Long-Term Nutrient Removal and Nutrient Mass Balance of a Free Water Surface Constructed Wetland Polishing Municipal Lagoon Effluent

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Abstract

A large pilot-scale free water surface (FWS) constructed wetland polishing effluent from an annual (spring) discharge municipal lagoon was operated for ten years followed by eleven years of dormancy and then restarted with an increase in operating depth. No significant effect of system aging was observed on Biological Oxygen Demand (BOD), total phosphorus (TP), and soluble reactive phosphorus (SRP) removal efficiencies, although internal TP water column concentrations in the first wetland and pond cells increased with time due to resuspension of accumulated sediments. Nitrate and ammonium removal efficiencies were higher during the start-up period due to plant establishment, while organic nitrogen and nitrate removal efficiencies increased during the restart period, likely due to a combination of the increased operating depth and accumulated sediments. No seasonal temperature effect was observed for nitrate or BOD removal efficiency, however, TP removal efficiencies increased with increasing influent concentrations due to seasonal algae growth. TSS removal efficiency increased significantly during the restart period, most likely due to an increase in the operating depth. Phosphorus was found to be mostly stored in the soil, followed by sediment and plants, while nitrogen was found to be stored more in plants, followed by soil and sediment. The wetland system was shown to be effective at the long-term removal of organic matter (BOD_5 < 10 mg/L) and TP (87% average removal efficiency), while TSS removal efficiency increased to 97% with an increase in operating depth from 25 to 50 cm.

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ANOVA	Analysis of Variance
BOD_5	5 days - Biological Oxygen Demand (mg/L)
C*	Background concentration (mg/L)
COD	Chemical Oxygen Demand (mg/L)
CW	Constructed Wetland
DO	Dissolved Oxygen (mg/L)
EPA / USEPA	Environmental Protection Agency (of the United States)
ET	Evapotranspiration (mm)
FWS	Free Water Surface (Constructed Wetland)
HLR	Hydraulic Loading Rate (m^3/m^2/yr)
HRT	Hydraulic Retention Time (days)
k	First-order areal rate constant (m/yr)
NH_3/NH_4^+	Unionized Ammonia/Ammonium (mg/L)
NO_3^-	Nitrate (mg/L)
o-PO_4	Orthophosphate (mg/L)
pH	logarithmic scale of the concentration of hydrogen ions
PPT	Precipitation (mm)
SRP	Soluble Reactive Phosphorus (mg/L)
T	Temperature (^oC)
TN	Total Nitrogen (mg/L)
TOC	Total Organic Carbon (mg/L)

TP	Total Phosphorus (mg/L)
TS	Total Solids (mg/L)
TSS	Totals Suspended Solids (mg/L)
VS	Volatile Solid (mg/L)

CHAPTER: 1

Introduction

1.1 Background

Constructed wetland (CW) systems can find important niche applications in rural wastewater treatment due to their low capital and operating costs but high space requirements compared to other treatment technologies (Kadlec & Wallace, 2008; Vymazal, 2010). CWs have grown in popularity as a wastewater treatment method during the last 50 years. This technique is also a viable alternative for domestic wastewater treatment technologies (Gikas & Tchobanoglous, 2009; Zhang et al., 2014).

Free Water Surface (FWS) CWs are similar to natural wetlands so that they have an exposed water surface, a bed with rooted emergent aquatic vegetation, a layer of soil serving as a rooting medium, a liner (which is not always present) that prevents infiltration and exfiltration, and inlet and outlet structures that regulate water flow in and out (Kadlec & Wallace, 2008; Reed et al., 2005; Tchobanoglous & Burton, 1991). FWS systems can remove a wide range of contaminants from wastewater, including biological oxygen demand (BOD), total suspended solids (TSS), total nitrogen (TN), total phosphorus (TP), *E.coli,* and metals, by microbial degradation, substrate adsorption, plant absorption, sedimentation, filtering, and biological predation (Gorgoglione & Torretta, 2018). The primary functions of plants in surface wetlands are to provide a physical effect through their stems and roots, facilitating some physical treatments in a wetland such as filtering, velocity reduction, sedimentation promotion, and decreased resuspension (Vymazal, 2013). Plant dormancy can affect biological processes around or below-freezing temperatures, so the seasonal influence on wetland treatment performance can be especially crucial in a cold climate (Faulwetter et al., 2009). In addition to vegetation and soil, other factors that affect surface wetland performance include wastewater quality, climatic conditions, flow rate, hydraulic retention time, and water depth (Stefanakis & Headley, 2020).

Lagoon-based sewage treatment systems are one of the earliest types of wastewater treatment and are still widely used in Canada to treat wastewater in small communities (Smith & Finch, 1985). In 2017, a total of 1048 lagoon-based wastewater treatment systems were reported to operate in Canada, representing nearly half of the total number of treatment plants (Environment and Climate Change Canada, 2017). In Canada, municipal lagoon system discharges are regulated by Provincial governments and more recently by the federal government through the Wastewater Systems Effluent Regulations, which stipulates maximum concentrations of 25 mg/L for TSS and $cBOD_5$ and 1.25 mg/L NH_3 at 15°C ± 1°C (WSER, 2012). Discharge criteria are also set by the appropriate Provincial Department for each system based on the attenuation capacity of the receiving water body, which can be more stringent than the federal criteria. Challenges for lagoon systems to meet increasingly stringent discharge limits include TSS, due to algae production, TP even with batch coagulation before discharge, and NH_4^+ in winter months and during spring for seasonal discharge lagoons (Federation of Canadian Municipalities, 2004). Polishing technologies are necessary to meet these challenges and constructed wetlands can provide an appropriate solution for seasonal discharge lagoon systems which do not discharge during winter. Meeting higher discharge limits can facilitate approval for extending the discharge period(s) which can have the important benefit of increasing the hydraulic capacity of a seasonal discharge lagoon.

1.2 Study objectives

It is hypothesized that a period of dormancy for a mature FWS constructed wetland polishing municipal lagoon effluent will not impact treatment performance while increasing the operating depth and the addition of a grass filter can improve TSS removal efficiency.

This study aims to characterize and evaluate the FWS constructed wetland system and vegetated filter performance. This research will provide new knowledge on the long-term performance of a FWS wetland system polishing the effluent from an annual discharge municipal lagoon system. The specific research objectives are as follows:

I. Evaluate the performance of a FWS constructed wetland and vegetated filter to polish municipal lagoon effluent.
II. Evaluate the effect of increasing operating depth on system performance in general and on TSS removal in particular.
III. Evaluate the effect of system aging and seasonal temperature on wetland system performance.
IV. Determine removal rates and kinetic rate constants and make design recommendations.
V. Quantify the N and P storage reservoirs in the system and conduct a system mass balance for N and P.
VI. Determine the relative importance of soil, sediment, and plant reservoirs for nutrient storage and the impacts on system longevity.

1.3 Thesis layout

Chapter 1 presents background information on wetland system performance. This chapter also includes the objectives of this research study. Chapter 2 provides a comprehensive literature review including the performance and mechanisms for pollutant and nutrient removal in surface-flow constructed wetlands.

Chapter 3 presents the results of the wetland system and vegetated filter system concentrations for different nutrients. The chapter also addresses the aging and temperature effects on the wetland system. Finally, the wetland performance and removal rates over time using the P-k-C* model were investigated.

Chapter 4 describes a complete system mass balance for phosphorus and a partial mass balance for nitrogen. The chapter also explains the storage of nutrients in the soil, sediment, and plant zone of the wetland system.

Finally, Chapter 5 presents all the conclusions resulting from this research study and recommendations for future research work in the area.

1.4 References

Environment and Climate Change Canada. (2017). Wastewater Systems Effluent Regulations. Annual Report: Section 2. https://www.canada.ca/en/environment-climate-change/services/wastewater/publications/wastewater-systems-effluent-regulations-2017-annual-report/reporting.html

Faulwetter, J. L., Gagnon, V., Sundberg, C., Chazarenc, F., Burr, M. D., Brisson, J., Camper, A. K., & Stein, O. R. (2009). Microbial processes influencing performance of treatment wetlands: A review. Ecological Engineering, 35(6), 987–1004. https://doi.org/10.1016/J.ECOLENG.2008.12.030

Federation of Canadian Municipalities. (2004). OPTIMIZATION OF LAGOON OPERATION. https://fcm.ca/sites/default/files/documents/resources/guide/infraguide-optimization-lagoon-operations-mamp.pdf

Gikas, P., & Tchobanoglous, G. (2009). The role of satellite and decentralized strategies in water resources management. Journal of Environmental Management, 90(1), 144–152. https://doi.org/10.1016/J.JENVMAN.2007.08.016

Gorgoglione, A., & Torretta, V. (2018). Sustainable management and successful application of constructed wetlands: A critical review. Sustainability (Switzerland), 10(11). https://doi.org/10.3390/SU10113910

Kadlec, R., & Wallace, S. (2008). Treatment wetlands.

Reed, S. C., Crites, R. W., & Middlebrooks, E. J. (2005). Natural Wastewater Treatment Systems. 576.

Smith, D. W., & Finch, G. R. (1985). A Critical Evaluation of the Operation and Performance of Lagoons in Cold Climate. Environmental Engineering Technical Report 85-2, Department of Civil Engineering, University of Alberta.

Stefanakis, A., & Headley, T. (2020). FWS WETLANDS. https://www.researchgate.net/publication/339472902_FWS_WETLANDS

Tchobanoglous, George., & Burton, F. (1991). Wastewater engineering: treatment, disposal, and reuse (3rd ed. /). McGraw-Hill.

Vymazal, J. (2010). Constructed Wetlands for Wastewater Treatment. Water 2010, Vol. 2, Pages 530-549, 2(3), 530–549. https://doi.org/10.3390/W2030530

Vymazal, J. (2013). Emergent plants used in free water surface constructed wetlands: A review. Ecological Engineering, 61, 582–592. https://doi.org/10.1016/j.ecoleng.2013.06.023

Wastewater Systems Effluent Regulations (2012). https://laws-lois.justice.gc.ca/eng/regulations/sor-2012-139/fulltext.html

Zhang, D. Q., Jinadasa, K. B. S. N., Gersberg, R. M., Liu, Y., Ng, W. J., & Tan, S. K. (2014). Application of constructed wetlands for wastewater treatment in developing countries--a review of recent developments (2000-2013). Journal of Environmental Management, 141, 116–131. https://doi.org/10.1016/J.JENVMAN.2014.03.015

CHAPTER: 2

Literature Review

2.1 Facultative discharge lagoons and treatment

Millions of cubic meters of wastewater are discharged into city sewer systems every day from residences, businesses, institutions, and industry. Municipal wastewater is one of the most significant polluters of Canada's surface waters with typical characteristics described in Table 2.1. Wastewater must be treated before it can be released into the environment (Canadian Environmental Sustainability Indicators, 2020).

Table 2.1: *Typical concentrations of water quality contaminants in North American untreated domestic wastewater*

Parameter	**Concentration (mg/L)**		
	Low	Medium	High
TSS (mg/L)	100	220	350
COD (mg/L)	250	500	1000
BOD_5 (mg/L)	110	220	400
TN (mg/L)	20	40	85
TP (mg/L)	4	8	15

Adapted from: Tchobanoglous & Burton (1991)

Lagoons are particularly well suited to treat wastewater from rural communities as they are less expensive to build and operate than other systems. However, they take up more land than other wastewater treatment systems, although land in rural areas is usually more available and less expensive. Lagoon systems can be designed for continuous discharge or controlled discharge. Controlled discharge lagoons can either discharge once a year (annual discharge) when flow in the receiving water body is highest during spring, or multiple times a year (seasonal discharge), usually during spring and fall, as lagoons can experience high ammonia effluent concentrations during winter and high algal solids during summer (Federation of Canadian Municipalities, 2004).

The removal of BOD_5 from facultative lagoons can be as high as 95% with effluent BOD < 30 mg/L; however, effluent TSS can range from less than 30 mg/L to greater than 100 mg/L, depending on algal densities and discharge structure design (USEPA, 2002). Ammonia nitrogen removal can also be significant (up to 80%) depending on the temperature, pH, and detention duration in the system, although, the removal cannot be maintained over cold periods (USEPA, 2002). The chemical addition of alum or ferric chloride can reduce typical TP concentrations to less than 1.0 mg/L with continuous dosing or between 0.5-1.0 mg/L with batch dosing (Federation of Canadian Municipalities, 2004). Increasingly stringent TP discharge limits will likely require phosphorus polishing technologies for municipal lagoon systems.

2.2 Constructed wetlands

Constructed wetlands (CWs) have several advantages over conventional systems, particularly when they are to be installed in small communities, including low construction and operation costs, ease of operation, and efficiency and reliability in pollutant removal. As a result, they are becoming increasingly popular in both developed and developing countries (Kadlec & Wallace, 2008; Reed et al., 2005; Vymazal, 2010). A constructed wetland is a shallow basin that may be filled with some form of filter material (substrate), commonly sand or gravel, and flora that can withstand saturated conditions. Wastewater enters the basin and flows over the surfaces or through the substrate before being discharged through a device that regulates the depth of the wastewater in the wetland.

The hydraulics, chemical, and biochemical processes in constructed wetlands can be considerably influenced by a cold winter environment, as is the case in many parts of Canada. Reduced physical and biological activity because of dormant plants and the delayed reaction rate of soil or aquatic microbes at cold temperatures may impact system performance on a seasonal basis (Mæhlum et al., 1995). As a result, many of Canada's constructed wetland systems are designed for seasonal discharge (Pries, 1994).

2.2.1 Types of constructed wetlands

Constructed wetlands for wastewater treatment can be divided into four categories based on the dominant macrophyte's life form: emergent, submerged, free-floating, and rooted with floating leaves (Vymazal, 2010). Further classifications could be developed based on wetland hydrology as free water surface and sub-surface wetlands. Wetland systems with a free water surface (FWS) are those where the water surface is open to the environment. A subsurface flow wetland is normally built as a bed or channel with an appropriate media (Iqbal et al., 2022). The subsurface wetlands can be categorized based on the direction of the flow as horizontal wetlands and vertical wetlands (Omondi & Navalia, 2020; Vymazal & Kröpfelová, 2008).

2.3 Free surface wetlands

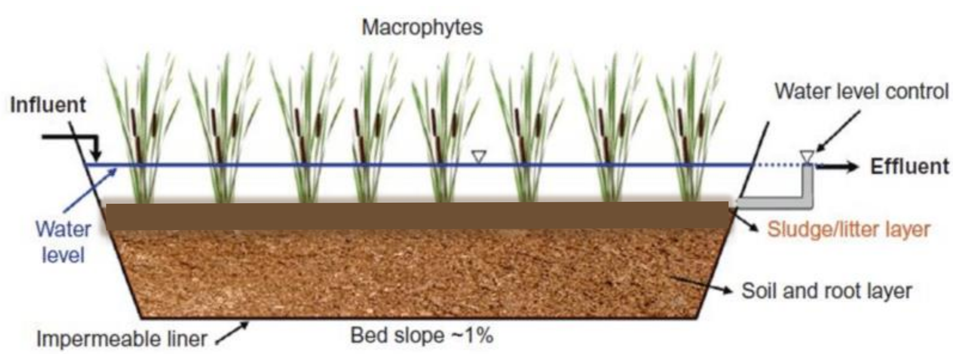


Figure 2.1: *Free water surface wetland characterization* (Stefanakis et al., 2014)

The very first fully constructed wetland with a free water surface (FWS) was developed in 1967 in the Netherlands (de Jong, 1976). During the 1970s, the FWS wetlands were established in North America (*Constructed Wetlands - Water Canada*, 2009). Free surface wetlands are very popular and used in the United States of America, Canada, and Australia (Omondi & Navalia, 2020; Vymazal, 2010; Zeng et al., 2019). The shallow water column can have a high oxygen concentration and promote nitrification while the sediment or sludge layer is often anoxic and can promote denitrification (Hassan et al., 2021; Vymazal, 2010).

