Natural Wetlands as Additional Wastewater Treatment for Phosphorus Removal in First Nations Communities in Manitoba

by

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ABSTRACT

At least 60% of First Nation communities in Manitoba, including the Lake Manitoba First Nation, are located in wetland areas. Wetlands can be used as an effluent-polishing step in removing phosphorus from wastewater. 47% of First Nations communities in Manitoba served by facultative lagoons failed to achieve the total phosphorus concentration of 1 mg/L in proposed regulations for effluent discharge into the environment. The Lake Manitoba First Nation community facultative lagoon system treats domestic wastewater and seasonally discharges effluent into a wetland that connects to Lake Manitoba. This research was performed to estimate phosphorus removal efficiency through the natural wetland during the vegetation growing season. The secondary treated wastewater was discharged at a mean flow rate of 2,823 m^3/day, resulting in a hydraulic loading rate of 22 cm/d with total phosphorus loading of 5 kg/day to an adjacent natural wetland. The average total phosphorus concentration reduction utilizing the observed treatment area of 1.3 ha was more than 70%, achieving the desired total phosphorus (TP) concentration below 1 mg/L. Data analysis showed that significant increase occurred in TP concentrations only in the Meleb series soil, in 20-40 cm depth interval in comparison to 0-20 cm and 40-60 cm depth intervals at the end of vegetation growing season. Hence, the main accumulation of TP occurred in the 20-40 cm soil depth. These short-term study results indicate the potential of natural wetland treatment applications under cold continental climate conditions, as an effluent polishing step to satisfy regulatory requirements for phosphorus reduction in smaller First Nations communities.

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DEDICATION

To Nikola and to my parents, I love you guys and have the biggest thanks…Život je lijep!

TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

CHAPTER 1 - INTRODUCTION

1.1 Sanitation in First Nations Communities in Manitoba

The primary objectives of wastewater treatment are the protection of public health and the environment. The effectiveness of treatment can be measured in terms of the reduction of carbonaceous biochemical oxidation demand (cBOD), total suspended solids (TSS) and un-ionized ammonia to a level that meets the effluent discharge regulations set by the Canadian Council of Ministers of the Environment within the Fisheries Act in 2009 (CCME, 2009). In addition, nutrient (phosphorus and nitrogen) control is also progressively required from wastewater treatment, mainly due to the problem of eutrophication in lakes (Schindler et al., 2012), (IJC, 2014).

In Manitoba, there are 63 First Nations communities, of which 55 communities are served by wastewater treatment systems (INAC, 2014b), (INAC, 2011b). Recently, many stakeholders have advocated for improved water and sanitation security in the First Nations Communities in Manitoba (Fallding, 2010), (CREATE H2O Program, 2013), (WCCR, 2010.). As of September 30, 2016, 138 communities, out of over 600 First Nations communities in Canada, were under drinking water advisories requiring residents to either boil their water or to heed “do not drink warnings”. 10 of these communities are in Manitoba, including Lake Manitoba First Nations Community (Health Canada, 2016). Low levels of sewage management serve as an inter-related factor responsible for environmental problems. Concerns of this topic dates back to 1975, when Gillies and Sherwood (Gilies and Sherwood, 1975) completed reports focused largely on remote Northern Manitoba, which prompted Environment Canada (today: Environment and Climate

Change Canada) to create an inventory of environmental problems in 28 First Nations reserves (Environment Canada, 1980). Among the investigated subject areas were wastewater treatment systems. Within this inventory it was revealed that wastewater treatment systems were built to serve schools, nursing stations, teacher's residence and sometime RCMP stations rather than personal residences (Environment Canada, 1980). In all of the communities, personal residences were using privies, with the exception of two communities where residences were serviced by sewage collection systems.

Wastewater systems managing, in First Nations communities is the responsibility of the Chief and the Band Councillors as well as territorial governments. At the federal level the responsibility is within three departments: Indigenous and Northern Affairs Canada (INAC) (previous: The Departments of Indian Affairs and Northern Development), Health Canada and Environment and Climate Change Canada. Their summarized roles are:

Indigenous and Northern Affairs Canada (INAC) provide funding to First Nations for the provision of water and wastewater infrastructure projects to First Nations communities, including capital construction, upgrading and a portion of operating and maintenance cost (80 %). Furthermore, oversees the design, construction and maintenance of drinking water/wastewater facilities (Simeoni, 2010)

Health Canada provides advice, guidance and recommendations about safe disposal of onsite domestic sewage and reviews wastewater infrastructure project proposals from a public health perspective on reserves located south of the 60th parallel, either directly or in an oversight role (White et al., 2012)

Environment and Climate Change Canada is involved in source water protection through its powers to regulate wastewater discharge into federal waters or into water generally where water quality has become a matter of national concern, and to enforce effluent discharge standards into water throughout Canada (Simeoni, 2010).

First Nations communities, through their Chiefs and Band Councils, are responsible for providing 20% of the cost for the design, construction, operation and maintenance of their wastewater treatment systems. In addition, they are responsible for ensuring that the wastewater treatment systems are operated by trained operators. Due to a lack of financial assets and having available land, these small communities are driven to build and operate less expensive treatment technologies such as lagoon treatment systems (White et al., 2012), (Gloyna, 1971), (Heinke et al., 1990).

The majority of First Nations communities in Manitoba are located around and between the cities of Dauphin and Thompson, which have average number of 199 and 239 days below 0°C, respectively (Weather Base n.d.). Temperatures substantially way below 0°C cause lagoons that are not aided by mechanical aeration to freeze (during often winter months), thereby decreasing treatment efficacy. Consequently, the lagoon's mode of operation in the majority of these communities tends to be a long-term storage as opposed to lagoons in locations of warmer conditions (Schneitner et al., 1993).

Krkosek et al. (2012) describe the domestic wastewater management in Northern communities, to have a holding tank at each residence or have a "honey-bag" - a plastic bag holding feces and urine. Wastewater is then pumped or picked up and transported to the local lagoon where the wastewater is dumped. They also noted that most of the lagoons in cold climatic regions operate as a storage during the winter months, with intermittent discharging once or twice per year in the spring, during break up, or in the fall, just before freeze-up. In addition, some lagoons do not have impervious berm or liner which classifies them as long detention time continuous discharge lagoons in warmer spring and summer months.

In 2011, a national assessment of First Nations water and wastewater systems was made available to the public, referred as a "Report" throughout the rest of the thesis (INAC, 2011b). The Report points out "..the average per capita demand ranges from 10 L/p/d to 420 L/p/d, with an average per capita demand of approximately 176 L/p/d.", whereas in 2011, the average daily Canadian residential uses of fresh water per capita was 251 L stated by Environment and Climate Change Canada (ECCC, 2011). Lagoon influents can be characterized as undiluted or high strength (e.g. 300 mg/L of biochemical oxygen demand, BOD), due to low water use. On the other hand, high water use (e.g. 420 L/p/d) may occur because of the practice of "bleeding" the water lines to prevent freeze-up, which results in greatly diluted or weak sewage with organic content as low as 50 mg/L BOD (Townshend and Knoll, 1987). These practices need to be identified and considered in designing of lagoon systems.

The Report addressed Manitoba's 61 wastewater systems that treat community wastewater, serving 55 First Nations, of which 57 wastewater systems comprises of lagoons, mechanical plants and septic systems and 4 wastewater systems provide treatment of community wastewater through a Municipal type agreement. Wastewater treatment through lagoons systems is used in 31 communities, which makes up 56 % of overall assessed treatment type. The Report indicates implementation of facultative lagoons in 21 First Nations communities while aerated lagoons are implemented in 10 communities. Facultative lagoons are wastewater stabilization ponds that are designed for biological treatment of raw domestic wastewater (USEPA, 2011).

Facultative lagoons in First Nations communities provide secondary wastewater treatment level with no installed disinfection unit and with no sludge treatment option. All facultative lagoons are designed having primary treatment cells and at least one storage cell. All of the facultative lagoon systems in Manitoba are designed as intermittent discharge systems that seasonally discharge effluents between June 15 and October 31, whereas 50 % of aerated lagoons among First Nations communities in Manitoba continuously discharge their treated wastewater into receiving bodies (INAC, 2014a). The First Nations communities' wastewater systems follow the INAC's Centralized Wastewater Protocol (INAC, 2010a) and strive to comply with Federal legislation. Relevant Federal legislation include Wastewater System Effluent Regulations within The Fisheries Act and The Canadian Environmental Protection Act , as well as, other applicable legislation, including Manitoba's provincial legislation (The Water Protection Act),(CG, 2012), (INAC, 2014c).

Overall, First Nations communities' wastewater systems in the Report are assessed by Neegan Burnside Ltd. based on criteria determined by Department of Aboriginal Affairs and Northern Development 's (todays INAC) Risk Level Evaluation Guidelines, which include five categories, namely: effluent receiver, design, operation and maintenance, reporting, and operators (INAC, 2011b), (INAC, 2010b). The overall risk score for wastewater systems is the average of the component risk scores. The risk for each of the five categories is ranked numerically from 1 to 10, with 1 to 4, 4.1 to 7.0, and 7.1 to 10, representing low risk, medium and high risk, respectively.

The first of the five categories - effluent receiver risk, is evaluated based on the receiving environment and the extent to which it is required for other human uses, such as fishing, recreation or drinking water. The assessed risk for treatment of community wastewater through aerated lagoons and facultative lagoons based on effluent receiver is 6.1 and 5.0, respectively, which translates to medium risk for both. 43 % of 37 First Nations communities in Manitoba, are under medium risk of surface water use as their source of drinking water after specific treatment. The Report indicates 14 communities use wetlands as a treated wastewater effluent receiver.

The second risk category evaluation reported is the design of aerated lagoons and facultative lagoons. It is assessed based on the extent to which the design has: (1) inappropriate treatment processes; (2) poor system reliability; (3) lacks the flexibility to meet future growth; (4) exceeded the design capacity; (5) inappropriate waste management. The assessed risk for treatment of community wastewater through aerated lagoons and facultative lagoon based on the design risk is 2.8 and 3.2, respectively, which means it poses low risk for both (INAC, 2011b). However, the final risk score is not in agreement with the following observations. In the Aboriginal Demography - Population, Household and Family Projections report, INAC stated that significant population growth is expected in all regions of Canada from 2001 to 2026, particularly in the prairie region, with overall growth in Manitoba of 53% (INAC, 2011a). In five communities where the maximum daily flow exceeds the design capacity, results imply that exceeding the capacity leads to hydraulic and organic overloading which could reduce efficient treatment of sewage. In addition, there is no sludge treatment within lagoon systems. Lagoon operation involves a net sludge build-up during winter months and a net sludge decrease during summer months. Sludge accumulates faster in colder climate conditions due to overall inhibition of anaerobic microorganism's activity. Under these conditions, there is a very little decomposition of organic matter. Consequently, the solids accumulate faster than they can be decomposed, resulting in a net sludge build-up (Schneitner et al., 1993). In their study about sludge from cold regions lagoons in North America, Schneitner et. al (1993) cite an accumulation rate in facultative lagoons of approximately 3 cm annually. Data obtained from the Report reveals construction year of lagoons for almost every system. For example, the Sandy Bay First Nations community's lagoon dates from 1988, which means 23 years of sludge accumulation until 2011, the year that the Report was published. Therefore, it can be assumed the accumulated sludge is approximately 69 cm. In addition, it has been demonstrated that the sludge deposit thicknesses are larger around the inlet of the cell, indicating that sludge is not distributed equally. Changing the shape of the bottom surface, may result in reducing the pond effective volume and shortening hydraulic residence times, which reduce efficient treatment of wastewater (Pena et al., 2000). Heinke et. al (1990) suggested a sludge removal frequency of 5 to 10 years for short-retention lagoon cells in cold climates, while long retention-time lagoons with seasonal or annual discharges may not need to be de-sludge for longer periods (Heinke et al., 1990). Lagoon de-sludging, as well as improvement of the inlet-outlet arrangements would increase the removal efficiency of TSS and decrease cBOD. Therefore, a sludge management plan should be incorporated in the management of each lagoon.

The third risk category evaluation reported in the Report is the operation risk of aerated lagoons and facultative lagoons. It is assessed based on extent to which: (1) inadequate maintenance logs are maintained; (2) general maintenance is not being performed adequately; (3) emergency response plans are not in place or not being used; (4) operations & maintenance manuals are not available or not in use. The assessed risk for treatment of community wastewater through aerated lagoons and facultative lagoon based on operation risk is 5.5 and 7.1 (medium and high risk), respectively. In addition, 18% of overall wastewater treatment systems including aerated lagoons and facultative lagoons fail to meet federal effluent quality guidelines due to operations. Half of First Nations communities in Manitoba use surface water as their source for drinking water supply after specific treatment from which 16 communities with lagoons systems discharge their effluent into surface waters (such as a lakes, river, or creek) that are assessed to pose as medium or high risk to water quality and human health (INAC, 2011b).

The fourth risk category evaluation reported in the Report is the reporting risk of lagoons. It is assessed based on evaluation and maintenance of effluent-testing and system-monitoring records and is noted that little record keeping is required for lagoons except for keeping general maintenance logs and sampling before discharging effluents. The assessed risk for treatment of community wastewater through aerated lagoons and facultative lagoon based on reporting risk is 4.8 and 3.6, posing medium and high risk, respectively (INAC, 2011b).

The fifth assessed risk for treatment of community lagoons wastewater based on the operator's risk is assessed as a low risk (INAC, 2011b). Whether the operator has adequate certification is assessed by this risk. Results showed 44% of the all 57 wastewater systems, did not have a fully certified primary operator and 89% did not have a fully certified backup operator. This can contribute why many authors imply lack of operational skills and knowledge as point of advantage in implementing lagoon process systems in northern remote communities and using natural treatment options (Heinke et al, 1990), (Gloyna, 1971).

As aforementioned, the INAC developed Risk Level Evaluation Guidelines to assess how well the wastewater systems are managed through the multi-barrier approach (INAC, 2010b). For example, using the data reported in the Report, the assessment risk evaluation of the Lake Manitoba First Nation community wastewater system based on the guidelines is provided in Table 1.1.

Table 1.1 The example of risk evaluation calculations for Lake Manitoba First Nations community (Surveyed data provided from the Report and weighted data resulted from INAC, 2010b)

Lake Manitoba First Nation Community	*Effluent Risk*	*Design Risk*	*Operations Risk*	*Report Risk*	*Operator Risk*	*Final Risk Score*
Surveyed data	4	2	7	1	1	3.3
Weighted results	0.4 * 10%	2*25%	7*25%	1*10%	1*20%	3.0

It should be noted that the summary of percentage for wastewater systems does not equal 100 percent as it does for water treatment systems calculations, and it is hard to calculate the same score as presented in the Table 1. Levangie (2009) in her thesis identified possible drawbacks and critiques of the guidelines, such as the weightings of specific categories in the INAC tool. For example, the operation and maintenance section was over-weighted; while the design and source protection section, as well as the integration between water and wastewater were inadequate.

Several upgrades or new investments of wastewater systems have been implemented since the release of the Report, such as, the implementation of new aerated lagoons, or expansion of an existing lagoon and/or implementing Submerged Attached Growth Reactors (SAGR) treatment methodologies (INAC, 2015), (Nelson Environmental Inc., 2016). As aforementioned, all systems follow the INAC's Centralized Wastewater Protocol and strive to comply with relevant federal and provincial legislation (INAC, 2010a).

The majority of wastewater treatment systems used by First Nations in Manitoba are lagoon systems (INAC, 2011b). Lagoon systems were designed to decrease levels of $cBOD_5$ and TSS to achieve secondary treatment levels (Government of Ontario, 2008). Currently, more stringent regulations in wastewater effluent quality require nutrient removal as well. Review of the Summary report of Manitoba First Nations wastewater systems effluent data provided by INAC (Daniel Benoit, personal communication, Nov 7, 2014), indicate that 47 % of facultative lagoons in Manitoba First Nations communities in 2014, did not achieve reduction of total phosphorus concentrations in effluent to 1 mg/L, with average TP concentration of 1.83 ± 2.09 mg/L (raw data presented in Appendix 4) (Figure 1.1). In addition, results of aerated lagoons, some improved with addition of chemical salt-aluminium, among First Nations Communities in Manitoba in 2014, revealed that only 38 % of communities achieved the proposed less than 1 mg/L total phosphorus concentrations in effluent (Figure 1.2).

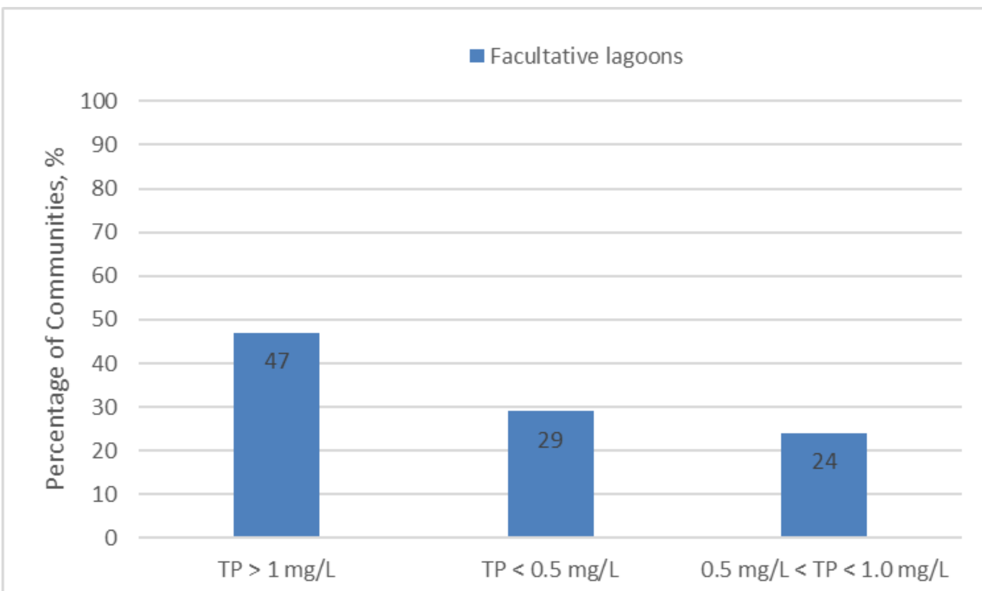


Figure 1.1 Percentage (%) of First Nations Communities using facultative lagoons as a wastewater treatment system and the achieved total phosphorus (TP) concentrations in effluent 2014 (Results provided courtesy of INAC (Daniel Benoit, personal communication, Nov 7, 2014))

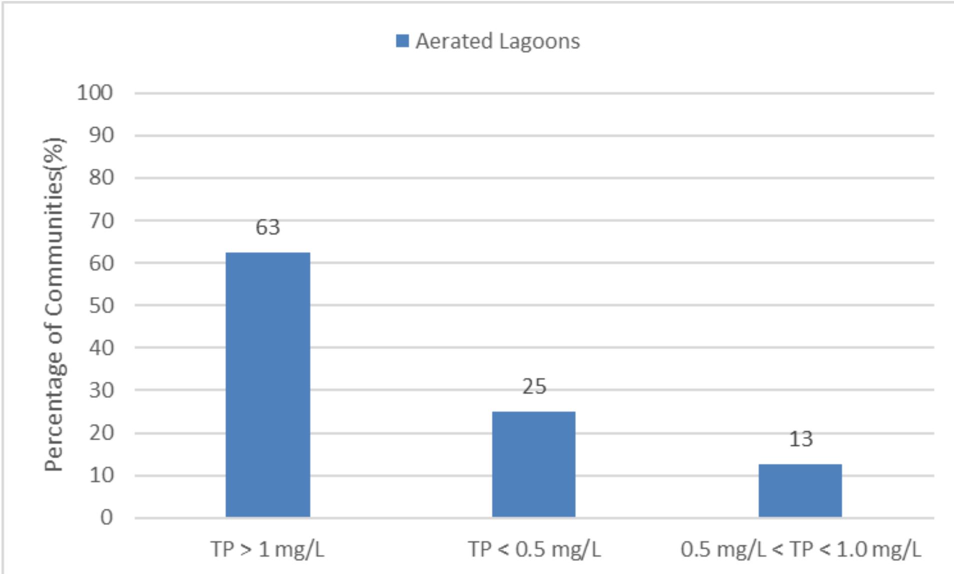


Figure 1.2. Percentage (%) of First Nations Communities using aerated lagoons as a wastewater treatment system and the achieved total phosphorus (TP) concentrations in effluent 2014 (Results provided courtesy of INAC (Daniel Benoit, personal communication, Nov 7, 2014))

Nevertheless, lagoon systems in other parts of Canada also omit phosphorus levels reductions and are not capable of meeting the nutrient discharge standards which are often lower than 1 mg/L (Kadlec et al, 2012). Kadlec et. al (2012) found a solution in the addition of treatment wetlands downstream of the lagoon and proved the ability of natural wetlands to seasonally remove phosphorus from lagoon effluents for 30 consecutive years in Michigan, USA.

