berm separations. All five ponds are approximately 1.5 m in depth at the centre with shallow (10-50 cm) 2-3 m wide ledges around the edge. The ledges are dominated by *Typha latifolia* (common cattail) growth. The ponds can be described as a series of free water surface-flow wetlands, which are similar in appearance to natural marshes.

Impervious surfaces in the vicinity of the ponds include: a paved road that crosses between ponds three and four; a paved parking lot directly north of pond three; a gravel parking lot directly north of pond four; a paved landing north of pond two that hosts contaminated soils; and a gravel road that encompasses the landfill.

Figure 2-1. Aerial photograph of the Napanee landfill and surrounding area. Figure used with the permission of Waste Management Inc.

2.2 Landfill Operation

The Napanee Landfill receives approximately 125 tonnes of residential (50%), industrial, commercial, institutional, construction, and demolition waste per year. The landfill has been receiving waste in an official capacity since 1988 (Waste Management, 2005).

The ponds are operated as a batch flow system with emptying occurring on an irregular schedule. No plant harvesting or dredging occurs to manage the plant uptake of pollutants or sediment build-up. Since the pond expansion in 2009, no structural maintenance has occurred other than the construction of a new outlet discharge management system in 2010.

Fertilizer has been used on the landfill property to the north and northwest of ponds one and two and to the south of ponds four and five. The grounds where fertilizer was used were maintained grass. Fertilizer was used to the edge of the ponds. Fertilizer was used multiple times in June and July of 2010 but dates of application and exact quantities of fertilizer used are unknown.

2.3 Meteorological Data

Meteorological data for the study period was taken from the Environment Canada Trenton weather station (44°07'00.000" N, 77°32'00.000"W) historical data. Mean temperature and total precipitation values were observed.

2.4 Sampling Procedures

Water samples were taken at the inflow, outflow, and where the rock berms separate the impoundments (Figure 2-2). Samples were collected using 500 mL wide-mouth plastic bottles and the bottle submersion method (Byrnes, 2009). Samples were retrieved monthly during 2010 and 2011. Samples were retrieved in 2010 on May 14, July 15, August 16, September 19, and October 11. Samples were retrieved in 2011 on July 19, August 9, September 1, September 27, and October 10.

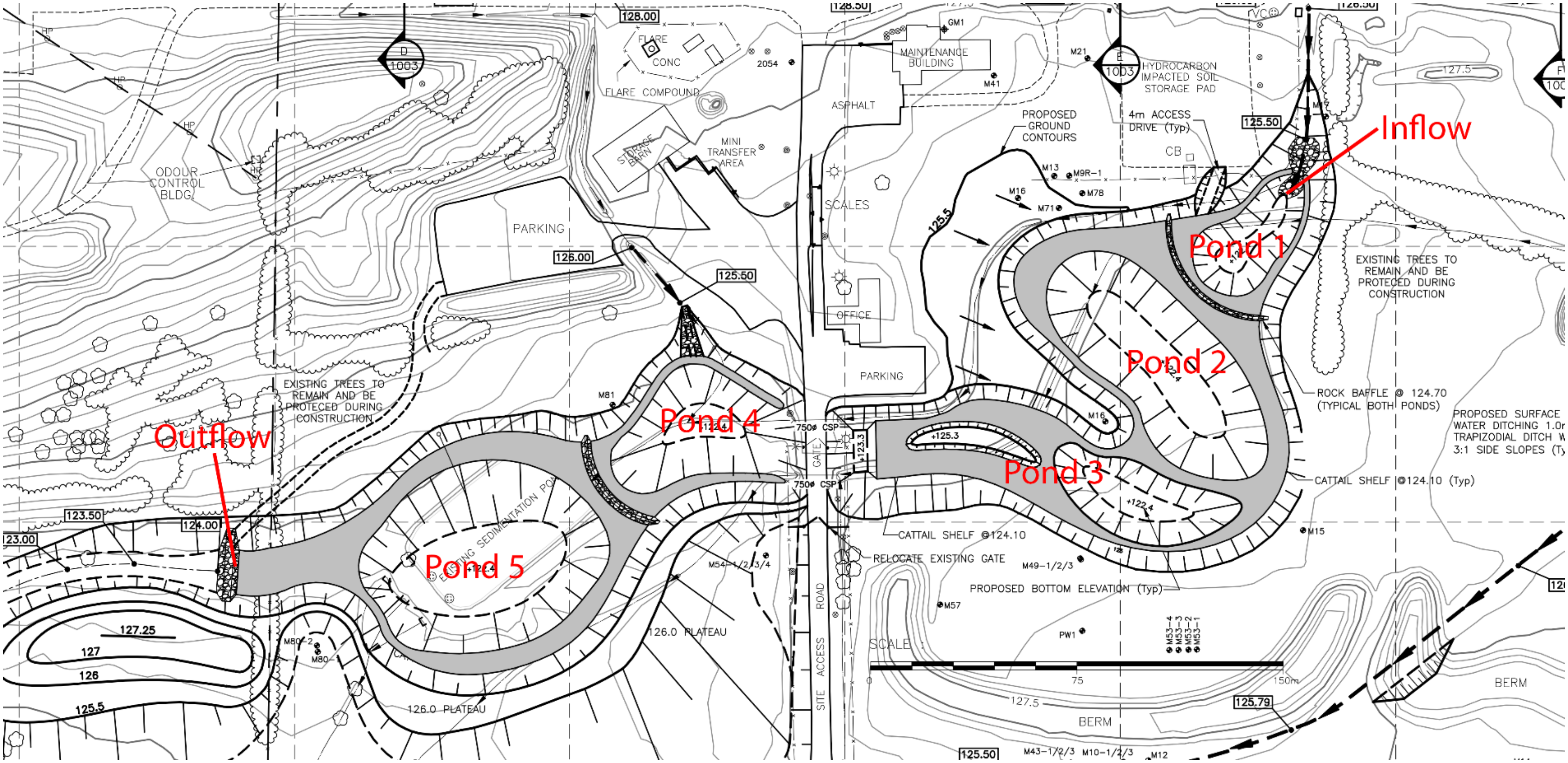

Figure 2-2. Napanee Landfill treatment ponds with the inflow and outflow identified. Figure used with the permission of Waste Management Inc.

2.5 Water Quality Parameters Analysis

I measured dissolved oxygen, pH, conductivity, temperature, phosphorus, sulphate, nitrate, ammonia (total ammonia and un-ionized ammonia), and chloride. Dissolved oxygen, pH, conductivity, and temperature were measured on-site at each sampling point using a Symphony SP70D multimeter. Phosphate concentrations were measured using a Hach DR 3800 spectrophotometer and the amino acid method (Eaton et al., 2005). Phosphorus concentrations were calculated by multiplying the phosphate results by 0.3322 (EPA, 1997). Sulphate concentrations were measured using a Hach DR 3800 spectrophotometer and the sulphate by turbidity method (EPA, 1978). Nitrate concentrations were measured using a Hach DR 3800 spectrophotometer and the cadmium reduction method (Eaton et al., 2005). Total ammonia concentrations were measured using a Hach DR 3800 spectrophotometer and the salicylate method (Reardon et al., 1966). Un-ionized ammonia was calculated using the equation:

$$\text{Un-ionized ammonia} = \frac{\text{Total ammonia}}{1+10^{pK-pH}} \times \frac{17}{14}$$

where

$$pK = 0.09018+2729.2/T^{o}K$$

(Emerson et al., 1975). Chloride concentrations were measured using the silver nitrate burette titration method (Eaton et al., 2005).

2.6 Statistical Analysis

SPSS software (IBM Corp, 2011) was utilized to conduct statistical analysis comparing inflow and outflow concentrations for the Napanee Landfill constructed wetland. Shapiro-Wilk with a significance threshold of $p = 0.05$ tests showed that results for pH, conductivity, dissolved oxygen, nitrate, sulphate, un-ionized ammonia, total ammonia, and chlorides did not have a normal distribution; only results for phosphorus were a normal distribution. The combined non-normal distribution and a constrained sample size for the data led me to use nonparametric Mann-Whitney U tests with a significance threshold of $p = 0.05$ to determine differences between inflow and outflow values for pH, conductivity, dissolved oxygen, nitrate, sulphate, ammonia, and chlorides. Mann-Whitney U tests with a significance threshold of $p = 0.05$ were used to compare sampling value differences between 2010 and 2011 for pH, conductivity, dissolved oxygen, nitrate, sulphate, ammonia, and chlorides.

Outliers were identified at the outflow on one of the sampling days for both nitrate and phosphorus. The outliers were a result of fertilizer being applied adjacent to the wetland. Figures were shown with outliers excluded.

2.7 Water Quality Guidelines

The data for the water quality parameters observed were compared to the Ontario Provincial Water Quality Objectives (PWQO), the British Columbia Environment (BCENV) guidelines, and the Canadian Water Quality Guidelines for the Protection of Aquatic Life (CWQG). The PWQO, BCENV, and CWQG guidelines provide concentration limits for pollutants based on studies analyzing the toxicity of pollutants to aquatic life.

3.0 Results

3.1 Meteorological Results

Based on 1971-2000 data from the Environment Canada Trenton weather station in Trenton, Ontario, Canada, the average daily temperature for May-October was 15.55 ^{o}C and the average daily precipitation was 2.44 mm (Table 3-1). Average temperature and average daily precipitation were higher than the May-October 1970-2000 averages for both 2010 and 2011 (Figure 3-1). There was minimal precipitation on the three days prior to sampling except for the dates July 25, 2010, September 19, 2010, and July 19, 2011 (Table 3-2; Figure 3-1; Figure 3-2).

Table 3-1. Precipitation and temperature data for the months of May to October for 1971-2000, 2010 and 2011. Data is from the Environment Canada Trenton weather station in Trenton, Ontario, Canada.

	Avg Temperature for 1971-2000 (^oC)	Avg Daily Precipitation for 1971-2000 (mm)	Avg Temperature for 2010 (^oC)	Avg Daily Precipitation for 2010 (mm)	Avg Temperature for 2011 (^oC)	Avg Daily Precipitation for 2011 (mm)
May	12.7	2.31	15.5	1.74	14.3	2.6
June	17.6	2.65	18.6	5.04	18.9	1.59
July	20.5	1.81	22.8	2.29	22.5	1.13
August	19.4	2.49	20.8	1.49	20.3	3.47
September	14.8	2.92	15.8	3.2	16.7	2.67
October	8.3	2.45	9.5	1.66	10.3	1.77
Mean	15.55	2.44	17.15	2.55	17.24	2.83
Total				470		492.4

Table 3-2. Precipitation and temperature on the days sampling was conducted and total precipitation for the 3 days prior to sampling. Data is from the Environment Canada Trenton weather station in Trenton, Ontario, Canada.