Free surface wetlands are typically utilized for runoff from roads and highways, agricultural farms, and urban territories (Kadlec & Wallace, 2008; Okurut, 2000; Vymazal, 2013). FWS is also used for dairy wastewater, agricultural wastewater, municipal wastewater, landfill leachate, abattoir factory, fish hatcheries, and wood waste leachate (Cameron et al., 2003; Kadlec & Wallace, 2008; Okurut, 2000; Vymazal, 2013; Vymazal & Kröpfelová, 2008). A shallow sealed basin or sequence of basins with 20–30 cm of rooting soil and a water depth of 20–40 cm is characteristic of an FWS CW with emergent macrophytes (Kadlec, 1995). Plants are rarely harvested, and the litter provides a source of organic carbon for denitrification, which can occur in anaerobic zones within the soil surface (Vymazal, 2010). In FWS wetlands, water can be lost totally in some circumstances due to evapotranspiration and seepage within the system (U.S.EPA, 2000).

FWS wetlands passively provide effective treatment, reducing the need for mechanical equipment, energy, and trained operators (USEPA, 2000). FWS wetlands are potentially less expensive to build, run, and maintain than subsurface treatment systems (Hassan et al., 2021). Moreover, these systems are a valuable addition to a community's "green space," as they provide wildlife habitat as well as public recreational opportunities (Omondi & Navalia, 2020; Stefanakis & Headley, 2020; USEPA, 2000; Vymazal, 2010). The removal of BOD, TSS, COD, metals, and organics in municipal wastewater can be effectively removed with sufficient detention time. Phosphorus and nitrogen removal can also be effectively reduced with longer retention times (Gorgoglione & Torretta, 2018; USEPA, 2000). However, FWS wetlands can require a lot of area, especially if nitrogen or phosphorus treatment is necessary (Hassan et al., 2021; Vymazal, 2010). Surface wetlands can be a nuisance source of mosquitoes, ticks as well as other vectors (Gorgoglione & Torretta, 2018; USEPA, 2000).

2.4 Removal Mechanisms of Pollutants in Surface Wetlands

Pollutants including suspended particles, metals, BOD_5, nutrients, hydrocarbons, and pathogens can be found in municipal wastewater, urban runoff, and agricultural and industrial activities, all of which can harm aquatic species and ecosystem health. Global environmental cycles have been significantly influenced by human activity. Humans have increased the input of nutrients into biogeochemical cycles, particularly nitrogen (Vitousek et al., 1997) and phosphorus (Reckhow & Simpson, 2011), through farming methods, urbanization, industrialization, and other changes (Picard et al., 2005). The effectiveness of the CW in improving water quality is dependent on a variety of dynamic and interconnected processes that can be loosely grouped into three categories: physical, chemical, and biological (Bosnina, 2021).

Eutrophication or nutrient enrichment of aquatic environments can increase the growth of algae and aquatic plants, as well as the loss of component species and ecosystem function (Smith et al., 1999). Eutrophication is the world's most serious water quality issue (Carpenter et al., 1998). For example, EPA criteria for total nitrogen and total phosphorus were not met by 61% of 2048 water bodies in the United States (Picard et al., 2005). Wetlands have been studied as a potential solution to the world's eutrophication and water quality issues (Fraser et al., 2003). In treatment wetlands, total nitrogen and total phosphorus removal can range from 3–98% to 31–99%, respectively, however, average removal across multiple studies is only approximately 50% (Dotro et al., 2017; Kadlec & Wallace, 2008).

The removal mechanisms of wastewater pollutants in constructed wetlands were shown in Table 2.2 below.

Table 2.2: *Pollutant and nutrient removal mechanisms of wastewater in constructed wetlands*

Pollutant	**Process or mechanism of removal**
TSS	Sedimentation and/or filtration
BOD_5	The aerobic or anaerobic degradation process Sedimentation of organic particles
Nitrogen	Ammonification Settling of organic N Nitrification and denitrification Nitrogen uptake by roots and plants Process of volatilization of ammonia and ANAMMOX
Phosphorus	Reactions of adsorption Settling of particulate P Phosphorus absorption by roots and plants Precipitation with some cations
Pathogenic organisms	Sedimentation and filtration or UV ray action Excretion of antibiotics by plants and other bacteria

Adapted from: Water Environment Federation (1994)

2.4.1 Suspended solids

Suspended solids are predominantly removed through flocculation, sedimentation, filtration, and interception in wetland systems (Kadlec & Wallace, 2008). Death of invertebrates, fragmentation of detritus from plants, generation of plankton and microorganisms within a water column or attached to plant surfaces, and formation of chemical precipitates such as iron sulfide are all examples of suspended solid formation within the wetland (Abou-Elela, 2019). TSS in effluents may be reduced to 5-15 mg/L by using free-surface wetlands as a secondary treatment (Reed et al., 2005). Empirical relationships across numerous studies suggest a 75% removal rate for influent TSS concentrations greater than 20 mg/L (Kadlec & Wallace, 2008).

2.4.2 Organic matter

The primary mechanisms for BOD removal in constructed wetlands are typically particulate settling as well as aerobic and anaerobic microbial degradation (Karathanasis et al., 2003). Atmospheric oxygen diffusion, convection (wind effect), and macrophyte root transfer into the plant rhizosphere are all routes for supplying oxygen for aerobic breakdown, while anoxic to anaerobic conditions may exist in the sediment layer (Abou-Elela, 2019). The removal of BOD is well documented and follows first-order kinetics (Kadlec & Wallace, 2008).

2.4.3 Nitrogen

Nitrogen compounds are among the most significant constituents in wastewater, because of their role in eutrophication and their impact on oxygen levels in receiving water bodies. These compounds also increase plant development, which significantly improves the biogeochemical cycles of the wetland. Optimizing the most fundamental chemical transformations of this element is a difficulty in ecological engineering because the wetland nitrogen cycle is so diverse.

Ammonia (NH_4^+), nitrite (NO_2), nitrate (NO_3^-), nitrous oxide (N_2O), and dinitrogen gas (N_2) are the most important forms of nitrogen in wetlands treating municipal or domestic wastewater (Kadlec & Wallace, 2008; Lee et al., 2009). Volatilization, ammonification, nitrification/denitrification, plant absorption, and matrix adsorption are some of the nitrogen removal mechanisms in constructed wetlands as shown in Figure 2.2. Microbial nitrification/denitrification is the primary removal mechanism in most constructed wetlands.

Ammonification is the first step in the process of converting organic nitrogen to inorganic nitrogen. Microbial growth is aided by the energy generated during the ammonification process (Vymazal, 2007). Nitrifying bacteria in aerobic zones convert ammonia to nitrite and then to nitrate. Denitrifying microorganisms in anoxic and anaerobic zones convert nitrates to dinitrogen gas (Cooper et al., 1996; Watson et al., 2020). Organic nitrogen is also invariably present in FWS wetlands and is the main N component of sediments and plant tissue (Kadlec & Wallace, 2008). Plants and seasonal temperature changes are two factors that have a significant impact on the system's nitrogen removal efficiency (Picard et al., 2005).

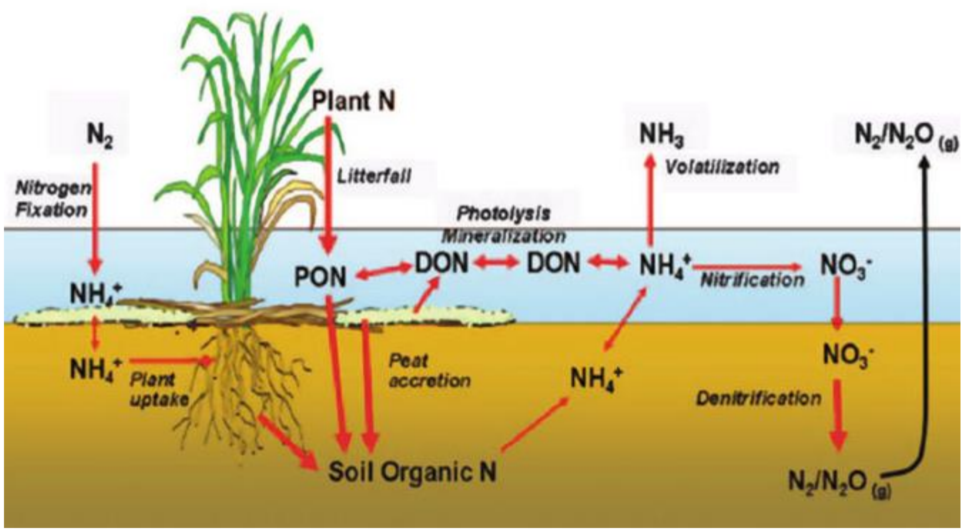


Figure 2.2: *Nitrogen transformation in FWS constructed wetland (Reddy & DeLaune, 2008)*

Note: PON = Particulate organic N; DON = Dissolved organic N.

2.4.4 Phosphorus

Wetlands can accumulate phosphorus for both short and long-term periods using biological, chemical, and physical mechanisms. In constructed wetland systems, phosphorus is removed by microbial and plant uptake, integration into organic matter, substratum adsorption, and chemical precipitation as shown in Figure 2.3 (Kadlec & Wallace, 2008; Mann, 1997; Richardson, 1985). Phosphorus can be dissolved in water, a solid mineral in the soil structure, or a solid organic in biomass. Mostly organic phosphorus and orthophosphate enter most treatment wetlands; however, significant organic phosphorus is transformed into orthophosphate during organic matter degradation (Dotro, Langergraber, et al., 2017).

Flow rate, flow direction, water and soil chemistry, pH, and oxidation-reduction potential all affect sorption. Soil sorption capacity is minimal, although it rises when there is a significant proportion of mineral cations in the soil (Richardson, 1985). Because macrophytes' decomposition reintroduces the nutrient back into the system, their uptake is considered short-term and temporary (Kadlec, 2010). Metal cations such as calcium, iron, and aluminum may abiotically coprecipitate with dissolved phosphorus in the water column. Particulate phosphorus can be incorporated into the sediment by sedimentation and can also be accreted into the soil through soil formation. Soil accretion is responsible for the system's long-term and permanent P storage (Richardson & Marshall, 1986).

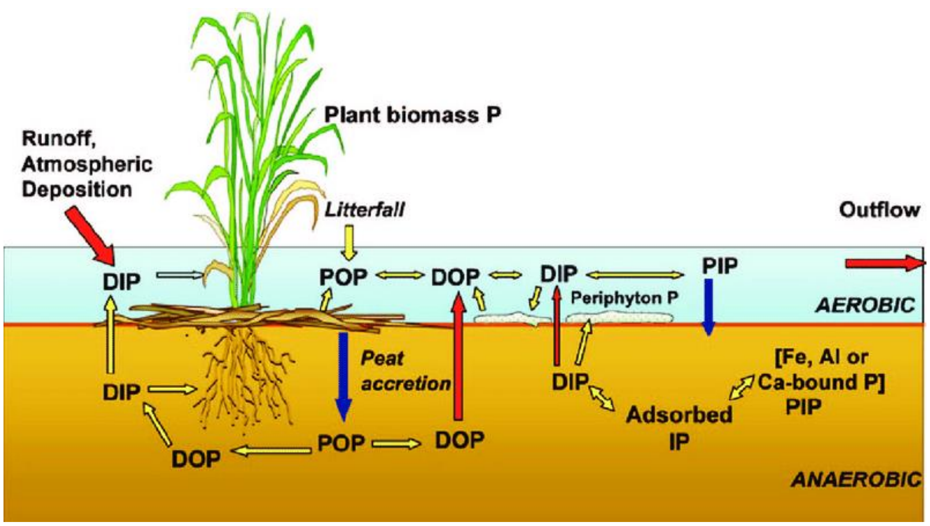


Figure 2.3: *Phosphorus transformation in FWS constructed wetland (Reddy & DeLaune, 2008)*

Note: POP = Particulate organic P; PIP = Particulate inorganic P; DIP = Dissolved inorganic P; DOP = Dissolved organic P; Al = Aluminum; Fe = Iron; Ca = Calcium.

2.5 Factors affecting FWS wetland performance

2.5.1 Climatic conditions

Wetlands are heavily influenced by climatic conditions since they are shallow bodies of water exposed to the atmosphere. Seasonal temperature ranges, freezing periods, and precipitation patterns must be considered during system design (Bendoricchio et al., 2000; Bosnina, 2021). Significant interactions with the atmosphere via precipitation and evapotranspiration influence water circulation in wetlands. These elements have an impact on the treatment process in treatment wetlands as precipitation dilutes concentrations and increases flow, whereas evapotranspiration increases concentrations while lowering flow (Kadlec & Wallace, 2008; Okurut, 2000; Vymazal, 2010).

2.5.2 Soil and geological conditions

The site's soils should be identified for planning purposes. Soils are divided into groups depending on a variety of physical and chemical features (Kadlec & Wallace, 2008). Depth of seasonal high groundwater, depth of confining layers of clays, soil textures, and chemical composition are examples of soil characteristics that may be useful during project design, particularly for bank building or leaking into the groundwater. In some applications, such as phosphorus or metal removal, the sorption potential of the soils will be a design variable (Bendoricchio et al., 2000; Stefanakis & Headley, 2020).

2.5.3 Hydrological factors

The factors such as the source of water, depth of water, flow rates, residence time, etc., influence the wetland hydrodynamics and the physical and chemical properties of the wetland substrate sediments (Stefanakis & Headley, 2020). The hydrodynamics in a wetland determine the types of flora and fauna that emerge in each region and the nutrient dynamics and biological processes that take place. The more time water spends in a wetland, the more likely it is for waterborne contaminants to interact with the wetland environment (Kadlec & Wallace, 2008; Okurut, 2000)

2.6 Design considerations for FWS wetland

The design criteria and guidelines for various parameters of FWS wetlands are established by several researchers. The size of FWS wetlands is usually determined by their area or volume (Stefanakis & Headley, 2020).

2.6.1 Volumetric and Areal loading-based kinetic model for wetland performance

The reaction kinetics for the contaminant of interest are often used to quantify the complicated decomposition, transformation, and removal processes that occur in treatment wetlands. Wetland designers often use chemical kinetic theory to represent the kinetics and hydraulic behavior of wetland treatment. The design of CW is based on two first-order models: the volumetric-based model by Reed et al. (2005) and the areal loading-based model by Kadlec & Wallace (2008). The volumetric-based model (Eq.2.1) is based on the rate constant and a plug-flow approach, while the areal-based model (Eq.2.2) used P-C*-k values with the tank-in-series approach.

$$\ln\left(\frac{C_i}{C_o}\right) = K_T \frac{V * n}{Q}$$ **Equation 2.1**

Where, C_i = inlet concentration (mg/L)
C_o = outlet concentration (mg/L)
K_T = volumetric first-order kinetic rate constant, d^{-1}
V= treatment volume of the wetland, m^3
n = porosity (percent, expressed as a decimal fraction)
Q = average flow rate through the wetland, m^3/d

$$\frac{C_o - C_*}{C_i - C_*} = \frac{1}{(1+k/Pq)^P}$$ **Equation 2.2**

Where, C_* = background concentration (mg/L),
k = first-order areal rate constant (m/yr),

P = apparent number of tanks in series,
q = hydraulic loading rate (m/yr).