1.2 Wetlands

Natural wetlands have served as a receiving body for wastewater disposal for as long as wastewater has been collected, which date back to the beginning of the 20th century in North America. There are many existing locations where natural and human-made wetlands serve as a discharge receiver, although most of them haven't been designed for wastewater or storm water treatment. There are 14 First Nations communities in Manitoba where natural wetlands are being used to dispose treated wastewater, some occurring since the 1970s (INAC, 2011b). Integrated results of existing natural systems lead to a better understanding of natural wetlands and their role as an ecosystem for pollutant assimilation which initiated the design of new natural wastewater treatment systems (Pries, 1994). It is considered that the research using constructed wetlands for wastewater treatment started based on observations made on the apparent treatment capacity of natural wetlands. Research insights gained from several studies on natural wetlands and implementation considerations of constructed wetlands for wastewater treatment happened during the 1980s and 1990s in Canada (Herskowitz, 1986),(Miller, 1989), (Serodes and Normand, 1999), (Dubuc et al., 1986), (Lakshman, 1983), (Kent, 1987), (Kadlec, 1983b).

During the 1990s, there were 67 treatment wetland systems used for municipal wastewater treatment across Canada, which included treatment of wastewater from agricultural practices, stormwater treatment and mining acid drainage treatment (Pries, 1994). Of those described systems, 31 consisted of lagoons and wetlands, some of which served as passive wastewater treatment systems and functioned on a seasonal basis.

Hierarchical methods to distinguish natural wetlands have been established using criteria to classify them into to categories of class, form and type. Within the class grouping, natural wetlands can be defined as a bog, fen, marsh, swamp or shallow open water. Wetlands are classified into forms described based on their surface form and pattern, water type and soils, while classified into types are determined based on vegetation. The possible combination of these classifications comprises more than 100 different classification of wetlands in Canada (National Wetland Working Group, 1997). Although there are many differences among wetlands they all share three common attributes. Firstly, soils saturated with water are an essential constituent of them all. Whether soils are saturated by either surface water or ground water for long periods or for much of the growing season (i.e. presence of water table at or less than 30 cm of the surface [rooting zone] for a period of 14 consecutive days during normal years), specific vegetation types, called hydrophitic plants are grown (Brix, 1994), (Vepraskas and Craft, 2016). These plants have specific structures, able to transport atmospheric gases, including oxygen, through their leaves and stems to their roots for respiration. These same structures also transport respiratory by-products and other gases generated in the soil back up the roots, stem and leaves for release to the atmosphere, thereby reducing the potentially harmful accumulation of gases in the region of the growing roots. Emergent plants obtain phosphorus primarily from soil, floating plants directly from the water, and rooted submerged plants from both soil and water (Nichols, 1983). Secondly, when soils become water-saturated, microbial communities promoting biochemical reactions consume oxygen changing soil conditions into anaerobic conditions. These conditions cause many elements and compounds to occur in reduced forms creating characteristic colours, textures and compositions typical of hydric soils. In addition, they influence nutrient cycling, pH change, sediment and organic matter accumulation, decomposition, and metal concentrations in the sediment and water (Kadlec and Knight, 1996). Thirdly, wetlands share the ability to be among the most productive ecosystem in the world. The high productivity in wetlands promotes high microbial activity and plant growth, which, in turn, results in a high capacity to decompose organic matter and recycling, and in rapid biomass turnover rates. Furthermore, the interaction between microbial communities and hydrophitic plants in wetlands results in high rates of nutrient transfer between these components (Berezowsky, 1995).

Due to these three attributes, numerous biogeochemical processes are performed in wetlands, enhancing transformation and retention of nutrients which leads to better quality of surface water and subsurface water (Hogan et al., 2004). Research results show that wetlands using chemical, physical and biological processes remove a variety of contaminants from water and effluent flowing through them (Kadlec et al., 2009).

1.3 Wetlands and phosphorus retention

A natural wetland, like any other biological wastewater treatment process, will exhibit variability in nutrient removal efficiency (Crites and Tchobanoglous, 1998). Removal efficiency is the percentage of a nutrient retained and removed from the overlying water column within the wetland ecosystem or released into the atmosphere (Hunter et al., 2009). Pries (1994), reported 55% as the average removal efficiency for total phosphorus, based on the operational data from 97 sites in North America (including both natural and constructed wetlands). Fisher and Acreman (2004) compiled results from studies examining total phosphorus removal efficiency in 57 natural wetlands and reported that 84 % of natural wetlands exhibited some total phosphorus retention while 6 % of them resulted in no net change in the concentration of total phosphorus of waters entering the wetlands and waters draining the wetlands. Most of these natural wetlands were riparian wetlands (in between land and river or stream), whereas five of them were classified as swamp/marsh, and served to treat wastewater effluent flowing through them. Some summary results of total phosphorus retention based on the input-output nutrient “black box” model within particular attributes are presented in Table 1.2.

Table 1.2 Summary of reference studies and their removal of P from wastewaters applied to natural wetlands

Location	*Wetland type*	*Size, ha*	*Total Phosphorus Loading rate, g/m²/y*	*Total Phosphorus Retention, %*	*Reference*
Wisconsin, USA	Marsh	156	15.2	32	(Spangler et al., 1977)
New Zealand	Marsh	NA	34	30	(Cooke, 1994)
Florida, USA	Marsh/Swamp	204	0.9	87	(Boyt et al., 1977)
Australia	Marsh/Swamp	60	3.8	94	(Patruno and Russell, 1994)
Louisiana, USA	Swamp	231	2.45	66	(Zhang et al., 2000)

In addition, Fisher and Acreman (2004) assessment showed that among the factors that influenced phosphorus retention, sediment oxygen content and redox potential had the most impact; being described as important in increasing the binding capacity of iron (Fe) and aluminum (Al) with phosphorus and enabling it to be buried by sedimentation. Hydraulic retention time, hydraulic loading and uptake by vegetation processes were also found to be important factors in phosphorus retention (Fisher and Acreman, 2004). Spangler et al. (1977) noted a low wetland phosphorus retention. However, by assessing the vascular growth of vegetation they noted that about 0.6-1.6 g P /m^2 would be removed by two or more harvests during the growing season. In addition, they assessed the concentration of phosphorous in soil throughout the wetland systems which comprised of wastewater treatment plant area and wetland area. They noticed no difference in concentration between inlet and outlet at the wetland area. However, they noticed a difference between the soils at the wetland inlet and the area where the wastewater treatment plant discharged effluent to the system. They concluded that a major portion of phosphorus was being removed by precipitation processes and retained by sedimentation in that region. Cooke (1994) tested several hypotheses of phosphorus transport and storage mechanisms in a temperate marsh wetland after a decade of receiving treated secondary wastewater from an aerated lagoon containing high phosphorus load. Sediment deposition was recorded in the wetland area with up to 30 g P/m^2/day. It was found that the deposition was enhanced by the mixing of sewage impacted wetland water and the natural wetland water. Furthermore, mixing conditions measured in the field together with laboratory experiments revealed that phosphorus deposition was predominately P removal by adsorption to Fe and/or Al containing substances. Boyt et al. (1977) asessed a wetland's (marsh and hardwood swamp) effectiveness in phosphorus uptake during wet and dry climatic conditions under low phosphorus loading. During dry conditions, a reduction in total phosphorus concentrations of 98% was achieved between the first and the last sampling sites. Boyt et al. (1977) found that the concentration of phosphorus was diluted during wet conditions, however, both conditions reduced phosphorus concentrations to values equal to or less than those found in the control site or in the lake (below 1 mg/L). In addition, Boyt et al. (1977) reported total phosphorus levels in the soil throughout the wetland system. The sampling sites closest to the wastewater plant outlet exhibited the highest concentrations of total phosphorus, while the control stations exhibited slightly less concentrations than at the first station. There was no evidence of a greater buildup of phosphorus in the soil of the wetland than in the control sites over the period of 20 years of receiving treated municipal wastewater. Patruno and Russell (1994) commenced a water sampling program in a wetland receiving wastewater effluent which showed a high average retention of phosphorus. Furthermore, dry weather conditions or low rainfall produce no discharge from the wetland into the receiving water. They also addressed possible concerns to the long term viability of a wetland under such loads, as the loading was expected to increase due to a forecasted increase of population. However, no predominant mechanism of phosphorus removal was addressed. Zhang et al. (2000) reported the average total phosphorus result over a two-year period in the inlet, treatment area and a control site of a swamp wetland and indicated that phosphorus removal was due to plant uptake, microbial assimilation, and soil adsorption. Moreover, they reported total phosphorus reduction of 11% within a 100 m distance from the inlet of wetland, and concluded that the reduction was mainly due to dilution, based on changes in the concentrations of chloride ions.

Additional review of natural wetland studies receiving wastewater by Nichols (1983) indicated that nutrient removal efficiencies rapidly declined with increasing loading rates. The P removal efficiencies of these systems fell below 50% at P loads greater than 8 g P/ m^2/ year. Nutrient removal or removal efficiency of a treatment wetland is inversely related to loading rate into a wetland. Richardson and Nichols (1985) demonstrated that phosphorus removal efficiency is typically low at higher loading rates (more than 15 g P/m^2/yr) and high at lower loading rates (e.g., less than 5 g P/m^2/yr). Thus, the removal efficiencies of natural treatment wetlands presented in Table 1.2 are in agreement with these results. The general rule of a thumb for wetlands intercepting non-point source pollution is that those wetlands can consistently retain phosphorus in the amount of 0.5 – 5 g P/m^2/year (Mitsch and Gosselink, 2015). Extended research was conducted for three decades in a lightly loaded natural wetland and showed the ability to seasonally provide 94 % of P removal from treated lagoon wastewater (Kadlec, 2009b), (Kadlec and Bevis, 2009), (Kaldec, 2009a).

Based on the above mentioned research observations a number of artificially or constructed wetlands have been modified or new constructed for purpose of water quality improvement, including for waters from urban runoff, municipal, industrial, agricultural practices, acid mine and agricultural drainage (Herskowitz, 1986), (Juston and DeBusk, 2005), (Carleton et al., 2001), (Yates et al., 2012), (Nichols, 1983), (Kadlec et al., 2012), (Sobolewski, 1999), (Dunne et al., 2005), (Jamieson et al., 2007), (Braskerud, 2002).

There are two main types of constructed wetlands: subsurface flow (SSF) and free water surface (FWS). SSF wetlands, also known as vegetated submerged bed systems, are made of a bed of media, with substrate such as crushed rocks, gravel, small stones or soil, and flow occurs beneath the surface of the media and is not visible, nor is it available to wildlife if properly designed. Wittgren and Maehlum,(1997) suggested that SSF wetlands are better suited for cold climate regions such as Canadian climates because of their ability to insulate microbial communities from frigid air and cold water temperatures. In terms of phosphorus removal, types of SSF wetlands are being researched with different selections of substrates and their ability to adsorbed phosphorus (Drizo et al., 2002). However, over time, potential P adsorption sites in the substrates can become saturated (Drizo et al.,2002), (Tattrie, 2006) or clogged with biofilm and organic matter (Weber et al., 2007), (Austin et al., 2007) resulting in a decrease in P removal. A system that is similar to a SSF wetland become operational in Manitoba in 2009, at the Village of Dunnottar (Anderson et al., 2015). Their passive filter system continuously treats secondary treated municipal wastewater from September to November each year. The passive filter consists of two vertical flow (25 m x 50 m) cells with a total bed volume of 3000 m^3. At average design flow rates of 250 m^3/d, the filter rate is 0.004 m/hr. The system can accommodate flows up to 500 m^3/d. The cells are lined with geomembrane liner filled with natural media (select sizes of rocks, sand, and gravel) and are covered with organic soil which is planted with local plants that contribute to the nutrient reduction and removal processes. The wastewater inflow is achieved through perforated piping that distributes the flow across the filter bed. Over its four operating seasons (2009 -2012) the system demonstrated up to 70 % removal of total phosphorus.

Free Water Surface (FWS) wetlands, contain aquatic plants that are rooted in a soil layer on the bottom of the wetlands and water flows through the vegetation consisting of leaves and stems which closely resembles natural wetlands. They typically have impervious clay or synthetic liners and engineered structures to control flow direction, liquid detention time and water level (USEPA, 2000). There are at least four FWS type of wetlands that provide polishing of lagoon effluents to reduce total phosphorus concentrations in Manitoba (Government of Manitoba, 2016). The systems, characteristics and phosphorus removal efficiency are summarized in Table 1.3.

Table 1.3. Examples of wastewater systems with wetlands as an additional treatment system in Manitoba, together with their characteristics and phosphorus removal efficiency

Location system	*Wastewater system*	*Treatment System*	*P Treatment Removal efficiency, %*	*Reference*
St Clements, Grand Marais	Primary/Secondary Lagoon Cell	Free surface Wetland	78-92	(Anderson, et al. 2013)
Riding Mountain NP	Primary/2 Secondary Lagoon Cells	Modified Natural Wetland	61-78*	(Belke and McGinn, 2003)
LGD of Pinawa	Primary/4 Secondary Lagoon Cells	Serpentine ditch-type constructed wetland	NA	(Conservation and Water Stewardship, 2014)
RM of Roblin	Primary/Secondary Lagoon Cell	FSW / Hybrid Poplar Plantation	100**	(Conservation and Water Stewardship, 2005)

*- Results of orthophosphate removal efficacy, %

** - Operated as of zero emission every two years of three years due to evapotranspiration loss in northern North America (Kadlec et al., 2009)

Outside of Canada, in northern regions with harsh climate where small sized communities persist several studies and reviews of treatment wetlands have been conducted (Jenssen et al., 1993), (Mander and Jenssen, 2003), (Wittgren and Maehlum, 1997), (Kennedy and Mayer, 2002). These studies demonstrate that wetlands have become accepted as a complement to more traditional methods. Chouinard et al. (2015) provided results of two case studies conducted in the Canadian Arctic using a passive wastewater system (wastewater lagoon plus wetland) as polishing step in lagoon system with seasonal discharge. Doku and Heinke (1995) suggested constructed wetlands be used in Northern Canada, as there is adequate land space and a need for an efficient and inexpensive treatment in many small Northern communities. Some of the studies revealed the economic benefits as a major advantage of using natural wetlands for wastewater treatment. Breaux et al. (1995) provided an example of economic benefit at a Louisiana natural wetland which was used to provide tertiary treatment of secondary treated effluents with substantial financial cost savings in comparison to the sand filtration method for treating approximately 15,000 m^3/day wastewater resulting in a hydraulic loading rate of 0.4 cm/day secondary treated effluent (Breaux et al.,1995). At that point using 9 % of discount rate and estimated 30-year life of conventional plant, the capitalized cost savings of using wetlands with UV disinfection resulted in average of $476000 or $2213 per hectare. The cost of piping and distribution structures, engineered improvements and restricted uses from previous un-impacted site conditions were included in the potential costs. In his study, Kirby (2002) noted that for FWS wetland and un-landscaped marsh both receiving treated wastewater, the use of a marsh land resulted in a 82 % reduction in cost.

With respect to wastewater treatment using wetlands, primary treatment of the wastewater is required since there is a potential for human and wildlife contact within the influent (Kadlec et al., 2009). In the United States, utilizing natural wetland treatment is limited to providing further treatment of secondary effluent (USEPA, 1987). In Canada, Alberta regulations include natural wetlands as an effluent receiver if it is evaluated and designed in accordance with the Alberta Environmental Protection publication entitled Guidelines for the Approval and Design of Natural and Constructed Treatment Wetlands for Water Quality Improvement (AEP, 2013). The publication recommends treatment wetlands for providing tertiary treatment of the municipal wastewater. In addition, recommendations are made with respect to hydraulic loading rate (3 cm/d or 3.3 ha area per 1000 m^3/day), as well as, hydraulic retention time (14 to 20 days) for a natural treatment wetland. A decade ago, the Lake Winnipeg Stewardship Board addressed among its recommendations the use of natural wetlands as nutrient abatement options, undertaking an in-depth review of the effectiveness of natural wetlands to reduce nutrient loading to Lake Winnipeg (Lake Winnipeg Stewardship Board, 2006). In addition, several strategies for nutrient reduction from wastewater facilities were provided and the emphasis was placed on phosphorus removal through means other than chemical precipitation (Marbek, 2010). Reasons for that are that chemical precipitation generates more sludge, which results in higher cost for sludge management. In addition, sludge stabilization, thickening, dewatering and transportation are highly energy-intensive processes. In terms of lagoons, using chemical precipitants generates sludge in which phosphorus is in bound with iron or aluminum oxides and hydroxides and not as readily available to growing plants if the sludge was land applied. Moreover, usage of coagulants leads to a higher content of inorganic salts in lagoon effluents which may adversely impact the germination of plants if the effluent was land applied (Marbek, 2010).

The sustainable use of natural wetlands requires proper management, considering the carrying capacity of the wetland ecosystem to ensure that the ecological and socio-economic functions of the wetlands are maintained for the long term in a sustainable manner. When a natural wetland is present as a wastewater receiver, it provides an opportunity for environmental enhancement and significant cost savings (Doku and Heinke, 1995), (Nõges and Järvet, 2002).

An adequate pretreatment and hydraulic loading rates, collection of monitoring information to assess system performance and knowledge of successful operation strategies drive effective natural wetland treatment performance (Kadlec and Knight, 1996).

1.4 Phosphorus forms and sources in wetlands

Phosphorus is often present in natural waters and wastewater as part of a phosphate molecule ($PO_4^{3-}, PO_4^{2-}, PO_4^{-}$). They are classified as: orthophosphate, complex or condensed (pyro-, meta- and poly-) phosphate and organically bound phosphate and their summation in water represents total phosphorus (TP). The common forms of phosphorus entering or leaving the wetlands are in soluble (dissolved) or insoluble forms of phosphorus. In aquatic environments, phosphates are typically measured as total phosphorus (TP), total dissolved phosphorus (TDP), soluble reactive phosphorus (SRP), particulate phosphorus (PP) and dissolved organic phosphorus (DOP) which can be computed from the measured components (Clesceri, Greenberg and Eaton 1998). The dissolved forms of phosphorus refer to the portion of the total phosphorus content that is present in the water after filtration through a 0.45 µm filter.

Soluble reactive phosphorus (SRP), or orthophosphate exists as an anion with distribution in the forms of $H_2PO_4^{-}, HPO_4^{2-}$ and PO_4^{3-}, at pH’s of 2-7, 8-12 and >13, respectively (Holtan et al., 1988). Therefore, the mobility of phosphorus is controlled by the pH of the medium. Mono- and dibasic phosphates ($H_2PO_4^-, HPO_4^{2-}$) are the most dominant forms in wetlands, with pH values ranging from 4 to 9 (Moler and Hering, 1993). Organically bound phosphorus refers to phosphorus bound to organic matter and can be within living organisms as phospholipids, co-enzymes, and nucleotides (deoxyribonucleic acid (DNA), and ribonucleic acid (RNA)), as well as associated with detrital particulate organic matter, including inorganic phosphorus compounds irreversibly sorbed onto organic matter. Wetland can be seen as consisting of several storages: water, plants, microbiota, litter and soil (Kadlec et al., 2009), and provides an environment for the interconversion of different forms of phosphorus (Figure 1.3).

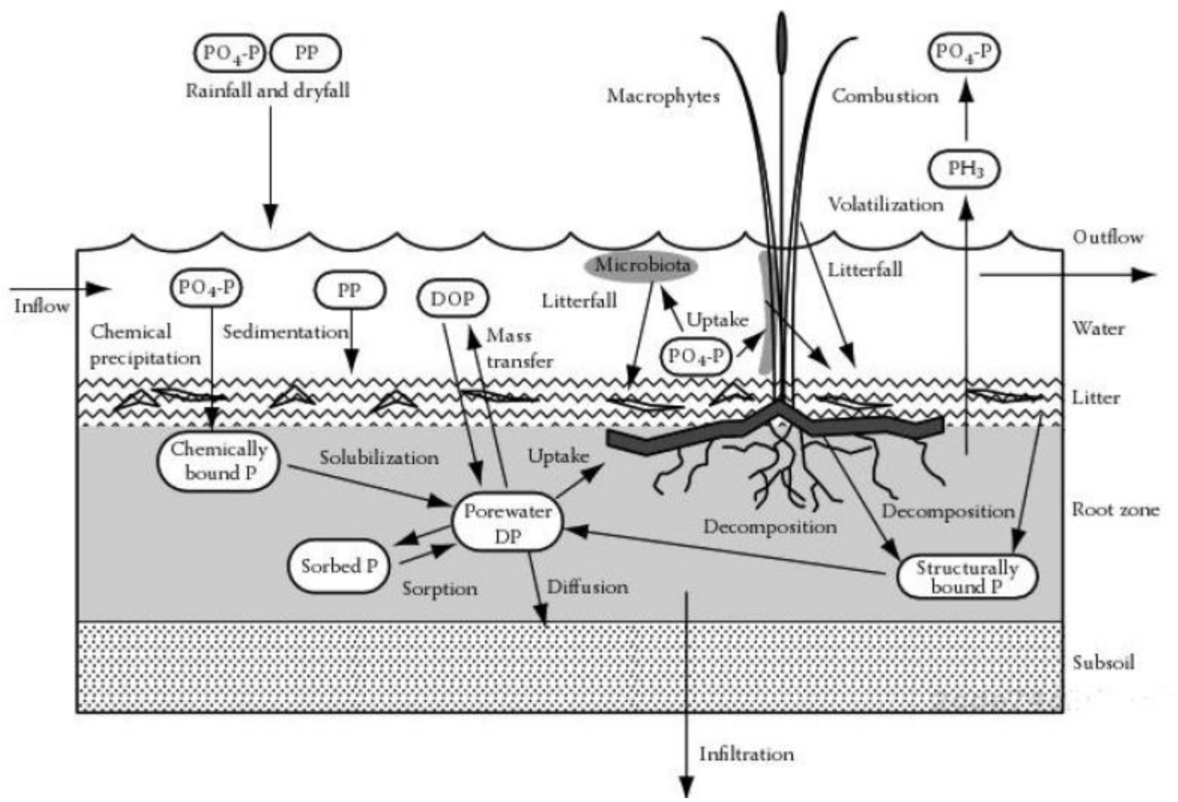


Figure 1.3. Phosphorus interconversions and storages in wetlands (Adopted from Kadlec and Knight, 1996)

Through decomposition of organic material and through the chemical weathering of soil and rocks, the phosphorus compounds are released in dissolved forms (DP). Particulate forms (PP) of phosphorus are released through the physical weathering of soils, more specifically, in the form of iron hydroxides, carbonates, or through plant tissue decomposition and release of cellular components (Vepraskas and Craft, 2016). As a growth-limiting nutrient for microbiota and plants in aquatic ecosystems, the forms of DP are not built up in the water column, but are instead taken up readily (Fisheries and Oceans Canada, 2015), (Richardson and Vaithiyanathan, 2009). In the root zone, DP occurs in porewater mainly as orthophosphate (PO_4 -P) while PP is mainly a component of soil particles (Valsami-Jones, 2004). Phosphorus availability of PP form is controlled by dissolution, precipitation reactions and sorption reactions which are controlled by the pH in the medium (Frossard et al., 1995).