Sampling Day	Total precipitation for previous 3 days (mm)	Precipitation on sampling day (mm)	Temperature on sampling day (°C)
14-May-2010	9.5	0	14.2
25-Jul-2010	17.4	0	22.4
16-Aug-2010	1	7.4	22
19-Sep-2010	19.2	0	14.5
11-Oct-2010	0	0	10.9
19-Jul-2011	48.6	0	22.5
9-Aug-2011	4.4	6.6	18.1
1-Sep-2011	0	9	21.6
27-Sep-2011	0	0	19.1
10-Oct-2011	0	0	16.7

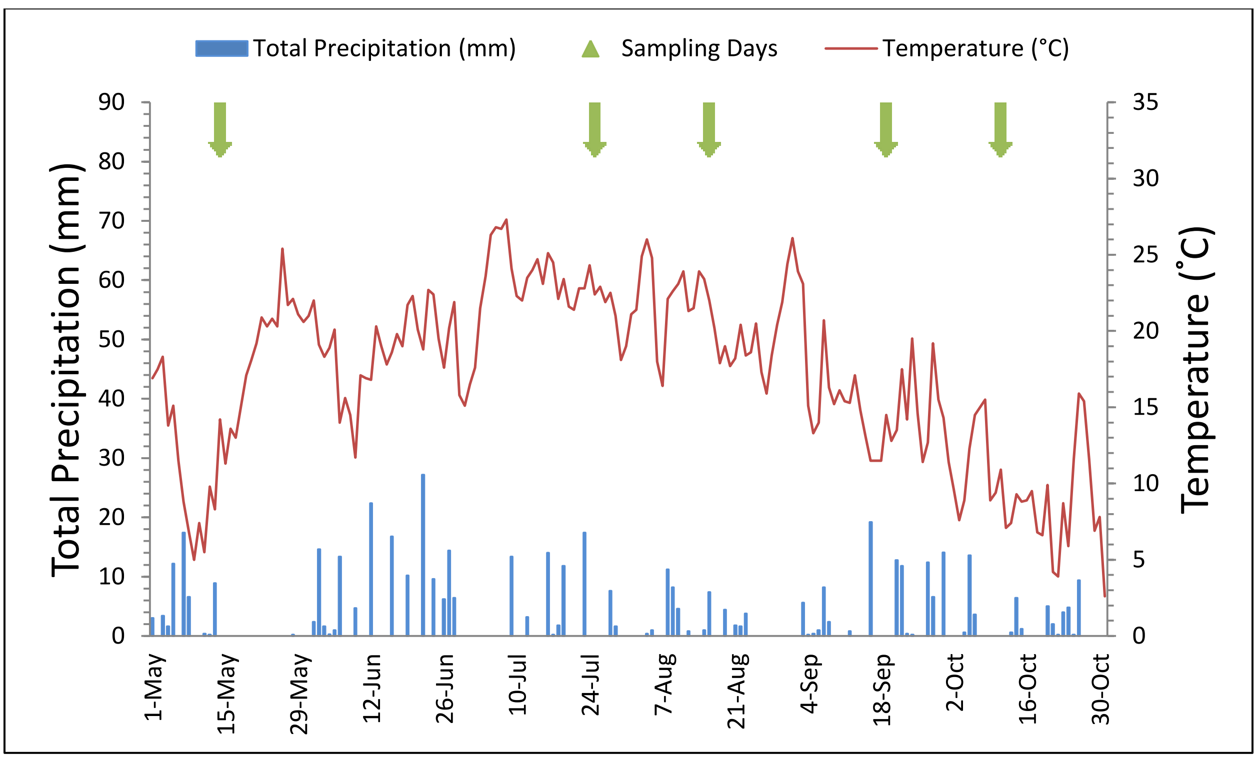

Figure 3-1. Temperature and precipitation for 2010 between May 1 and October 30. Data is from the Environment Canada Trenton weather station in Trenton, Ontario, Canada.

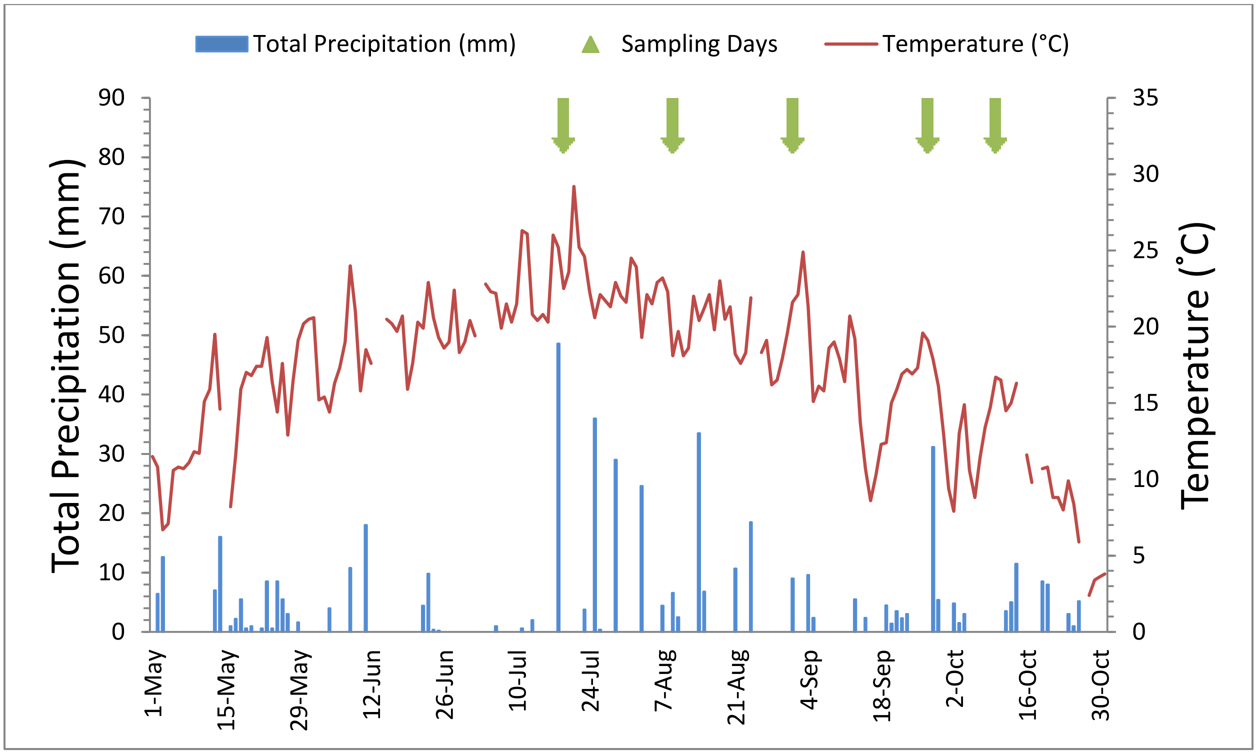

Figure 3-2. Temperature and precipitation for 2011 between May 1 and October 30. Data is from the Environment Canada Trenton weather station in Trenton, Ontario, Canada.

3.2 Temperature

The mean water temperature for the constructed wetland was 19.1 ^{o}C for 2010 and 19.8 ^{o}C for 2011 (Table 3-3). For all sampling days, the mean temperature for the wetland was 19.5 ^{o}C (Table 3-3).

Table 3-3. Water Temperatures for the constructed wetland at the Napanee Landfill taken on sampling days

Year	**2010**				**2011**				
Sampling Date	July 25	Aug 16	Sept 19	Oct 11	July 19	Aug 9	Sept 1	Sept 27	Oct 10
Temperature (^{o}C)	23.9	22.7	16.8	12.9	24.1	23.3	20.3	17.8	13.4

3.3 pH, Conductivity, and Dissolved Oxygen

The pH for the constructed wetland had a median pH of 7.85 ± 0.63 for all samples taken (n = 70). There was no significant difference between inflow (n = 10, M = 7.76, SD = 0.42) and outflow (n =10, M = 8.05, SD = 0.82) pH values (p = 0.075) (Table 3-3). There was an increase for pH of 3.74% from the inflow to the outflow (Table 3-3). High levels of precipitation on July 25, 2010, September 19, 2010, and July 19, 2011 did not result in a change in pH from median values (Figure 3-3). The pH values for the inflow and outflow showed small variation from the median for 2010 and 2011 (Figure 3-3).

Median conductivity for all samples taken (n = 70) was 830.5 ms cm^{-1} ± 463.03. There was no significant difference between inflow (n = 10, M = 937 ms cm^{-1}, SD = 461.43) and outflow (n = 10, M = 707.5 ms cm^{-1}, SD = 390.75) conductivity values (p = 0.151) (Table 3-3). There was a decrease for conductivity of 24.49% from the inflow to the outflow (Table 3-3). High levels of precipitation on July 25, 2010, September 19, 2010, and July 19, 2011 did not result in a change in conductivity from median values (Figure 3-4). Conductivity was higher in 2011 than 2010 for both inflow and outflow values, with the exception of the October 10, 2011 values (Figure 3-4).

Median dissolved oxygen for all samples taken (n = 59) was 7.52 mg L^{-1} ± 3.97. There was no significant difference between inflow (n = 10, M = 8.09 mg L^{-1}, SD = 4.7) and outflow (n = 10, M = 8.6 mg L^{-1}, SD = 3.84) dissolved oxygen concentrations (p = 0.834) (Table 3-3). There was a decrease for dissolved oxygen of 6.3% from the inflow to the outflow (Table 3-3). High levels of precipitation on July 25, 2010, September 19, 2010, and July 19, 2011 did not result in a change in dissolved oxygen concentrations from the median (Figure 3-5). Dissolved oxygen was only sampled for three times in 2010 but showed higher concentrations for the inflow and outflow than 2011 (Figure 3-5).