The Tank-in-series model created by Kadlec & Wallace (2008) has been the most often used modeling system to characterize treatment performance. The significance of influent concentrations, HLR, and HRT on first-order k-values has been proven in investigations. This model assumes steady-state conditions: no infiltration, no evapotranspiration, and constant flow. Even though this assumption does not accurately reflect field conditions, the TIS and P-C*-k models have been used to estimate discharge concentrations in treatment wetlands with acceptable performance (Kadlec, 2003). Rate coefficients, which vary for each treatment performance parameter, are used to complete wetland design sizing and estimations for pollutant removal.

Table 2.3: *P, C* (mg/L) and k (m/yr) values for different constituents*

Constituents	P [a]	C* (mg/L) [a]	k (m/yr) [a]	P [b]	C* (mg/L) [b]	k (m/yr) [b]
BOD	1	2	33	1	1	23
TN	3	1.5	12.6	3	0.4	3.9
NH_3	3	0	14.7	3	0	3.8
NO_3^-	3	0	26.5	3	0.02	101

Source: [Treatment wetlands] by Kadlec & Wallace (2008),
[b] Brighton, Ontario wetland by Kadlec et al. (2012)

The value of parameters C^* and P used in the investigation of different wetlands are as follows. For most pollutants, background concentrations, or C* values are close to zero for NH_3 and NO_3^-, however, Brighton wetland used C*=0.02 mg/L for NH_3-N. Most of the FWS wetlands used P=1 for BOD and P=3 for nitrogen constituents. The irreducible background concentrations, C*, were also calculated using the lowest effluent concentrations recorded in the Constructed wetlands for each contaminant (Trang et al., 2010). The average annual k- values for ten years in Brighton wetland treating municipal lagoon effluent are 23 m/yr for BOD, 3.9 m/yr for TN, 28.5 m/yr for ammonification, 3.8 m/yr for nitrification, 101 m/yr for denitrification (Kadlec et al., 2012).

Moreover, the average annual k- values by Kadlec & Wallace (2008) at the 50th percentile was observed as 33 m/yr for BOD, 12.6 m/yr for TN, 14.7 m/yr for ammonia, and 26.5 m/yr for nitrate.

Most design criteria given in the literature are based on personal experience rather than scientific evidence, making them difficult to apply across a range of influent concentrations, flow rates, and soil conditions. Long-term monitoring projects at different wetlands sites would allow the estimation of acceptable rate coefficients.

2.6.2 Cell configurations and retention time

The number and arrangement of individual wetland cells (in parallel and series), as well as their dimensions, are important design steps after determining the FWS wetland area. Multiple wetland cells can provide the advantages of allowing for more design and operation flexibility, as well as improving overall system performance (Bendoricchio et al., 2000). The average water depth in CW ranges from 0.1 to 0.6 m (Bendoricchio et al., 2000; Reed et al., 2005). The retention time should range from 2 to 3 days for each wetland cell (USEPA, 2000). Bendoricchio et al. (2000) suggest retention times should be 5 to 10 days for BOD and TSS removal and 8 to 14 days or longer for nitrogen removal, while Crites (1994) suggests a retention time of 15 to 25 days for phosphorus removal.

The length-to-width ratio is highly significant in wetland design, because of its effect on flow distribution and hydraulic short-circuiting. The effectiveness of pollution removal is increased when the hydraulic performance is good, which is achieved by the design of the shape and hydraulic structures (Bendoricchio et al., 2000). Typically, the aspect ratio is between 2:1 to 5:1 (Economopoulou & Tsihrintzis, 2004) but some studies also resulted in a good performance with a length: width ratio of 10:1 (Hammer, 1989). Deep open water ponds within constructed wetlands are beneficial to facilitate mixing and reduce short-circuiting between wetland cells, providing sedimentation of smaller particles, and enhancing the wetland system's visual and recreational value (Bendoricchio et al., 2000). Also, deep water ponds allow oxygenation and increase the overall retention time of the wetland system. The inclusion of open water areas in FWS treatment wetland systems has been divided into two categories (a) Deep zones inside the wetland and (b) Ponds preceding wetlands. Ponds help settle incoming TSS, but they also produce TSS through algal cycling. Pre-treatment ponds are essential for systems with high-influent solids. For example, APAI (1995) reported that although the settling ponds only took up 15%-25% of the total area, they were responsible for 94%–97% of the solid removal.

2.7 Effects of temperature and age on FWS wetland

Even in reasonably cold weather, FWS wetlands can operate during the winter because water may be regulated to flow under ice (Kadlec & Wallace, 2008). As temperatures drop, however, complete freeze-up occurs, resulting in over-ice water flows and ice buildup. Microbial processes are greatly reduced by cold water temperatures, and the plant cycle is dormant in freezing conditions. Ice and snow obstruct oxygen transmission to under-ice water. The overall observed treatment slowdown is the result of these separate process effects combined in complicated ways (Kadlec et al., 2012).

Uusheimo et al. (2018) investigated wetlands that received nitrogen-rich wastewater. During the ice-free period, the highest removal efficiency was found at 79% for ammonium-nitrogen (NH_4^+-N), 71% for nitrate-nitrogen (NO_3^--N), and 88% for phosphate-phosphorus (PO_4^{3-}-P). Truu et al. (2009) found that temperature also affects nitrogen efficiency in wetland systems. Nitrogen removal and nitrification will be reduced when temperatures fall below 10°C (Bendoricchio et al., 2000). However, winter performance in constructed wetlands in Denmark, Sweden, and North America was not significantly lower than in other seasons (Jenssen et al., 1993).

Kadlec et al. (2012) studied a FWS wetland located in Brighton, Ontario, Canada treating municipal lagoon wastewater for ten years from 2001 to 2010. The study presents monthly averages for ten years and found no clear effect of system age on treatment. They found that total phosphorus (TP) entering the wetland follows a strong seasonal pattern with higher inlet concentrations in late summer (July–September) and lower inlet concentrations in winter months, which corresponded to lower removal in late summer and higher removal in winter. Also, TSS and BOD resulted in higher concentrations and higher removal in winter and early spring. However, nitrogen had higher removal in the spring and summer periods as compared to the winter season (Kadlec et al., 2012). The age and temperature effects on nitrogen removal over ten years from 2002-2011 were studied in a FWS wetland treating post-tertiary sewage for the municipality of Eskilstuna, Sweden (Waara et al., 2015). A temperature effect on wetland performance was observed with average temperatures between 8-10ºC (April to October) removing 70-100% of TN, which temperatures below 8ºC (November to March) removing 0-30% of TN. However, no effect of system aging was observed on TN reduction.

2.8 Plant effects in FWS wetlands

Plants use nutrients for metabolic growth. Plant uptake can be a significant source of nutrient removal in wetland systems, particularly at low loading rates (Gottschall et al., 2007). They may also absorb trace compounds found in the root zone, which are then retained or released as gases in some situations. The roots are primarily responsible for uptake, and they are most typically found in wetland soils, while adventitious roots can occasionally be discovered in the water column (Kadlec & Wallace, 2008). The purpose of aquatic plants in FWS systems is to stabilize the surface of the beds, create good conditions for physical filtration, and provide a large surface area for associated microbiological development (Brix, 1994). Decomposing plant biomass may produce the organic carbon needed for denitrification (Vymazal, 2013). Additionally, the macrophyte helps to stabilize the wetland media, lowers clogging, enhances hydraulic conductivity, creates a suitable environment for bacteria to grow, absorbs nutrients, and oxygenates the water (Stefanakis et al., 2014).

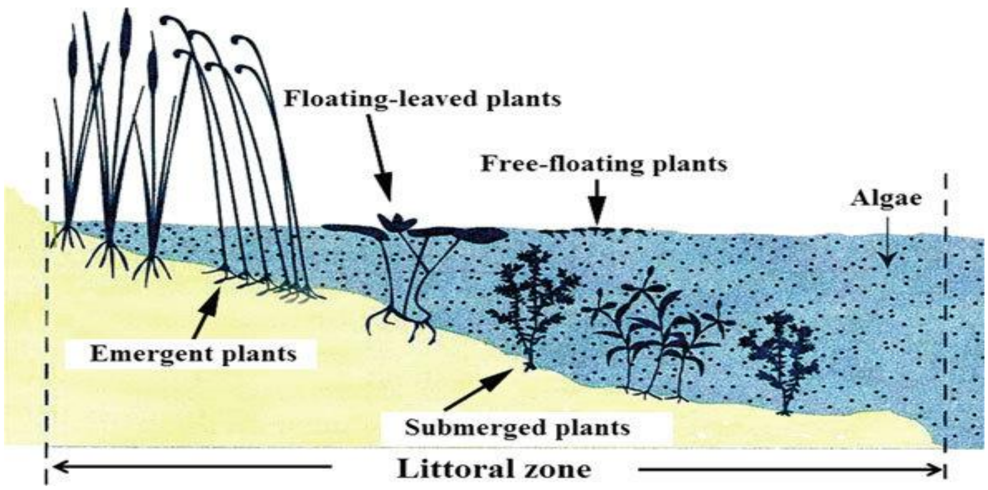


Figure 2.4: *Various aquatic plants found in the littoral zone of the wetland (Rana & Maiti, 2020)*

The cattail species is a perennial freshwater wetland macrophyte that can reach a height of three meters or more and the leaves are thick, ribbon-like structures with a spongy cross-section that shows air passages (Vymazal, 2013). Typha latifolia has flat, arching, pale grayish-green leaves and the stems are sharply triangular or gently rounded and softly angled, reaching up to 3 m tall in certain species or even higher. Roots penetrate down to 70–80 cm, allowing for more aeration in the root zone. Scirpus validus roots, on the other hand, hardly penetrate 10–30 cm in manmade wetlands (Vymazal, 2013). However, unlike other standing aquatic species, the plants have a great capacity for nutrient removal, especially in wetlands receiving nutrient-rich water (Okurut, 2000). The N and P standing stock values range from 12.5 to 585 g m^{-2} and 1.8 to 112.5 g m^{-2}, respectively (Kadlec & Wallace, 2008). Previous research has found that macrophytes play a significant role in pollutant removal. COD and BOD reductions of about 89% were found in planted systems, on the other hand in control systems the reduction rate is about 85% (Karathanasis et al., 2003). Bachard and Horne (1999) found higher removal of nitrogen in a Typha sp. wetland (565 mg N $m^{-2}d^{-1}$) compared to a Scirpus sp. wetland (261 mg N m^{-2} d^{-1}) in California.

2.9 Review of various FWS wetland systems treating or polishing municipal wastewater

There has been significant research done on using wetlands to treat wastewater (Hammer, 1989; Jørgensen & Mitsch, 1989; Kadlec & Wallace, 2008; Moshiri, 2019; Reed et al., 2005; Sedlak, 1991; Tchobanoglous & Burton, 1991). For more than 35 years, treatment wetlands have been implemented in Canada to clean wastewater and stormwater (Kadlec et al., 2012). In North America, free-surface wetlands are the most common type of wetland used for wastewater treatment (Brix, 1994).

Table 2.4: Summary of inlet concentrations and percentage removal for FWS wetlands

Location		TSS	BOD	COD	TN	TP	HRT	References
Full-scale studies								
Brighton, Ontario [a]	IN (mg/L)	13.2	5.4		13.6*	0.378	9.4 d	(Kadlec et al., 2012)
	%Removal	45%	41%		18%*	33%		
Garip, Italy [b]	IN (mg/L)	92	292	616	36	6.6	29 d	(Gunes et al., 2012)
	%Removal	66%	90%	92%	50%	35%		
Pompea, Greece [b]	IN (mg/L)	36	39	100	25*	9.1	5 -14 d	(Tsihrintzis et al., 2007)
	%Removal	84%	80%	82%	28%*	32%		
Tianjin, China [a]	IN (mg/L)	97	119.9		39.6	3.30	1.5 - 3 d	(Li & Jiang, 1995)
	%Removal	80%	85%		51%	70%		
Pilot-scale studies								
Truro, NS [b]	IN (mg/L)	48.3	132.9		36.6*	3.5	25 d	(Boutilier et al., 2010)
	%Removal	78%	69%		46%*	39%		
Sri Lanka [b]	IN (mg/L)	162.7	60.4			1.68	18 hr	(Jinadasa et al., 2006)
	%Removal	76%	54%			15%		
Miho, Japan [c]	IN (mg/L)	150	200	140	100	10	6 d	(C. Liu et al., 2009)
	%Removal	98%	90%	82%	47%	53%		
Laboratory-scale study								
Kerala, India [c]	IN (mg/L)	400	511.1	537.6	128*		16 d	(Midhun et al., 2016)
	%Removal	94%	84%	87%	37%*			
This study [a]	IN (mg/L)	116		126	6.6	1.2	40 d	
	%Removal	97%		57%	51%	83%		

[a] FWS wetland after lagoon treatment
[b] FWS wetland after septic tank treatment
[c] FWS wetland using raw wastewater
* Nitrogen in the form of TKN

A long-term study was conducted in Brighton, Ontario treating continuous discharge municipal lagoon effluent with a FWS wetland (Kadlec et al., 2012). The system had an average flow rate of

3072 m^3/d and 9.4 days of hydraulic retention time and was operated under an ice layer during the winter months. The inlet concentrations were observed as 13.2 mg/L TSS, 5.4 mg/L BOD, 13.6 mg/L TKN, and 0.378 mg/L TP. The wetland resulted in 45% TSS, 41% BOD, 18% TKN, and 33 % TP removal efficiency. Also, they discovered that the constructed wetland may be used throughout the year in a relatively cold region, but the spring growing season saw the most removals except TP. The cold temperatures slow down the treatment due to a lack of microbial processes and a lack of oxygen transfer under ice and snow. Also, vegetation plays important role in the spring and summer seasons.

Gunes et al. (2012) conducted a full-scale study of the FWS wetland situated in Garip village near Lake E ˘girdir, Turkey with a flow rate of 462 m^3/d and around 29 days of HRT. The average concentrations at the FWS wetland inlet were 92 mg/L for TSS, 292 mg/L for BOD, 616 mg/L for COD, 36 mg/L for TN, and 6.6 mg/L for TP. The study revealed that the system removed approximately 66%, 90%, 92%, 50%, and 35% of TSS, BOD, COD, TN, and TP from the high-strength domestic wastewater, respectively. The full-scale FWS wetland system located in Pompeo, Crete, South Greece was investigated by Tsihrintzis et al. (2007). The flow rate of the wetland system was 144 m^3/d and retention time was varying between 5-14 days. The influent concentrations were 36 mg/L, 39 mg/L, 100 mg/L, 25 mg/L, and 9.1 mg/L for TSS, BOD, COD, TKN, and TP respectively. The system resulted in 84% TSS, 80% BOD, 82% COD, 28% TKN, and 32% TP removal efficiency. They also concluded that the FWS system performed well in BOD, COD, and TSS removal compared to a vertical subsurface wetland system.