The interconversions among different phosphorus forms in wetlands occur at the different rates for each forms (Kadlec, 2016). For example, average TP mass removal rate was reported lower then average PP mass removal rate for a period of records (2004 -2012) in the Lake Apopka, Florida marsh systems (Dunne et al., 2015). Furthermore, Kadlec (2016) was noted that the quicker reduction of SRP than DOP or PP could be observed from results of measured internal wetland P speciation over multiple years in the Florida Everglades wetlands (Juston and DeBusk, 2011).

Generally, phosphorus enters natural waters through surface and subsurface waters, and atmospheric depositions. Research conducted in the Interlake area of Manitoba during spring time reported the total phosphorus concentration of atmospheric deposition and run-off from non agricultural land at 0.314 mg/L (Green, 1996). Typical background total phosphorus (TP) concentrations in pristine natural wetlands are less than 0.1 mg/L. However, studies showed that they are influenced by nonpoint discharges and are now ranging from 0.1- 0.7 mg/L (Kadlec and Knight, 1996). Many research studies have shown that the storage and release of phosphorus in the water column are governed by physical, chemical and biological processes (Bostrom et al., 1988),(Reed and Brown, 1995), (Moshiri, 1993),(Cooper and Findlater, 1990). From soils and litter, those processes are affected by soil texture and composition, pH and redox status, hydrology, and biological activity (Richardson and Vaithiyanathan 2009). For instance, in the Houghton Lake natural wetland, with receiving water containing 3 mg/L TP, the TP concentration declined from 1268 to 1180 mg/kg in 0-5 cm vs. 5-10 cm of depth (Kadlec et al., 2009). Other results suggest that in wetlands, which have been exposed for several years to higher concentrations of phosphorus, TP in organic soil ranges in between 1000 - 2000 mg/kg. A research study in the Florida Everglades, showed a vertical profile of TP that declines with depth (Reddy et al., 1991). Thus, the first layer of soil typically contains the majority of roots and the highest uptake of P. However, that layer exhibits a cycle of root growth, death, and decomposition, therefore, the return flux of biomass available P is also the greatest. In addition, the top layer of soil is also the newest accretion site. Mayer et al (Mayer et al., 1999) reported similar TP results in Canada wetlands that exhibit decline within 40 cm of depth, ranging from TP around 1200 mg/kg on the water-sediment interface to around 700 mg/kg at 40 cm depth.

Phosphorus is often the first nutrient to limit biological productivity in freshwater aquatic ecosystem (Kadlec et al., 2009), with even small increases in P levels having a significant effect on plant and algal productivity and progress leading to eutrophication (Chambers, 2001). Since gaseous release in the form of phosphine (PH_3) has been found in ultra-trace amounts and it is considered less common (Devai and DeLaune 1995), phosphorus is either building up in the wetland soil or is leaving the wetland through surface outflows or infiltration to groundwater (Vepraskas and Craft, 2016).

1.5 Phosphorus removal mechanisms in wetlands

As mentioned in the previous chapter, P from wastewater entering in a wetland can be retained by physical, chemical and biological processes (Bostrom et al., 1988), (Kadlec et al., 2009). Physical removal of phosphorus occurs through association with suspended solids and in precipitates followed by sedimentation to the bottom of wetlands. The settable and floatable solids may combine with many of soluble and insoluble forms of phosphorus in wetlands (Kadlec et al., 2009). The sedimentation process is a key factor in removal of particulate P during overland flow and floodplain inundation (Hoffmann et al., 2009). Generally, the deposition efficiency of particles is influenced by range of factors, such as, hydraulic load, surface area of wetland, water temperature, as well as, particle size and other sediment characteristics. With water infiltration capacities and porosity being particularly relevant to the latter factor. Additional physical factors in wetlands are the presence of vegetation, their form and stem density, and wind effects; both of which influence particle deposition and phosphorus storage (Kadlec, 1990). These two influences interplay with the velocity and volume of flow which determine the residence time, the time necessary for particle sedimentation and water infiltration. Reinhardt et al. (2005) identified the key factor for dissolved reactive phosphorus (DRP) retention in constructed wetlands as residence time and suggested a minimum water residence time of 7 days to retain DRP by 50 %. However, the turbulence inducing effects, including hydrological events may lead to resuspension and a subsequent release of retained solid phase phosphorus back into solution (Stephan et al., 2010), (Kadlec et al., 2009). Hence, further chemical or biological transformations of phosphorus into stable biological or geochemical compounds are considered necessary for long-term P retention.

Precipitation and sorption processes occur as chemical processes in wetlands, occurring in the soil, sediments, soil pore water and water column (Reddy et al., 1999). Predominantly, precipitation processes occur in the water column where soluble ionic species, such as, soluble iron, aluminum, magnesium or calcium cations bond with soluble phosphate to form an insoluble lattice structure (Baldwin et al., 2002), (Cooke, 1992). Thus, the portion of phosphorus in these precipitates may eventually be removed from the water column through physical processes into sediments. Sorption is the process of a substance adhering to hydrated soil minerals (Kadlec and Knight, 1996). The process varies with parameters of: dynamics of adsorption/desorption (depends on the strength of attraction), adsorption site availability (depends on the particle size), substance abundance and pH of the system (Hoffmann et al., 2009). The relationships between the amount of phosphorus adsorbed per unit of soil and an equilibrium pore water phosphorus concentration at a constant temperature is described by sorption isotherms. A number of different sorption isotherms have been proposed, including Langmuir and Freundlich isotherms proposed as a predictor in treatment wetlands design (Kadlec et al., 2009). The P adsorption isotherms are useful in comparisons of different soil types and to determine the following indicators of P retention/release capacity: the equilibrium P concentration (EPC_0), maximum sorbed concentration of soils (S_{max}) for binding P and the P binding energy (k) (Meissner et al., 2008). A general comparison of the P sorption capacity among different soil types indicates that wetlands vary greatly in terms of P that can be removed, and shows that wetlands with mineral soils, specifically with high concentrations of Al and Fe, have the highest adsorption capacity (Richardson and Vaithiyanathan, 2009). However, isotherm based sorption capacities for P in soils predicted under laboratory settings using batch incubation experiments were much lower than the field based sorption capacities, ascribing to slow mineralization, fixing or precipitation of phosphate and regeneration of P sorption sites with alternate wetting and drying of soils during loading under field conditions (Lin and Banin, 2006). Many papers in the literature address the EPC_0 at which concentration P sorption equals P desorption stating that if water at the EPC_0 is added to the soil, there is neither sorption nor desorption (Hoffmann et al., 2009). If the water has a higher solute concentration of P than the EPC_0 adsorption will occur and the soil will serve as a sink, whereas, water with lower concentrations of P than the EPC_0 results in desorption and the soil will release P (Kadlec et al., 2009), (Hoffmann et al., 2009). Other authors use the degree of P saturation (DPS) of the sediments to estimate the potential of P leaching or potential of P loss from soil to water which is based on specific sequential analysis of soil (Litaor et al., 2003), (Akinremi et al., 2007). Numerous studies have shown that in wetlands with acidic to neutral soil conditions (pH 4-7), phosphorus in the forms of $H_2PO_4^-, HPO_4^{2-}$ exits the water column by adsorbing mainly on crystalline or amorphous hydrous oxides of iron and aluminium in forms of insoluble iron- phosphates (Fe-P) or aluminium- phosphates (Al-P) (Richardson, 1985), (Moore and Reddy, 1994), (Schlichting et al., 2002), (Litaor et al., 2004). These studies demonstrated that the amount of P that soil can absorb in a wetland is positively correlated with the amount of amorphous Fe and Al oxides. It has also been found that the type of solid also plays a role in the sorption capacity for P with Williams et al. (1971) discovering that crystalline Fe and Al oxides having a much lower sorption capacity. The redox reactions in wetland systems play a role in the regulation of soluble P concentrations due to the flooding and/or dry-out cycles in natural wetlands. Several studies have reported increased P and Fe concentrations in soil solution following the changes from aerobic (dry) to anaerobic (wet) conditions (KR 1994), (Pant et al., 2002), (Litaor et al., 2005), (Meissner et al., 2008). Furthermore, in wetlands under lower pH (< 4) and reduced conditions (which may occur during flooding or by an elevated groundwater table), the Fe and Al hydroxides dissolve, and the P bonded to them can be released back into the soil solution (Lindsey, 1979). Flooding events in wetland soil with readily available organic matter cause anaerobic conditions, further alter the oxidation-reduction conditions and the soil pH, and thus directly or indirectly influence the solubility and sorption/desorption of P (Racz, 2006), (Reddy et al., 1998).However, there are studies that have shown little correlation between redox potential and P release (Khalid et al., 1977), (Vadas and Sims, 1998). When wetland soil is in reduced conditions and undergoes drainage with subsequent soil oxidation, three processes can generally occur: precipitation of phosphates, re-adsorption of released P onto remaining non-reduced Fe hydoxides, or adsorption onto redox stable components, such as, calcium carbonate ($CaCO_3$), phyllosilicate clay, and Al hydroxides may also occur (Darke and Walbridge, 2000), (Murray and Hesterberg, 2006).

Thomas et al. (2006) found that although P was released during dry out and rewetting conditions in wetland microcosms, the regrowth of microbial communities led to the reabsorption of that phosphorus within a matter of days. Kadlec et al. (2009) showed results in a stormwater treatment area, STA6, at the Florida Everglades treatment wetland where the release of phosphorus occurred while rehydration was occurring following a dry out period. Pulse P release happened when waters first entered the system, however the release of P declined as the wetland refilled, prior to any outflow. Working on batch-filling mode, and a fill time of about one month, treatment wetlands exhibited P retention. The Florida Everglades wetland is comprised of soil under alkaline conditions, pH > 7.0, where phosphate precipitation occurs resulting in insoluble compounds such as Ca-P or Mg-P and these have been found to be the dominant forms of P transformation and retention.

Another important process in wetlands is phosphorus co-precipitation governed by photosynthesis and respiration, where carbon dioxide depletion initiates changes by increasing the pH in the water column which affects the solubility of compounds (Tadesse et al., 2004). Algae can uptake dissolved CO_2 (as carbonic acid) faster than it can be replaced by atmospheric compensation, leading to an increase of pH. These processes are important mechanisms for P removal and retention in alkaline wetlands (Qualls and Richardson, 1995). For example, at pH 8.3 when the saturation point of calcium carbonate ($CaCO_3$) is reached, its formation simultaneously provides a sorption site for phosphorus. Depending on the further pH increase and buffering capacity of the water column, a significant portion of dissolved phosphorus can precipitate as calcium phosphate (Reddy et al., 1999). The P sorption processes are more important during subsurface flow conditions, whereas sedimentation processes may be the major P retention mechanisms during overland flow of wastewater in treatment wetlands (Kadlec and Knight, 1996).

Phosphorus is retained biologically in wetlands through uptake by microbial communities and plants (Johnston, 1993), (Kadlec et al., 2009). Numerous studies have measured phosphorus removal in wetlands by plant assimilation while receiving treated sewage water (Spangler et al., 1977), (Prentki et al., 1978) ,(Dolan et al., 1981), (Kadlec and Bevis, 2009), (Herskowitz, 1986), (Nolte and Associates, 1998). A range of 0.1- 0.64 % phosphorus concentration in live tissue plants, assessed in 41 marshes, was reported by Bedford et al.(1999). It was also found that live plant tissue in a discharge area exhibited higher P concentration than an area not receiving discharge (Kadlec et al., 2009). In addition, the live tissues of wetland plants have higher P concentrations than their dead counterparts. It has also been observed that with an increase in phosphorus availability in wetlands, such as through the addition of wastewater, there is an evident increase in the standing crop (Meulemana et al., 2002). Moreover, measurements revealed large differences in the phosphorus content in different parts of the plants based on the growing season and climate (Bayly and O'Neill, 1972). One part of P storage in plants is translocation of phosphorus to the belowground plant rhizomes during the late summer to early autumn period. However, studies revealed different results on the location of P storage, depending on vegetation type as well as growth conditions (Kadlec et al., 2009). In Manitoba, under low nutrient status marshes, measured phosphorus translocation for mixed stands of vegetation ranged from 29-36 % (Murkin et al., 2000). Whereas, treatment wetlands with higher P loading (TP load of 340 kg P/ha/year) result in higher percent of phosphorus (e.g. 50 %) translocated (Meulemana et al., 2002). Overall, phosphorus storage in plants, shows a repeatable pattern with an increase of storage during the growing season leading to maximum storage and a decrease in storage through the senescence season resulting in minimum storage. During the senescence period, the decomposition and leaching of dead material back into the water column occurs, with residual accreting as new sediment and soils (Kadlec et al., 2009). One alternative remedy for excess phosphorus release has been reported in carefully managing the growing season and harvesting the aboveground plant tissue (Herskowitz, 1986), (Grosshans, 2014). For example, studies conducted on harvesting stands of *Typha* spp. in a natural wetland in Manitoba showed that the P content was 0.32% of dry weight, and the removal potential through an annual harvest was 3 to 6 grams of P/m^2/yr or 30-60 kg P /ha/yr (Cicek et al., 2006). Kadlec and Bevis (2009) reported vegetation and associated biota uptake, in treatment wetlands, being as high as 7.7 g/m^2/yr in irrigation areas, resulting in 14% of the added phosphorus sequestered in a new, larger standing crop during their study. Generally, within treatment wetlands the aboveground harvestable portion of phosphorus ranges from 1 to 5 $g/m^2/yr$ (Kadlec aet al., 2009). Being a sustainable phosphorus removal mechanism in wetlands receiving treated wastewater, the burial of P in the form of new accretion of sediment/peat and soils has been reported as a dominant removal mechanism in a 30 year long study of a full-scale operation of a marsh-wetland treating wastewater (Kadlec, 2009b), (Kadlec et al., 2009). The non-decomposable fractions of dead microflora and microfauna, remnant macrophyte stem and leaf debris, and remnants of dead roots and rhyzomes are constituents of the new sediment (Kadlec et al., 2009). In addition, as a new accretion of sediment the non-decomposable fractions immobilise organically bound phosphorus, acting as barrier to nutrient release (Tanner et al., 1998). Studies indicate that as the distance from the inflow of a wastewater increases, the accretion rate decreases (Reddy et al., 1993), (Kadlec, 2009b). Kadlec (2009b) measured the accumulation rate in the discharge zone of a wastewater treatment marsh-wetland to be 2 cm/yr, where as, in the broader area outside of the inflow the rate was 1.33 cm/yr. This rate has been found to vary based on the location and type of wetlands, considering other reported values, however consistent between the studies is a decrease in the accretion rate as measurements are taken farther away from the inflow zone. For example, in a study on the Everglades, Florida, the storm water runoff treatment wetlands, Reddy et al. (1993) reported an accretion rate of sediment at 1.1 cm/yr within the inflow area, which decreased away from the inflow area to 0.25 cm/yr. Mustafa (Mustafa and Scholz, 2011) reported a very high sediment accretion rate of 6.4 cm/yr in the first cell of an integrated constructed wetland treating agricultural wastewater. The P storage in sediments depends on the vegetation type, with rates reported higher in an area with *Typha spp.* with rates ranging from 0.54 - 1.14 g $P/m^2/yr$ in the Everglades, Florida wetlands study (Reddy et al., 1993) where as in northern temperate wetlands, Kadlec (2009b) reported average rate of 1.5 g P/m^2/yr. Murkin et al.(2000) estimated the burial flux as an annual accretion rate ranging from 0.73 to 0.95 g P/m^2/yr in low nutrient natural wetlands with mixed vegetation in Manitoba.

To summarize, phosphorus is sequestered in treatment wetlands by the microbial community, sorption reactions and vegetation. Among theses compartments, phosphorus moves and mostly resides in wetland soil in the form of refractory phosphorus compounds and buried phosphorus. The initial compartments have a limited capacity for storage of phosphorus and can become saturated, consequently the additional loads of phosphorus move further into the wetland. These additions create a moving front of phosphorus saturation and other ecological effects involving larger wetland area of treatment. At locations, farther from inlets, the rate of phosphorus loss to the final compartment (such as soil accretion) can equal the rate of phosphorus supply (Lindsey, 1997). Thus, the background concentrations will continue to prevail. This theory is found to be relevant for Lake Houghton treatment wetland where the ecological effects on vegetation change due to phosphorus input diminish in the direction of water flow away from the point of wastewater discharge. Research conducted for three decades of this natural wetland's ability to seasonally remove P from treated lagoon wastewater (Kadlec, 2009b) revealed that the majority (94 %) of P was removed by P storage in new biomass, new increased soil sorption, and the accretion of new soils and sediments. The transformation and retention of phosphorus by natural wetlands are important processes regulating nutrient transport to surface waters, however the prediction of P attenuation is extremely challenging because of the aforementioned complex processes and their interactions.

1.6 Phosphorus discharge regulations

Phosphorus is considered a pollutant in the context of adverse environmental and ecological impact. It can originate from a point source, such as wastewater lagoon effluents, and can cause increase of a water body's primary productivity due to nutrient enrichment causing eutrophication (Chambers, 2001). Many studies indicated that phosphorus inputs directly control algal blooms in lakes and recommend reduction of phosphorus loadings to reduce harmful algal blooms (Schindler et al., 2012), (IJC, 2014). In Manitoba, the discharge of phosphorus and other nutrients into the environment is presently regulated according to two principal environmental bylaws:

- the Environmental Act 1987-1988;
- the Water Protection Act 2015;
    - Manitoba Water Quality standards, Objectives and Guidelines 2011;
    - the Nutrient Management Regulation 2008

The Environmental Act is the principal pollution control statute in Manitoba and is used in conjunction with the Water Protection Act to address point sources of pollution. The statutes have been used to regulate the operation of municipal wastewater systems; the discharge limits of phosphorus are explicitly regulated within Manitoba Water Quality standards, Objectives and Guidelines under the Water Protection Act (Water Science and Management Branch, 2011). All wastewater facilities discharging more then 820 kg TP per year or new or expanding facilities discharging less then 820 kg TP per year are required to meet the 1 mg/L phosphorus discharge limit as of January 1, 2016. Others needs to demonstrate nutrient reduction strategy, which will be evaluated on a site-specific basis. In addition, the Environmental Act regulates the discharge of polluting material, including phosphorus, in or near water and mandates the requirement of regulatory approvals for all sewage works. Nutrient Management Regulations, among others, encourage responsible nutrient planning and regulate or prohibit wastewater facilities in areas where water bodies or groundwater are sensitive to its impact; and regulate applications to land of materials containing phosphorus. Using the Manitoba Conservation Public Registry database, one can be observed that there are at least five approved wastewater treatment systems in rural municipalities that utilize natural or constructed wetlands or passive filter treatment as alternative for P reduction under the Environment Act (Government of Manitoba, 2016). For example, population of Village of Dunnottar in Manitoba utilize a wetlands treatment system (Anderson et al., 2015).