Table 3-4. Summary of inflow medians, outflow medians, significant difference between inflow and outflow values, and the percent change between the inflow and outflow medians for pH, conductivity, and dissolved oxygen.

	n	Inflow median	n	Outflow median	p-value	% Change
pH	10	7.76 ± 0.42	10	8.05 ± 0.82	0.075	3.74
Conductivity (ms/cm)	10	937 ± 461.43	10	707.5 ± 390.75	0.151	24.49
Dissolved Oxygen (mg/L)	10	8.09 ± 4.7	10	8.6 ± 3.84	0.834	6.3

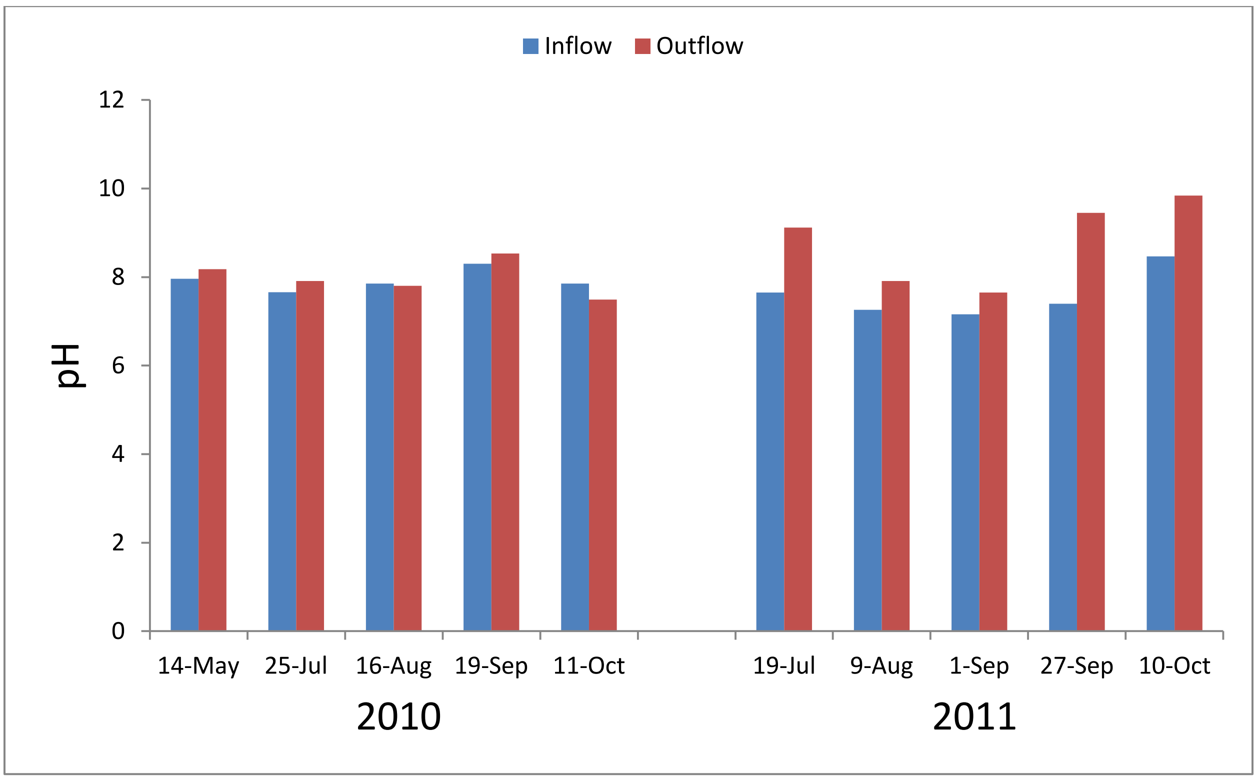

Figure 3-3. pH values for the inflow and outflow.

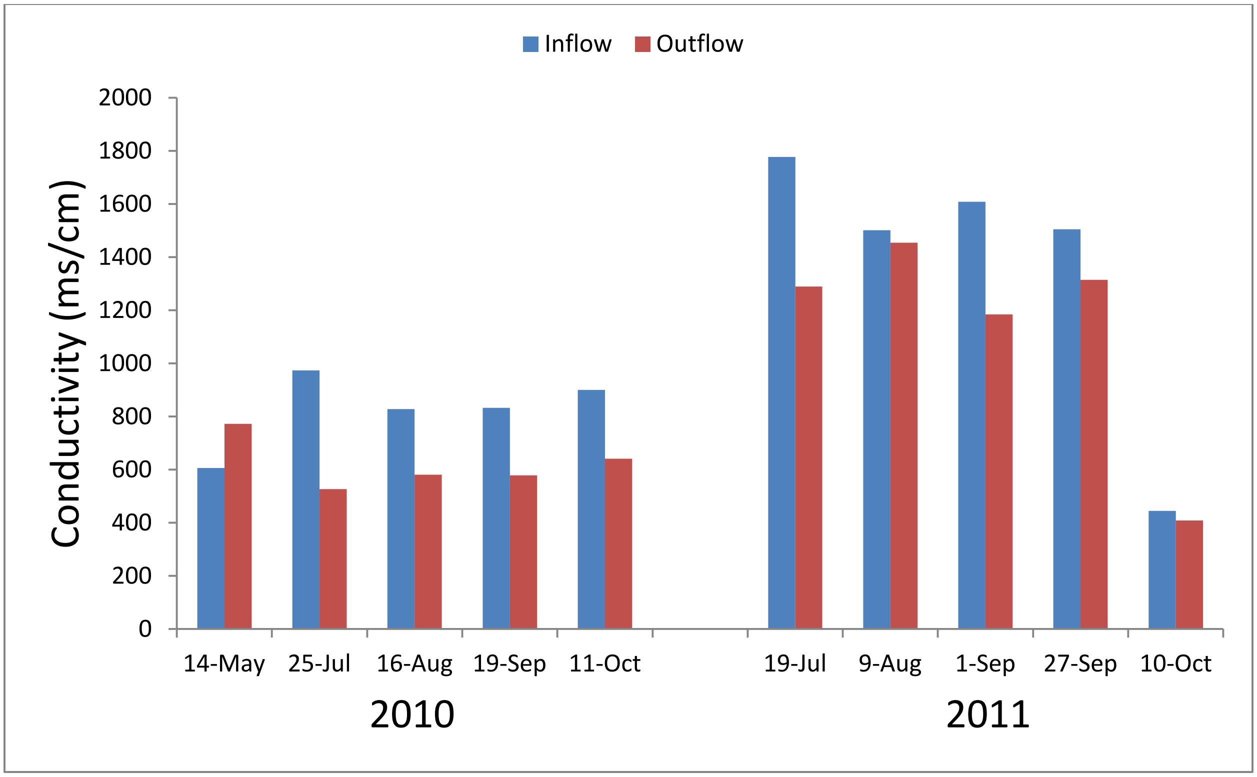

Figure 3-4. Conductivity values for the inflow and outflow.

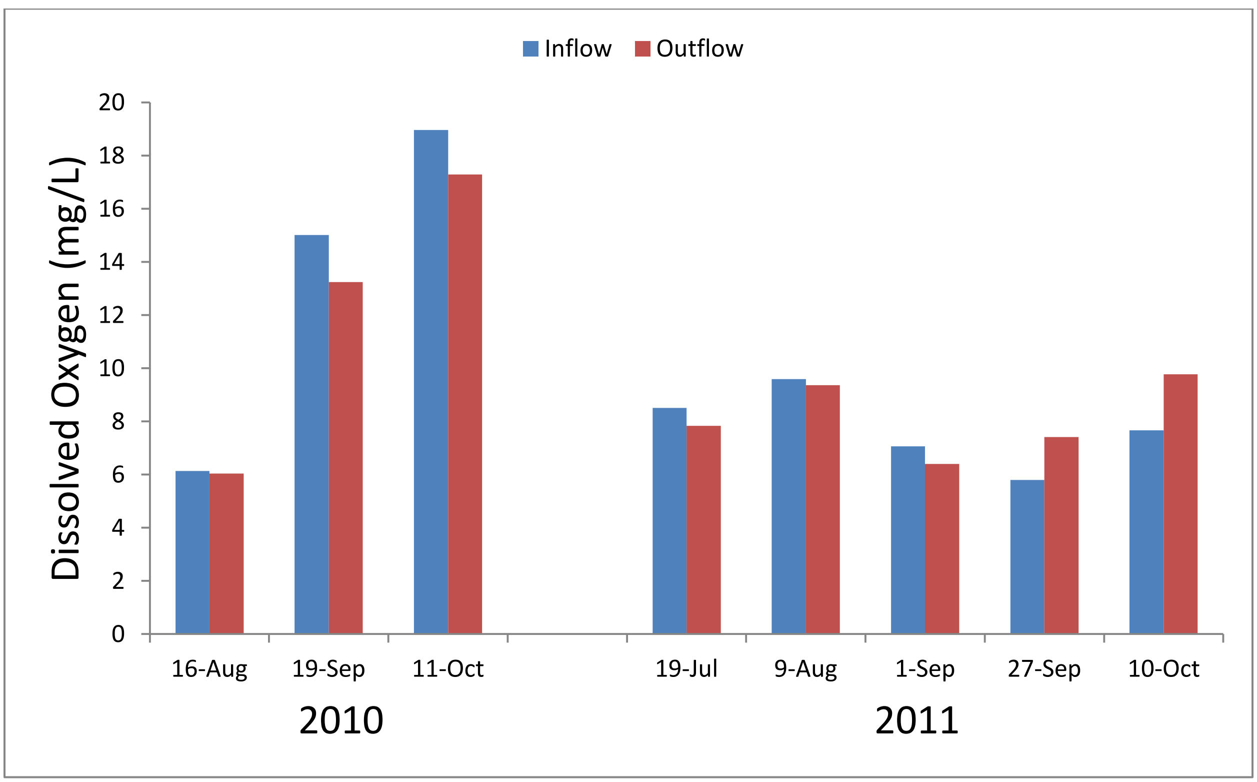

Figure 3-5. Dissolved oxygen concentrations for the inflow and outflow.

3.4 Nitrate and Ammonia Treatment Performance

The median nitrate concentration for all samples taken (n = 70) was 0.85 mg L^{-1} ± 1.31. There was no significant difference between inflow (n = 10, M = 0.7 mg L^{-1}, SD = 0.56) and outflow (n = 10, M = 1.05 mg L^{-1}, SD = 3.14) nitrate concentrations (p = 0.053) (Table 3-4). There were high levels of precipitation on July 25, 2010, September 19, 2010, and July 19, 2011; nitrate concentrations at the inflow were highest for these two dates out of all samples days (Figure 3-5). Nitrate outflow concentrations were similar for 2010 and 2011. The 2011 inflow concentrations were lower than those for 2010 (Figure 3-6).