Two small-scale FWS domestic wastewater treatment wetland located in Truro, Nova Scotia was investigated by Boutilier et al. (2010) between October 2007 and April 2009. Each wetland had a 100 m^2 surface area, was loaded with 1,400 L/d of domestic septic tank effluent and had an estimated hydraulic retention time (HRT) of 25 days. The average influent concentrations were observed as 48 mg/L TSS, 133 mg/L BOD, 37 mg/L TKN, 3.5 mg/L TP, and 2.2 mg/L SRP. Both wetlands in total were reduced by 78% TSS, 69% BOD, 46% TKN, 39% TP, and 35% SRP.

The Tianjin Institute for Environmental Protection located in China explored the effectiveness of FWS CW for municipal sewage treatment. 97 mg/L TSS, 120 mg/L BOD, 40 mg/L TN, and 3.3 mg/L TP were observed at the inlet of the wetland. The treatment results for CW in Tianjin for TSS, BOD, TN, and TP are respectively 80%, 85%, 51%, and 70% at a flow rate of 200 m^3/d and a retention time of 1.5 to 3.0 days (Li & Jiang, 1995). Jinadasa et al. (2006) examined the performance of three pilot-scale free water surface constructed wetlands in treating domestic wastewater in tropical climatic conditions. The wetland was built to treat domestic wastewater in Sri Lanka with a flow rate of 13 m^3/d and 18h of HRT. The raw influent wastewater concentrations were 162.7 mg/L for TSS, 60.4 mg/L for BOD, 13.3 mg/L for NO_3, 1.8 mg/L for TP, and 1.68 for SRP. The study resulted that the system has the potential for removing 76% TSS, 54.3% BOD, 58.6% NO_3, 38.8% TP, and 14.9% SRP. Furthermore, Liu et al. (2009) studied a pilot-scale FWS wetland located in Miho village, Japan, and operated using the influent wastewater flow of 500 L/d with a hydraulic retention time (HRT) of six days. The influent concentrations were observed as 150 mg/L, 200 mg/L, 140 mg/L, 100 mg/L, 88 mg/L and 10 mg/L for TSS, BOD, COD, TN, NH_4^+, and TP respectively. The removal efficiency for TSS, BOD, COD, TN, NH_4^+, and TP was 97.7%, 90.1%, 81.7%, 46.7%, 46.9%, and 52.5%, respectively.

Özengin & Elmaci (2014) investigated an FWS wetland on a laboratory scale in Turkey with weak domestic wastewater (COD-250 mg/L) and strong domestic wastewater (COD-500 mg/L) with 3 days of retention time. Both wastewaters were examined for planted (with *Lemna minor L.*) and unplanted systems in a reactor with a surface area of 1200 cm^2 and a water depth of 6 cm. The average removal efficiencies obtained from the evaluation of the system in the weak domestic wastewater treatment were 60.29 and 57.88 % for COD, 16.78 and 11.73 % for BOD, 31.09 and 13.1 % for total nitrogen, 36.61 and 17.53 % for total phosphorus, 36.4 and 16 % for orthophosphate in planted and unplanted reactors, respectively. However, the average removal efficiencies obtained from the evaluation of the system in the strong domestic wastewater treatment were 91.42 and 87.67 % for COD, 79.09 and 69.92 % for BOD, 72 and 63.6 % for total nitrogen, 66.62 and 54.79 % for total phosphorus, 61.82 and 51.7% for orthophosphate in planted and unplanted reactors, respectively. The wastewater treatment efficiency of a constructed wetland system using a laboratory-scale model with 16 days of HRT and treating domestic wastewater was examined by Midhun et al. (2016) in Kerala, India. The inlet concentrations were 400 mg/L for TSS, 511.1 mg/L for BOD, 537.6 mg/L for COD, 128 mg/L for TKN, and 91.4 mg/L for NO_3. The surface flow constructed wetland system showed a removal efficiency of 94%, 84%, 87%, 37%, and 98% respectively for TSS, BOD, COD, TKN, and NO_3.

2.9.1 Discussion

The removal percentages of TSS and BOD are often high in full-scale FWS-constructed wetlands that treat domestic wastewater, although the removal percentages of nutrients (N and P) are typically low and mostly variable (Doku & Heinke, 1995; Gunes et al., 2012; Kadlec et al., 2012; Li & Jiang, 1995; Tsihrintzis et al., 2007). However, the research conducted in Brighton wetlands resulted in only 45% of TSS removal due to very low inlet concentrations and they are also hitting natural background concentrations (Kadlec et al., 2012). The removal of TSS in all other full-scale and pilot-scale studies listed above was around 80%. The full-scale studies in Canada resulted in a variation in the BOD removal rate from 34% to 98% (Boutilier et al., 2010; Doku & Heinke, 1995). While the full-scale studies besides Canada resulted in better performance in terms of BOD removal rates ranging from 84 to 94% (Gunes et al., 2012; Li & Jiang, 1995; Tsihrintzis et al., 2007). Very few researchers investigated the performance of COD removal in FWS wetlands. The full-scale study in Mediterranean countries observed above 90% of COD removal (Gunes et al., 2012; Tsihrintzis et al., 2007), while 75% of COD removal resulted in research investigated by Doku & Heinke (1995) in North-west territories in Canada. The laboratory studies resulted in better performance for BOD and COD removal efficiency (Midhun et al., 2016).

The N removal in full-scale and pilot-scale FWS wetlands by most of the authors was lower than 50% (Kadlec et al., 2012; Liu et al., 2009; Midhun et al., 2016). A very high range of TP removal efficiency (33-97%) in full-scale FWS wetlands was found in the literature (Doku & Heinke, 1995; Gunes et al., 2012; Kadlec et al., 2012; Li & Jiang, 1995; Tsihrintzis et al., 2007). The laboratory-scale study conducted by Özengin & Elmaci (2014) concluded that FWS wetland performed better, and the removal efficiency of nutrients was also high with a higher range of domestic wastewater as compared to weak domestic wastewater. The pilot-scale and laboratory studies observed a low amount of N and P removal (Jinadasa et al., 2006; Midhun et al., 2016; Özengin & Elmaci, 2014), while full-scale wetlands performed well in P removal.

2.10 Nutrient sinks and mass balance in FWS wetlands

The attenuation of organic compounds in wetland systems includes complex interactions between physical, chemical, and biological processes (Imfeld et al., 2009). All these activities take place in the many wetland compartments, including water, plants, algae, bacteria, sediments, and soil. Both nitrogen and phosphorus can exist in a variety of states (particulate, dissolved, organic, inorganic), and the various activities that occur inside the wetland compartments react differently to each of these states. Mass balance methods can be used to characterize the accumulation of N and P in the different wetland storage compartments.

The deposit of nitrogen as organic matter in the sediments may lead to nitrogen storage in constructed wetlands (Bastviken, 2006). The settling of solid particles, mortality of the below-ground biota, and litterfall cause the deposition and accumulation of nitrogen in constructed wetland soils (Lee et al., 2014). Wetland soils and sediments usually contain most of the phosphorus that is present. The remaining phosphorus is found in plants and litter, with very little mass being found in microorganisms, algae, or water (Kadlec & Wallace, 2008; Vymazal, 2007). The long-term storage of phosphorus mainly depends on soil and sediment accumulation (Dolan et al., 1981; Reddy et al., 1999). N accumulation in different wetland sediments ranged from 15.9 to 26.6 g/m^2 (Wu et al., 2012). Moreover, Dunne et al. (2007) studied that in comparison to the sum of all other ecosystem compartments (above-ground plant biomass, plant litter, and below-ground plant biomass), phosphorus storage in an isolated emergent marsh wetland topsoil (0-10 cm) was highest and more than 87 %.

A mass balance study was conducted by Silbernagl (2017) on a three-cell surface flow wetland following the primary treatment of treating domestic wastewater located in the Eastern Cape province of South Africa. The inlet and outlet concentrations were recorded for four months from April to September. The nitrogen storage in the wetland was 450 kg with 138 kg in plants and 312 kg in sediments, while the phosphorus storage was 57 kg with 13 kg in plants and 44 kg in sediments (Silbernagl, 2017). Lee et al. (2014) studied the nitrogen mass balance in the FWS system for 4 years from 2008 to 2012 treating piggery wastewater. The average nitrogen concentration at the influent was 37,819 kg/year, and roughly 45% of that quantity was left in the effluent. This resulted in denitrification accounting for 34% of the net nitrogen input, while sediment accretion made up approximately 7% of the total input. Only 1% of the total nitrogen load was held in the biomass of plants (Lee et al., 2014). The FWS wetland treating a slightly polluted river resulted in nitrogen removal by plants being 8-34 % of nitrogen input, whereas sediment storage provided 21-34% of the nitrogen removal, according to the mass balance concept (Wu et al., 2012). Research on a FWS wetland treating piggery wastewater reported a phosphorus mass balance for 2 years from 2008 to 2010. During the monitoring period, the average inflow and outflow phosphorus loads were 1,167 kg/yr and 408 kg/yr, respectively. The average phosphorus retention rate was 65%, and the main accumulation was in sediments with 30% of the total. Less than 1% of phosphorus was taken up by plants (Lee et al., 2012). However, no similar and detailed study treating municipal lagoon wastewater was found for phosphorus and nitrogen mass balance.

2.11 Summary

Small rural communities need low-cost and easy-to-maintain wastewater treatment solutions. Constructed wetlands may prove to be a cost-effective, low-maintenance, natural alternative that is both aesthetically pleasing and provides habitat for wildlife. Several studies investigated that free surface wetlands are useful to treat municipal wastewater at some level. The studies on FWS wetlands demonstrate that suspended solid and BOD removal percentages are generally high, however nutrient (N and P) removal percentages are often low and more inconsistent. Moreover, few studies analyzed the COD and TN removal efficiency in FWS wetlands treating domestic wastewater. Aquatic plants uptake nutrients, metals, and other contaminants and play a significant role in reducing nutrient concentrations. The literature review suggests that considerable lack of research on free surface wetlands based on temperature and system age effects. As well, there are few published reports on CW applications in cold climates.

This research is capturing effects and changes in surface wetland removal efficiency due to age and temperature and is characterizing long-term storage and nutrient fractioning in the different wetland compartments. This study may increase our understanding of the ability and performance of treatment wetland systems to remove nutrients in a seasonal polishing wetland operating in a colder climate region.

2.12 References

Abou-Elela, S. I. (2019). Constructed Wetlands: The Green Technology for Municipal Wastewater Treatment and Reuse in Agriculture. Handbook of Environmental Chemistry, 75, 189–239. https://doi.org/10.1007/698_2017_69

APAI. (1995). The use of constructed wetlands for protection of water quality in water supply reservoirs.

Bastviken, S. (2006). Nitrogen removal in treatment wetlands-Factors influencing spatial and temporal variations. Dissertation, Linköping University.

Bendoricchio, G., Cin, L. D., & Persson, J. (2000). Guidelines for free water surface wetland design. EcoSys Bd., 8, 51-91.

Bosnina, M. (2021). Characterising and modeling pollutant dynamics in urban stormwater constructed wetlands.

Boutilier, L., Jamieson, R., Gordon, R., Lake, C., & Hart, W. (2010). Performance of surface-flow domestic wastewater treatment wetlands. Wetlands, 30(4), 795–804. https://doi.org/10.1007/S13157-010-0067-1

Brix, H. (1994). Functions of Macrophytes in Constructed Wetlands. Water Science and Technology, 29(4), 71–78. https://doi.org/10.2166/WST.1994.0160

Cameron, K., Madramootoo, C., Crolla, A., & Kinsley, C. (2003). Pollutant removal from municipal sewage lagoon effluents with a free-surface wetland. Water Research, 37(12), 2803–2812. https://doi.org/10.1016/S0043-1354(03)00135-0

Canadian Environmental Sustainability Indicators. (2020). MUNICIPAL WASTEWATER TREATMENT CANADIAN ENVIRONMENTAL SUSTAINABILITY INDICATORS. www.canada.ca/en/environment-climate-change/services/environmental-indicators/municipal-

Carpenter, S. R., Caraco, N. F., Correll, D. L., Howarth, R. W., Sharpley, A. N., & Smith, V. H. (1998). Nonpoint pollution of surface waters with phosphorus and nitrogen. http://repository.si.edu/xmlui/handle/10088/18030

Constructed Wetlands - Water Canada. (2009). https://www.watercanada.net/feature/constructed-wetlands/

Cooper, P. F., Job, G. D., Green, M. B. and, & Shutes, R. B. E. (1996). Reed Beds and Constructed Wetland for Wastewater Treatment. WRc Swindon, UK.

Crites, R. W. (1994). Design Criteria and Practice for Constructed Wetlands. Water Science and Technology, 29(4), 1–6. https://doi.org/10.2166/WST.1994.0144

de Jong, J. (1976). The purification of wastewater with the aid of rush or reed ponds. In: Tourbier J, Pierson RW (eds) Biological control of water pollution. Pennsylvania University Press, Philadelphia, pp 133–139

Doku, I. A., & Heinke, G. W. (1995). Potential for Greater Use of Wetlands for Waste Treatment in Northern Canada. Journal of Cold Regions Engineering, 9(2), 75–88. https://doi.org/10.1061/(ASCE)0887-381X(1995)9:2(75)

Dolan, T. J., Bayley, S. E., Zoltek, J., & Hermann, A. J. (1981). Phosphorus Dynamics of a Florida Freshwater Marsh Receiving Treated Wastewater. The Journal of Applied Ecology, 18(1), 205. https://doi.org/10.2307/2402490

Dotro, G., Langergraber, G., Molle, P., Nivala, J., Puigagut, J., Stein, O., & von Sperling, M. (2017). TREATMENT WETLANDS.

Dotro, G., Molle, P., Nivala, J., Puigagut, J., & Stein, O. (2017). Treatment Wetlands - Biological Wastewater Treatment Series - Volume 7. 7(November), 172.

Dunne, E. J., Smith, J., Perkins, D. B., Clark, M. W., Jawitz, J. W., & Reddy, K. R. (2007). Phosphorus storages in historically isolated wetland ecosystems and surrounding pasture uplands. Ecological Engineering, 31(1), 16–28. https://doi.org/10.1016/J.ECOLENG.2007.05.004

Economopoulou, M. A., & Tsihrintzis, V. A. (2004). Design Methodology of Free Water Surface Constructed Wetlands. In Water Resources Management (Vol. 18). Kluwer Academic Publishers.

Federation of Canadian Municipalities. (2004). OPTIMIZATION OF LAGOON OPERATION. https://fcm.ca/sites/default/files/documents/resources/guide/infraguide-optimization-lagoon-operations-mamp.pdf

Fraser, L. H., Bradford, M. E., & Steer, D. N. (2003). Global supply of freshwater: The role of treatment wetlands. International Journal of Environment and Sustainable Development, 2(2), 174–183. https://doi.org/10.1504/IJESD.2003.003327

Gorgoglione, A., & Torretta, V. (2018). Sustainable management and successful application of constructed wetlands: A critical review. Sustainability (Switzerland), 10(11). https://doi.org/10.3390/SU10113910

Gottschall, N., Boutin, C., Crolla, A., Kinsley, C., & Champagne, P. (2007). The role of plants in the removal of nutrients at a constructed wetland treating agricultural (dairy) wastewater, Ontario, Canada. Ecological Engineering, 29(2), 154–163. https://doi.org/10.1016/J.ECOLENG.2006.06.004

Gunes, K., Tuncsiper, B., Ayaz, S., & Drizo, A. (2012). The ability of free water surface constructed wetland system to treat high strength domestic wastewater: A case study for the

Mediterranean. Ecological Engineering, 44, 278–284. https://doi.org/10.1016/j.ecoleng.2012.04.008

Hammer, D. A. (1989). Constructed wetlands for wastewater treatment. Constructed Wetlands for Wastewater Treatment.