1.7 Phosphorus removal predictions in natural wetlands

Among the most used methods to predict wetland treatment performance are models using a concentration reduction percentage (CRP) or the areal phosphorus load reduction method (Kadlec, 2016). The CRP method does not reflect the flow rate the wetland receives as the areal P load reduction model does. (Kadlec, 2016). Natural treatment wetlands include the same compartments as constructed wetlands: an inlet distribution system, the wetland basin, the natural vegetation, sediments and sometimes outlet design features (Kadlec and Knight, 1996). Generally, it can result in the same design as used for the design of the free water surface (FWS) wetlands (Kadlec and Knight, 1996). Treatment behaviour in wetlands is often considered to be a figurative black-box based on input and output data (Rousseau et al., 2004) Conventional chemical reactor principles have been used as one of the methodologies to design treatment wetlands (Jamieson et al., 2007). It assumes that the concentration profile of P along the transect of the wetland decreases exponentially to non-zero background wetland concentrations (C^*). Thus, the rate of P removal is then calculated assuming a first order plug flow system which incorporates hydraulic conditions (Kadlec and Knight, 1996) as follows:

$$\ln \frac{C_0 - C^*}{C_i - C^*} = -\frac{ky}{q} \qquad 1.1$$

Where k is the areal removal rate constant (m/day), *Co* is the outlet concentration (mg/l), C^* is the background wetland concentration (mg/l), *Ci* is the inlet concentration (mg/l), q is the hydraulic loading rate (HLR) (m/day) and y is the fractional distance from inlet to outlet (y=x/L where x is point in distance length; L is wetland (distance) length, for example y = 1.0 at the outlet). It also assumes that the system is mature, i.e. the inflow is steady and the influent concentration of phosphorus is uniform (Kadlec and Knight, 1996). In addition, the model assumes that no groundwater or surface water infiltrates into the wetland and that there is no atmospheric phosphorus deposition, or in another words, it quantifies for the situation of no significant water gains or losses. The model further assumes a rectangular shaped wetland cell with no spatial variation in time-averaged flow (Sikdar, 2008). The first order model is widely accepted as the preferred design approach (Braskerud, 2002), (Rousseau et.al., 2004), even if it doesn't account for the complex reactions and interactions that occur in wetlands. Insufficient and variable data have been a barrier to an accurate determination of k. Hydraulic loading information is rarely presented along with the performance data, making estimations of k difficult. The importance of areal rate coefficients is the forecasting of the longitudinal variation in P concentrations, which relates hydraulic loading with the distance (Kadlec, 2016). Kadlec and Knight (1996) were able to do a comparison of several SSF constructed wetlands and determined k to be 11.7 (± 4.2) m/yr. A study by Tanner et al. (1998) determined a k of 8.8 m/yr in a gravel based SSF wetland. Braskerud (2002) determined a *k* of 214 m/yr for treatment of agricultural wastes. The significant variation of *k* shows the problem associated with modeling phosphorus removal in wetlands. Variation stems from differences in design which includes sizing and media, environmental conditions, data presentation and dynamics of phosphorus transformations (Mitchell et al., 1995). However, being a highly complex ecosystem, designing treatment wetlands requires multi-disciplinary input including wetland hydrology that sometimes requires an understanding beyond the site-scale (Baker et al., 2009), (Tilton and Kadlec, 1979). Therefore, several authors have proposed more sophisticated wetland performance models, which simulate non-ideal hydraulics, either using a tanks in series (TIS) model or plug flow with dispersion (PFD) approach, and going into more complex computer models in order to predict fate and the transport of phosphorus in vertical movement wetlands (Landgrabber et al., 2009, Landgrabber and Simunek, 2005).

As aforementioned, predicting a treatment wetlands effectiveness in retaining phosphorus requires a fundamental understanding of the factors that influence solute transport including hydraulic factors such as water flow path and residence time as well as an understanding of all water storages. The basic mass balance equation, change in storage = inputs – outputs, is used to express the hydrologic processes in a wetland and is often referred to as a water budget (Owen, 1995). In a natural wetland, there is a great deal of uncertainty over the hydrologic budgets. A number of studies have identified the hydrological pathways and fluxes operating within natural wetlands. A study by Kadlec (Kadlec, 1983b) in the south basin of Lake Manitoba identifies hydrological fluxes, namely: precipitation, surface water inflow, groundwater inflow, evapotranspiration, surface water outflow and groundwater outflow; and calculated the annual water budget. This study revealed that the interaction between groundwater and surface water was the most difficult flux to estimate since it was relatively small in comparison to other components of mass water balance. In a natural wetland, there is a great deal of uncertainty over the hydrologic budgets, whereas in a constructed wetland, although the uncertainty still exists, it is reduced due to a better control of some of the components. Large variations in storage are possible due to the strong daily and seasonal difference in evapotranspiration. Similarly, precipitation usually occurs with seasonal cycles during summer months. As aforementioned the hydraulic loading rate (HLR) and the hydraulic residence time (HRT) are common parameter to consider in designing a treatment wetland (Kadlec and Knight 1996). The HLR, or *q* is defined as “the rainfall equivalent of whatever flow is under consideration” (Kadlec et al., 2009). The HRT in wetlands is the average amount of time that is taken for water and conservative solutes to move through the wetland (Kadlec and Knight 1996). The ideal loading rate found to ensure a wetland performs as a phosphorous sink over the long-term is between 1-5 g/m^2, or 10-50 kg/ha, of phosphorous per year (Mitsch, 1992), (Mitsch et al., 1995), (Mitsch and Gosselink,2015). Many studies determined HRT in natural wetlands by conducting tracer tests (Headley and Kadlec, 2007), (Stern et al., 2001), (Hayward et al., 2014). Furthermore, some used chloride ion concentration of the stream inflows and outflows in estimation of the water budget in addition to measurements of electrical conductivity based on their difference in wastewater and wetlands incoming waters (Kadlec, 1983a). Generally, better treatment performance is achievable within a wetland with a longer HRT, as well as with lower hydraulic loading rates (Blahnik and Day, 2000). However, Know et al. (2008) did not find significant relationships between hydraulic loading rate, total phosphorus loads and percent retention.

1.8 Research objectives

This research was motivated by the growing awareness of water and sanitation security issues in the First Nations Communities in Manitoba, in part due to the Report that provides an inventory and a risk assessment of these systems. The broad objective of this research is to contribute to improving sanitation conditions in First Nations communities and to better understand the potential role of passive wastewater treatment, moreover the potential role of natural wetlands in reducing phosphorus discharge levels.

Specific objectives of this research are:

1. To investigate First Nations lagoon systems in Manitoba through a review of available literature and to select an appropriate location to conduct a field study.
2. To conduct research to evaluate:
    a. the phosphorus mass load entering into a natural treatment wetland following discharge of treated secondary wastewater
    b. the phosphorus mass balance by comparing concentrations in the inflow and outflow to assess the P concentration removal efficiency of secondary treated wastewater through the natural wetland during vegetation growing season
    c. the phosphorus content in soils to understand water movement and how total phosphorus concentrations in soils are impacted by receiving secondary treated effluents during vegetation growing season.

CHAPTER 2 - MATERIAL AND METHODS

2.1 Study site

The investigation of lagoon systems and the examination of site topography among First Nations communities was initially done using the Report of inventory of water and wastewater systems in Manitoba. After preliminary investigation and with Chief and Band Council approval, the wastewater treatment facility and the receiving wetland in the Lake Manitoba First nation community was chosen. The Lake Manitoba First Nation community (50°54’30” N, 98°35’50” W) is located on the northeast shore of the south basin of Lake Manitoba, and is bordered by the Rural Municipality of Eriksdale and the Rural Municipality of Siglunes. The community has a total population of 680, with an average growth rate of 2.04 % between 2006 and 2011 (Statistics Canada, 2016). However, P. Swan, the Lake Manitoba First Nation community`s wastewater operator describes that the population in the community has almost doubled in the last 4 years, (P. Swan, personal communication, May 20, 2016). At the study site, long-term climate data are lacking; however, climate data was recorded in Vogar, a neighboring community, 5 km northwest of the Lake Manitoba First Nation community. In the past 30-year period the daily average temperature was 2.5°C with the daily maximum reaching 7.2°C and the daily minimum being -2.2°C. The average annual precipitation consists of 387 mm as rainfall and 159 mm as snowfall, which account for 502.9 mm of total precipitation (Government of Canada, 2015). The total precipitation in the town of Fisher Branch, which is 70 km East of the Lake Manitoba First Nation community, was reported as 381.4 mm for 2015. In addition, during the study period, from May to October 2015, an average total precipitation of 285.0 mm was determined from recordings made in Fisher Branch.

The population of the Lake Manitoba First Nation community is served by a 41-year-old wastewater treatment facility (labelled "Dog Creek 46"), which is facultative lagoon comprised of two cells. It was constructed in 1975, and was designed to serve the school and teachers residence facilities (an estimated 12 homes piped) of approximately 60 people. Designed flow capacity was 53 m^3/d, yet today, according to personal communication with a community wastewater operator, the lagoon receives a flow of 198 m^3/d. As designed, the surface area of the primary lagoon cell at the full supply level (f.s.l.) of 824 ft is 5261 m^2, and the surface area of the secondary cell at its f.s.l. is 8094 m^2. Today, the primary cell also receives wastewater directly from sewage trucks. Consequently, the retention time and flow capacity within the primary cell is not well defined.

The secondary cell effluent is to be discharged once per year after treatment results for a number of parameters are determined to fall within the acceptable range of the wastewater system effluent regulations (CG, 2012). During the last two years, the capacity of the secondary lagoon cell has been overloaded, resulting in the effluent being discharged twice a year (P. Swan, personal communication, May 7th, 2015). Originally, the secondary cell's effluent was naturally drained by swale onto a marsh (wetland) area of approximately 51 ha that connects to Lake Manitoba. The newly developed pond and an earthen embankment constructed in 2014 divided the treatment wetland area into two areas, labelled as: area 1 and area 2 (Figure 2.1).

Figure 2.1. A modified aerial photo of the Lake Manitoba First Nation community wastewater treatment facility and the wetland that receives effluent from the treatment facility.

The secondary cell effluent is discharged onto a swale, approximately 38 m from the secondary cell through a solid PVC sewer drain pipe, with a diameter of 20 cm. It is then naturally drained through culvert A (diameter 33 cm) onto Area 1 of the wetland. Area 1 with a surface area approximately 3000 m^2 is separated from the rest of the wetland (Area 2) by an earthen embankment on the south, west and east sides. Culvert B located on the southwestern side of Area 1 connects it to Area 2 allowing water to flow from Area 1 into Area 2. Area 2 which has a surface area of approximately 16000 m^2, is closed on its western side by a newly developed pond, while the southern and eastern sides are closed with earthen embankments. The combination of embankments and the pond hydrologically isolates the treatment area from rest of the wetland. In the late August (Aug. 24th) the Lake Manitoba First Nations community constructed a culvert C under the southern earthen embankment where water exited the treatment wetland.

The topography of the wetland treatment area (Areas 1 and 2) in the Lake Manitoba First Nation community is characterized as flat sloping downward from west to south within Area 1, changing to a very flat sloping topography that slopes downward from north to south in area 2. The elevation at the wetland inlet (outlet of culvert A) denoted by the circle Culvert A icon in Figure 2.1 is 2260 mm relative to embankment height. The elevation of the wetland outlet (inlet of culvert C) denoted by the circle Culvert C icon in Figure 2.1 is 2771 mm. The straight-line distance between (A) and (C) is 120 m, which yields a site slope of approximately 0.6%.

The vegetation in the southern portion of Area 1. were found to be predominately by meadow grasses and bulrush (*Spartina Pectinata*, *Elymus Canadensis*, *Scirpus spp.*) with small scattered areas of cattails (*Typha spp.*). In contrast, the southern part of Area 2., a narrow zone approximately 20 m wide, was dominated by cattails (*Typha spp.*), while the rest of the area was dominated by meadow grasses and sedges. In addition to the recording the present vegetation, the soil in both areas was characterized as the a Meleb series marsh with shallow soils (Goverment of Canada, 2013). The Meleb series is represented by carbonated phase soils developed on very strongly to extremely calcareous, stony loam to clay glacial till. It can also consist of thin peat layer between of 0 to 45 cm depth underlain by textures ranging from loam to clay till. In addition to the Meleb series, different soil type named Berm soil type was established in the immediate vicinity of the new earthen embankments, found in both the inlet and outlet of both wetland areas.

2.2 Sampling period

Sampling trips were conducted considering the growing season of vegetation. The initial sampling trip occurred in May 2015 before the growing season and involved background soil and water sampling in the wetland area. The wastewater flow measurements took place during the discharge period in June, 2015. During the discharging of the lagoon effluent, water samples were taken from wetland Area 1 and 2, as well as from an adjacent non-effluent receiving wetland as a background control. Final soil and water sampling to evaluate treatment performance was conducted at the end of the vegetation growing season in October, 2015.

2.3 Wastewater flow measurements

Single point velocity measurements were taken along a cross section of the outflow of culvert A and at the inflow of culvert B to determine the mean flow rate as a measurement of the inflow and outflow, respectively, to the marsh (Figure 2.1). Single point velocity measurements were taken using a FlowTracker handheld Acoustic Doppler Velocimeter (ADV) with 2D side looking probe Sontek™. The velocity distribution was determined at each culvert by taking velocity measurements in the cross section at a depth equivalent to 60% of the total depth from the water surface. The position of the FlowTracker's probe was oriented so the axis of the transmit transducer was roughly perpendicular to the direction of flow. Measurements were taken at each culvert twice a day during the discharge. The mean flow rate was then calculated by multiplying the area of the cross section by its mean velocity.

2.4 Soil and water sampling

Ten sites, in total, were chosen along a transect along both wetland areas based on the flow path of historic wastewater discharge from the secondary cell onto natural swale including the newly developed earthen embankments as depicted in the Figure 2.2 into consideration. The geographical location coordinates, including others, are provided in the Appendix 2.A. Table I. Water samples were taken using a 5 m long telescopic sampling pole with an attached polypropylene bottle and transferred into acid washed polypropylene bottle which were pre-rinsed with sample water from all ten sites. After which, at the eight locations, the soil was sampled at three depth intervals of 0-20, 20-40 and 40 -60 cm using a dutch auger. For comparison purposes, additional samples were obtained adjacent to the wetland as a control (CR), representing water samples and soil that had not received treated wastewater loading, as well as water samples from newly developed ponds (P1 and P2), following the same procedure. During the discharge period, in wetland Area 2, every day at certain times (reported in the Appendices 3), wetland water sampling was conducted following the same procedures at specific sampling stations (Figure 2.2).The sites are coded as B0, L30, L60, L90, W30, W60, W90 shown in Figure 2.2, where L refers to east-west direction towards the Lake Manitoba and W refers to south-north direction towards the woods, and the numerical values refer to distance in meters from culvert B which was the inflow for wetland treatment Area 2.

Figure 2.2. An aerial photo of the sampling locations in wetland area and adjacent wetland (taken in April, 2015).

2.5 Analytical Methods

Background wetland area water samples and the water treatment performance samples were transported in insulated coolers with ice packs to the laboratory, stored in a refrigerator at 4 °C upon arrival at the laboratory and analyzed within 24 hours. The dissolved oxygen (DO) was measured in situ at each sampling location by a Thermo Scientific Orion 3 star DO meter and accompanying probe. Each water sample was tested for an array of parameters including: pH, conductivity, TP, SRP, chemical oxygen demand (COD), and TSS. A pH/conductivity meter (Accumet AP85, Fisher Scientific, Singapore) was used to measure pH and conductivity. The TP samples were analyzed using the Hach® TP Test ‘N Tube™ and processed according to the procedure described in Hach for low range samples (ranging from 0.06 to 3.50 mg/L PO_4^{3-}),whereas high range samples (ranging from 1.0 to 100.0 mg/LPO_4^{3-}) (HACH, 2014a) were analysed following the procedure in the high range Hach (HACH, 2014b). SRP and TSS were measured according to standard methods (APHA, 1998). The COD was analyzed using he Hach®COD Digestion Vials ULR (HACH, 2014c).

Soil samples were air-dried and ground using a pulveriser to pass through a 2-mm sieve. TP were extracted from the soil samples using a wet oxidation method (Akinremi et al., 2003). TP concentrations were measured in the digested solution using a colorimetric method (Murphy and Riley, 1962).

2.6 Data analysis

For each water quality parameter tested, the minimum, maximum and average values are provided, as well as a Box and Whisker graph for the total phosphorus (TP) concentrations to more easily denote important patterns. The TP mass loading (kg/day) was calculated by multiplying the average TP concentration (mg/L) in the inflow by the average inflow rate (m^3/day) at the inlet of treatment wetland Area 2. The instantaneous TP mass loading (kg/year) was calculated by multiplying the TP mass loading (kg/day) by duration of the discharge per year (day/year). The treatment performance of the wetland was calculated based on concentration reduction percentage as the percent (%) reduction for TP using Equation 2.1. as follows:

$$\%[TP]\ reduction = \frac{(C_{outlet} - C_{inlet})}{C_{inlet}} * 100 \qquad 2.1.$$

Where:

C_{inlet} – average TP concentration at the inlet of treatment wetland Area 2 (mg/L)

C_{outlet}- average TP concentration at the outlet of treatment wetland Area 2 (mg/L)

The TP data in soil samples were analysed using the GLIMMIX procedures performing a repeated measures analysis of variance in SAS (SAS Institute, 2012.). The TP concentrations in the soil samples were not normally distributed based on the Shapiro-Wilk test and were natural log-transformed prior to performing the variance analysis. The TP concentrations in soil before and after discharge treated wastewater, among different soil types, and different depths were compared using the Tukey multiple comparison procedure. Changes were considered significant if P < 0.10.

CHAPTER 3 - RESULTS AND DISCUSSION

3.1 Selection of study site

Recently, work of Watchorn et al. (2012) delineated wetlands around the coast of Manitoba's Great Lakes following a classification system developed by the Great Lakes Coastal Wetlands Consortium (Albert et al., 2005) based on hydrogeomorphic classes. ArcMap, the geographical information system (GIS) tool was used to overlay the layer of delineated coastal wetlands (Watchorn et al., 2012) with the layer of First Nations communities in Manitoba (NRC, 2016), in order to better represent the location of First Nations communities which are situated on the coasts of the Lakes Winnipeg, Manitoba and Winnipegosis (Figure 3.1). There are 19 First Nations communities located in the coastal areas with the coastal areas being classified as coastal wetlands, barrier-protected or lacustrine wetlands.

In regions of Manitoba, four other First Nations communities are located in high and low subarctic wetland regions with another 19 First Nations communities situated in high and mid boreal wetland regions. Therefore, at least 42 community locations are potentially suitable for carrying out wetlands research study, which amounts to 67 % of the total 63 First Nations communities in Manitoba (Watchorn et al., 2012), (Halsey et al., 1997).

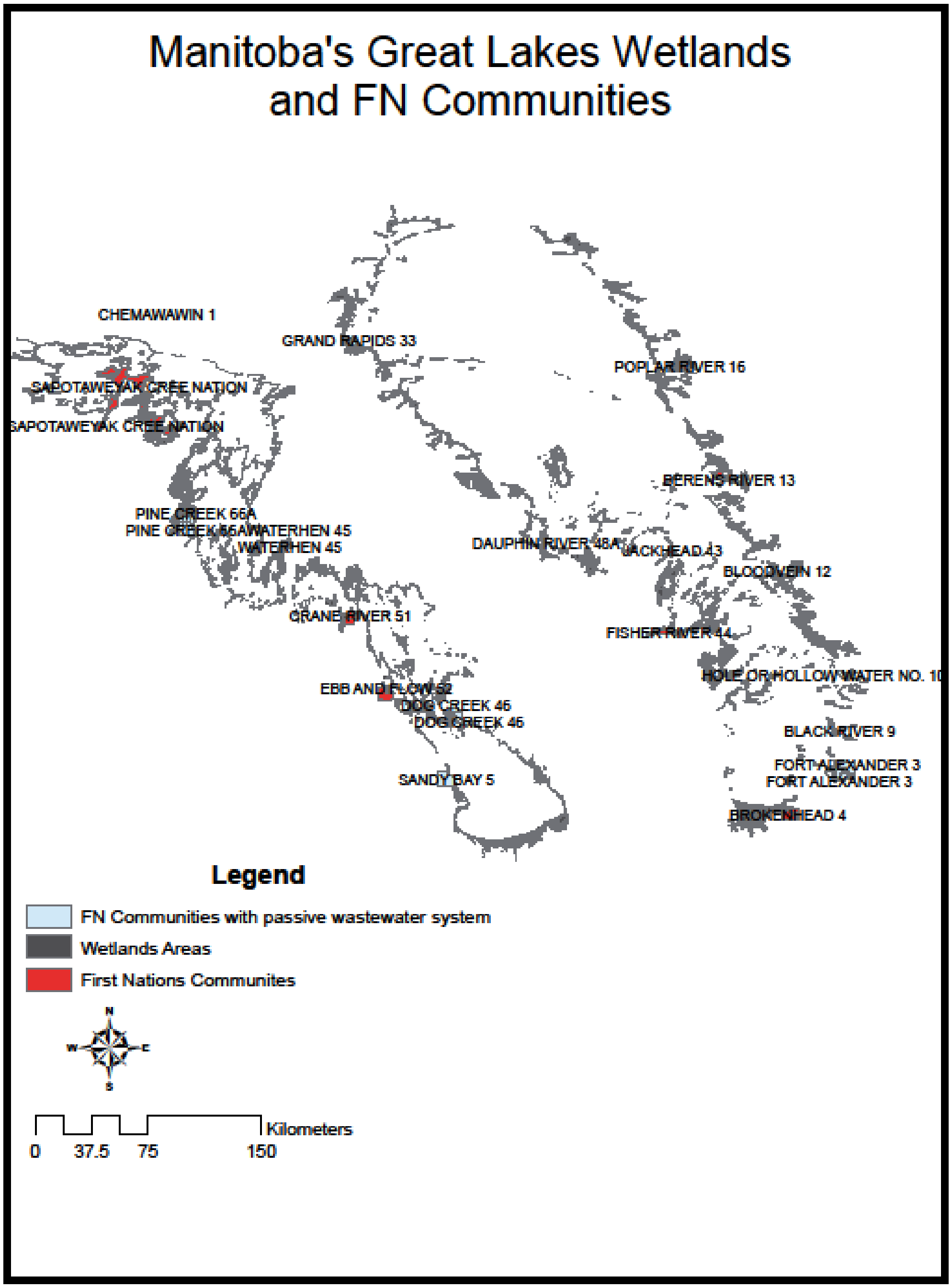


Figure 3.1. Location of First Nations (FN) communities on Manitoba's Great Lakes shores with wetlands areas delineated.