The median un-ionized ammonia concentration for all samples taken (n = 63) was 0.051 mg L^{-1} ± 0.047. There was no significant difference between inflow (n = 9, M = 0.017 mg L^{-1}, SD = 0.079) and outflow (n = 9, M = 0.011 mg L^{-1}, SD = 0.12) un-ionized ammonia concentrations (p = 0.69) (Table 3-4). Un-ionized ammonia concentrations were higher in 2011 compared to 2-10 with elevated concentration at the inflow and outflow for July 19, 2011, September 27, 2011, and October 10, 2011 (Figure 3-7).

The median total ammonia concentration for all samples taken (n = 63) was 0.16 mg L^{-1} ± 0.17. There was no significant difference between inflow (n = 9, M = 0.19 mg L^{-1}, SD = 0.20) and outflow (n = 9, M = 0.19 mg L^{-1}, SD = 0.17) total ammonia concentrations (p = 0.86) (Table 3-4). Total ammonia concentrations were similar for 2010 and 2011 with the exception of elevated concentration at the inflow and outflow for July 19, 2011 and September 1, 2011 (Figure 3-7).

Table 3-5. Summary of inflow medians, outflow medians, significant difference between inflow and outflow values, and the percent change between the inflow and outflow medians for nitrate, un-ionized ammonia, and total ammonia.

	n	Inflow median	n	Outflow median	p-value	% Change
Nitrate (mg/L)	10	0.7 ± 0.56	10	1.05 ± 3.14	0.053	50
Un-ionized Ammonia (mg/L)	9	0.017 ± 0.079	9	0.011 ± 0.12	0.69	35
Total Ammonia (mg/L)	9	0.19 ± 0.2	9	0.19 ± 0.17	0.86	0

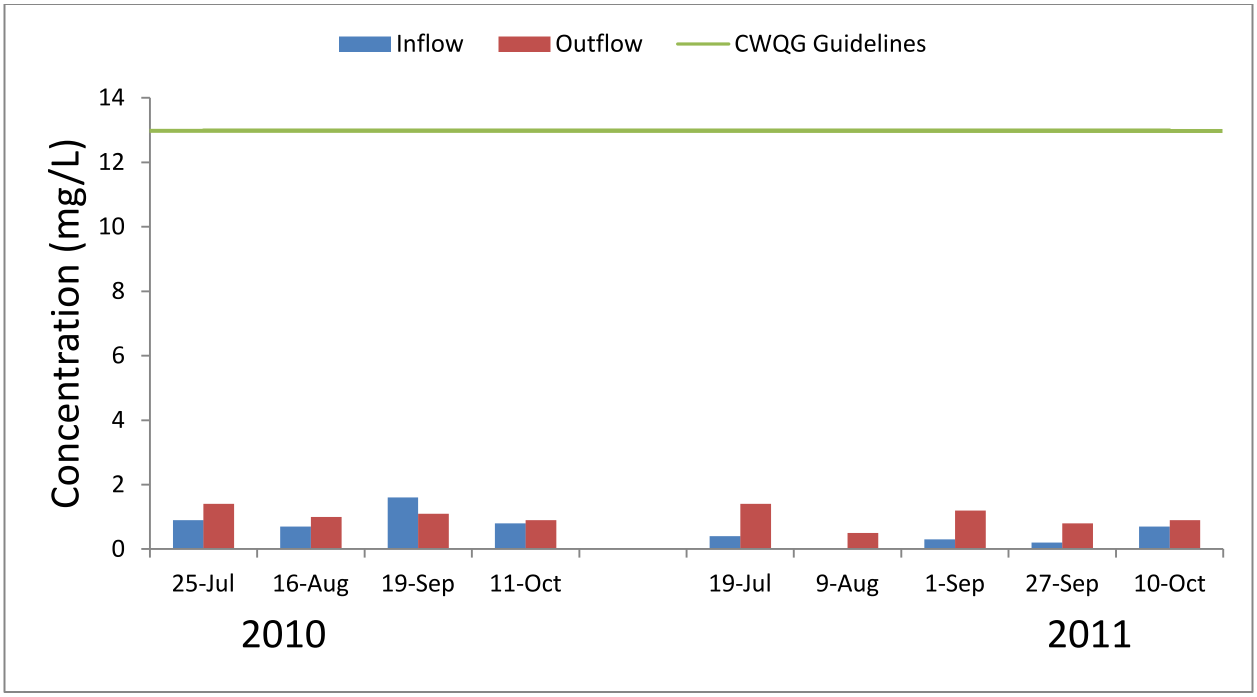

Figure 3-6. Nitrate concentrations for the inflow and outflow.

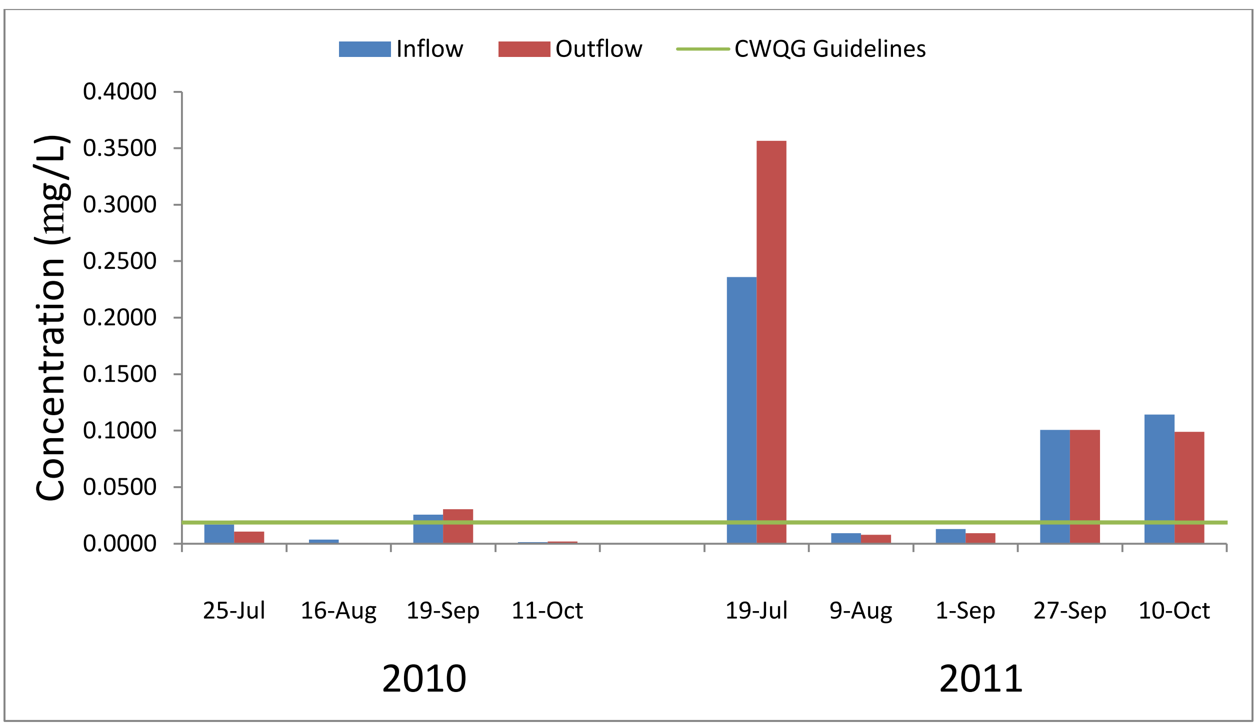

Figure 3-7. Un-ionized-ammonia concentrations for the inflow and outflow.

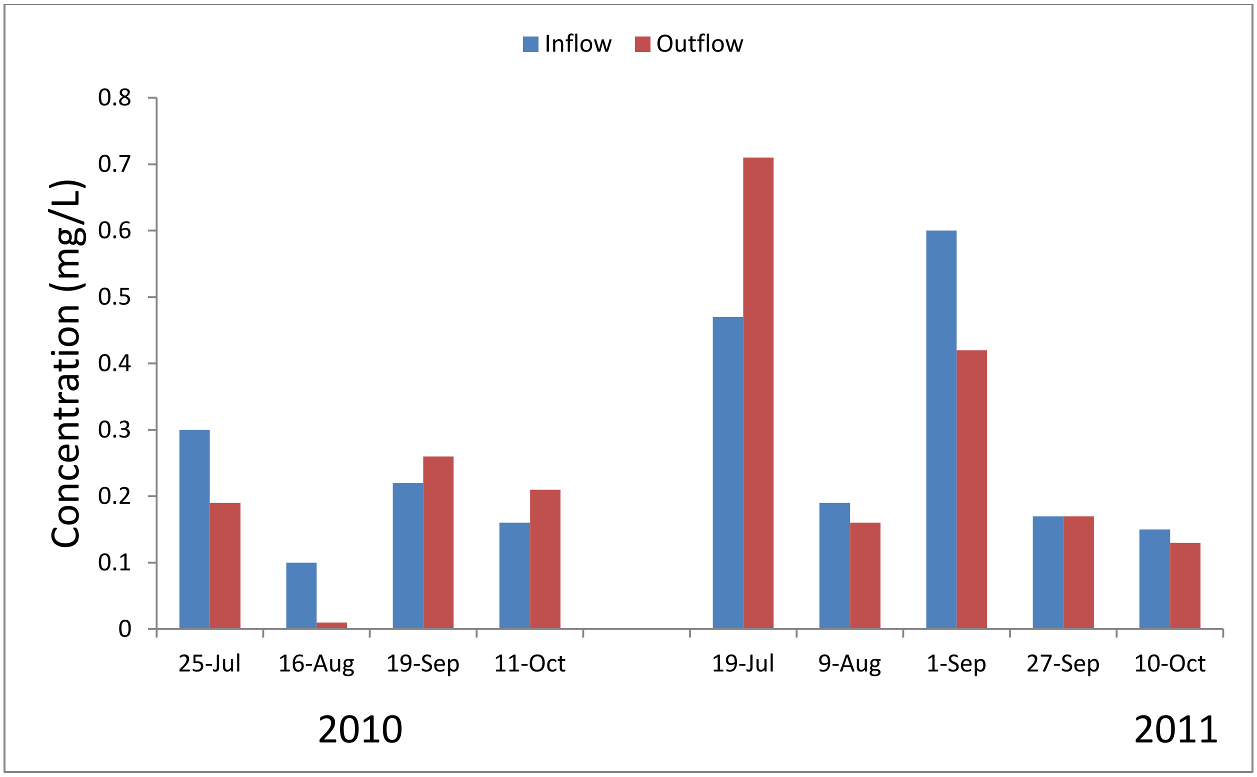

Figure 3-8. Total ammonia concentrations for the inflow and outflow.