Hassan, I., Chowdhury, S. R., Prihartato, P. K., & Razzak, S. A. (2021). processes Wastewater Treatment Using Constructed Wetland: Current Trends and Future Potential. https://doi.org/10.3390/pr

Imfeld, G., Braeckevelt, M., Kuschk, P., & Richnow, H. H. (2009). Monitoring and assessing processes of organic chemicals removal in constructed wetlands. Chemosphere, 74(3), 349–362. https://doi.org/10.1016/J.CHEMOSPHERE.2008.09.062

Iqbal, K., Sharma, N., Takkar, S., Shukla, S., Shukla, K., Varma, A., & Mishra, A. (2022). Integrated CO2 sequestration, wastewater treatment, and biofuel production by microalgae culturing: Needs and limitations. Integrated Environmental Technologies for Wastewater Treatment and Sustainable Development, 217–240. https://doi.org/10.1016/B978-0-323-91180-1.00027-2

Jenssen, P. D., Maehlum, T., & Krogstad, T. (1993). Potential Use of Constructed Wetlands for Wastewater Treatment in Northern Environments. Water Science and Technology, 28(10), 149–157. https://doi.org/10.2166/WST.1993.0223

Jinadasa, K. B. S. N., Tanaka, N., Mowjood, M. I. M., & Werellagama, D. R. I. B. (2006). Chemistry and Ecology Free water surface constructed wetlands for domestic wastewater treatment: A tropical case study Free water surface constructed wetlands for domestic wastewater treatment: A tropical case study. Chemistry and Ecology, 22(3), 181–191. https://doi.org/10.1080/02757540600658849

Jørgensen, S. E., & Mitsch, W. J. (1989). Ecological engineering principles. Ecological Engineering: An Introduction to Ecotechnology, 21–37.

Kadlec, R. H. (1995). Overview: Surface flow constructed wetlands. Water Science and Technology, 32(3), 1–12. https://doi.org/10.1016/0273-1223(95)00599-4

Kadlec, R. H. (2003). Effects of pollutant speciation in treatment wetlands design. Ecological Engineering, 20(1), 1–16. https://doi.org/10.1016/S0925-8574(02)00118-0

Kadlec, R. H. (2010). Phosphorus Removal in Emergent Free Surface Wetlands. Journal of Environmental Science and Health, 40(6–7), 1293–1306. https://doi.org/10.1081/ESE-200055832

Kadlec, R. H., Pries, J., & Lee, K. (2012). The Brighton treatment wetlands. Ecological Engineering, 47, 56–70. https://doi.org/10.1016/J.ECOLENG.2012.06.042

Kadlec, R., & Wallace, S. (2008). Treatment wetlands. https://books.google.com/books?hl=en&lr=&id=hPDqfNRMH6wC&oi=fnd&pg=PP1&ots=k8M07QbT6N&sig=sfZ5zY6zFfG83Xs3yYO1Eok28Rs

Karathanasis, A. D., Potter, C. L., & Coyne, M. S. (2003). Vegetation effects on fecal bacteria, BOD, and suspended solid removal in constructed wetlands treating domestic wastewater. Ecological Engineering, 20(2), 157–169. https://doi.org/10.1016/S0925-8574(03)00011-9

Lee, C. Gyun., Fletcher, T. D., & Sun, Guangzhi. (2009). Nitrogen removal in constructed wetland systems. Engineering in Life Sciences, 9(1), 11–22. https://doi.org/10.1002/ELSC.200800049

Lee, S., Maniquiz-Redillas, M. C., Choi, J., & Kim, L. H. (2014). Nitrogen mass balance in a constructed wetland treating piggery wastewater effluent. Journal of Environmental Sciences (China), 26(6), 1260–1266. https://doi.org/10.1016/S1001-0742(13)60597-5

Lee, S. Y., Maniquiz, M. C., Choi, J. Y., Kang, J. H., & Kim, L. H. (2012). Phosphorus mass balance in a surface flow constructed wetland receiving piggery wastewater effluent. Water Science and Technology, 66(4), 712–718. https://doi.org/10.2166/WST.2012.231

Li, X., & Jiang, C. (1995). Constructed wetland systems for water pollution control in North China. Water Science and Technology, 32(3), 349–356. https://doi.org/10.2166/WST.1995.0157

Liu, C., Xu, K., Inamori, R., Ebie, Y., Liao, J., & Inamori, Y. (2009). Pilot-scale studies of domestic wastewater treatment by typical constructed wetlands and their greenhouse gas emissions. https://doi.org/10.1007/s11783-009-0155-8

Liu, Y., Jiang, M., Lu, X., Lou, Y., & Liu, B. (2017). Carbon, Nitrogen and Phosphorus Contents of Wetland Soils in Relation to Environment Factors in Northeast China. Wetlands, 37(1), 153–161. https://doi.org/10.1007/S13157-016-0856-2

Mæhlum, T., Jenssen, P. D., & Warner, W. S. (1995). Cold-climate constructed wetlands. Water Science and Technology, 32(3), 95–101. https://doi.org/10.1016/0273-1223(95)00609-5

Mann, R. A. (1997). Phosphorus adsorption and desorption characteristics of constructed wetland gravels and steelworks by-products. Soil Research, 35(2), 375–384. https://doi.org/10.1071/S96041

Midhun, G., Divya, L., George, J., Jayakumar, P., & Suriyanarayanan, S. (2016). Wastewater treatment studies on free water surface constructed wetland system. Environmental Science and Engineering (Subseries: Environmental Science), 9783319272269, 97–109. https://doi.org/10.1007/978-3-319-27228-3_9/FIGURES/11

Moshiri, G. A. (2019). Constructed wetlands for water quality improvement. CRC Press, Taylor & Francis Group. https://www.routledge.com/Constructed-Wetlands-for-Water-Quality-Improvement/Moshiri/p/book/9780367449681

Okurut, T. (2000). A PILOT STUDY ON MUNICIPAL WASTEWATER TREATMENT USING A CONSTRUCTED WETLAND IN UGANDA. http://www.balkema.nl

Omondi, D. O., & Navalia, A. C. (2020). Constructed Wetlands in Wastewater Treatment and Challenges of Emerging Resistant Genes Filtration and Reloading. Inland Waters - Dynamics and Ecology. https://doi.org/10.5772/INTECHOPEN.93293

Özengin, N., & Elmaci, A. (2014). Free water surface (FWS) system for the treatment of domestic wastewater: A comparative study. Asian Journal of Chemistry, 26(20), 6957–6963. https://doi.org/10.14233/AJCHEM.2014.17485

Picard, C. R., Fraser, L. H., & Steer, D. (2005). The interacting effects of temperature and plant community type on nutrient removal in wetland microcosms. Bioresource Technology, 96(9), 1039–1047. https://doi.org/10.1016/J.BIORTECH.2004.09.007

Pries, J. H. (1994). Wastewater and stormwater applications of wetlands in Canada. North American Wetlands Conservation Council (Canada). Environment Canada. Canadian Wildlife Service., 66.

Rana, V., & Maiti, S. K. (2020). Municipal and Industrial Wastewater Treatment Using Constructed Wetlands. 329–367. https://doi.org/10.1007/978-3-030-00099-8_10

Reckhow, K. H., & Simpson, J. T. (2011). A Procedure Using Modeling and Error Analysis for the Prediction of Lake Phosphorus Concentration from Land Use Information. Https://Doi.Org/10.1139/F80-184, 37(9), 1439–1448. https://doi.org/10.1139/F80-184

Reddy, K. R., Kadlec, R. H., Flaig, E., & Gale, P. M. (1999). A Review Phosphorus Retention in Streams and Wetlands: A Review. Critical Reviews in Environmental Science and Technology, 29(1), 83–146. https://doi.org/10.1080/10643389991259182

Reed, S. C., Crites, R. W., & Middlebrooks, E. J. (2005). Natural Wastewater Treatment Systems. 576. https://books.google.com/books/about/Natural_Wastewater_Treatment_Systems.html?id=K6k8yE8rw0MC

Richardson, C. J. (1985). Mechanisms Controlling Phosphorus Retention Capacity in Freshwater Wetlands. Science, 228(4706), 1424–1427. https://doi.org/10.1126/SCIENCE.228.4706.1424

Richardson, C. J., & Marshall, P. E. (1986). Processes Controlling Movement, Storage, and Export of Phosphorus in a Fen Peatland. Ecological Monographs, 56(4), 279–302. https://doi.org/10.2307/1942548

Sedlak, Richard. (1991). Phosphorus and nitrogen removal from municipal wastewater: principles and practice. 240.

Silbernagl, R. (2017). An assessment of the effectiveness of the Crossways Farm Village constructed wetland in the treatment of domestic wastewater.

Smith, V. H., Tilman, G. D., & Nekola, J. C. (1999). Eutrophication: impacts of excess nutrient inputs on freshwater, marine, and terrestrial ecosystems. Environmental Pollution, 100(1–3), 179–196. https://doi.org/10.1016/S0269-7491(99)00091-3

Stefanakis, A., Akratos, C. S., & Tsihrintzis, V. A. (2014). Vertical Flow Constructed Wetlands: Eco-engineering Systems for Wastewater and Sludge Treatment. Vertical Flow Constructed Wetlands: Eco-Engineering Systems for Wastewater and Sludge Treatment, 1–378. https://doi.org/10.1016/C2012-0-01288-4

Stefanakis, A., & Headley, T. (2020). FWS WETLANDS. https://www.researchgate.net/publication/339472902_FWS_WETLANDS

Tchobanoglous, George., & Burton, F. (1991). Wastewater engineering: treatment, disposal, and reuse (3rd ed. /). McGraw-Hill.

Trang, N. T. D., Konnerup, D., Schierup, H. H., Chiem, N. H., Tuan, L. A., & Brix, H. (2010). Kinetics of pollutant removal from domestic wastewater in a tropical horizontal subsurface flow constructed wetland system: Effects of hydraulic loading rate. Ecological Engineering, 36(4), 527–535. https://doi.org/10.1016/J.ECOLENG.2009.11.022

Truu, M., Juhanson, J., & Truu, J. (2009). Microbial biomass, activity, and community composition in constructed wetlands. Science of The Total Environment, 407(13), 3958–3971. https://doi.org/10.1016/J.SCITOTENV.2008.11.036

Tsihrintzis, V. A., Akratos, C. S., Gikas, G. D., Karamouzis, D., & Angelakis, A. N. (2007). Performance and cost comparison of a FWS and a VSF constructed wetland system. Environmental Technology, 28(6), 621–628. https://doi.org/10.1080/09593332808618820

USEPA. (2000). Free Water Surface Wetlands. Wastewater Technology Fact Sheet.

U.S.EPA. (2000). Wastewater Technology Fact Sheet Free Water Surface Wetlands.

USEPA. (2002). Facultative Lagoons. Wastewater Technology Fact Sheet.

Uusheimo, S., Huotari, J., Tulonen, T., Aalto, S. L., Rissanen, A. J., & Arvola, L. (2018). High Nitrogen Removal in a Constructed Wetland Receiving Treated Wastewater in a Cold Climate. Environmental Science and Technology, 52(22), 13343–13350. https://doi.org/10.1021/ACS.EST.8B03032/SUPPL_FILE/ES8B03032_SI_001.PDF

Vitousek, P. M., Aber, J. D., Howarth, R. W., Likens, G. E., Matson, P. A., Schindler, D. W., Schlesinger, W. H., & Tilman, D. G. (1997). HUMAN ALTERATION OF THE GLOBAL NITROGEN CYCLE: SOURCES AND CONSEQUENCES. In Ecological Applications (Vol. 7, Issue 3).

Vymazal, J. (2007). Removal of nutrients in various types of constructed wetlands. Science of the Total Environment, 380(1–3), 48–65. https://doi.org/10.1016/J.SCITOTENV.2006.09.014

Vymazal, J. (2010). Constructed Wetlands for Wastewater Treatment. Water 2010, Vol. 2, Pages 530-549, 2(3), 530–549. https://doi.org/10.3390/W2030530

Vymazal, J. (2013). Emergent plants used in free water surface constructed wetlands: A review. Ecological Engineering, 61, 582–592. https://doi.org/10.1016/j.ecoleng.2013.06.023

Vymazal, J., & Kröpfelová, L. (2008). Wastewater Treatment in Constructed Wetlands with Horizontal Sub-Surface Flow. 14. https://doi.org/10.1007/978-1-4020-8580-2

Waara, S., Gajewska, M., Cruz Blázquez, V., Alsbro, R., & Norwald, P. (2015). Long term performance of a FWS wetland for post-tertiary treatment of the sewage-the influence of flow, temperature and age on nitrogen removal. 13–18. http://www.diva-portal.orghttp://urn.kb.se/resolve?urn=urn:nbn:se:hh:diva-30137

Wallace, S., & Knight, R. (2006). Small-Scale Constructed Wetland Treatment Systems: Feasibility, Design Criteria, and O&M Requirements. Feasibility, Design Criteria, and O&M Requirements, 304.

Watson, J. T., Reed, S. C., Kadlec, R. H., Knight, R. L., & Whitehouse, A. E. (2020). Performance Expectations and Loading Rates for Constructed Wetlands. Constructed Wetlands for Wastewater Treatment, 319–351. https://doi.org/10.1201/9781003069850-31

Wu, H., Zhang, J., Wei, R., Liang, S., Li, C., & Xie, H. (2012). Nitrogen transformations and balance in constructed wetlands for slightly polluted river water treatment using different macrophytes. https://doi.org/10.1007/s11356-012-0996-8

Zeng, L., Zhang, H. W., Wu, Y. H., Li, C. F., & Wang, P. (2019). Theoretical and numerical analysis of vertical distribution of active particles in a free-surface wetland flow. Journal of Hydrology, 573, 449–455. https://doi.org/10.1016/J.JHYDROL.2019.03.085

CHAPTER: 3

Long-term Performance of a Free Water Surface Constructed Wetland and Vegetated Filter Treating Municipal Lagoon Effluent

3.1 Abstract

A Free Water Surface (FWS) constructed wetland was established to polish lagoon effluent from an annual (spring) discharge lagoon in the Town of Alfred, Ontario to extend the discharge period from June to November. The wetland system commenced operation in 2000 and operated seasonally for ten years followed by an 11-year dormant period. In 2021, the wetland was restarted with an increased wetland operating depth from 25 to 50 cm. Wetland performance was compared between the Startup Period (Years 1-3), the Mature operation Period (Years 4-10), and the Restart Period. No significant effect of aging or the dormant period was observed on the treatment of organic matter and total phosphorus (TP), but the removal rate of organic nitrogen and nitrate (NO_3^--N) was improved after a dormant period in 2021. The total suspended solids (TSS) removal improved significantly during the Restart Period, likely due to the increase in operating depth. The wetland system resulted in removing 97% TSS, 57% chemical oxygen demand (COD), 83% Organic nitrogen, 37% NO_3^--N, 18% ammonia (NH_4^+-N), 83% TP, and 87% soluble reactive phosphorus (SRP) in the restart period. No significant effects of seasonal temperature were observed on biological oxygen demand (BOD) and NO_3^- removal, while TP removal efficiency was significantly higher from August to October compared with both early summer (June) and late fall (November). First-order kinetic removal rate constants (k) were determined for BOD and nitrogen species. The averaged k values for BOD, TN, NH_4^+, and NO_3^- were 45.8 m/yr, 7.9 m/yr, 15.5 m/yr, and 3.0 m/yr, respectively.