According to the Report, the national inventories of water and wastewater systems in First Nations communities, there are 12 First Nations communities in Manitoba with lagoon systems that use natural wetlands as an effluent receiving body (INAC, 2011b). Moreover, eight of these communities treat their domestic wastewater passively using facultative lagoons adjacent to a natural wetland (Table 3.1).

Table 3.1. Names and other characteristics of the eight First Nations communities that utilize passive wastewater treatment system in Manitoba. (Data taken from Report (INAC, 2011b)).

Name of community	*Wastewater treatment system*	*Receiver*	*Design Capacity/ Max daily flow(m^3/day)*	*Discharge season*	*Sludge Treatment*
Hollow Water First Nation	Facultative lagoon	Wetland	176/437	Spring/fall	No
Lake Manitoba First Nation	Facultative lagoon	Wetland	53/22	Spring/fall	No
Little Saskatchewan First Nation	Facultative lagoon	Wetland	-	Spring/fall	No
O-Pipon-Na-Piwin Cree Nation (South Indian Lake)	Facultative lagoon	Wetland	627/132	Spring/fall	No
Sandy Bay First Nation	Facultative lagoon	Wetland	2242/1599	Spring/fall	No
York Factory First Nation	Facultative lagoon	Wetland	364/153	Spring/fall	No
Pinaymootang First Nation	Facultative lagoon	Wetland	467/-	Spring	No
Pine Creek First Nation	Facultative lagoon	Wetland	71/-	Spring	No

Alberta regulations and its manual for obtaining an approval for a treatment wetland, mentioned in introduction, among others, emphasis on wetland treatment applications in First Nations communities as viable additional effluent treatment alternative. The document also highlights additional benefits of using wetlands as an additional treatment, such as employment opportunities and wildlife habitat (AEP, 2013). Similar regulations have also been introduced in British Columbia for First Nations communities (Government of Canada, 2013).

Following an inventory of First Nations water and wastewater systems in Manitoba (INAC, 2011b), conducting preliminary site investigations among several First Nations communities and obtaining the Chief and Band Council approval, the Lake Manitoba First Nation community's wetland/lagoon treatment system was chosen in order to examine the objectives of this research.

3.2 Background wetland water quality

Summaries of wetland water quality parameters from the treatment wetland (Area 1 and Area 2), control wetland and values from the literature wetlands are provided in Tables 3.2., 3.3. and 3.4., respectively (raw data are reported in Appendix 3.A. Table I.). The pH of the surface water within the treatment wetlands were alkaline, ranging from 7.66 to 8.74. The pH in the control wetland was also alkaline with similar values to the treatment wetlands (Table 3.2. and 3.3). Salinity, measured as electrical conductivity in the treatment wetland ranged from 560 μS/cm to 1650 μS/cm which places it into the classifies of a slightly brackish wetlands type (AESRD, 2015). The salinity in the control wetland was similar to the treatment wetland with the values falling within the range measured in the treatment wetland (Table 3.2. and 3.3). A comparison of the conductivity values measured in this study with literature wetlands show much higher values from the control and treatment wetlands from this study which indicates a greater concentration of total dissolved solids (TDS) in the area (Table 3.2. and 3.4). The higher concentration of TDS could be attributed to it being a historical discharge area. The wide range of conductivity values along the treatment area can be attributed to the varying range of water depths, as shallow areas may cause the back-diffusion of salts from the concentrated pore water (Kadlec et al., 2009). The electrical conductivity measurements are also affected by biological conditions in wetlands. Hayward (2013) observed changes in the electrical conductivity in a tundra treatment wetland between seasons; a higher average temperature (11°C) during mid - summer increased metabolism of ionic species within the wetland due to algal growth and potentially increased microbial activity resulting in decreased conductivity values. The effect of dilution, driven by rainfall or runoff, and evaporation may also result in substantial changes in conductivity measurements (Cooke et al., 1990).

Table 3.2. Summary of water quality parameters over the wetland treatment area measured at the beginning of the vegetation growing season in May, 2015.

Parameter	*Average*	*Max*	*Min*	*Standard Deviation (SD)*	*Number of samples*
TP (mg/L)	0.58	1.08	0.20	0.25	10
Electrical Conductivity (µS/cm)	918	1650	560	329	11
COD(mg/L)	47	52	36	5	10
Temperature (°C)	8.4	9.8	7.5	0.7	11
DO (mg/L)	10.65	12.97	7.45	1.60	11
pH	8.14	8.74	7.66	0.26	11
TSS (mg/L)	43	92	6	29	11
Water depth (m)	0.12	0.16	0.08	0.03	11

Table 3.3. Summary of water quality parameters over the control wetland measured at the beginning of vegetation growing season in May, 2015

Parameter	*Average*	*Max*	*Min*	*Standard Deviation (SD)*	*Number of samples*
TP (mg/L)	0.09	0.10	0.07	0.02	2
Electrical Conductivity (µS/cm)	1016	1153	878	195	2
COD (mg/L)	50	51	48	2	2
Temperature (°C)	8.1	8.6	7.6	0.7	2
DO (mg/L)	11.22	11.24	11.20	0.03	2
pH	7.65	7.75	7.55	0.14	2
TSS (mg/L)	41	48	34	9.90	2
Water depth (m)	0.21	0.23	0.18	0.04	2

Table 3.4 Summary of water quality parameters in natural wetlands from the literature

Parameter	*Literature values ± SD from Louisiana natural wetland (Zhang et al., 2000)*	*Literature values ± SD from Manitoba natural wetland (Anderson et al., 2013)*
TP (mg/L)	0.34 ± 0.28	0.030 ± 0.002
Electrical Conductivity (µS/cm)	113 ± 21	420
Temperature (°C)	NA	14.8
DO (mg/L)	3.3 ± 1.9	7.4
pH	7.60 ± 0.52	NA
TSS (mg/L)	97.2 ± 100	8.2 ± 0.8
Water depth (m)	0.33 ± 0.12	0.40 – 0.60

The total phosphorus (TP) concentration in the treatment wetlands was relatively high with concentrations ranging from 0.20 mg/L to 1.08 mg/L with an average concentration of 0.58 mg/L (Table 3.2). According to Manitoba Water Quality Standards, Objectives and Guidelines (Water Science and Management Branch, 2011) TP >0.025 mg/L in any reservoir, lake, or pond, or in a tributary at the point where it enters such bodies of water, may contribute to the nuisance growth of algae. A similar observation can be made for the TP in the control wetland water (Table 3.3) where the measured TP was lower in comparison with the treatment areas and the natural wetland from Louisiana (Zhang et al., 2000), yet was higher in comparison with a natural wetlands in other parts of Manitoba (Anderson et al., 2013), (Table 3.4). The average TP in the treatment wetland areas is higher than the TP reported for both the Louisiana wetland (Zhang et al., 2000), and the Manitoba study (Anderson et al., 2013). These differences can be explained in two ways. Firstly, the treatment areas bear legacy phosphorus (Kadlec et al., 2009); these treatment wetlands have received at least 40 treated wastewater discharge loads in the past which may have resulted in long term P storage in the sediment (Kadlec, 2009b). Taking into account the climatic conditions of the day prior to sampling and the day of water sampling (May 07, 2015) may also provide some insights into the elevated TP concentrations. With sampling day being windy and rainy (total of 28.2 mm precipitation for Fisher Branch, MB) (Government of Canada, 2015), which can result in some runoff, and resuspension of sediments, as well as changes in redox conditions, due to water inundation, the release of P from the soil could have contributed to elevated concentrations of TP. Several studies have reported increased P concentrations in soil solution following changes from aerobic (dry) to anaerobic (wet) conditions (Moore and Reddy, 1994), (Pant et al, 2002), (Litaor et al., 2005), (Meissner et al., 2008), (Reddy et al. 1998). Secondly, the elevated values of TP in these treatment areas may indicate possible leaking of wastewater from the lagoon outflow through a discharge pipe equipped with gate valve. Similar observations and speculation was reported in a treatment wetland of Riding Mountain National Park, Manitoba (Belke and McGinn, 2003). A decrease in the TP concentration with an increasing the distance away from the lagoon site along the wetland supports both assumptions (Figure 3.2).

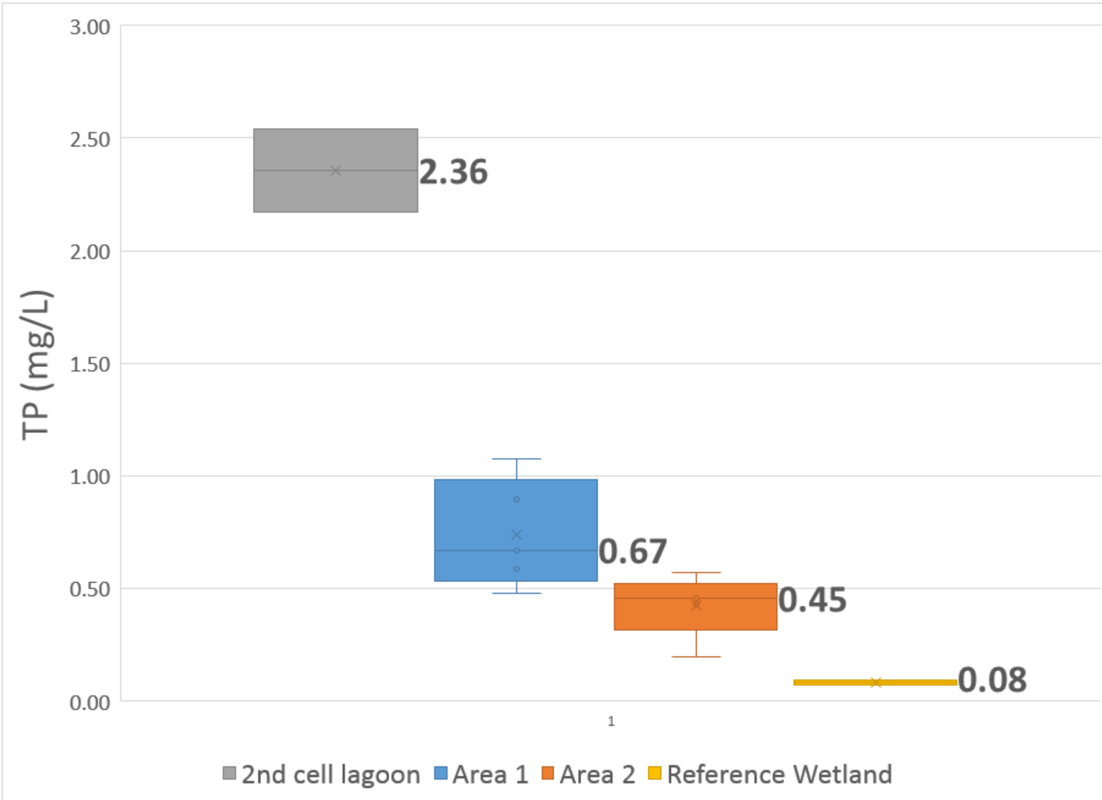


Figure 3.2 A Box and Whisker Plot of the total phosphorus (TP) concentrations measured in the lagoon and along the wetland (areas 1 and 2) in May, 2015. The plot displays the dissipation of TP from the lagoon through area 1, area 2 and control wetland. The average TP for each is provided beside each box and indicated by the vertical line within each box, with the maximum and minimum values represented by the ends of the whiskers, the x inside the box denotes median.

3.3 Discharge of secondary treated wastewater

A summary of the treatment performance of the secondary lagoon cell at Lake Manitoba First Nations community from 2014 is presented in Table 3.5. An environmental compliance approval is required for almost all point discharges of water and wastewater into surface waters in Canada (CCME, 2009) under the Fisheries Act. They define the compliance limits of specific wastewater constituents that have to be met before issuing approval to discharge. The data in the Table 3.5 show that un-ionized ammonia (NH_3-N) and total phosphorus (TP) concentrations are higher than values proposed by regulation whereas carbonaceous biochemical oxidation demand (cBOD5), total suspended solids (TSS) and Total Coliforms in the treated secondary lagoon wastewater samples were in compliance with regulatory limits.

The carbonaceous biochemical oxidation demand ($cBOD_5$), total suspended solids (TSS) and Total Coliforms in the treated secondary lagoon wastewater samples were in compliance with regulatory limits in 2015 discharge (P.Swan, personal communication, June 17th, 2015).

Table 3.5. A summary of treated wastewater quality results from Lake Manitoba First Nations community lagoon, 2014 in comparison to regulative values established by two Acts (Data courtesy of NWETS Ltd -CRTP Serving Manitoba First Nation Communities-Ken Mattes, personal communication, Oct 30th, 2014).

Parameter	*Regulative maximum results*	*Lake Manitoba First Nations results*
$CBOD_5$ (mg/L)	25*	2
Total Coliforms (MPN/ 100mL)	1500*	430
TSS (mg/L)	25*	<4
NH_3-N (mg/L)	1.25*	3.8
TP (mg/L)	1**	2.05

*Wastewater Systems Effluent Regulations under Fisheries Act (CG, 2012)

**Manitoba Water Quality standards, Objectives and Guidelines (Water Science and Management Branch, 2011)

In June,2015 there was no visible water (surface water) in the treatment wetland Area 1, however it did have saturated sediments, whereas in Area 2 there was visible water along the southern part of the basin, a narrow zone, approximately 20 m wide, with a measured level of 0.09 m of water. The discharge of the secondary lagoon commenced on June 17th, 2015 at 11.50

AM and finished on July 19th at 00:38 AM. The water depth in the secondary lagoon was recorded by measuring the water level on the measurement rod stacked in the bottom of the lagoon, next to the discharge pipe. The time when the discharge period ended was computed based on the 28 hours needed to discharge the 0.64 m height of the secondary lagoon wastewater plus hours needed for 0.20 m diameter of discharge pipe. The total discharge period was 36 hours and 48 minutes. The velocity of the treated wastewater at culvert A was difficult to measure because of high turbulence, where as, the measured mean velocity at culvert B was 0.276±0.019 m/s during the discharge period which resulted in an average flow of 2823±197 m^3/day. Considering the original surface area of the secondary cell was 8094 m^2 and the depth of the discharged lagoon was measured at 0.84 m, this resulted in a total discharge volume of 6783 m^3 into the natural treatment wetland areas. The total volume that entered in Area 2 was 4330 m^3. Considering the measured elevation points, earthen embankments and observed distance of discharged volume from the inlet, the wetland treatment Area 2 was estimated as 1.26 ha. Considering the size of the treatment area and the average treated wastewater flow, the hydraulic loading rate was calculated as 22 cm/day. The total phosphorus (TP) at the upstream/downstream of the culvert B, which represented the inlet of the treatment Area 2, had an average concentration of 1.74 mg/L which resulted in the total phosphorus mass loading of 5 kg/day (raw data are reported in Appendices 3.C). Thus, considering that treated wastewater discharge occurred once in 2015, the instantaneous phosphorus loading was 8 kg/year or the instantaneous areal loading rate was 0.6 g TP /m^2/year during the vegetation growing period. In Alberta, the recommended hydraulic loading rate for natural treatment wetland systems is 3 cm/day (AEP, 2013), which is similar to the recommended value of 2.5 cm/day for polishing the treated effluents (Kadlec and Knight, 1996). On the other hand, Mitsch and Gosselink (2015) summarised loading rates from many surface flow constructed wetland studies treating small municipalities wastewater reporting an average loading rate of 5.4 cm/day. Thus, the treatment wetland Area 2 in Lake Manitoba FN operated under a high hydraulic loading rate with relatively low P loadings. Other parameters measured during the discharge period in the lagoon, at the inlet of both treatment areas, the control site and adjacent ponds are summarized in Table 3.6.

Table 3.6. The average ± SD of the water quality parameters measured from the Lake Manitoba First Nations community lagoon inside 2nd cell and at the outflow (Lagoon) and in wetland treatment areas (Area 1 and 2) together with results from control wetland (Control wetland) and adjacent ponds (Pond 1 and 2) during the discharge period in 2015 (raw data are reported in Appendices 3.C)

Mean ± SD	*Lagoon*	*Area 1*	*Area 2*	*Control wetland*	*Pond 1*	*Pond 2*
TP (mg/L)	1.67 ± 0.28	1.74 ± 0.08	1.59 ± 0.37	0.03 ± 0.01	0.48 ± 0.00	0.60 ± 0.20
Electrical Conductivity, EC (µS/cm)	939 ± 53	974 ± 91	1562 ± 1144	1482 ± 308	1365 ± 0	1903 ± 233
COD (mg/L)	50 ± 2	49 ± 1	49 ± 2	50 ± 1	51 ± 0	50 ± 2
Temperature (°C)	21 ± 2	24 ± 1	24 ± 1	21 ± 2	na	na
pH	8.4 ± 0.6	8.4 ± 0.4	8.1 ± 0.4	7.9 ± 0.3	8.7 ± 0	8.1 ± 0.1
TSS (mg/L)	15 ± 18	47 ± 52	15 ± 2	24 ± 0	62 ± 0	na

SD – standard deviation; *na – not measured*

Results of total suspended solids (TSS) of wastewater in the lagoon showed compliance with the wastewater system effluent regulations (CG, 2012) where as, the total phosphorus (TP) concentration exceeded the limits of TP concentrations as specified in the Water Protection Act (Water Science and Management Branch, 2011). The pH of the surface water during the treated wastewater discharge period within the treatment wetlands was alkaline averaging 8.4 and 8.1 in Areas 1 and 2, respectively. The pH measured in a control wetland was also alkaline with an average of 7.9. The pH from the treatment wetland areas and the control wetland were similar to those measured in May. During the discharge period the salinity increased due to the addition of dissolved salts and other dissolved solids from the treated wastewater. However, the average salinity for both treatment areas were still within the range to indicate the same salinity type, (slightly brackish wetlands) (AESRD, 2015) as prior to the discharge period. A comparison of the results between the May sampling, representing background conditions in treatment and control wetland, and during the discharge period in June, does not indicate changes in chemical oxidation demand (COD) concentrations. The similar pattern as observed within the background wetland water quality results can be drawn from dissipation of results for total phosphorus in wetland treatment Area 2 indicating consequences of long term P storage, built up in the sediments during, at least the last 40 discharge loads combined with dry out and rewetting conditions. Higher values of TP and salinity in both ponds in comparison to the lagoon and the treatment areas, may indicate wetland hydrological conditions characterized by lateral groundwater flow that in the process of flowing into the ponds may dissolve salts and transport them into these ponds (Nachshon et al., 2013). Consequently, considering the greater concentration of TP in surrounding soils, under specific conditions, higher TP concentrations could be expected in these ponds. The observed TP concentration in pond 2 was 0.46 mg/L after 11 hours' post-discharge which was slightly greater than the reported TP concentration of 0.31 mg/L in a research study conducted in the nearby Interlake area of Manitoba from non-agricultural land (Green, 1996).

In order to observe and better understand the treatment wetland Area 2, the trends of P speciation profiles were presented 11 hours post-discharged (Figure 3.3 and 3.4). The internal trends of orthophosphates and remaining forms of P forms (which consisted of total dissolved P minus orthophosphates plus particulate P), demonstrated that orthophosphates were reduced by 61% at a location of 90 m distance from inlet discharge into Area 2 to east-west direction toward Lake Manitoba (Figure 3.3). This orthophosphate reduction can be related to rapid conversion of orthophosphates through biological processes utilized by microorganisms, algae and macropytes (Fisheries and Oceans Canada, 2015), (Richardson and Vaithiyanathan, 2009). Furthermore, the orthophosphate reduction or conversion is consistent with observations from other studies, including those by (Zhang et al., 2000) and (Juston and DeBusk, 2011). Although the higher TP concentrations were found at sampling locations of 30 m (L30) and 60 m (L60) wetland length in comparison with the initial value (B0), the final location at 90 m (L90) from inlet discharge point, the TP was reduced for 33%, which was greater than Zhang et al. (Zhang et al., 2000) reported in their 100 m long transect study of a swamp. Different removal values of TP and orthophosphates coincided with other observed studies (Herskowitz, 1986), (Kadlec, 2016).

With an average TP inflow concentration of 1.74 mg/L, 11 hours of residence time provided a TP positive removal efficiency of 12 %, 5 %, 41 % at initial (inlet), 0.6 (60 m) and 0.9 (90 m) fractional distances from inlet, respectively. Whereas a location at 30 m from inlet or 0.3 fractional distance exhibit TP negative removal efficiency of -2 % , indicating a TP release. Overall, the reduction of TP concentrations in the water column may indicate that P removal occurs through sedimentation and bio-accretion which is known to be spatially non-uniform (Kadlec, 2016).