3.5 Sulphate Treatment Performance

The median sulphate concentration for all samples taken (n = 70) was 27 mg L^{-1} ± 16.93. There was a significant difference between inflow (n = 10, M = 32.5 mg L^{-1}, SD = 13.83) and outflow (n = 10, M = 20 mg L^{-1}, SD = 9.38) sulphate concentrations (p = 0.008) (Table 3-5). High levels of precipitation on July 25, 2010, September 19, 2010, and July 19, 2011 did not result in a change in sulphate concentrations from the median (Figure 3-8). Sulphate concentrations for the outflow were similar for 2010 and 2011 (Figure 3-8). Inflow sulphate concentrations were higher for 2010 (Figure 3-8).

Table 3-6. Summary of inflow median, outflow median, significant difference between inflow and outflow values, and the percent change between the inflow and outflow medians for sulphate.

	n	Inflow median	n	Outflow median	p-value	Change (%)
Sulphate (mg/L)	10	32.5 ± 13.83	10	20 ± 9.38	0.008	38.46

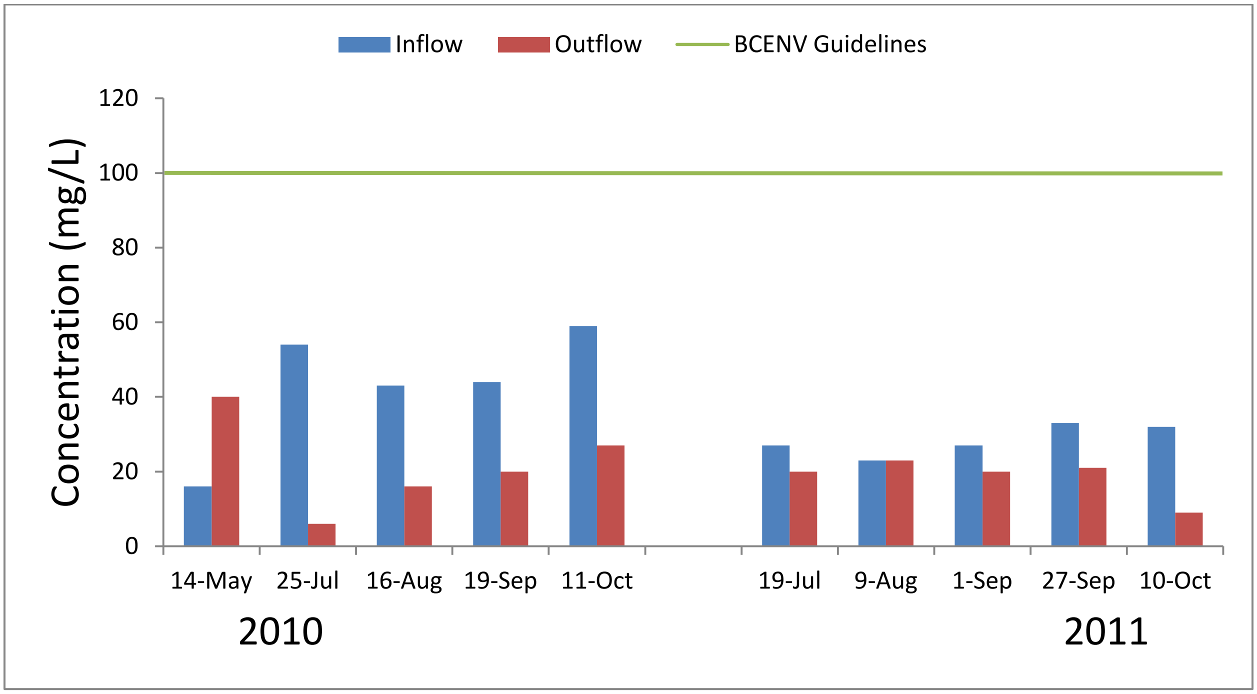

Figure 3-9. Sulphate concentrations for the inflow and outflow.

3.6 Phosphorus Treatment Performance

The median phosphorus concentration for all samples taken (n = 70) was 0.13 mg L^{-1} ± 0.41. There was no significant difference between inflow (n = 10, M = 0.13 mg L^{-1}, SD = 0.11) and outflow (n = 10, M = 0.16 mg L^{-1}, SD = 1.069) phosphorus concentrations (p = 0.88) (Table 3-6). High levels of precipitation on July 25, 2010, September 19, 2010, and July 19, 2011 did not result in a change in phosphorus concentrations from the median (Figure 3-9). Phosphorus concentrations for the outflow were similar for 2010 and 2011 (Figure 3-9). Inflow phosphorus concentrations were higher for 2010 for three of the five sampling days (Figure 3-9).

Table 3-4. Summary of inflow median, outflow median, significant difference between inflow and outflow values, and the percent change between the inflow and outflow medians for phosphorus.

	n	Inflow median	n	Outflow median	p-value	% Change
Phosphorus (mg/L)	10	0.13 ± 0.11	10	0.16 ± 1.069	0.88	23.08

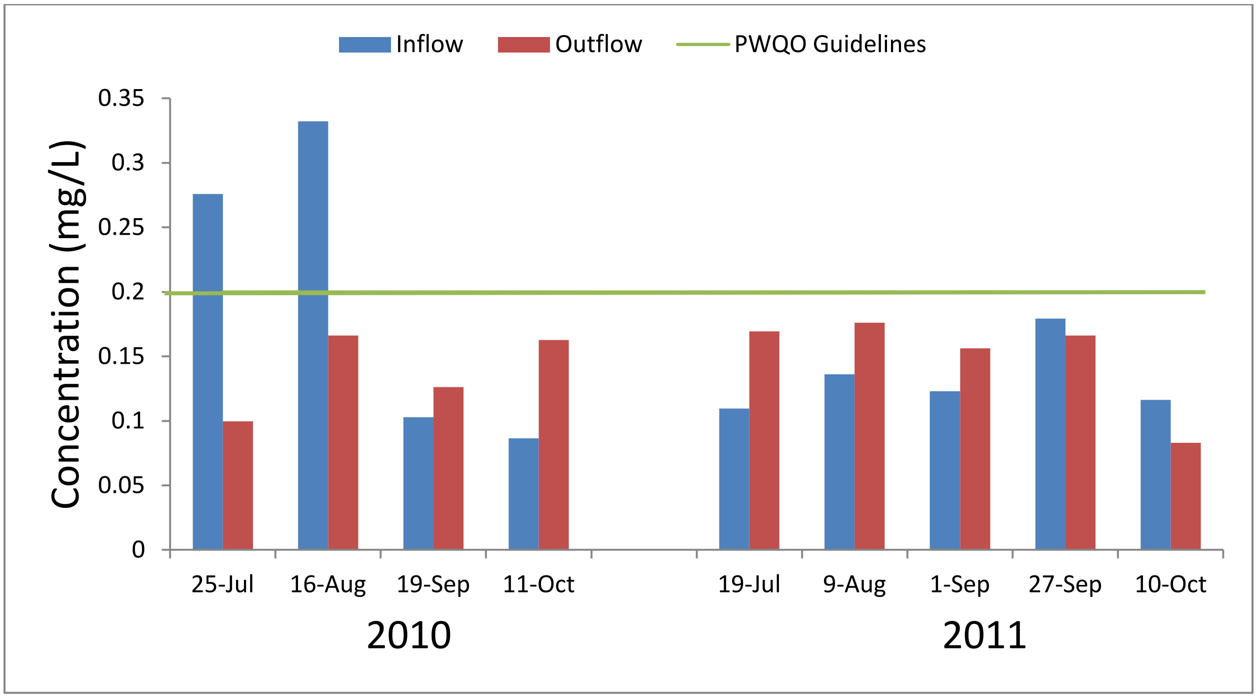

Figure 3-10. Phosphorus concentrations for the inflow and outflow.

3.7 Chloride Treatment Performance

The median chloride concentration for all samples taken (n = 63) was 120 mg L^{-1} ± 27.51. There was a significant difference between inflow (n = 9, M = 120 mg L^{-1}, SD = 20.88) and outflow (n = 9, M = 120 mg L^{-1}, SD = 35.86) chloride concentrations (p = 0.56) (Table 3-7). High levels of precipitation on July 25, 2010, September 19, 2010, and July 19, 2011 did not result in a change in chloride concentrations from the median (Figure 3-10). The chloride values for the inflow and outflow showed small variation from the median for both 2010 and 2011 (Figure 3-10).