Keywords: FWS wetland, Vegetated filter, Long-term performance, Temperature effects, Kinetic removal rate

3.2 Introduction

Lagoons are particularly well suited to treat wastewater from rural communities as they are less expensive to build and operate than other systems (Steinmann et al., 2003). In Canada, municipal lagoon system discharges are regulated by Provincial governments and more recently by the federal government through the Wastewater Systems Effluent Regulations, which stipulates maximum concentrations of 25 mg/L for TSS and $cBOD_5$ and 1.25 mg/L NH_3 at 15°C ± 1°C (WSER, 2012). However, facultative lagoons alone are often unable to meet increasingly stringent water quality criteria for pollutants (USEPA, 2022). One potential solution is to establish a treatment wetland to polish lagoon effluent (Polprasert et al., 2005). A combined system in North America typically consists of facultative lagoons for primary and secondary treatment, followed by a constructed FWS wetland system (Kadlec et al., 2012).

Wetlands are designed and constructed to treat municipal, industrial, and agricultural wastewater at a low cost by using natural processes including microbial transformation, plant uptake, and soil adsorption. Many studies show the efficiency and performance of treatment wetlands all over the world (Kadlec & Wallace, 2008). Constructed wetlands (CW) are considered among the most appropriate solutions for rural and small communities (Wood et al., 1995), and have been utilized for wastewater treatment in North America since the 1970s (Kadlec et al., 2009; Vymazal, 2010). CWs can remove a variety of pollutants from wastewater, including BOD, COD, TSS, TN, TP, total coliforms, and metals, by microbial degradation, plant absorption, substrate adsorption, filtering by packed media, and biological predation (Kadlec & Wallace, 2008; Saeed & Sun, 2011). Removal efficiencies greater than 80% are typical for TSS and BOD (Jenssen et al., 1993), while large variability has been reported for N and P removal from systems treating domestic and municipal wastewater (Jenssen et al., 1993; Wu et al., 2011) and in several cases failing to meet nitrogen and phosphorus treatment objectives (Fraser et al., 2004a; Greenway, 2005). Changes in macrophyte species and density, media, wastewater type, retention times, loading rates, climatic conditions, temperature, design, and size may also play an important role in the differences between results (Brisson & Chazarenc, 2008; Gearheart, 1992; Tanner, 2001). Liu et al. (2016) suggested that the increases in water depth of a constructed wetland from 20 cm to 60 cm have little effect on improving nutrient removal efficiency. However, the study on FWS wetland reported that a water depth of 20 cm presented the lowest TSS, TN, and TP removal rate and observed better treatment efficiency between 30 to 50 cm of water depth (Guo et al., 2017).

The lack of long-term wetland studies is an important data gap as nutrient removal, and particularly phosphorus removal is not clearly defined over short time frames (Land et al., 2016; Vymazal, 2018). The effects of wetlands maturation (Nilsson et al., 2020b) and seasonality (Land et al., 2016), short-term studies may be insufficient to specify any general conclusions (Nilsson et al., 2020a). Kadlec et al. (2012) studied a FWS wetland located in Brighton, Ontario, Canada treating municipal lagoon wastewater for ten years from 2001 to 2010 with no clear evidence found on the effects of system age. Also, the long-term study from 2002 to 2011 on a FWS wetland treating post-tertiary sewage for the municipality of Eskilstuna, Sweden reported no indication of variation due to the age of the wetland system (Waara et al., 2015). The seasonal and temperature effects on wetland performance remain contentious (Vymazal, 2011). Picard et al. (2005) reported that temperature has a greater impact on nitrogen removal in wetlands than they do on phosphorus removal, which is mostly dependent on sediment adsorption instead of biological activities. However, Mæhlum et al., (1999) reported no effect of temperature on nitrogen, phosphorus, and

BOD treatment efficiency. Also, Jenssen et al., (1993) found no seasonal effects in wetlands situated in Denmark, Sweden, and North America.

A pilot FWS-constructed wetland system was established at an annual discharge lagoon (spring-controlled discharge) for the Town of Alfred, ON, Canada. The objective of the wetland is to polish lagoon effluent to discharge to a receiving brook from June to November, thus increasing storage capacity in the lagoon. The pilot system was operated for 10 years while meeting treatment objectives for BOD and TP but not for TSS, followed by 11 years of dormancy, before re-starting the system in 2021 at a higher operating depth.

The objectives of this study are: (i) to evaluate the performance of a FWS constructed wetland and vegetated filter to polish municipal lagoon effluent (ii) to evaluate the effect of increasing operating depth on system performance in general and on TSS removal in particular, (iii) to investigate the effect of system aging from 1-10 years of operation and the effect of an 11-year dormant period on system performance, (iv) evaluate the effect of seasonal temperature on system performance, and (v) recommend design criteria for seasonal lagoon polishing wetlands.

3.3 Materials and Methods

3.3.1 Site description and operation

The pilot-constructed wetland and vegetated filter are located at the municipal lagoon in the Town of Alfred, Ontario, Canada approximately 70 km east of Ottawa (approx. population 1350) (45º32’36” N, 74º51’32” W). The Town’s wastewater is treated in a two-cell facultative controlled discharge lagoon that discharges once a year in the spring to the Azitica Brook, which later flows into the Ottawa River north of the Town. The lagoon cells were built in the early 1970s. The wetland system was constructed in 1999.

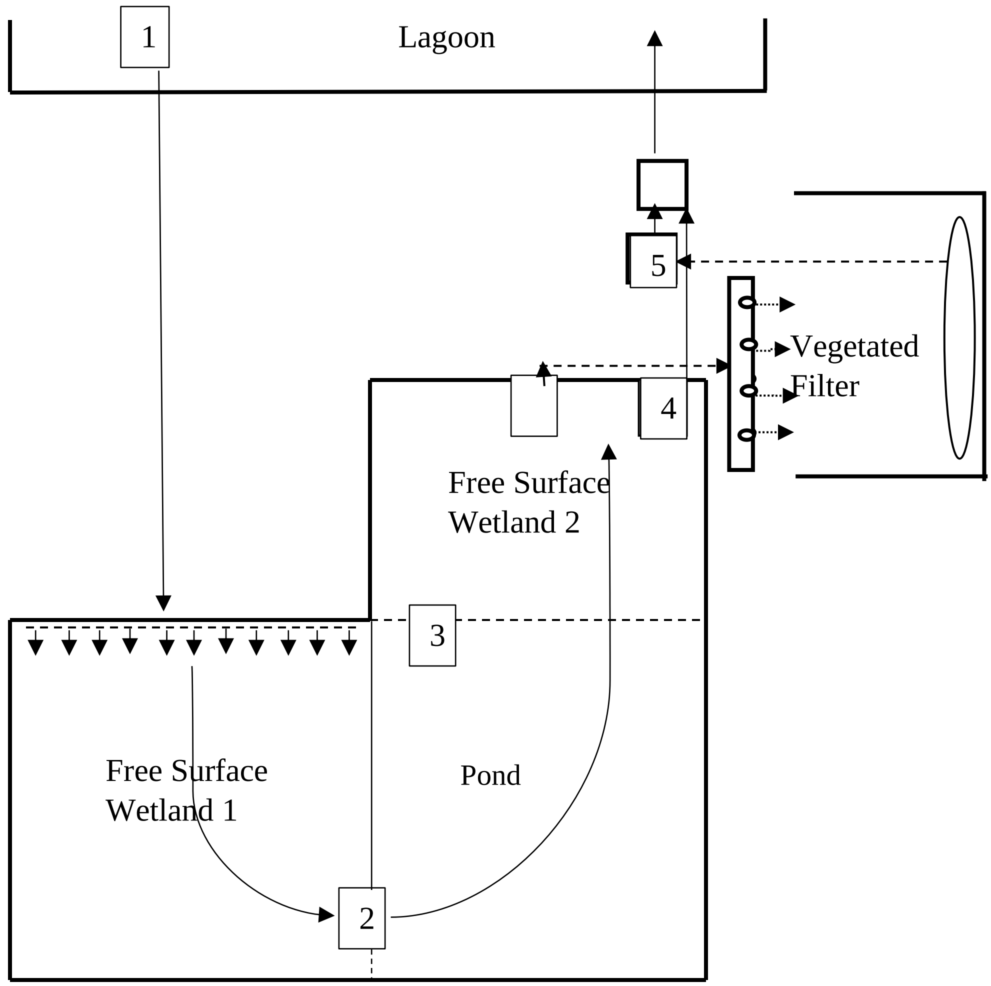


Figure 3.1: *Schematic diagram of Alfred Constructed Wetland System*

The pilot-constructed wetland system consists of three cells in a marsh-pond-marsh configuration. The wetland cells, FSW I and FSW II are shallow basin surface-flow wetlands operating at 25 cm water depth from the year 2000 to 2009. The depth of the wetland was increased to 50 cm in the restart year 2021 after a long dormant period to account for accumulated sediment and to improve TSS effluent quality. The FSW I is 78m in length x 40m in width and FSW II is 39m in length x 40m in width. The pond dimensions are 78m in length x 40m in width. The depth of the pond was 95 cm from 2000-2009 and it averaged 120 cm in the year 2021. A vegetated filter was also established as a post-wetland polishing system. The vegetated strip is 30m length x 20m width at a 2.5% slope. The runoff from the vegetated filter strip was collected at the end of the system in a pond with dimensions of 20m in length x10m in width x 0.5m in depth.

The system was operated from 2000 to 2009 and then after eleven years, it was re-started in 2021. The wetland was dosed from the lagoon with a ¾ HP (Myers WHR7-21C) sewage pump via 7.5 cm ABS piping to a 7.5 cm PVC perforated header pipe across the inlet of the first wetland cell. The vegetated filter strip was dosed with the wetland outlet via 7.5 cm ABS piping to a 5.0 cm PVC perforated pipe across the top of the filter. The wastewater runoff from the vegetated filter was collected in a manhole and pumped back to the lagoon.

3.3.2 Sampling and analysis

3.3.2.1 2021 Operating Season:

The system was restarted at the beginning of July 2021 and operated until freeze-up at the beginning of November. Water samples were collected weekly from five different locations across the system flow profile: Lagoon (L), Wetland 1 outlet (W1), Pond dock, Wetland 2 outlet (W2), and Vegetated filter return (VF) (Figure 3.1).

Grab samples were collected from the middle of the water column using a Nasco swing sampling pole. Grab samples were *in-situ* analyzed for pH, dissolved oxygen, and temperature with a HACH-HQ40D multimeter (LDO101 probe for DO and PHC101 probe for pH).

The collected samples were transported to the Environmental Engineering Research Laboratory for analysis at the University of Ottawa. The samples were stored in the refrigerator at $4 \pm 2^{o}C$. All grab samples were analyzed using the following standard methods for the Examination of Water and Wastewater (APHA, 2012). Total suspended solids (TSS) were analyzed as per method 2540

D. Total phosphorus (TP) and ortho-phosphates (PO_4^{3-}-P) were measured as per method 4500-P B & D (stannous chloride). Ammonia (NH_4^+ -N) and nitrate (NO_3^- -N) were measured according to methods 4500-NH_3 D and 4500- NO_3^- D respectively. Total nitrogen (TN) and Chemical oxygen demand (COD) were measured using the HACH method 10071 and HACH method 8000 respectively.

3.3.2.2 2000-2009 Operating Seasons:

The monitoring program was conducted by the Ontario Rural Wastewater Centre, Campus d'Alfred, University of Guelph. The data remains unpublished except for the first operating year 2000 (Cameron et al., 2003). Grab samples were collected from the middle of the water column using a Nasco swing sampling pole. Grab samples were *in-situ* analyzed for pH, dissolved oxygen, and temperature with a YSI Pro1020 multimeter.

Grab samples were collected weekly to bi-weekly depending on the year and analyzed at the ORWC's environmental quality laboratory at Campus d'Alfred, University of Guelph. The following Standard Methods (APHA, 1998) were used: 5-day BOD (BOD_5) as per method 5210 B; total Kjeldahl nitrogen (TKN) according to method 4500-N_{org} B; NH_4^+ according to method 4500-NH_3 D; NO_3^- according to method 4500-NO_3^- D; TP according to method 4500-P B; orthophosphate (O-PO_4^{3-}) as per method 4500-P E; TSS as per method 2540 D.

3.3.2.3 Monitoring Intensity:

Monitoring program intensity varied across the operating years. Influent (Lagoon) and wetland outlet (W2) samples were collected for all operating years. Additionally, samples were collected at the end of Wetland 1 (W1), Pond, and Vegetated Filter (VF) from 2000 to 2004 and during 2021. Complete datasets were available for BOD_5, TSS, TP, NH_4^+, and NO_3^-, with COD replacing

BOD_5 in 2021 as BOD levels were below the method detection limit. Additionally, OPO_4^{3-} data was available from 2000 to 2002 and 2021, and TN data from 2000-2003 and 2021.

3.3.3 Data Analysis

The correlation between the lagoon and W2 over the monitoring period was analyzed for TSS, BOD, NO_3^-, NH_4^+, and TP. The correlation was not calculated for COD, TN, and SRP due to a lack of data series. The data were divided into three phases to compare any effect of aging on system performance: the Start-up Phase (2000-2002), the Mature Operating Phase (2003-2009), and the Re-start Phase (2021). The aging effect was observed as per the available data for each nutrient. The wetland performance for the aging effect was calculated in terms of concentrations removal (%R) as:

$$\%R = \frac{C_{in} - C_{out}}{C_{in}} \times 100 \qquad \textbf{Equation 3.1}$$

where C_{in} and C_{out} were the concentration of a given contaminant in the inflow and outflow sample.

Moreover, the data collected in the year 2021 at various wetland segments were compared to the historical data from 2000-2004 as this period had a complete dataset including for the VF.

The effect of temperature was considered for BOD_5, TP, and NO_3^--N, the dominant nitrogen species. The Lagoon and W2 data for all monitoring years were divided into three groups of increasing temperature from June – August (15-19, 20-24, and 25-29ºC) and four groups of decreasing temperature from September to November (20-24, 15-19, 10-14, and 0-9ºC) to compare equivalent temperature and seasonal effects.

Wetland kinetic rate constants (k values) were determined using the P-k-C* tank in series (TIS) model by Kadlec & Wallace (2008).

$$\frac{C_o - C^*}{C_i - C^*} = \frac{1}{(1+k/Pq)^P} \quad \textbf{Equation 3.2}$$

Where, C_o = outlet concentration (mg/L), C_i = inlet concentration (mg/L), C^* = background concentration (mg/L), k = first-order areal rate constant (m/yr), P = apparent number of tanks in series, and q = hydraulic loading rate (m/yr).