Observations in orthophosphates concentrations at sampling points towards the woods did not demonstrate similar orthophosphate patterns, but rather exhibit TP releases along the fractional wetland distance (Figure 3.4).

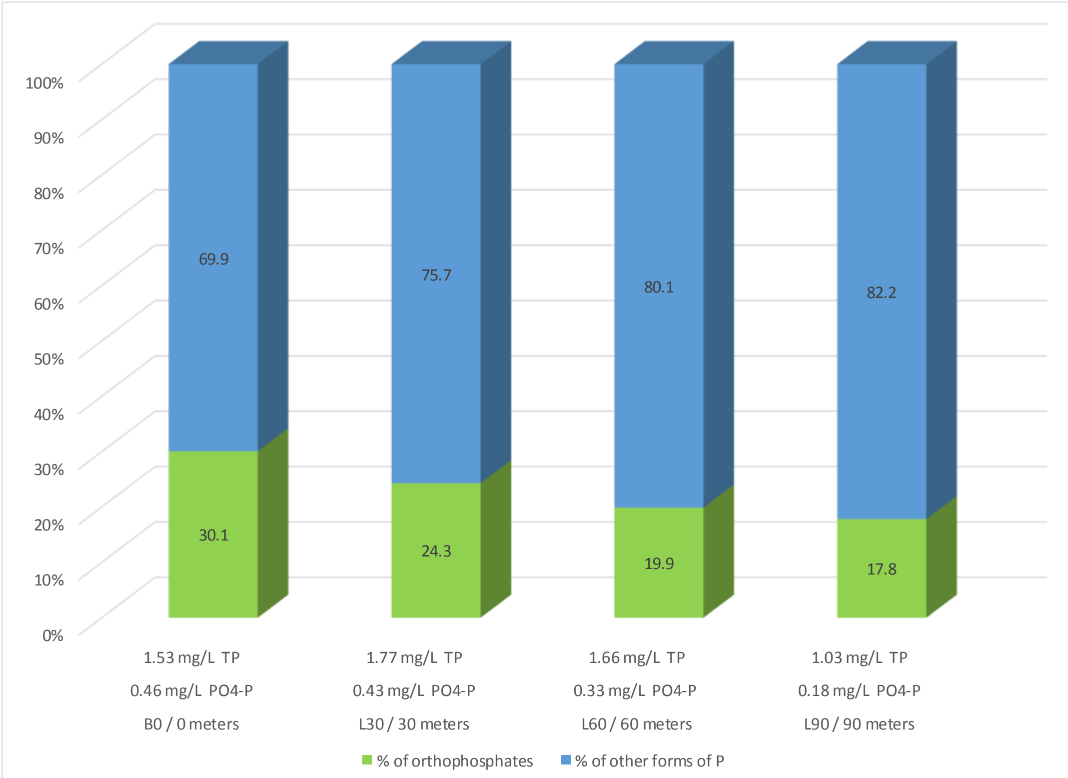


Figure 3.3 P speciation within fractional distance to east-west direction toward Lake Manitoba (B0, L30, L60, L90 were sampling stations; L refers to an east-west direction toward the lake and the numerical values refer to the distance in meters from culvert B)

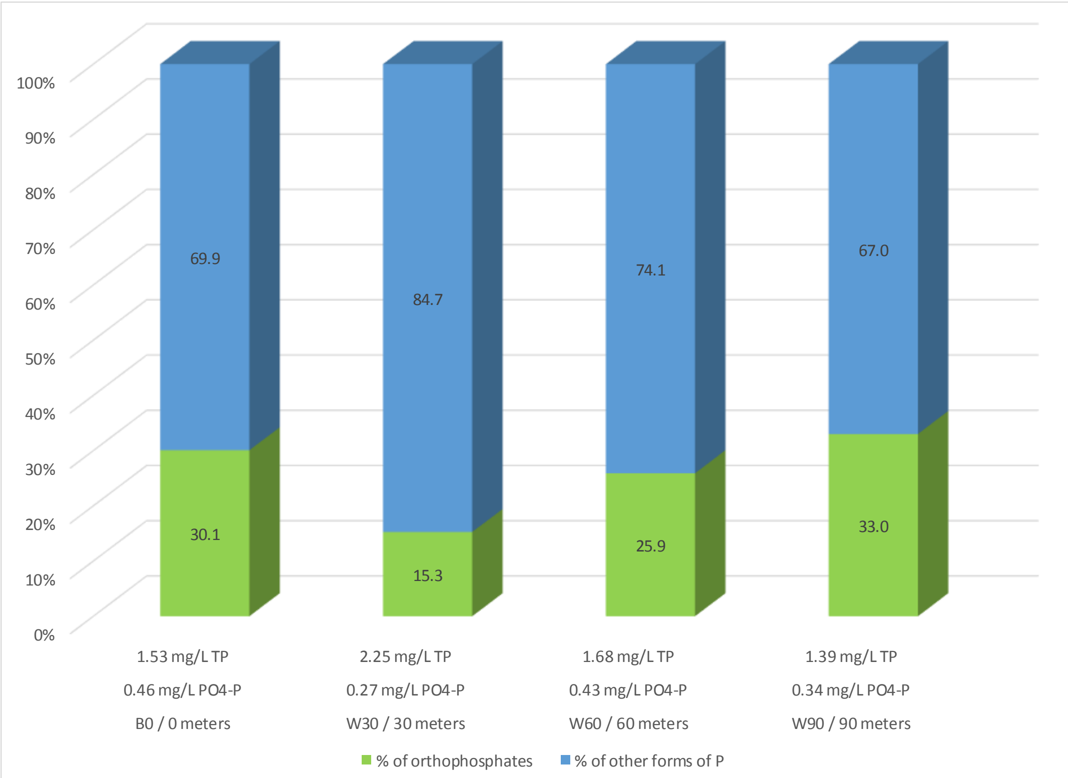


Figure 3.4 P speciation within fractional distance to south-north direction toward the woods (B0, W30, W60, W90 were sampling stations; W refers to a south-north direction toward the woods and the numerical values refer to distance in meters from the culvert B)

During surveying of the wetland Area 2, differences between vegetation types and elevation points were observed among the southern portion, the east-west side and the rest of the northern area. Where the southern part is more uniform, consisting of cattails along the east west pathway, the northern direction was dominated by meadow grasses and sedges except for where the old swale used to be (approximately 40 m north from culvert C) that consisted of cattails. These observations may explain the observed difference among P speciation between the east-west and south-north direction of sampling stations (Kadlec et al., 2009). Considering the higher PP forms removal rates then TP removal as demonstrated in a wetland study Apopka, Florida, the decomposition of settled PP forms may release dissolved forms, such as orthophosphates, back to the water column (Dunne et al., 2015) which may explain the spatial increment of orthophosphates in the water column. Furthermore, at the northern transection there was no surface water as observed in the southern part before the discharge began. Thus, at specific locations the re-flooding could impede conditions favourable to release of dissolved P forms back to water column. However, there are studies that have shown that when wetland soil are in reduced conditions and undergo drainage with subsequent soil oxidation, there is an overall removal of P from the pore water due to the following processes: precipitation of phosphates, re-adsorption of released P onto remaining a non reduced Fe hydoxides, or adsorption onto redox stable components, such as, calcium carbonate ($CaCO_3$), phyllosilicate clay, and Al hydroxides may also occur (Darke and Walbridge, 2000) , (Murray and Hesterberg, 2006).

3.4 Treatment performance

The length of growing season for the location of Lake Manitoba First Nations is determined to be between 140 and 150 days by Climate Change Scenario: 2010 – 2039 for agricultural purposes (AAFC, 2014). The growing season is defined as "the portion of the year when soil temperatures at 19.7 in. below the soil surface are higher than biologic zero (5°C)" (U.S. Army Corps of Engineers, 1987). The growing season can be approximated by the number of frost-free days. The natural wetland treatment performance of secondary treated wastewater for TP concentration during 123 days of vegetation growth was 78 % (Raw data reported in Appendices 3.C.). The wetland performed treatment under high hydraulic loading rate of 22.2 cm/day with low instantaneous P loadings of 8 kg/year occupying area of 12600 m^2 resulting in instantaneous loading rate of 0.6 g TP /m^2/ year. During the final sampling trip October, 19th 2015, the treatment wetland area did not have visible water however had saturated sediments, whereas at the wetland outflow, downstream of culvert C the average water depth was 12 cm and average TP concentration was 0.38 ± 0.14 mg/L. The relatively high standard deviation of TP results can be explained by observed non-uniform vegetation in the area of culvert C downstream and by the newly developed soil layer as per implementation of the culvert. Natural wetlands have been used in many locations for polishing secondary effluent with different removal efficiency reported (Table 3.7). Phosphorus removal has been particularly challenging in wetland systems with widely variable removal efficiencies and loading rates that are reported for the design and operation of a treatment wetland.

Table 3.7. A summary of reference studies and their removal of phosphorus from wastewaters together with its type, size and loading rate applied to natural wetlands.

Location	*Wetland type*	*Size, ha*	*Total Phosphorus Loading rate, g/m²/y*	*Total Phosphorus Retention, %*	*Reference*
Houghton Lake	Marsh	100	1.76	94	(Kaldec, 2009a)
Wisconsin, USA	Marsh	156	15.2	32	(Spangler et al., 1977)
New Zealand	Marsh	NA	34	30	(Cooke, 1994)
Florida, USA	Marsh/Swamp	204	0.9	87	(Boyt et al., 1977)
Australia	Marsh/Swamp	60	3.8	94	(Patruno and Russell 1994)
Louisiana, USA	Swamp	231	2.45	66	(Zhang et al., 2000)

The general rule of a thumb for wetlands intercepting non-point source pollution with low phosphorus loadings, suggest that such wetlands can consistently retain phosphorus in the amount of 0.5 – 5 g P/m^2/year (Mitsch and Gosselink, 2015). This commensurate with an estimated average accretion rate of 0.84 g P/m^2/year in low nutrient natural wetlands with mixed vegetation in Manitoba (Murkin et.al, 2000). Extended research conducted for three decades in a seasonally lightly loaded natural wetland Houghton Lake (Kadlec, 2009b), as well as, short-term studies such as a study in Louisiana (Zhang et al., 2000) and low P loaded marshy/swampy areas in Florida (Boyt et al., 1977) (Table 3.7) fit in these ranges and are in wide agreement with the current study. The natural wetland adjacent to the secondary lagoon in Lake Manitoba First Nation community performed TP removal to a point where the municipal by-law limit of having effluent concentration less of 1 mg TP/L was met (average TP concentration of 0.38 ± 0.14 mg/L at location of downstream culvert C).

3.5 TP in Soil

The effect of discharging treated wastewater into a natural wetland on soil TP concentrations was examined in two soil types. The changes in TP concentrations prior to the discharge of treated wastewater (at the beginning of vegetation growing season) and after discharge (at the end of vegetation growing season) was significant only in the 20-40 cm depth interval for Meleb soil type which is the main soil type in this wetland (Table 3.8). Raw data are reported in Appendix 3.E. Table I.

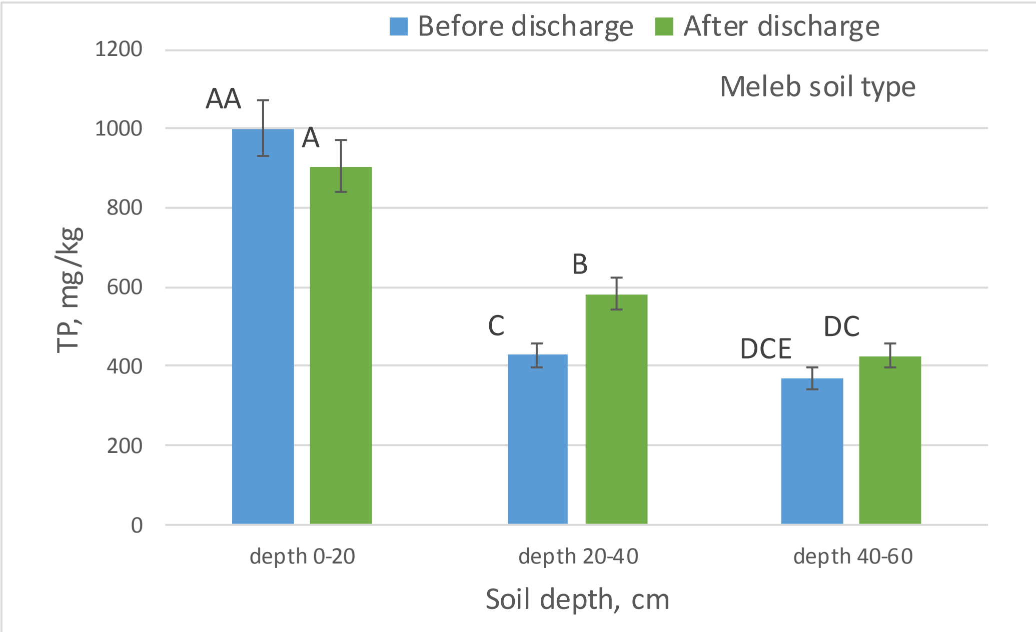
Table 3.8. Effect of discharging treated wastewater (before and after discharge), wetland soil types (Soil 1 – Berm soil and Soil 2 – Meleb series) and depth intervals (0-20 cm, 20-40 cm and 40-60 cm) on soil TP concentration (mg/kg)

Depth intervals	Berm Soil Average soil TP (mg/kg)		Meleb Soil Average soil TP (mg/kg)	
	Before discharge	After discharge	Before discharge	After discharge
0-20 cm	339 DE	404 DCE	1000 AA	902 A
20 - 40 cm	335 E	381 DCE	428 C	583 B
40-60 cm	348 E	394 DCE	370 DCE	426 DC

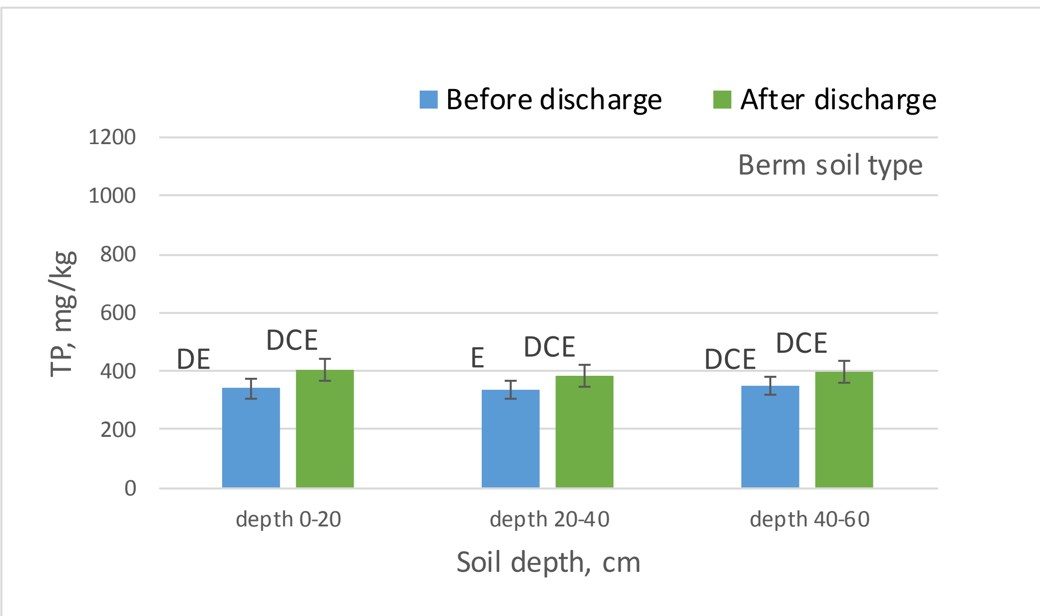
Analysis of variance (ANOVA) applied to ln transformed values	Degrees of freedom	Probability > F value
Discharge effect	8	0.0010
Soil types	8	< 0.0001
Depths	16	< 0.0001
Discharge effect*Soil types	8	0.6172
Discharge effect*Depths	16	0.0635
Soil type*Depths	16	< 0.0001
Discharge effect*Soil types*Depths	16	0.0170

†Average TP values followed by the same uppercase letter are not significantly different

At the 20-40 cm soil interval for the Meleb soil, the TP concentration was significantly higher at the end of the vegetation growing season after the discharge of treated wastewater compared to the initial TP concentration measured prior to the beginning of the vegetation growing season. In contrast, TP in the deeper layers (40-60 cm depth interval), of both soil types did not exhibit a significant difference in TP concentrations (Figure 3.5, Table 3.8). For the Meleb soil, at the end of growing season, the difference in TP concentrations among the depth intervals 0 - 20 cm and 20 - 40 cm was significant with TP concentrations decreasing in the former layer and increasing in latter layer. Thus, storage of phosphorus occurred in the layer 20-40 cm of Meleb series soil. This results may indicate that vertical water movement occurred until 40 cm depth interval with no further vertical water movement deeper in either soil types. During the field work and laboratory analysis it was observed that samples from the depth intervals 40 - 60 cm were clayic. Similar notation of a clayic layer was observed by Boyt et al. (1977) in their study of a natural wetland treatment system and assumed that it prevented the vertical movement of water. TP results in the soil profiles in this study, were generally higher in the surface layer and decreased with depth (Figure 3.5) which agrees with observations at other treatment wetlands (Kadlec, 2009b), (Reddy et al., 1993), (Craft and Richardson, 1993).



A



B

Figure 3.5. TP concentrations in two different soil types: (A) Meleb soil type and (B) Berm soil type within different depth intervals (0-20 cm, 20-40 cm and 40-60 cm) before and after discharge of treated wastewater (Error bars represent standard error of the means. Bars with the same letter are not significantly different ($P > 0.10$) per the Tukey-Kramer multiple comparison test)

The concentrations of phosphorus varied with depth for the Meleb series soil (Table 3.8). The 0-20 cm depth interval had relatively high concentrations of 1000 ± 148 mg TP/kg soil, before vegetation growing season started. This concentration was similar to the TP concentration (1043 ± 117 mg/kg) measured at a control site next to the newly developed pond (P2) (Figure 2.2). The corresponding concentrations decreased with depth (20 - 40 cm) to 428 ± 124, and 483 ± 17 mg/kg for the Meleb soil in the treatment wetland and P2, respectively. Approximately 400 m from the inlet to treatment wetland, at a control wetland closer to Lake Manitoba (CR) (Figure 2.2), lower TP concentration (539 ±17 mg /kg soil) was measured in a soil depth interval 0-20 cm, and the higher TP concentrations was measured for of 524 ±62 and 424± 61 mg /kg soil in depth intervals 20-40 and 40-60 cm, respectively. The lower TP values at the control wetland could be attributed to the difference in soil types and the distance from the inlet to the treatment wetland. Similar observations have been reported in other treatment wetlands where phosphorus values decreased within distance from inflow to the treatment wetland (Kadlec, 2009b), (Reddy et al., 1993).

After the discharge of treated wastewater and at the end of vegetation growing season, the TP values at the control site next to the newly developed pond (P2), were 1207 ± 93 and 453 ± 84 mg TP /kg soil at depth intervals 0-20 cm and 20 -40cm, respectively. In comparison, the TP in depth interval 0-20 cm of Meleb series soil in treatment wetland, was 902 ± 158 mg TP /kg soil which was relatively low compared to the values at control point next to the newly developed pond (P2). Whereas, relatively high TP values were measured in the soil depth interval 20-40 cm, at the treatment area averaging 583 ± 114 mg TP /kg Meleb series soil comparable to the same depth values in P2 (453 ± 84 mg /kg soil).

This short-term (pre and post vegetation growing season) TP soil measurements in the natural treatment wetland showed a downward phosphorus moving front, which represented phosphorus accumulation in depth interval 20-40 cm of Meleb soil. This could be attributed to the carbonated rich soil type overlying limestone bedrock (AFFC, 2013) which may indicate P sorption as the dominant phosphorus retention process. Furthermore, considering that at least 40 years of TP loads discharged into the treatment area, and the potential of new solid accretion at rates of 1.3 cm/year (Kadlec 2009) would result in approximately 50 cm of new soil layer could have accumulated. This soil layer may serve as a long-term sink for phosphorus, as it may consist of the indecomposable fractions which may immobilise organically bound phosphorus, acting as barrier to phosphorus release (Tanner et al., 1998). This could serve as a sustainable phosphorus removal mechanism in wetlands receiving treated wastewater (Kadlec, 2016).