Table 3-7. Summary of inflow median, outflow median, significant difference between inflow and outflow values, and the percent change between the inflow and outflow medians for chloride.

	n	Inflow median	n	Outflow median	p-value	% Change
Chloride (mg/L)	9	120 ± 20.88	9	120 ± 35.86	0.56	0

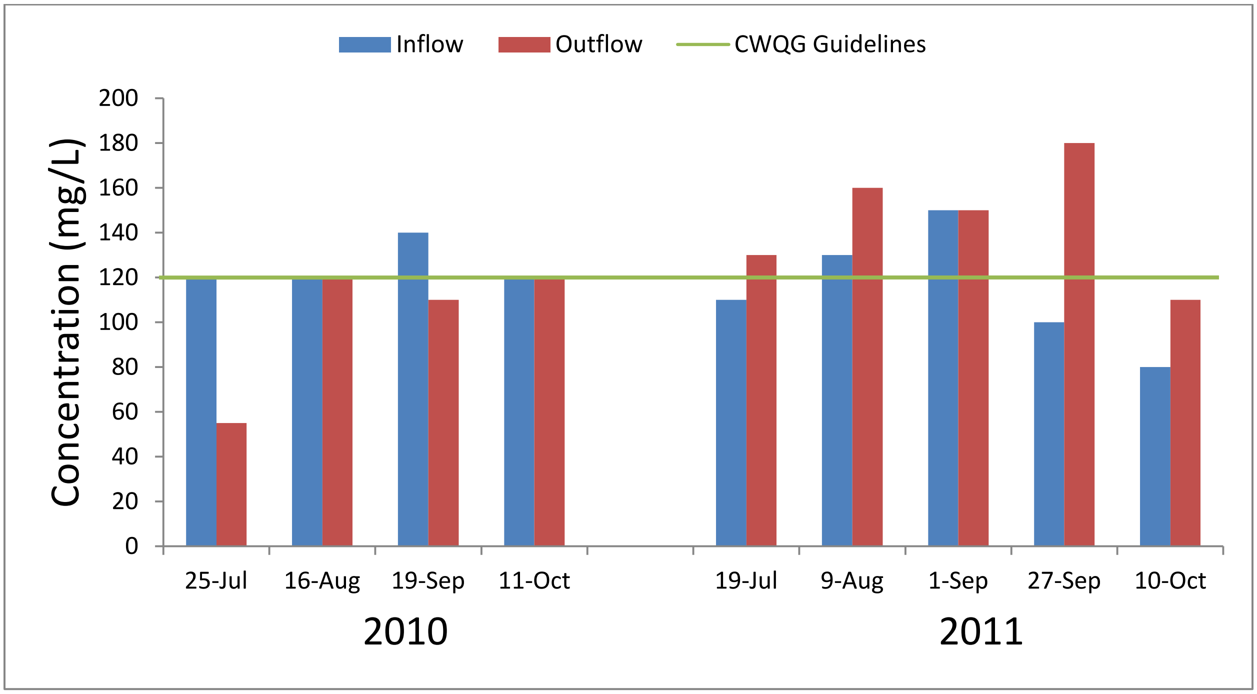

Figure 3-11. Chloride concentrations for the inflow and outflow.

4.0 Discussion

Inorganic macrocomponents and heavy metals are commonly found in landfill surface-water runoff (Marques and Hogland, 2001). While Snow et al., (2008) studied the ability of a surface-flow wetland to treat heavy metals found in landfill surface-water runoff, there have not been any studies focussing on the inorganic macrocomponents found in landfill surface-water runoff. Inorganic components like nitrate, un-ionized ammonia, sulphate, phosphorus, and chloride can be dangerous if exposed to the environment. It is important to characterize the inorganic macrocomponents in landfill surface-water runoff to determine potential environmental risks. Constructed wetlands are a treatment tool that have been shown to successfully treat inorganic macrocomponents and could be useful in the treatment of landfill surface-water run-off (Kropfelova, 2006; Gu, 2008; Beutel et al., 2009; Kadlec and Wallace, 2009; Sims et al., 2012).

4.1 Characterization of Landfill Surface-Water Runoff

The values and concentrations for conductivity, sulphate, and phosphorus found in the Napanee Landfill surface-water runoff were comparable to the low-end ranges of concentrations found in landfill leachate (Table 4-1). Total ammonia and chloride concentrations were lower than those found in landfill leachate (Table 4-1). Landfill surface-water runoff and landfill leachate differences can be attributed to the way runoff and leachate move through landfill materials. Leachate percolates vertically, picking up dissolved materials from landfill waste (Bulc, 2006). Landfill surface-water runoff flows along the surface of the landfill and makes contact with top layers of waste and daily cover.

For the surface-water runoff coming from the Napanee Landfill, none of the inflow median concentrations were above Canadian federal or provincial guidelines (Table 4-1). Some of the pollutants sampled did exceed Canadian federal or provincial guidelines on specific sampling days. Un-ionized ammonia concentrations exceeded CWQG guidelines at the inflow and outflow on four of the nine sampling days. Phosphorus concentrations exceeded PWQO guidelines at the inflow on two sampling days. The days where phosphorus concentrations exceeded PWQO guidelines were following instances where fertilizer was used on the surrounding grounds, which could explain the higher concentrations. Chloride concentrations exceeded CWQG guidelines at the inflow on three of the nine sampling days.

Table 4-1. Summary of Napanee Landfill constructed wetland median inflow concentrations compared to literature landfill leachate ranges and federal and provincial water quality guidelines.

	Unit	**Inflow Median**	**Landfill Leachate Range**	**Canadian Water Quality Guidelines**	**Ontario Provincial Water Quality Objectives**	**British Columbia Environment Guidelines**
pH	-	7.76	4.5 - 9.0[a]	6.5 - 9.0	6.5 - 8.5	-
Conductivity	ms/cm	937	230-3500[ab]	-	-	-
Dissolved Oxygen	mg/L	8.09	-	> 6.5 - 9.5	> 5 - 8	> 5 - 8
Nitrate	mg/L	0.7	-	13	-	3
Un-ionized ammonia	mg/L	0.017	-	0.02	0.019	-
Total ammonia	mg/L	0.19	30-450[a]	-	-	-
Sulphate	mg/L	32.5	22-650[b]	-	-	100
Phosphorus	mg/L	0.13	0.13 - 4.0[b]	-	0.20	0.005-0.015
Chloride	mg/L	120	150 – 4900[ab]	120	-	150

[a] Kjeldsen et al., (2002)

[b] Oman and Junestedt (2008)

4.2 Evaluation of Treatment Performance

4.2.1 Evaluation of Nitrate Treatment

There was no significant change in nitrate concentrations between the inflow and outflow of the treatment wetland. While there was statistically significant change, median nitrate concentrations increased by 50% from the inflow to the outflow. Nitrate concentrations were higher at the outflow for all sampling occurrences except May 14, 2010 and September 19, 2010. The increase in nitrate concentrations from the inflow to the outflow was greater in 2011. Since total ammonia concentrations did not change, the increase in nitrate concentrations are most likely due to the nitrification of nitrite.

The studies that use surface-flow constructed wetlands to treat nitrate show removal efficiencies in the range of 40-95 % (Kozub and Liehr, 1999; Reilly et al., 2000; Lin et al., 2008; Beutel et al., 2009; Kadlec, 2010). Nitrate availability, water temperature, organic carbon availability, and dissolved oxygen concentrations have been identified as limiting factors for nitrate removal (Beachamp et al., 1989; Bachand and Horne, 2000). The nitrate concentrations for the Napanee Landfill are low when compared to other constructed wetlands and may be a limiting factor for nitrate removal (Kozub and Liehr, 1999; Reilly et al., 2000; Lin et al., 2008; Beutel et al., 2009). A study by Cameron et al. (2003) used a surface-flow constructed wetland to treat a variety of pollutants and found that nitrate removal was non-existent with 0.1 mg L^{-1} nitrate concentrations in their wetland. Denitrifying bacteria populations may be too small for significant nitrate concentration changes to occur in surface-flow constructed wetlands when nitrate concentrations are low (Kadlec, 2010). Low water temperatures lead to reduced denitrifying bacteria activity and, as a result, less nitrate removal (Bachand and Horne, 2000; Kadlec, 2010). Nitrate concentrations did not fluctuate with changes in water temperature at the Napanee Landfill constructed wetland and is likely not a limiting factor for nitrate removal. Organic carbon is a requirement for denitrification with approximately one gram of carbon needed per gram of nitrate (Kadlec, 2010). Organic carbon concentrations in the Napanee Landfill constructed wetland were greater than 6 mg L^{-1} during the 2010-2011 sampling period (Genivar, 2011; Genivar, 2012), suggesting that carbon was also not a limiting factor for denitrification. Dissolved oxygen concentrations in the Napanee Landfill constructed wetland were high. Generally, high dissolved oxygen concentrations limit denitrification (Kadlec, 2010), but oxygen gradients and oxygen zonal differences in a constructed wetland water body can lead to areas of low dissolved oxygen concentrations where denitrification can occur (Phipps and Crumpton, 1994; van Oostrom and Russell, 1994). The areas where low oxygen and denitrification processes can be found in a constructed wetland are areas with little water movement, deep water, and a lack of vegetation (Kadlec and Wallace, 2009). While the Napanee Landfill constructed wetland was built with areas that would usually have low dissolved oxygen, the batch flow operation of the wetland may be sufficiently oxygenating the water to limit denitrification (Venterink et al., 2002).

Even with the increases in nitrate and apparent lack of nitrate removal, concentrations at the outflow were 66% lower than the lowest federal or provincial water quality guideline; as a result, nitrate at concentrations observed is not a threat to the health of the environment being exposed to water emitted from the Napanee Landfill surface-water runoff treatment wetland.

4.2.2 Evaluation of Ammonia Treatment

Total ammonia and un-ionized ammonia did not show any significant change in concentrations between the inflow and outflow at the Napanee Landfill constructed wetland. Un-ionized ammonia showed a decrease of 35% between the inflow and outflow for all sampling days. The decrease in un-ionized ammonia concentrations is contrary to expectations considering the increase in pH between the inflow and outflow. As pH increases, the ratio of ammonium to un-ionized ammonia in water decreases (Emerson et al., 1975). There were spikes in un-ionized ammonia concentrations on July 19, 2011, September, 27, 2011, and October 10, 2011 which can be attributed to higher pH on those days. In order to reduce un-ionized ammonia concentrations independent of temperature and pH, total ammonia concentrations need to be reduced. There was no percentage change in total ammonia concentrations from the inflow to outflow at the Napanee Landfill constructed wetland.

Surface-flow constructed wetlands have been shown to successfully reduce total ammonia concentrations, with removal efficiencies ranging from 8-96% (Carleton et al., 2000;

Wu et al., 2001; Cameron et al., 2003; Kotti et al., 2010; Sims et al., 2012). The primary mechanism for total ammonia concentration reduction is nitrification. Nitrification is limited by temperature, pH, dissolved oxygen, total ammonia concentration, and microbial populations (Vymazal, 2007). The optimal temperature for nitrification bacteria activity is 25-35 ^{o}C, but bacterial growth and activity will occur at temperatures as low as 5 ^{o}C (Vymazal, 2007). While the mean water temperature for the Napanee Landfill constructed wetland was not in the optimal temperature range for nitrification, the mean temperature was within ranges where nitrification bacteria can be active. The optimal pH range for nitrification is 6.6-8.0 (Vymazal, 2007), which the median pH for the Napanee Landfill constructed wetland fell within. Nitrification requires oxygen to proceed, with approximately 4.3 mg of oxygen needed per mg of ammonia (Vymazal, 2007). While overall dissolved oxygen concentrations at the Napanee Landfill constructed wetland were greater than that needed for nitrification, oxygen may have been limited to the nitrifying bacteria through competition with sulphate reducing bacteria (Wiessner et al., 2008). Sulphate reducing bacteria and nitrifying bacteria populations are often found in close proximity leading to competition for available oxygen (Wiessner et al., 2008). Studies where total ammonia concentrations were greater than 1 mg L^{-1} had results with the greatest removal efficiencies (Kotti et al., 2010; Sims et al., 2012), while studies where total ammonia concentration were less than 1 mg L^{-1} showed removal efficiencies around 7-8% (Wu et al., 2001; Cameron et al., 2003). The median concentration for total ammonia at the Napanee landfill constructed wetland was 0.16 mg L^{-1}, which would result in lower removal efficiencies if treatment at the Napanee Landfill constructed wetland followed trends set by other studies. Nitrifying bacteria require soil substrate or plant materials for growth (Wu et al., 2001); bacteria populations may have been limited by available surfaces for growth, which would have led to low removal efficiencies.