The equation was used to determine average k values BOD, TN, NH_4^+, and NO_3^- removal. The background concentration (C*) values were used as minimum concentrations observed in this study as 1 mg/L for BOD, 0.6 mg/L for TN, and 0.05 mg/L for NO_3^-. While the inlet concentrations were very low, the C* value for NH_4^+=0 mg/L was found in the literature (Kadlec et al., 2012; Kadlec & Wallace, 2008). Also, the values for P=1 for BOD and P=3 for TN, NH_4^+, and NO_3^- were used for the kinetic model and observed by other studies (Kadlec et al., 2012; Kadlec & Wallace, 2008).

Average data from the meteorological station of Ottawa Intl airport and Montreal Intl airport was used to approximate the precipitation rate and daily mean temperature during the study period. The hydraulic loading rate (HLR) and hydraulic retention time (HRT) were different for both wetland and vegetated systems. The average inflow of wetland was 141 ± 5 m^3/day in all monitoring years resulting in an HLR of 6.6 m^3/m^2/yr. While the VF system was dosed at 44 ± 6 m^3/day from the year 2000 to 2009 and 33 m^3/day in the year 2021. The flow for both systems was assumed independent of the frequency of precipitation rate. The retention times were different before and after the dormant period as the depth increased in the year 2021. From the year 2000 to 2009 the HRT was 15 days and in the year 2021 HRT was 40 days for the wetland system.

3.3.4 Statistical Analysis

Significant differences in parameter concentrations were tested using a one-way analysis of variance (ANOVA) test. Differences were tested between treatment components, operating years (system aging), and time periods (temperature and season). Pearson's Correlation Coefficient was used to compare inlet and outlet data sets as well as the relationship between water quality parameters.

3.4 Results and Discussion

The system was operated from June to November (2000-2009) and restarted in 2021 after an eleven-year dormant period. The inlet and outlet data are presented as time series to observe any long-term trends and are also grouped into three phases of operation to evaluate the effect of the startup period (plant establishment), mature operation, and restart after a dormant period on wetland performance. Longitudinal data across the wetland system is also presented to evaluate the internal performance within the system.

3.4.1 Climatic conditions

The averaged meteorological data from the Ottawa and Montreal Intl Airports were used to approximate the conditions at the research site, as it is roughly equidistant between the two airports. The average monthly precipitation and temperature over the study period are presented in Figure 3.2. Precipitation was quite variable but with consistent average values during the wetland operating period. Average temperatures ranged from a low of -9 ± 3 ºC in January to a high of 22 ± 1ºC in July.

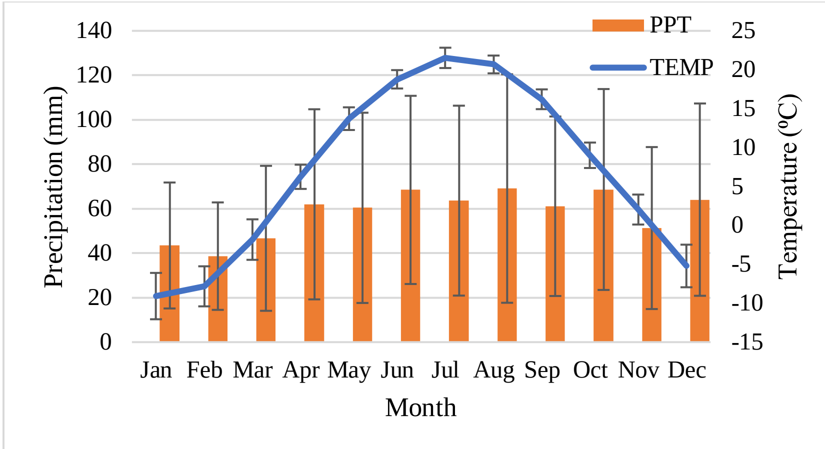


Figure 3.2: *Monthly Average ± STDEV precipitation (mm) and temperature (°C) from 2000 to 2021 (Average of two meteorological climate stations, Ottawa International Airport and Montreal International Airport; Environment and Climate Change Canada)*

3.4.2 Dissolved Oxygen and pH

The water quality parameters of dissolved oxygen (DO) and pH were measured during each sampling event at each sampling location. Average values for the entire study period are presented in Table 3.1 as no significant change was observed over time for either parameter. The inlet (lagoon) and wetland water column remained highly aerobic ranging from 4.7 to 8.9 mg/L, while the pH was neutral to slightly basic ranging from pH 7.4 to pH 8.3. This is consistent with FWS wetlands, where outlet water's pH was just above neutrality in the open water zone providing a high pH environment due to high algal activity. The pH was in the range of $6.5 < pH < 9.0$ required for water discharge limits (Kadlec & Wallace, 2008).

Table 3.1: *DO and pH observations (Mean ± STDEV) at various locations*

Location	DO (mg/L)	pH
Lagoon	4.7 ± 1.2	7.4 ± 0.3
W1	8.2 ± 2.9	8.0 ± 0.5
P	7.9 ± 2.8	8.1 ± 0.4
W2	6.2 ± 1.8	7.5 ± 0.3
VF	8.9 ± 2.9	8.3 ± 0.6

3.4.3 Total Suspended solids (TSS)

No clear trend in TSS removal or TSS outlet concentrations with time was observed (Figure 3.3). Inlet and outlet concentrations varied widely from between 16 ± 3 and 152 ± 84 mg/L for the lagoon and between 3 ± 1 and 55 ± 36 mg/L for the wetland outlet, with outlet concentrations exceeding inlet concentrations for 2 years. No correlation was observed between inlet and outlet concentrations (R^2=0.02). In comparison to the solids entering the system, the solids leaving the system are frequently created from plant detritus and the resuspension of previously settled solids in the wetland (Kadlec & Wallace, 2008).

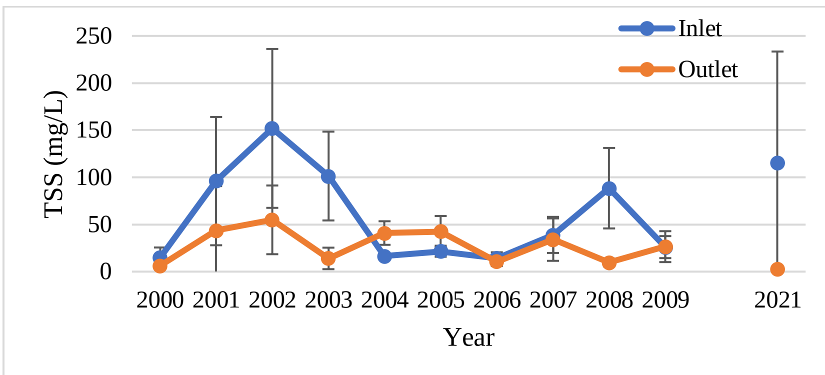


Figure 3.3: *Wetland System Inlet and Outlet TSS (Mean ± 95%CI) concentrations by Year*

No significant differences in outlet TSS concentrations were observed between Startup and Mature Phases with 35 ± 29 and 26 ± 13 mg/L, respectively; however, a significant decrease in TSS outlet concentration was observed in the Restart Phase with 3 ± 1 mg/L ($P<0.05$) (Table 3.2). This is the only year, except during the first year of operation, where the treatment objective of 10 mg/L was met and is likely due to the increase in operating depth in the wetland cells from 25 to 50 cm, reducing the resuspension and export of sediment. Removal efficiencies increased from 41 and 60% during the Startup and Operating Phases to 97% with the increased operating depth during the Restart Phase. Empirical relationships across numerous studies suggest a 75% removal rate for influent TSS concentrations greater than 20 mg/L (Kadlec & Wallace, 2008). In the FWS wetland located at Truro, Nova Scotia treating septic tank effluent with HLR of 5.1 $m^3/m^2/yr$ and influent TSS concentrations of 48.3 mg/L resulted in 78% treatment efficiency (Boutilier et al., 2010). Also, a FWS wetland treating secondary effluent with TSS inlet concentrations of 92 mg/L and HLR of 59.4 $m^3/m^2/yr$ observed 66% TSS removal (Gunes et al., 2012). This study with a hydraulic loading rate of 6.6 $m^3/m^2/yr$ exhibited lower TSS removal efficiencies as compared to literature values during the Startup and Mature periods but higher removal efficiencies during the Restart Phase with an increase in operating depth from 25 to 50 cm.

Table 3.2: *TSS percentage removal at three different phases of wetland*

	Lagoon (Mean ± 95 CI)	**W2 (Mean ± 95 CI)**	**Removal Efficiency (%)**
Startup (2000-2002) †	88 ± 54	35 ± 29	60
Mature (2003-2009) †	44 ± 19	26 ± 13	41
Restart (2021) †	116 ± 118	3 ± 1	97

†Lagoon and W2 difference statistically significant (P<0.05) using ANOVA test

TSS removal through the system compartments exhibited similar trends through all operating phases with a large initial reduction in the first wetland cell of 73-80% followed by further reductions in the Pond cell to achieve 85-90% removal efficiencies (Figure 3.4). This is expected as most TSS should be physically removed through sedimentation and filtration early in the system (Kadlec & Wallace, 2008). However, TSS concentrations increased in the second wetland cell during Startup and Mature Phases, indicating a resuspension of sediment at the 25 cm operating depth. This contrasts with the much lower average concentration observed during the Restart Phase, where the operating depth was increased to 50 cm, again suggesting that increasing the operating depth significantly improved TSS effluent quality.

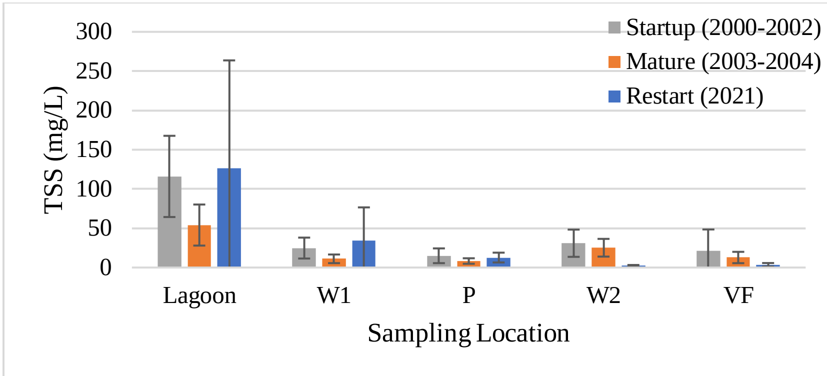


Figure 3.4: *Comparison of TSS (Mean ± 95%CI) concentrations at various wetland segments*

The performance of the vegetated filter was quite variable during both the Startup and Mature Phases with TSS concentrations of 22 ± 27 and 13 ± 7 mg/L, respectively, compared with 4 ± 2 mg/L during the Restart Phase. During the Startup and Mature Phases, barley was grown and harvested from the VF, while during Restart Phase the VF was operated as a grass filter. These results indicate that the barley filter was not capable of reducing TSS to the objective of 10 mg/L, possibly from the soil being disturbed during seeding and harvesting, while the grass filter maintained very low TSS concentrations, suggesting a grass filter could provide an effective TSS polishing step. It is reported that TSS may be reduced to 5-15 mg/L in free-surface wetlands used for secondary treatment of municipal wastewater (Reed et al., 2005) while this study observed an effluent concentration below 15 mg/L only in the Restart phase with a wetland operating depth increased to 50 cm.

3.4.4 Organic Matter

No trend in inlet or outlet BOD concentrations was observed over time with relatively low average lagoon BOD_5 concentrations varying between 3 ± 2 and 21 ± 12 mg/L and wetland outlet concentrations varying between 2 ± 2 and 9 ± 1 mg/L (Figure 3.5). Inlet and outlet concentrations demonstrated a strong correlation (R^2=0.69), with wetland outlet concentrations consistently meeting the treatment objective of 10 mg/L.

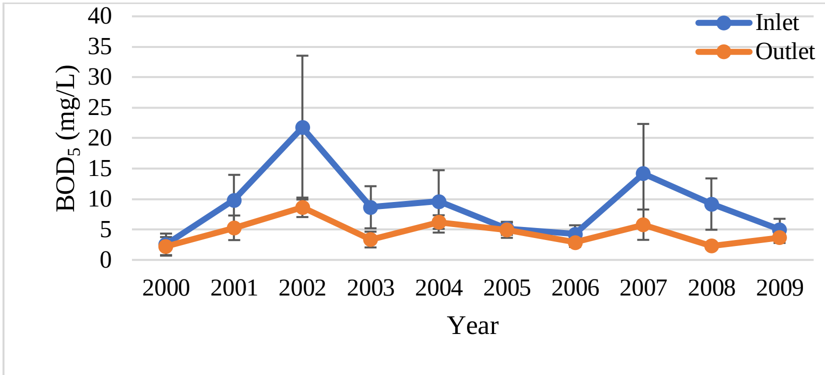


Figure 3.5: *Wetland System Inlet and Outlet BOD_5 (Mean ± 95%CI) concentrations by Year*

No differences in inlet or outlet concentrations were observed between the Startup and Mature Phases with removal efficiencies of 44 and 50%, respectively (Table 3.3). BOD_5 outlet concentrations approached background water quality levels, which limit the removal efficiencies observed. The Restart Phase demonstrated a similar level of treatment with a COD removal efficiency of 57%. A similar lagoon polishing FWS system from 2000 to 2010 in Brighton, ON reported 40% BOD removal with a low inlet concentration of 5.4 mg/L (Kadlec et al., 2012). FWS wetlands treating domestic wastewater typically remove over 80% BOD_5 (Doku & Heinke, 1995; Gunes et al., 2012; Tsihrintzis et al., 2007) and 75% to 92% COD (Gunes et al., 2012; Tsihrintzis et al., 2007), however, with higher inlet concentrations than is the case with this study.

Table 3.3: *BOD_5/COD percentage removal at different phases of wetland*

	Parameter	**Lagoon (Mean ± 95 CI)**	**W2 (Mean ± 95 CI)**	**Removal Efficiency (%)**
Startup (2000-2002) †	BOD_5	11 ± 9	4 ± 2	44
Mature (2003-2009) †	BOD_5	9 ± 4	4 ± 2	50
Restart (2021) †	COD	126 ± 13	54 ± 4	57

†Lagoon and W2 difference statistically significant (P<0.05) using ANOVA test

The same trend in treatment was observed for all three operating phases with significant reductions in BOD or COD only occurring in the first wetland cell with concentrations approaching background levels (Figure 3.6). This indicates that one FWS cell would be sufficient to polish lagoon effluent to maintain BOD_5 less than 10 mg/L at the loading rates applied. No significant differences in any of the operating phases were observed between the wetland outlet and the vegetated filter.