CHAPTER 4 - CONCLUSIONS

A research project to study a natural wetland as additional wastewater treatment for phosphorus removal in First Nations communities in Manitoba was undertaken following an inventory of First Nations water and wastewater systems in Manitoba, conducting preliminary site investigations among several First Nations communities and obtaining the Chief and Band Council approval of Lake Manitoba First Nation community to examine the objectives of the research. The following conclusions are drawn:

1. Creating the ArcGIS map by combining a map of delineated coastal wetlands (Watchorn et al., 2012) with a map of First Nations communities in Manitoba (NRC, 2016) revealed 19 communities that are located on the coasts of Lake Winnipeg, Manitoba and Winnipegosis in the zones of those wetlands (Figure 3.1). In addition, four other First Nations communities are located in high and low subarctic wetland regions with another 19 First Nations communities located in high and mid boreal wetland regions. Therefore, at least 67 % of the total 63 First Nations communities in Manitoba could be suitable for further investigation of using wetlands as an effluent-polishing step in removing phosphorus from wastewater.
2. A natural treatment wetland in the Lake Manitoba First Nation community operated as a polishing step of lagoon treated wastewater under a high hydraulic loading rates of 22.2 cm/day and with low instantaneous P loading rates of 8 kg/year. The wetland occupied an area of 12600 m^2 an provided a TP removal efficiency of 78 % during vegetation growing season of 123 days. The by-law limit of 1 mg TP/L was met with average

concentration of 0.38 mg/L. The information gathered in this study suggest that adjacent natural wetlands to lagoons could be suitable as an effluent polishing step to satisfy regulatory requirements for phosphorus reduction in smaller First Nations communities despite high hydraulic loading rates and short vegetation growing season.

3. Statistical analysis of soils total phosphorus (TP) concentrations revealed that the effect of discharging lagoon treated wastewater into the natural wetland was significant only in the 20-40 cm for Meleb soil type. This comparison was made on the bases of analysing TP before discharge of treated wastewater (at the beginning of vegetation growing season) and after the discharge (at the end of vegetation growing season) at different soil depth layers. The accumulation of TP was observed in 20-40 cm soil depth interval, suggesting that P saturation of this layer could result in a net TP release from the wetland in the long term.

CHAPTER 5 - LIMITATIONS AND RECOMMENDATIONS FOR FUTURE WORK

1. A TP concentration reduction in this study was calculated based on concentration reduction percentage which does not reflect on the dynamic flow rate through the wetland. Implementing outflow settings for flow measurements, such as specific V-notch weirs and stilling wells with recorders, would enable better understanding of wetland removal efficiency over time (such as estimating phosphorus load removed (PLR)) and would provide better interpretation of observed operating data (Herskowitz, 1986), (Kadlec, 2016). Furthermore, the TP concentration reduction in this study was calculated for a wetland without significant water gains or losses, assuming that the discharge volume was the only water input in the area. A common method to assess overall TP input and output is to measure and/or compute other possible water storages such as groundwater recharge or discharge, precipitation and evapotranspiration (J. A. Kadlec 1983b). Quantifying total phosphorus (TP), dissolved phosphorus (DP), particulate phosphorus (PP), dissolved organic phosphorus (DOP), and orthophosphates (SRP) in those storages is recommended. It would enable to investigate the internal P speciation profile and to understand interconversions that may occur.
2. As a receiver body for treated wastewater over the last 40 years, the natural wetland in this study bears legacy phosphorus. Nevertheless, within this short-term study, a TP concentration reduction of secondary wastewater was observed. Determination of phosphorus pools in soil samples are recommended to determine the P adsorption characteristics, and to characterize indicators of soil P retention/release capacity

(Meissner et al., 2008). This would allow for the estimation of P saturation of a wetland and potential release of P from the wetland into the environment.

3. During this study the vegetation in the Lake Manitoba First Nations community's natural wetland adjacent to the wastewater lagoon was not harvested. Several studies indicated that carefully managing the growing season and harvesting the aboveground plant tissue may contribute to the overall removal of phosphorus (Grosshans, 2014, Cicek et al., 2006), (Herskowitz, 1986). Recommended harvesting could permanently remove P from the wetland system and the harvested biomass could be used as a substrate to produce biofuel pellets (Vaičekonytė et al., 2013/2014) or potentially as a substrate in anaerobic digester for the production of biogas (Herskowitz, 1986) and may eventually serve as a revenue source for First Nations communities. Further studies should determine P accumulation in aboveground plant tissue to investigate the practice of harvesting wetland plants and its beneficial use as feedstock for energy production while permanently removing and enhancing overall phosphorus reduction from treatment wetlands.
4. To get a complete picture of wetland treatment performance, an assessment of water residence time is recommended. Although nominal retention time can be calculated based on theoretical equations that do not take into consideration dead zones and/or preferential flow. The common method of assessing retention time in natural wetland is to conduct a hydraulic tracer study to determine a residence time distribution curve, showing distribution of times that parcels of water reside in the wetland (Hayward et al., 2014), (Lin et al., 2003). A tracer study can be conducted in a wetland by injecting a known amount of tracer into the inlet and monitoring tracer concentration at the

outlet, along with inflow and outflow rates (Kadlec and Knight 1996). A tracer study using Rhodamine water tracing dye (Lin et al., 2003) or bromide (Hayward et al., 2014) would be the most efficient for this site. Other tracer studies have used sodium chloride as a more cost-effective method (Chazarenc et al., 2003).

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APPENDICES

Appendix 2.A. Table I. The geographical location coordinates – longitude and latitude (decimal degrees, DD).

Name of location	Longitude (DD)	Latitude (DD)
T1 S4 / SW4	-98.60269	50.90575
T1 S3 / SW3	-98.60267	50.90556
T1 S2 / SW2	-98.60278	50.90544
T1 S1 / SW1	-98.60286	50.90531
T1 S0 / SW0	-98.60292	50.90517
T2 S0 / SW0	-98.60317	50.90514
T2 SW0	-98.60317	50.90514
T2 S1 / SW1	-98.60344	50.90536
T2 S2 / SW2	-98.60347	50.90547
T2 SW3	-98.60361	50.90569
T2 SW4	-98.60364	50.90581
Control (CR)	-98.60767	50.90361
Lagoon 2nd cell	-98.60175	50.90594
Lagoon outfall	-98.60219	50.90589
Culvert A	-98.60242	50.90581
Culvert B	-98.60311	50.90514
Culvert C	-98.60358	50.90481
Pond 1 (P1)	-98.60497	50.90506
Pond 2 (P2)	-98.60444	50.90742
L30	-98.60356	50.90506
L60	-98.60392	50.90494
L90	-98.60436	50.90481
W30	-98.60328	50.90539
W60	-98.60339	50.90567
W90	-98.60428	50.90599

Appendix 3.A. Table I. Water quality parameters of surface water samples sampled over the wetland treatment areas sampled on May 7th, 2015 in between noon and 3.30 PM.

Sampling locations ID	TP (mg/L)		EC (µS/cm)	COD (mg/L)		Temp. (°C)	DO (mg/L)	pH	TSS (mg/L)	*Water depth (cm)*
Area 1 - T1 S0 / SW0	1.03	0.761	646	49.0	49.5	8.0	10.42	8.13	6	10
Area 1 - T1 S1 / SW1	0.46	0.50	945	40.6	42.4	7.8	12.97	8.18	48	16
Area 1 - T1 S2 / SW2	0.58	0.78	649	48.9	49.6	7.8	10.90	8.13	64	14
Area 1 - T1 S3 / SW3	1.06	1.09	658	49.8	49.1	7.5	11.39	8.74	42	12
Area 1 - T1 S4 / SW4	0.60	0.58	660	51.2	51.1	8.7	8.50	8.12	6	10
Area 2 - T2 S0 / SW0	0.57	0.57	1650	38.7	39.0	8.0	9.59	7.90	42	8
Area 2 – T2 / SW0	0.43	0.43	1071	46.8	51.0	8.4	12.25	8.18	48	8
Area 2 - T2 S1 / SW1	0.42	0.49	1200	46.7	46.5	9.8	7.45	8.23	8	12.5
Area 2 - T2 S2 / SW2	2.31	2.04	1108	35.1	37.1	9.1	11.07	8.14	82	12.5
Area 2 - T2 SW3	0.44	0.50	947	52.2	51.7	9.1	11.07	8.16	92	12
Area 2 - T2 SW4	0.182	0.21	560	50.8	50.0	7.8	11.46	7.66	32	15
Control (CR)-a	0.07	0.07	878	8.6	49.4	46.7	11.21	7.75	48	23
Control (CR)-b	0.09	0.11	1153	7.6	51.1	50.7	11.24	7.55	34	18

Appendix 3.B. Table I. Characteristics of culvert A and computed cross sectional area

Diameter, m	0.3
Radius, m	0.15
Depth of water, m	0.1524
Water surface angle, ϴ, radians	3.1736
Cross sectional area, m^2	0.036

Appendix 3.B. Table II. Characteristics of culvert B and computed cross sectional area

Diameter, m	0.6350
Radius, m	0.3175
Depth of water, m	0.2540
Water surface angle, ϴ, radians	2.7389
Cross sectional area, m^2	0.1183

Appendix 3.B. Table III. Velocity values measured at culvert A.

DOWNSTREAM culvert A	17/06/2015 11.55 AM	17/06/2015 01.55 PM
Depth, cm	Velocity, m/s	Velocity, m/s
2 cm depth, cm/s	0.034	0.153
4 cm depth, cm/S	0.098	0.617
6 cm depth, cm/s	0.154	3.066
8 cm depth, cm/s	2.49	1.07
10 cm depth, cm/s	3.03	0.452
12 cm depth, cm/s	0.137	0.245
14 cm depth, cm/s	0.094	-

Appendix 3.B. Table IV. Velocity values measured at culvert B.

UPSTREAM culvert B	17/06/2015 00.03 PM	17/06/2015 02.30 PM	18/06/2015 11.30 AM	18/06/2015 02.10 PM
Depth, cm	Velocity, m/s	Velocity, m/s	Velocity, m/s	Velocity, m/s
2 cm depth, cm/s	NA	NA	0.120	NA
4 cm depth, cm/S	NA	NA	0.253	NA
6 cm depth, cm/s	0.199	0.201	0.178	0.161
8 cm depth, cm/s	0.297	0.276	0.231	0.231
10 cm depth, cm/s	**0.298**	**0.279**	**0.251**	**0.277**
12 cm depth, cm/s	0.301	0.312	0.244	0.333
14 cm depth, cm/s	0.337	0.331	0.212	0.252
16 cm depth, cm/s	0.337	0.342	0.288	0.269
18 cm depth, cm/s	0.354	0.361	0.421	0.323
20 cm depth, cm/s	0.332	0.371	0.325	0.357
22 cm depth, cm/s	0.417	0.396	0.298	0.387

Appendix 3.C. Table I. Water quality parameters of surface water samples sampled over the wetland areas and premises of the lagoon on June 17th, 2015.

Sample ID or/and explanation	Sampling time	TP (mg/L)	EC (μS/cm)	COD (mg/L)	pH	TSS (mg/L)
Lagoon 2nd cell	11.50 AM	1.81	897	49.0	8.09	2
Lagoon 2nd cell	02.00 PM	1.87	968	52.3	8.03	NA
Culvert A Downstream	00.00 PM	1.81	920	47.3	8.45	44
Culvert A Downstream	02.10 PM	1.67	873	49.9	8.46	NA
Culvert B Upstream	00.15 PM	1.63	938	50.4	9.05	NA
Culvert B Downstream	00.15 PM	1.79	956	NA	9.05	13
Lagoon outfall	11.00 AM	NA	1130	49.4	7.36	NA
Ditch to lagoon fence downstream of outfall pipe-before discharge	11.00 AM	NA	2060	48.2	9.04	NA
In the premises of lagoon, left to discharge pipe – before discharge	11.10 AM	0.72	2060	50.1	8.92	NA
In the premises of lagoon, right to discharge pipe – before discharge	11.20 AM	0.39	1860	48.3	8.80	NA
Lagoon outfall 5 min after discharge	11.55 AM	1.81	1079	50.2	7.55	NA
Culvert B Upstream	02.15 PM	NA	1074	50.9	7.95	NA
Culvert B Downstream	02.30 PM	1.73	1068	48.1	7.91	NA
L30 – 30 m to lake	01.00 PM	1.70	1092	47.9	8.15	NA
L60 - 60 m to lake	01.10 PM	1.81	1334	49.0	7.60	NA
L90 – 90 m to lake	01.20 PM	0.54	2120	NA	NA	NA
Control (CR)	01.40 PM	0.026	1608	NA	NA	NA
Control (CR)	01.41 PM	0.025	1602	48.1	8.15	24

NA – not measured

Appendix 3.C. Table II. Water quality parameters of surface water samples sampled over the wetland areas and premises of the lagoon on June 18th, 2015.

Sample ID or/and explanation	Sampling time	TP (mg/L)	SRP (mg/L)	EC (µS/cm)	COD (mg/L)	pH	TSS (mg/L)
Lagoon 2nd cell	02.45 PM	1.26	0.47	999	50.7	9.35	27
Culvert A downstream	11.00 AM	1.80	0.55	1075	49.5	8.06	13
Culvert B downstream	11.45 AM	1.80	0.54	1046	49.8	8.02	16
Culvert A downstream	03.10 PM	1.78	0.58	1065	48.6	8.41	9
Pond 1 (P1)	01.00 PM	0.483	0.10	1365	50.6	8.74	62
L30 - 30 m to lake	00.15 PM	1.75	0.50	1023	47.3	8.06	NA
L60 – 60 m to lake	00.30 AM	1.68	0.43	1019	51.2	8.14	NA
L90 – 90 m to lake	00.40 AM	1.07	0.34	1209	52.9	7.99	NA
Area 2 – W10 -10 m from Culvert B towards woods	00.00 PM	1.72	0.46	1608	48.5	8.56	NA
W30 – 30 m to woods	01.30 AM	1.68	0.34	5045	46	8.05	NA
W90 – 90 m to woods	01.50 AM	1.71	0.20	1228	52.6	7.85	NA
W60 – 60 m to woods	01.40 PM	1.69	0.30	NA	50.6	7.86	16
Control (CR)	01.15 PM	0.029	ND	1692	50.4	7.87	NA
Pond 2 (P2)	02.00 PM	0.747	0.15	1738	49.6	8.18	NA

NA – not measured; ND - below detection

Appendix 3.C. Table III. Water quality parameters of surface water samples sampled over the wetland areas and premises of the lagoon on June 19th, 2015.

Sample ID or/and explanation	Sampling time	TP (mg/L)	SRP (mg/L)	EC (µS/cm)	COD (mg/L)	pH	TSS (mg/L)
Lagoon outfall upstream	11.00 PM	1.71	0.39	1273	49.0	7.60	NA
Culvert A downstream	11.10 AM	1.60	0.54	1285	52.4	8.17	NA
Culvert B downstream	11.30 AM	1.53	0.46	1212	52.3	7.85	NA
W30 – 30 m to woods	00.10 PM	2.25	0.27	1774	51.3	7.76	NA
W90 – 90 m to woods	01.30 PM	1.39	0.34	2829	45.9	7.77	NA
W60 – 60 m to woods	00.50 PM	1.68	0.43	1794	49.1	8.13	NA
W30 – 30 m to woods and 45m inside	00.40 PM	1.78	0.33	1057	52.5	7.42	NA
L30 – 30 m to lake	02.00 PM	1.77	0.43	1133	52.3	7.98	NA
L60 – 60 m to lake	02.10 PM	1.66	0.33	929	52.6	7.91	NA
L90 – 90 m to lake	02.20 PM	1.03	0.18	1330	52.2	8.03	NA
Control (CR)	02.40 PM	0.038	ND	1024	50.4	7.60	NA
Pond 2 (P2)	01.40 PM	0.455	0.07	2068	48	8.05	NA

NA – not measured; ND - below detection

Appendix 3.C. Table IV.Water quality parameters of water samples sampled in lagoon on October 9th, 2015.

Sample ID or/and explanation	Sampling time	TP (mg/L)	SRP (mg/L)	EC (µS/cm)	COD (mg/L)	pH	TSS (mg/L)
Lagoon primary cell	03.00 PM	2.18	0.41	1171	48.5	8.52	128
Lagoon secondary cell	03.05 PM	3.23	1.03	1275	50.8	7.68	30
Lagoon secondary cell	03.10 PM	3.12	1.05	1259	50.6	7.54	50

Appendix 3.C. Table V. Water quality parameters of water samples sampled over the wetland treatment areas, and groundwater wells on October 19th, 2015.

Sample ID or/and explanation	Sampling time	TP (mg/L)	SRP (mg/L)	EC (µS/cm)	COD (mg/L)	pH	TSS (mg/L)
Groundwater Well #1 premises of School closer to the building (depth 1.1m, casing length above ground 0.56m)	11.45 AM	0.19	ND	552.3	37.6	7.45	NA
Groundwater Well #2 premises of Lorie Paul (depth 2.64m, casing length above ground 0.28m)	00.00 PM	0.14	0.03	564.2	7.5	7.48	NA
Piezometer water (at 30m 60m, Area2 depth 61 cm)	02.30 PM	0.23	ND	2424	48.5	8.50	NA
Piezometer water (at 60m 60m, Area2 depth 72 cm)	02.00 PM	0.21	0.05	1642	50.8	8.10	NA
Culvert C downstream (water depth 9 cm)	03.30 PM	0.21	ND	2754	50.8	8.37	NA
Culvert C downstream (water depth 12 cm)	03.40 PM	0.46	0.08	2700	48.2	8.57	NA
Culvert C downstream (water depth 13 cm)	03.45 PM	0.46	0.04	2424	47.5	8.50	NA
Pond 1	00.30 PM	0.23	ND	1394	45.1	8.13	NA
Pond 2	02.30 PM	0.45	0.03	1716	49.9	8.37	NA
Control	04.00 PM	0.18	ND	1503	43.2	8.52	NA

Appendix 3.E. Table I. TP concentration and electrical conductivity of soil samples sampled in wetland treatment areas.

Date of sampling	Type of Soil	Depth	Location	Replicates	TP, mg/kg soil	EC* µS/cm
May 7th, 2016	Berm soil	Depth 0-20 cm	T1 S4 / SW4	Rep1	269	712
May 7th, 2016	Berm soil	Depth 0-20 cm	T1 S4 / SW4	Rep2	258	691
May 7th, 2016	Berm soil	Depth 0-20 cm	T1 S4 / SW4	Rep3	294	697
May 7th, 2016	Berm soil	Depth 0-20 cm	T1 S0 / SW0	REp1	377	475
May 7th, 2016	Berm soil	Depth 0-20 cm	T1 S0 / SW0	Rep2	423	464
May 7th, 2016	Berm soil	Depth 0-20 cm	T1 S0 / SW0	Rep3	361	469
May 7th, 2016	Berm soil	Depth 0-20 cm	T2 S0 / SW0	Rep1	343	499
May 7th, 2016	Berm soil	Depth 0-20 cm	T2 S0 / SW0	Rep2	351	487
May 7th, 2016	Berm soil	Depth 0-20 cm	T2 S0 / SW0	Rep3	415	483
May 7th, 2016	Berm soil	Depth 20-40 cm	T1 S4 / SW4	Rep1	302	414
May 7th, 2016	Berm soil	Depth 20-40 cm	T1 S4 / SW4	Rep2	316	402
May 7th, 2016	Berm soil	Depth 20-40 cm	T1 S4 / SW4	Rep3	289	421
May 7th, 2016	Berm soil	Depth 20-40 cm	T1 S0 / SW0	REp1	304	605
May 7th, 2016	Berm soil	Depth 20-40 cm	T1 S0 / SW0	Rep2	358	585
May 7th, 2016	Berm soil	Depth 20-40 cm	T1 S0 / SW0	Rep3	317	577
May 7th, 2016	Berm soil	Depth 20-40 cm	T2 S0 / SW0	REp1	403	524
May 7th, 2016	Berm soil	Depth 20-40 cm	T2 S0 / SW0	Rep2	362	528
May 7th, 2016	Berm soil	Depth 20-40 cm	T2 S0 / SW0	Rep3	383	533
May 7th, 2016	Berm soil	Depth 40-60 cm	T1 S4 / SW4	REp1	261	436
May 7th, 2016	Berm soil	Depth 40-60 cm	T1 S4 / SW4	Rep2	269	436
May 7th, 2016	Berm soil	Depth 40-60 cm	T1 S4 / SW4	Rep3	286	435
May 7th, 2016	Berm soil	Depth 40-60 cm	T1 S0 / SW0	REp1	353	701
May 7th, 2016	Berm soil	Depth 40-60 cm	T1 S0 / SW0	Rep2	348	635
May 7th, 2016	Berm soil	Depth 40-60 cm	T1 S0 / SW0	Rep3	375	641
May 7th, 2016	Berm soil	Depth 40-60 cm	T2 S0 / SW0	REp1	412	514
May 7th, 2016	Berm soil	Depth 40-60 cm	T2 S0 / SW0	Rep2	438	518
May 7th, 2016	Berm soil	Depth 40-60 cm	T2 S0 / SW0	Rep3	451	514
May 7th, 2016	Meleb soil	Depth 0-20 cm	T1 S1 / SW1	REp1	1080	1283
May 7th, 2016	Meleb soil	Depth 0-20 cm	T1 S1 / SW1	Rep2	811	1333
May 7th, 2016	Meleb soil	Depth 0-20 cm	T1 S1 / SW1	Rep3	924	1293
May 7th, 2016	Meleb soil	Depth 0-20 cm	T1 S2 / SW2	REp1	1216	1327
May 7th, 2016	Meleb soil	Depth 0-20 cm	T1 S2 / SW2	Rep2	1031	1360
May 7th, 2016	Meleb soil	Depth 0-20 cm	T1 S2 / SW2	Rep3	1207	1344
May 7th, 2016	Meleb soil	Depth 0-20 cm	TI S3 / SW3	REp1	839	1203
May 7th, 2016	Meleb soil	Depth 0-20 cm	TI S3 / SW3	Rep2	729	1111
May 7th, 2016	Meleb soil	Depth 0-20 cm	TI S3 / SW3	Rep3	935	1188
May 7th, 2016	Meleb soil	Depth 0-20 cm	T1 S4 / SW4	REp1	981	1192