Un-ionized ammonia concentrations exceeded guidelines for four of the five sampling days. Total ammonia removal needs to be increased by addressing the growth surface availability for nitrifying bacteria and the availability of dissolved oxygen regardless of competition with other bacteria to reduce un-ionized ammonia concentrations at the outflow of the Napanee Landfill constructed wetland.

4.2.3 Evaluation of Sulphate Treatment

The Napanee Landfill constructed wetland was able to significantly reduce sulphate concentrations between the inflow and outflow. Since dissolved oxygen concentrations in the wetland were high, the removal of 38% of the sulphate that entered the system suggests that sulphate-reducing bacteria were utilizing the sulphate and converting it to sulphide and carbon dioxide. The sulphate removal efficiency for the Napanee Landfill corresponds with the results of other studies. Wiessner et al. (2005) observed a removal efficiency of 28% and Vymazal and Kropfelova (2006) reported a mean removal efficiency of 51% for five treatment wetlands in the Czech Republic.

The British Columbia Environment guidelines for sulphate have 100 mg L^{-1} as the upper limit for sulphate. The sulphate concentrations at the Napanee Landfill constructed wetland did not exceed the upper limit at the inflow or outflow.

While the removal of sulphate is seen as a positive, the fate of the sulphide (bi-product of sulphate reduction) is important. High sulphide concentrations can lead to reduced plant growth and plant mortality (Koch and Mendelssogn, 1989; Lamers et al., 1998; Wu et al., 2013).

4.2.4 Evaluation of Phosphorus Treatment

Phosphorus concentrations did not significantly change between the inflow and outflow at the Napanee Landfill constructed wetland. There was a decrease in phosphorus concentrations of 23%; this change is mostly due to three sampling days involving high concentrations of phosphorus on July 25, 2010, and August 16, 2010. The high phosphorus concentrations are likely a result of fertilizer being applied to the grounds surrounding the wetland prior to sampling. Phosphorus concentrations increased from the inflow to the outflow on five of the nine sampling days.

The phosphorus concentrations for the Napanee Landfill constructed wetland were not above Canadian federal or provincial water quality guidelines except for the two high concentration days. The primary mechanism for phosphorus removal in FWS wetlands is accretion, with sorption and microbial uptake acting as secondary removal mechanisms

(Vymazal, 2007). Other studies have shown that FWS wetlands are capable of 48-99% phosphorus removal (Serodes and Normand, 1999; Cameron et al., 2003; Andersson et al., 2005; Vymazal, 2007; Gu, 2008). The removal capabilities for phosphorus may have been saturated at the Napanee Landfill due to the loading brought on by fertilizer use on the surrounding grounds. Phosphorus saturation could explain the increase in phosphorus from the inflow to the outflow on many of the sampling days. Wetlands release phosphorus after periods of saturation followed by relatively low concentrations (Reddy et al., 1999), which was the case at the Napanee Landfill. Other potential explanations for the lack of phosphorus removal include: the use of a batch flow system and/or competition with sulphate for available iron. A batch flow system has reduced mixing of the water column compared to continuous flow systems (Doyle et al., 2003). It has been shown that mixing of the water column can lead to increased phosphorus removal due to increased interaction between the water column and sediments (Newbold et al., 1983; Doyle et al., 2003; Macrae et al., 2003). Sulphate-polluted wetlands have been shown to increase phosphorus mobilization and reduce phosphorus retention (Caraco et al., 1989; Lamers et al., 2002). Sulphate concentrations similar to those found in the Napanee Landfill constructed wetland trigger sulphur reducing bacteria to convert sulphate to sulphide (Lamers et al., 2002). The resulting sulphide disrupts phosphorus-iron binding, resulting in phosphorus mobilization in the water column (Lamers et al., 2002). Sulphide also competes with phosphorus for available iron by binding with iron to create iron sulphides, limiting the amount of phosphorus that can be retained through phosphorus-iron binding (Lamers et al., 2002).

Phosphorus management at the Napanee Landfill constructed wetland needs to begin with eliminating the introduction of phosphorus through fertilizer use on the surrounding grounds. During periods of fertilizer use, phosphorus concentrations tripled. Phosphorus retention would also be improved by increasing the interaction of the water column with soil and sediment material. Finally, reducing sulphate concentrations would mitigate the impact sulphide has on phosphorus mobilization and retention.

4.2.5 Evaluation of Chloride Treatment

Chloride concentrations at the Napanee Landfill constructed wetland did not significantly change between the inflow and outflow. The lack of chloride removal is consistent with other studies. A review conducted by Kadlec and Wallace (2009) observed that 8 out of the 9 wetlands with inflow chloride concentrations above 100 mg L^{-1} did not show significant chloride removal.

Chloride concentrations were above CWQG guidelines at the inflow on three sampling days and the outflow on four sampling days. Since constructed wetlands are limited in their ability to treat chloride, other mechanisms of removal need to be used or the source of chloride needs to be identified and mitigated.

4.3 Recommendations for Improvements

The Napanee Landfill constructed wetland needs improvements to the treatment of pollutants of concern ammonia, sulphate, and phosphorus so that guideline limits for the protection of aquatic life are not exceeded. Concentrations above the guidelines can lead to reduced fitness or mortality for many aquatic species. Management improvements for the pollutants of concern can be through pollutant concentration reduction for water entering the wetland and/or improvements to pollutant treatment/retention.

Pollutant concentration reduction for water entering the wetland is difficult to manage due to the constant nature of the pollutants found in the surface-water runoff of the landfill. If fertilizer with phosphorus use on the landfill grounds stopped, it would reduce surface-water runoff phosphorus concentrations and reduce the threat phosphorus being introduced to the surrounding environment poses. Improvements to pollutant treatment/retention for the Napanee Landfill constructed wetland can be obtained through changes to the layout/configuration and operation.

4.3.1 Improvements to Treatment by Changes to Layout and Configuration

Changes to the layout and configuration of the Napanee Landfill constructed wetland have the potential to improve the treatment/retention of ammonia, sulphate, and phosphorus. Changes that could be incorporated into the wetland are increased wetland size and the use of different constructed wetland types.

Increasing the size of the wetland will increase treatment performance for ammonia, sulphate, and phosphorus. Increased size allows for: greater retention time; more available substrate for sulphur reducing bacteria, nitrifying bacteria, and denitrifying bacteria to develop; and, more soil and sediment for phosphorus sorption (Kadlec and Wallace, 2009). It is important to manage retention time and flow when considering increasing the side of a constructed wetland. Increasing the size of a constructed wetland but not increasing the retention can lead to low water levels and part of the wetland not being utilized. If a constructed wetland has vegetation islands/shelves, then a low water level could lead to reduced vegetation. Increasing the size will also slow down the flow within the wetland if water volume input is not increased as well. Less flow would particularly impact phosphorus removal due to reduced mixing of the water column.

Surface-flow wetlands are a versatile solution that can treat many pollutants, but cannot treat large pollutant concentrations (Vymazal, 2007). Hybrid constructed wetland solutions are becoming increasingly popular for treating water pollutants (Vymazal, 2005). Subsurface-flow constructed wetlands provide increased substrate surface area for bacteria population development, which can result in greater removal efficiencies compared to surface-flow constructed wetlands (Vymazal, 2005). Horizontal subsurface-flow (HSSF) wetlands tend to have low dissolved oxygen available for biochemical reactions, while the nature of vertical-flow (VF) wetland operation results in high dissolved oxygen availability for biochemical reactions (Vymazal, 2002; Vohla et al., 2007). Due to the oxygen dynamics involved in each subsurface-flow constructed wetland type, VF wetlands better treat pollutants like ammonia and sulphate and HSSF wetlands better treat pollutants like nitrate (Vymazal, 2005). Phosphorus removal is increased in both subsurface-flow wetland types compared to surface-flow wetlands (Vymazal, 2005). Having a constructed wetland system with a VF, a HSSF and a surface-flow wetland linked in series respectively could result in greatly increased treatment performance for the Napanee Landfill constructed wetland system (O'Hogain, 2003; Melian et al., 2010).

4.3.2 Improvements to Treatment by Changes to Operation

The operation of a constructed wetland can improve treatment performance through the introduction of needed materials for biochemical reactions and the use of the best hydraulic flow regime for treatment goals.

Different biochemical reactions involved in the treatment of pollutants found in landfill surface-water runoff can be limited by some compounds. For the Napanee Landfill constructed wetland, it is suspected that iron availability was limiting phosphorus retention. Introducing an iron additive to the wetland could increase phosphorus retention (Ann et al., 2000). The sulphate cycle also requires iron and competes with phosphorus for available iron; adding iron can fulfill the needs of sulphur molecules and allow for increased phosphorus retention through biochemical reactions with iron (Caraco et al., 1989). Reducing sulphur molecules has the benefit of lowering pH, which results in lower un-ionized ammonia concentrations (Wiessner et al., 2008). Ann et al., (2000) found that iron chloride was the best iron additive for constructed wetlands. Stopping the use of fertilizer in proximity to the Napanee Landfill constructed wetland would reduce phosphorus concentrations and the concentrations are low for other pollutants that would see increased treatment from the use of an additive.

Hydraulic flow regimes have a significant impact on the oxygen concentrations in a constructed wetland (Tanner et al., 1999). Batch flow systems have higher oxygen concentrations throughout the water column and in the substrate compared to continuous flow systems (Tanner et al., 1999). There are trade-offs that occur when operating a system under batch or continuous flow regimes. Ammonia and sulphate biochemical removal processes require oxygen, which favors a batch flow regime; while nitrate removal through denitrification is most efficient at low oxygen concentrations, which favors a continuous flow system (Vymazal, 2007). The increased mixing that can occur in a continuous flow wetland can result in more phosphorus retention compared to a batch flow wetland (Doyle et al., 2003). The Napanee Landfill constructed wetland currently operates under a batch-flow hydraulic flow regime. Maintaining batch-flow is ideal for the Napanee Landfill constructed wetland in order to keep sustain nitrate and potentially increase ammonia treatment.

Fertilizer and other chemicals should not be used in proximity to the wetlands. There is the potential for the chemicals to flow into the wetland and compromise water quality. If large quantities of chemicals are being introduced to the wetland, the system could be overloaded and outflow pollutant concentrations could exceed water quality guidelines.

Records of operative procedures and collected data are important. Having a readily available database with records of activities such as batch-releases, water quality sampling, dredging, and plantings would help operators track trends and analyze operations for potential avenues for improvement. Current regulations require minimal record-keeping, which could lead to potentially important and environmentally damaging events to be overlooked. Treatment wetland systems need to receive more attention from operators and regulators, especially as they get increasingly used to treat more toxic pollutants at higher concentrations.

4.4 Future Considerations

While landfill surface-water runoff does not have pollutant concentrations as high as landfill leachate, I have shown that there is potential for some pollutants in landfill surface-water runoff to exceed Canadian provincial and federal guidelines. There are many factors that can influence pollutant concentration in landfill surface-water runoff. Influences from factors like landfill age and waste composition on landfill leachate pollutants are well documented (Kadlec and Wallace, 2009); having a similarly developed pool of data to draw on for landfill surface-water runoff would allow for better design and operation of landfill surface-water runoff management systems. My study was conducted on a landfill that has been recently closed. The capping procedures potentially reduced the exposure of waste to precipitation and might have reduced the concentrations and number of pollutants in the surface water runoff. Landfill surface water runoff needs to be characterized for landfills in different stages of operation, from newly open to having been closed for decades. The ideal project would collect data on landfills of different sizes from when they first open until well after they are closed. Considerations should also be made for seasonal variation in landfill surface water runoff pollutant types and concentrations.

Constructed wetlands are a promising management system to treat landfill surface-water runoff. Constructed wetlands in their various forms have been shown to successfully treat pollutants found in landfill surface-water runoff. More research is necessary to better determine the efficiency of constructed wetlands at treating the comparatively low concentrations of pollutants found as pollutants like un-ionized ammonia and phosphorus can be toxic at low concentrations. Hybrid constructed wetlands involving surface-flow and subsurface-flow modules have been shown to improve treatment performance over systems that solely use subsurface-flow or surface-flow wetlands. Rigorous studies on the use of hybrid systems with clear results could influence practitioners to build more hybrid systems.

Outside of their use for treating landfill surface-water runoff, treatment wetland can successfully treat pollutants found in landfill leachate (Kadlec and Wallace, 2009). The high concentrations of pollutants found in landfill leachate result in greater removal efficiencies (Oman and Junestedt, 2008). Systems that combine the treatment of landfill leachate and landfill surface water runoff could result in greater landfill surface water runoff pollutant removal since the low concentration limiting factor would be addressed.

4.5 Conclusions

Managing the environmental impact of landfills is important due to the variety and high concentrations of toxic pollutants that can be found in landfills. Landfill surface-water runoff has received little attention from researchers. My research goals for this study involved characterizing water chemistry of landfill surface-water runoff, evaluating constructed wetlands as a method for landfill surface-water runoff treatment, and using the gathered information to make recommendations to optimize the design, maintenance, and operation of the Napanee Landfill constructed wetland. Overall, median concentrations of pollutants did not exceed any Canadian federal or provincial guidelines. There were specific sampling day incidences where un-ionized ammonia and chloride exceeded CWQG guidelines at the outflow. Constructed wetlands have been shown to be successful in treating the pollutants observed in the Napanee Landfill constructed wetland, but sulphate was the only pollutant that saw a significant decrease in concentration from the inflow to outflow at the Napanee Landfill constructed wetland. The Napanee Landfill constructed wetland was not able to lower the concentrations of other pollutants. There are a variety of design and operation improvements that could be made to the Napanee Landfill constructed wetland in order to improve the treatment of incoming pollutants, including: increasing the size of the constructed wetland and/or incorporating subsurface-flow modules to the system. Incorporating a vertical-flow wetland would increase available surface area for nitrifying bacteria growth and would provide more oxygen for nitrification processes; both would increase the potential for significant ammonia treatment. Overall, the concentrations of the pollutants found in the surface-water runoff coming off of the Napanee Landfill constructed wetland did not pose a significant threat to the environment at the time of sampling.

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Appendix A – Photos

Figure 4-1. Pond 1

Figure 4-2. Pond 2 (left) and pond 3 (right)

Figure 4-3. Pond 4 (left) and pond 5 (right)


Figure 4-4. Pond 1

Appendix B – Memorandum of Understanding

Memorandum of Understanding for Research Projects Between
Waste Management of Canada Corporation University of Waterloo

This memorandum of understanding is made December 17 2009 between Waste Management of Canada Corporation (a Canadian Company) with its corporate offices at 117 Wentworth Court, Brampton Ontario L6T 5L4 (Waste Management) and the University of Waterloo with its address as 200 University Avenue West, Waterloo, Ontario, N2L 3G1

This memorandum of understanding cancels and replaces any previous agreements and memoranda between Waste Management and the University of Waterloo related to the project described in the attachment entitled, Assessment the Effectiveness and Performance of the Re-Constructed Surface Water impoundments at Napanee Landfill, Ontario dated 2009 12 17.

Whereas Professor Stephen Murphy intends to provide a research graduate student for Assessment of the Effectiveness of Constructed Wetlands at Napanee Landfill.

And whereas Waste Management has the experience, knowledge and ability to provide data that is public, and can provide that information to the research team and owns facilities that are of interest to this research

Therefore in consideration of the above the parties agree to:

1. Waste Management agrees to provide access and facilities as required for the research described above. Access to facilities, staff and sites to be specified in the project proposal attached to this Memorandum of Understanding.

2. Waste Management will provide all safety training to meet its corporate requirement to any accessing the site.

3. Waste Management agrees to provide material and labour to assist in the research to the extent described in the attached project proposal description of work. Such provision of services is not to impede the operational activities and environmental integrity of the site.

4. The University of Waterloo agrees to provide on campus facilities as required for the research. Funding from other agencies such as Ontario Centres of Excellence, NSERC and the University of Waterloo are expected to be applied for by Dr. S. Murphy but no guarantee is provided as to success, if any, of applications.

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5. The Client and the University agree that it is part of the University's function and policies to disseminate information and to make it available for the purpose of scholarship.

6. At any time during the term of this Agreement, the University will provide the Client with a draft copy of any proposed publication or disclosure of Research Result for its review at least sixty (60) days before submission for publication or disclosure. Upon the Client's written request, which shall be received by the University within the same sixty (60) day period, the University will:

    a. delete any Confidential Information of the Client from the proposed publication or disclosure; or

    b. delay publication, subject to section 7, up to a maximum of sixty (60) additional days for the purposes of filing for intellectual property protection on terms and conditions to be negotiated and agreed upon by the Client and the University.

7. Notwithstanding 6(b), the University retains the right to have the thesis reviewed and defended for the sole purpose of academic evaluation in accordance with the University's established procedures. The University will, in consultation with the student and the Client, determine if such a delay as set forth in 6(b) will be provided. The Client may request that a thesis defense be held in camera and that the members of the thesis examination board, including the external examiner(s), be required to sign a non-disclosure agreement. The University shall determine in its sole discretion if such request shall be granted.

8. Each of us will cooperate with the other in initiatives to publicize this arrangement, including specifically:
    a. Inclusion of the University of Waterloo logo on our academic entrance notice at the landfill and for no other purpose.
    b. An advertisement in the local newspaper for the information of the public
    c. Letter notification to the Council of the Town of Greater Napanee

9. The Memorandum of Understanding is terminable by either of us on 90 days notice to the other.

APPENDIX A

Assessment of the Effectiveness of Constructed Wetlands at Napanee Landfill

2009 12 17

Owner: Waste Management of Canada

Academic Affiliate: University of Waterloo, Waterloo, Ontario

Period: Three Years subject to review in 2012

Staff:

University of Waterloo	Dr. Stephen Murphy Professor Graduate Student TBA
Waste Management of Canada Corporation	Chris Prucha Hydrogeologist Randy Harris, Landfill Manager Linda Cooper, Community Relations Representative

Project Description:

The Project is to provide an assessment of the recently re-constructed surface water impoundments at Napanee Landfill. The impoundments were designed as wetlands and have some level of treatment inherent in the design. The performance of the impoundments to treat or attenuate upstream contaminants and landfill surface water contaminants is to be measured. Any conclusions from the research should be aimed at improvements of the system for treatment. Secondarily if improvements can be made for ecological purposes without impairing the treatment performance, they should be part of the research.

Facilities Provided:

- WMCC can provide, given adequate warning, of an office for research activities at the site. WMCC does not have laboratory processing ability at Napanee Landfill.
- WMCC will provide equipment to assist in the sampling program as needed and review of any data and interpretations on a regular basis.

Conditions:

- Safety Training will be provided by Waste Management of Canada to meet the company's mandate of safety at the site.
- All laboratory results are to be acknowledged as not-certified, so as not to conflict with any regulatory requirements imposed by the permits at the site.

- All University people are to sign in and sign out at the landfill gatehouse and to have the appropriate safety certification for WHMIS.

Costs:

- All costs related to the site would be born by WMCC.
- Our internal hydrogeologist and staff costs will be covered by WMCC.
- Costs of transportation to and from the site will be covered by University of Waterloo.
- Costs of sampling bottles and devices will be covered by University of Waterloo.

Funding:

- Waste Management of Canada Corporation is prepared to fund this project for a period of three years 2009-2012 in the amounts of $5,000 for 2009 $6,000 for 2010 and $6,000 for 2011. In kind equipment and staff are to be estimated by WMCC. The University of Waterloo agrees to provide on campus facilities as required for the research. Funding from other agencies such as Ontario Centres of Excellence, NSERC and the University of Waterloo are expected to be applied for by Dr. S. Murphy but no guarantee is provided as to success, if any, of applications.

Student Roster:

One Master's student primarily for the period 2010-2011