May 7th, 2016	Meleb soil	Depth 0-20 cm	T1 S4 / SW4	Rep2	970	1102
May 7th, 2016	Meleb soil	Depth 0-20 cm	T1 S4 / SW4	Rep3	1039	1193
May 7th, 2016	Meleb soil	Depth 0-20 cm	T2 S1 / SW1	REp1	1161	2260
May 7th, 2016	Meleb soil	Depth 0-20 cm	T2 S1 / SW1	Rep2	1040	2220
May 7th, 2016	Meleb soil	Depth 0-20 cm	T2 S1 / SW1	Rep3	1192	2179
May 7th, 2016	Meleb soil	Depth 20-40 cm	T1 S1 / SW1	REp1	303	765
May 7th, 2016	Meleb soil	Depth 20-40 cm	T1 S1 / SW1	Rep2	342	777
May 7th, 2016	Meleb soil	Depth 20-40 cm	T1 S1 / SW1	Rep3	331	834
May 7th, 2016	Meleb soil	Depth 20-40 cm	T1 S2 / SW2	REp1	665	1110
May 7th, 2016	Meleb soil	Depth 20-40 cm	T1 S2 / SW2	Rep2	580	1103
May 7th, 2016	Meleb soil	Depth 20-40 cm	T1 S2 / SW2	Rep3	740	1088
May 7th, 2016	Meleb soil	Depth 20-40 cm	TI S3 / SW3	REp1	449	381
May 7th, 2016	Meleb soil	Depth 20-40 cm	TI S3 / SW3	Rep2	392	426
May 7th, 2016	Meleb soil	Depth 20-40 cm	TI S3 / SW3	Rep3	418	398
May 7th, 2016	Meleb soil	Depth 20-40 cm	T1 S4 / SW4	REp1	407	606
May 7th, 2016	Meleb soil	Depth 20-40 cm	T1 S4 / SW4	Rep2	422	633
May 7th, 2016	Meleb soil	Depth 20-40 cm	T1 S4 / SW4	Rep3	430	631
May 7th, 2016	Meleb soil	Depth 20-40 cm	T2 S1 / SW1	REp1	397	1010
May 7th, 2016	Meleb soil	Depth 20-40 cm	T2 S1 / SW1	Rep2	355	968
May 7th, 2016	Meleb soil	Depth 20-40 cm	T2 S1 / SW1	Rep3	395	990
May 7th, 2016	Meleb soil	Depth 40-60 cm	T1 S1 / SW1	REp1	369	867
May 7th, 2016	Meleb soil	Depth 40-60 cm	T1 S1 / SW1	Rep2	423	937
May 7th, 2016	Meleb soil	Depth 40-60 cm	T1 S1 / SW1	Rep3	354	879
May 7th, 2016	Meleb soil	Depth 40-60 cm	T1 S2 / SW2	REp1	397	1090
May 7th, 2016	Meleb soil	Depth 40-60 cm	T1 S2 / SW2	Rep2	244	1074
May 7th, 2016	Meleb soil	Depth 40-60 cm	T1 S2 / SW2	Rep3	436	1066
May 7th, 2016	Meleb soil	Depth 40-60 cm	TI S3 / SW3	REp1	404	407
May 7th, 2016	Meleb soil	Depth 40-60 cm	TI S3 / SW3	Rep2	346	406
May 7th, 2016	Meleb soil	Depth 40-60 cm	TI S3 / SW3	Rep3	386	471
May 7th, 2016	Meleb soil	Depth 40-60 cm	T1 S4 / SW4	REp1	351	553
May 7th, 2016	Meleb soil	Depth 40-60 cm	T1 S4 / SW4	Rep2	384	561
May 7th, 2016	Meleb soil	Depth 40-60 cm	T1 S4 / SW4	Rep3	351	521
May 7th, 2016	Meleb soil	Depth 40-60 cm	T2 S1 / SW1	REp1	414	1000
May 7th, 2016	Meleb soil	Depth 40-60 cm	T2 S1 / SW1	Rep2	356	980
May 7th, 2016	Meleb soil	Depth 40-60 cm	T2 S1 / SW1	Rep3	376	912
May 7th, 2016	Meleb soil	Depth 0-20 cm	CONTROL	Rep1	526	NA
May 7th, 2016	Meleb soil	Depth 0-20 cm	CONTROL	Rep2	550	NA
May 7th, 2016	Meleb soil	Depth 20-40 cm	CONTROL	Rep1	480	NA
May 7th, 2016	Meleb soil	Depth 20-40 cm	CONTROL	Rep2	568	NA
May 7th, 2016	Meleb soil	Depth 40-60 cm	CONTROL	Rep1	363	NA
May 7th, 2016	Meleb soil	Depth 40-60 cm	CONTROL	Rep2	455	NA

May 7th, 2016	Meleb soil	Depth 0-20 cm	POND 2 (P2)	Rep1	1146	NA
May 7th, 2016	Meleb soil	Depth 0-20 cm	POND 2 (P2)	Rep2	914	NA
May 7th, 2016	Meleb soil	Depth 0-20 cm	POND 2 (P2)	Rep3	1068	NA
May 7th, 2016	Meleb soil	Depth 20-40 cm	POND 2 (P2)	Rep1	463	NA
May 7th, 2016	Meleb soil	Depth 20-40 cm	POND 2 (P2)	Rep2	491	NA
May 7th, 2016	Meleb soil	Depth 20-40 cm	POND 2 (P2)	Rep3	495	NA
Oct 19th, 2016	Berm soil	Depth 0-20 cm	T1 S4 / SW4	Rep1	392	400
Oct 19th, 2016	Berm soil	Depth 0-20 cm	T1 S4 / SW4	Rep2	353	384
Oct 19th, 2016	Berm soil	Depth 0-20 cm	T1 S4 / SW4	Rep3	328	389
Oct 19th, 2016	Berm soil	Depth 0-20 cm	T1 S0 / SW0	REp1	483	310
Oct 19th, 2016	Berm soil	Depth 0-20 cm	T1 S0 / SW0	Rep2	453	312
Oct 19th, 2016	Berm soil	Depth 0-20 cm	T1 S0 / SW0	Rep3	348	320
Oct 19th, 2016	Berm soil	Depth 0-20 cm	T2 S0 / SW0	Rep1	422	818
Oct 19th, 2016	Berm soil	Depth 0-20 cm	T2 S0 / SW0	Rep2	447	791
Oct 19th, 2016	Berm soil	Depth 0-20 cm	T2 S0 / SW0	Rep3	436	810
Oct 19th, 2016	Berm soil	Depth 20-40 cm	T1 S4 / SW4	Rep1	403	415
Oct 19th, 2016	Berm soil	Depth 20-40 cm	T1 S4 / SW4	Rep2	495	415
Oct 19th, 2016	Berm soil	Depth 20-40 cm	T1 S4 / SW4	Rep3	456	420
Oct 19th, 2016	Berm soil	Depth 20-40 cm	T1 S0 / SW0	REp1	327	544
Oct 19th, 2016	Berm soil	Depth 20-40 cm	T1 S0 / SW0	Rep2	344	527
Oct 19th, 2016	Berm soil	Depth 20-40 cm	T1 S0 / SW0	Rep3	388	512
Oct 19th, 2016	Berm soil	Depth 20-40 cm	T2 S0 / SW0	REp1	320	380
Oct 19th, 2016	Berm soil	Depth 20-40 cm	T2 S0 / SW0	Rep2	387	368
Oct 19th, 2016	Berm soil	Depth 20-40 cm	T2 S0 / SW0	Rep3	345	375
Oct 19th, 2016	Berm soil	Depth 40-60 cm	T1 S4 / SW4	REp1	465	453
Oct 19th, 2016	Berm soil	Depth 40-60 cm	T1 S4 / SW4	Rep2	446	534
Oct 19th, 2016	Berm soil	Depth 40-60 cm	T1 S4 / SW4	Rep3	436	536
Oct 19th, 2016	Berm soil	Depth 40-60 cm	T1 S0 / SW0	REp1	402	597
Oct 19th, 2016	Berm soil	Depth 40-60 cm	T1 S0 / SW0	Rep2	344	564
Oct 19th, 2016	Berm soil	Depth 40-60 cm	T1 S0 / SW0	Rep3	372	543
Oct 19th, 2016	Berm soil	Depth 40-60 cm	T2 S0 / SW0	REp1	375	462
Oct 19th, 2016	Berm soil	Depth 40-60 cm	T2 S0 / SW0	Rep2	370	470
Oct 19th, 2016	Berm soil	Depth 40-60 cm	T2 S0 / SW0	Rep3	358	468
Oct 19th, 2016	Meleb soil	Depth 0-20 cm	T1 S1 / SW1	REp1	1120	558
Oct 19th, 2016	Meleb soil	Depth 0-20 cm	T1 S1 / SW1	Rep2	1103	552
Oct 19th, 2016	Meleb soil	Depth 0-20 cm	T1 S1 / SW1	Rep3	1205	567
Oct 19th, 2016	Meleb soil	Depth 0-20 cm	T1 S2 / SW2	REp1	902	1400
Oct 19th, 2016	Meleb soil	Depth 0-20 cm	T1 S2 / SW2	Rep2	815	1423
Oct 19th, 2016	Meleb soil	Depth 0-20 cm	T1 S2 / SW2	Rep3	831	1429
Oct 19th, 2016	Meleb soil	Depth 0-20 cm	TI S3 / SW3	REp1	781	795
Oct 19th, 2016	Meleb soil	Depth 0-20 cm	TI S3 / SW3	Rep2	646	735

Oct 19th, 2016	Meleb soil	Depth 0-20 cm	TI S3 / SW3	Rep3	850	744
Oct 19th, 2016	Meleb soil	Depth 0-20 cm	T1 S4 / SW4	REp1	836	1030
Oct 19th, 2016	Meleb soil	Depth 0-20 cm	T1 S4 / SW4	Rep2	818	1186
Oct 19th, 2016	Meleb soil	Depth 0-20 cm	T1 S4 / SW4	Rep3	756	1135
Oct 19th, 2016	Meleb soil	Depth 0-20 cm	T2 S1 / SW1	REp1	1007	1139
Oct 19th, 2016	Meleb soil	Depth 0-20 cm	T2 S1 / SW1	Rep2	1011	1578
Oct 19th, 2016	Meleb soil	Depth 0-20 cm	T2 S1 / SW1	Rep3	1043	1189
Oct 19th, 2016	Meleb soil	Depth 20-40 cm	T1 S1 / SW1	REp1	681	816
Oct 19th, 2016	Meleb soil	Depth 20-40 cm	T1 S1 / SW1	Rep2	656	814
Oct 19th, 2016	Meleb soil	Depth 20-40 cm	T1 S1 / SW1	Rep3	818	820
Oct 19th, 2016	Meleb soil	Depth 20-40 cm	T1 S2 / SW2	REp1	529	1177
Oct 19th, 2016	Meleb soil	Depth 20-40 cm	T1 S2 / SW2	Rep2	511	1106
Oct 19th, 2016	Meleb soil	Depth 20-40 cm	T1 S2 / SW2	Rep3	622	1143
Oct 19th, 2016	Meleb soil	Depth 20-40 cm	TI S3 / SW3	REp1	512	573
Oct 19th, 2016	Meleb soil	Depth 20-40 cm	TI S3 / SW3	Rep2	463	549
Oct 19th, 2016	Meleb soil	Depth 20-40 cm	TI S3 / SW3	Rep3	572	561
Oct 19th, 2016	Meleb soil	Depth 20-40 cm	T1 S4 / SW4	REp1	470	2782
Oct 19th, 2016	Meleb soil	Depth 20-40 cm	T1 S4 / SW4	Rep2	490	2689
Oct 19th, 2016	Meleb soil	Depth 20-40 cm	T1 S4 / SW4	Rep3	454	2699
Oct 19th, 2016	Meleb soil	Depth 20-40 cm	T2 S1 / SW1	REp1	693	860
Oct 19th, 2016	Meleb soil	Depth 20-40 cm	T2 S1 / SW1	Rep2	732	861
Oct 19th, 2016	Meleb soil	Depth 20-40 cm	T2 S1 / SW1	Rep3	690	874
Oct 19th, 2016	Meleb soil	Depth 40-60 cm	T1 S1 / SW1	REp1	467	731
Oct 19th, 2016	Meleb soil	Depth 40-60 cm	T1 S1 / SW1	Rep2	487	742
Oct 19th, 2016	Meleb soil	Depth 40-60 cm	T1 S1 / SW1	Rep3	414	745
Oct 19th, 2016	Meleb soil	Depth 40-60 cm	T1 S2 / SW2	REp1	443	1010
Oct 19th, 2016	Meleb soil	Depth 40-60 cm	T1 S2 / SW2	Rep2	476	992
Oct 19th, 2016	Meleb soil	Depth 40-60 cm	T1 S2 / SW2	Rep3	404	999
Oct 19th, 2016	Meleb soil	Depth 40-60 cm	TI S3 / SW3	REp1	408	552
Oct 19th, 2016	Meleb soil	Depth 40-60 cm	TI S3 / SW3	Rep2	490	539
Oct 19th, 2016	Meleb soil	Depth 40-60 cm	TI S3 / SW3	Rep3	388	530
Oct 19th, 2016	Meleb soil	Depth 40-60 cm	T1 S4 / SW4	REp1	454	2361
Oct 19th, 2016	Meleb soil	Depth 40-60 cm	T1 S4 / SW4	Rep2	482	2382
Oct 19th, 2016	Meleb soil	Depth 40-60 cm	T1 S4 / SW4	Rep3	455	2366
Oct 19th, 2016	Meleb soil	Depth 40-60 cm	T2 S1 / SW1	REp1	362	609
Oct 19th, 2016	Meleb soil	Depth 40-60 cm	T2 S1 / SW1	Rep2	333	609
Oct 19th, 2016	Meleb soil	Depth 40-60 cm	T2 S1 / SW1	Rep3	374	592
Oct 19th, 2016	Meleb soil	Depth 0-20 cm	POND 2 (P2)	Rep1	1302	NA
Oct 19th, 2016	Meleb soil	Depth 0-20 cm	POND 2 (P2)	Rep2	1117	NA
Oct 19th, 2016	Meleb soil	Depth 0-20 cm	POND 2 (P2)	Rep3	1200	NA
Oct 19th, 2016	Meleb soil	Depth 20-40 cm	POND 2 (P2)	Rep1	440	NA

Oct 19th, 2016	Meleb soil	Depth 20-40 cm	POND 2 (P2)	Rep2	375	NA
Oct 19th, 2016	Meleb soil	Depth 20-40 cm	POND 2 (P2)	Rep3	542	NA

*- Electrical conductivity (1:2 soil: solution)

Appendix 4. Summary report of lagoons effluent 2014 (Daniel Benoit, personal communication, Nov 7, 2014)

Comm-unity	Facility Type	LC50	Total Ammonia	Un-ionised Ammonia		P	pH	TSS	CBOD	Temp	O&G	Coli -forms	FC	Chlor-ination	Pass-/Fail
	Maximum Concentrations	<5@96 hrs @100%	Total	WSER 1.25 mg/l	Acute Leth-ality 0.1-. 2 mg/l- Ten State	1mg/l		25 mg/l	25 mg/l	15oC +/- 1oC	15mg/l	1500 MPN /100ml	200 MPN/ 100ml	0.02 mg/l	
1	Aerated Lagoon with SAGR, Alum and UV	0	0.073	0.00244636	0.002446355	1.32	8.1	<4	<2	15	<1	9	<3		Pass
2	Aerated Lagoon with SAGR and Alum	NA	0.02	0.00160237	0.001602367	0.56	8.5	5	6	24	<2	29	<3		Pass
3	Aerated Lagoon	1	12	0.08244931	0.082449307	2.62	7.4	50	21	15	<1	15000	9300	<.02	Fail
4	Aerated Lagoon with Alum and UV	0	2	0.06702342	0.067023421	2.05	8.1	8.7	<2	14.8	<1	<3	<3		Pass
5	Aerated Lagoon		23			3.65		11.5	8		1	150000	23000		Pass
6	Aerated Lagoon	0	4.6	0.01999246	0.019992458	0.424	7.2	25.3	10	14.9	<1.0	11	<3		Fail
7	Aerated Lagoon with Alum and UV (with acid addition to	10	3.9	0.03367429	0.033674289	0.261	7.5	<4	3	15.5	1	<3	<3		Fail

	alter pH for un-ionised ammonia)														
8	Aerated Primary Lagoon with chlorination and dechlorination	1#	9.2	0.30830774	0.308307737	3.63	8.1	4	4	14.6	<1	23000	9300	*	Pass
9	STP Mechanical SBR	0	16	0.0552862	0.055286198	0.441	7.1	15.8	6	15	<1	23000	2300		Pass
10	STP Spiraflow Clarifier	0*	17	0.18437961	0.184379611	0.49	7.6	14	9	15.1	<1	4300	230		Pass
11	STP Mechanical Bioclear	0	0.021	0.00056289	0.000562885	4.36	8	5	10	14.5	<1	43000	43000		Pass
12	STP Mechanical SBR	10	4.6	0.03160557	0.031605568	0.679	7.4	22.5	43	14.9	1.9	23000	23000		Fail
13	STP Mechanical SBR	0	5	0.02732687	0.027326871	0.884	7.3	28.7	9	15.8	<1	>1100000	43000		Fail
14	STP Mechanical SBR		11			1.88		53	34	0	1.5	>1100000	150000		Fail
15	STP Mechanical SBR		21			2.66		49.5	94	0	4.2	>1100000	460000		Fail
16	STP Mechanical SBR		6			0.952		40.8	13		<1	7500	2300		Fail
17	STP Mechanical SBR	10	9.6	0.08289056	0.082890558	1.59	7.5	25	51	14.7	2	>1100000	460000	<.2	Fail
18	STP Mechanical SBR	10	8.9	0.00488823	0.004888227	1.97	6.3	22.3	67	14.3	4.2	>1100000	>1100000		Fail

19	STP Mechanical SBR	10	190	2.5870193	2.587019295	14.5	7.7	64.5	56	15	<1	>1100000	>1100000		Fail
20	STP Mechanical SBR	10	33	0.56367866	0.563678661	2.64	7.8	10.8	8	14.3	<1	23000	23000		Fail
21	STP Mechanical SBR	0	16	0.10993241	0.109932409	1.3	7.4	46.5	48	15.3	<1.0	43000	23000		Fail
22	STP Mechanical SBR	10	80	0.11027851	0.110278514	10.3	6.7	256	631	14.7	20	**	2E+06		Fail
23	Facultative Lagoon	0	0.042	1.16E-11	0	0.573		5.5	<2		<1.0	750	93		Pass
24	Facultative Lagoon		7.6	2.09E-09	2.1E-09	0.352		50	17		<1	15000	4300		Fail
25	Facultative Lagoon	0	2.9	0.03948608	0.039486084	0.245	7.7	<4	5	15	1.1	93	4		Pass
26	Facultative Lagoon	0	0.76	0.44072382	0.440723818	0.783	9.7	4.3	3	14.5	1	<3	<3		Pass
27	Facultative Lagoon	0	5.6	0.11989171	0.119891706	2.28	7.9	5.5	2	14.9	2.2	2300	15		Pass
28	Facultative Lagoon	0	0.11	0.01334246	0.013342455	0.63	8.7	<4.0	3	15	<1	75	9		Pass
29	Facultative Lagoon	0	7.2	0.07809019	0.078090188	3.02	7.6	<4	6	15.4	<1	2300	9		Pass
30	Facultative Lagoon	10	23	0.61649298	0.616492975	8.49	8	7.7	10	15.1	<1	23000	23000		Fail
31	Facultative Lagoon	0	0.13	0.08921903	0.089219033	0.092	9.9	62.8	12	15.5	<1	930	430		Fail
32	Facultative Lagoon		0.081			0.484		16.8	9		2	7500	<3		Pass
33	Facultative Lagoon		0.05	1.38E-11	0	0.895		<4	<2		<1	430	<3		Pass
34	Facultative Lagoon		3.5			2.51		<4	3		1.2	43	<3		Pass
35	Facultative Lagoon	0	0.51	0.13130687	0.131306865	1.35	9.1	19.3	9	15	<1	930	75		Pass

36	Facultative Lagoon	0	4.6	0.03971839	0.039718392	2.7	7.5	<4.0	2	15	<1.0	4300	23		Pass
37	Facultative Lagoon	10	16	0.27329875	0.273298745	4.13	7.8	11.3	41	14	<1	9300	1500		Fail
38	Facultative Lagoon	0	1.6	0.06692131	0.066921312	2.38	8.2	35.3	5	15.5	<1.0	4300	930		Fail
39	Facultative Lagoon	0	0.084	0.00437568	0.004375682	0.248	8.3	<4	3	15	1	430	4		Pass

*-Sodium Hypochlorite/ sodium metabisulfa

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