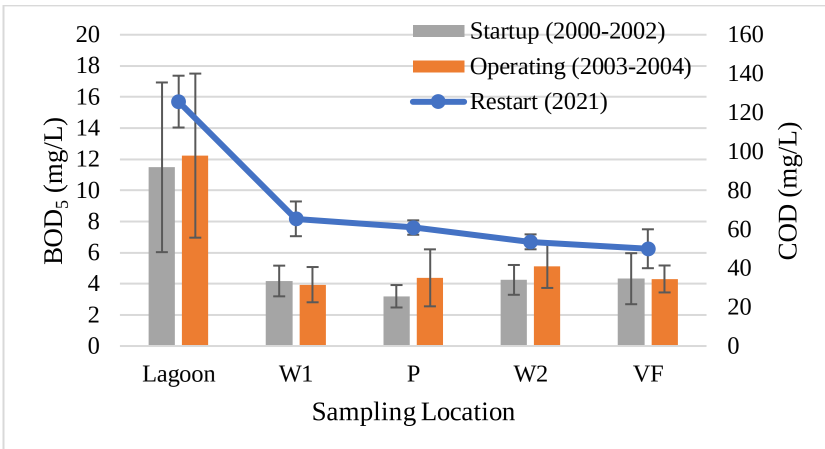


Figure 3.6: *BOD_5 (2000-2002 & 2003-2004) and COD (2021) (Mean ± 95%CI) concentrations at various wetland segments*

3.4.4.1 Seasonal temperature effects:

The seasonal temperature did not have a significant effect on BOD lagoon or wetland outlet concentrations as shown in Figure 3.7. However, BOD outlet concentrations ranged from 3-6 mg/L and were likely impacted by background concentration thresholds. Kadlec & Wallace (2008) suggested that BOD removal is not improved at higher wetland water temperatures, which is consistent with these findings. Conversely, the Brighton wetland system with an average inlet concentration of 5.4 mg/L performed better during winter while no significant removal was observed between June and November (Kadlec et al., 2012).

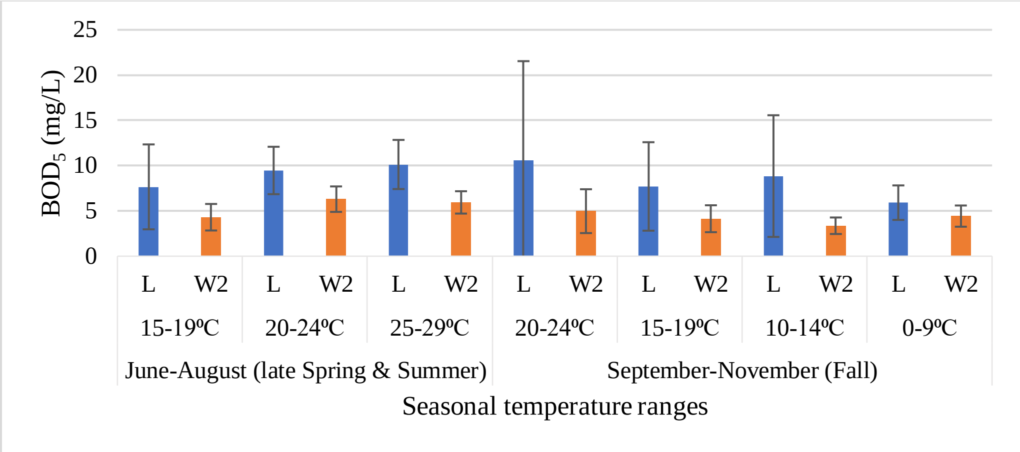


Figure 3.7: *Wetland System BOD_5 (Mean ± 95%CI) concentrations at various seasonal temperature ranges*

3.4.4.2 BOD removal rate coefficients:

The kinetic rate constant for BOD_5 removal for the first wetland cell was calculated using Eq. 3.2 from 2001-2004 with k = 45.6 ± 12.9 m/yr (P=1; C*=1 mg/L). The kinetic rate constant for COD removal was calculated for 2021 with k = 32.6 m/yr (P=1; C*=35). These values are similar to each other as well as to the 50% percentile k value for systems with BOD < 30 mg/L of 33 m/yr (Kadlec and Wallace, 2008). The Brighton wetland system reported a lower k value of 23 ± 5 m/yr which is likely due to the lower influent BOD concentration observed with that system (Kadlec et al., 2012). Given the similar k values to literature and the similar k values between COD and BOD observed, these values can be used with a degree of confidence for system design. Furthermore, the similar trends in organic matter reduction and k-values observed between the three Phases suggest that increasing the water depth at the Restart Phase had no significant effect on organic matter reduction in the system.

3.4.5 Nitrogen

The time series of inorganic nitrogen inlet and outlet concentrations are presented in Figure 3.8. Inlet nitrate concentrations increased from the detection limit value of 0.10 mg/L to 0.44 mg/L over the Startup Phase of the system, while outlet concentrations remained at or close to detection limit values, suggesting plant uptake during the macrophyte establishment phase. During the Mature Phase, a relatively constant but small reduction in nitrate concentration was observed at 0.12 ± 0.06 mg/L while inlet concentrations varied between 0.30 and 1.25 mg/L. This indicates that both plant uptake and denitrification played marginal roles in the net reduction of nitrate in the system and suggests that low-carbon FWS systems are not effective at denitrification. However, during the Restart Phase, nitrate concentrations were reduced by 1.45 mg/L from 3.90 to 2.45 mg/L, suggesting improved conditions in the sediment supporting denitrification. Inlet and outlet nitrate concentrations were strongly correlated (R^2=0.94) due to limited treatment in the system.

Ammonium inlet concentrations were low, varying from 0.08 to 0.84 mg/L but exhibited good removal throughout both the Startup and Mature Phases; however, no significant reduction was observed during the Restart Phase, indicating an effect of either the dormant period or the increased operating depth on net nitrification. Inlet and outlet ammonium concentrations showed a weak correlation of R^2=0.19, which was impacted by the almost complete NH_4^+ removal with varying inlet concentrations observed during many operating seasons.

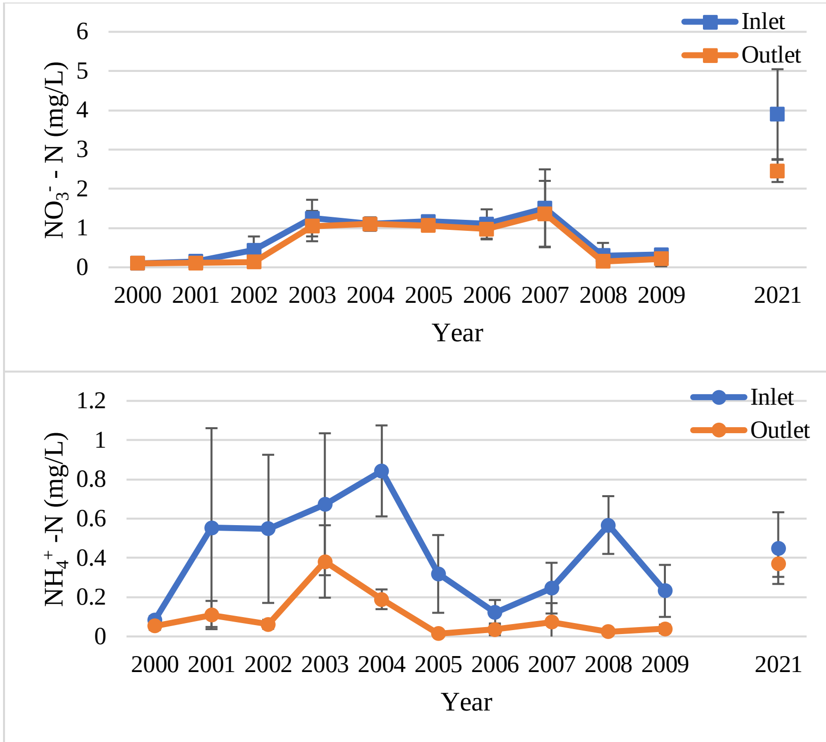


Figure 3.8: *Wetland System Inlet and Outlet NO_3^--N and NH_4^+-N (Mean ± 95%CI) concentrations by Year*

Organic N removal was high in both Startup and Restart Phases with 46 and 83%, respectively, reflecting removal through sedimentation (Table 3.4). Inlet nitrate concentrations increased sequentially through the three Phases, while treatment varied by operating phase. During Startup Phase 50% removal was observed, likely due to plant uptake of low concentrations. This is consistent with observations of a high N removal rate in starting phase of FWS systems due to initial vegetation planting (Nilsson et al., 2020b). During Mature Phase only 7% removal was observed, likely due to a lack of carbon to support denitrification, while during the Restart Phase 38% removal efficiency was observed, suggesting increased sediment levels provided better conditions for denitrification, while increased operating depth could also play a role in reducing oxygen transfer to the sediment. Ammonium reduction was high during both Startup and Mature Phases at 83 and 66%, respectively, while only 18% removal was observed during the Restart

Phase, suggesting that either the dormant period or the increased operating depth had a negative impact on nitrification.

Table 3.4: *Org N, NO_3^- and NH_4^+ percentage removal at different phases of wetland*

	Lagoon (Mean ± 95 CI)	**W2 (Mean ± 95 CI)**	**Removal Efficiency (%)**
Org N			
Startup (2000-2002) †	2.4 ± 0.6	1.3 ± 0.2	46
Restart (2021) †	2.3 ± 1.2	0.4 ± 0.2	83
NO_3^--N			
Startup (2000-2002)	0.24 ± 0.13	0.12 ± 0.03	50
Mature (2003-2009)	0.96 ± 0.14	0.89 ± 0.76	7
Restart (2021) †	3.90 ± 1.15	2.46 ± 0.24	37
NH_4^+-N			
Startup (2000-2002) †	0.42 ± 0.21	0.07 ± 0.03	83
Mature (2003-2009) †	0.43 ± 0.09	0.11 ± 0.03	66
Restart (2021)	0.45 ± 0.23	0.37 ± 0.07	18

†Lagoon and W2 difference statistically significant ($P<0.05$) using ANOVA test

Different trends in nitrogen reduction across the system were observed for different operating phases (Figure 3.9). During Startup Phase, the first wetland cell removed 46% of organic N and most of the nitrate and ammonium, with no further reductions observed in the system. During Mature Phase, a surprising reduction of organic N was observed only in W2 but increased again in the vegetated filter, no nitrate reduction was observed across the system, while ammonia reduction occurred on in the first wetland cell and was constant thereafter. During the Restart Phase, organic N was reduced in the first wetland cell, a trend of nitrate reduction was observed throughout the system, while no reduction in ammonium was observed.

A FWS wetland system for polishing municipal sewage observed no variation and aging effects in nitrogen removal over ten years (Waara et al., 2015). The Brighton FWS system treating post-lagoon effluent with an HLR of 19.3 $m^3/m^2/yr$ had removal efficiencies of 17% for organic N, 36% for NO_3^-, and 17% for NH_4^+ from respective inlet concentrations of 2.24 mg/L, 0.53 mg/L, and 11.31 mg/L and an average BOD_5 of 5.4 mg/L (Kadlec et al., 2012). Pilot-scale FWS studies reported a 39% removal efficiency for NO_3^- with an influent concentration of 1.8 mg/L (Jinadasa et al., 2006), and 42-59% removal efficiencies for NH_4^+ with influent concentrations above 10 mg/L (Boutilier et al., 2010; Jinadasa et al., 2006; Liu et al., 2009). However, this study with a loading rate of 6.6 $m^3/m^2/yr$ observed that the Startup phase resulted in better removal efficiency for NO_3^- and NH_4^+ as compared to the Mature phase, while the Restart phase after a long-dormant period performed well for Organic N and NO_3^- removal.

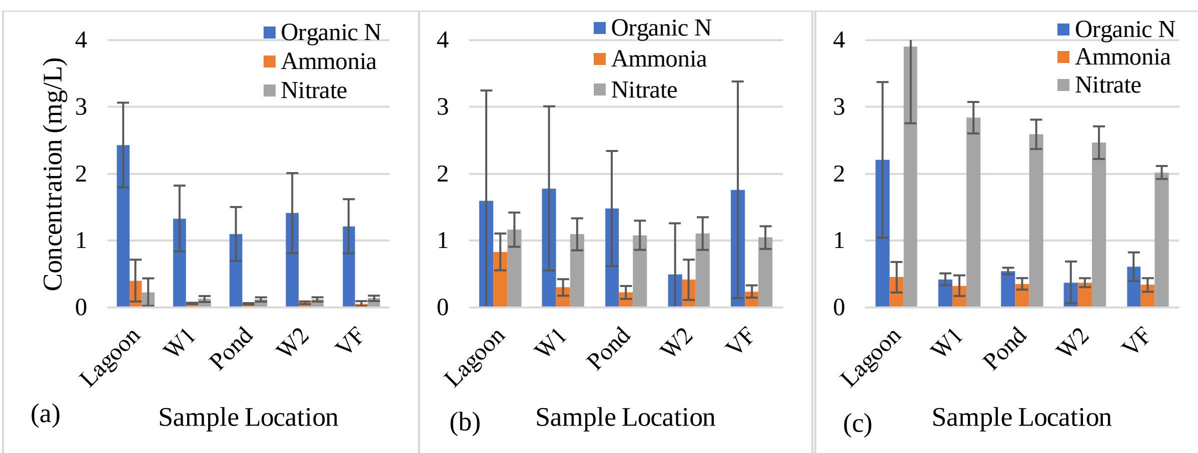


Figure 3.9: *Comparison of NO_3^-, NH_4^+, and organic N (Mean ± 95%CI) concentrations at various wetland segments by Operating Phase (a) Startup (2000-2002), (b) Mature (2003-2004), and (c) Restart (2021)*

3.4.5.1 Seasonal temperature effects:

Nitrate concentrations were independent of the seasonal temperature changes as shown in Figure 3.10, with no significant differences in the lagoon and W2 concentrations observed at any of the temperature conditions (P>0.05). The limited nitrate removal in the system likely resulted from a lack of a significant anoxic sediment layer and low influent carbon. This contrasts with a treatment wetland in Tres Rios, Arizona, where nitrate removal was better in warmer temperature ranges between 15-30ºC at influent concentrations from 1 to 4 mg/L (Kadlec & Reddy, 2001).

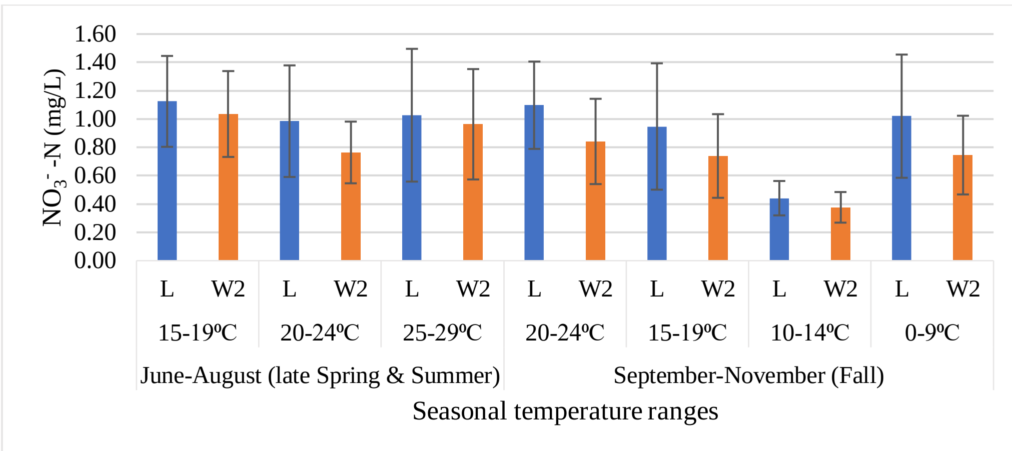


Figure 3.10: *Wetland System NO_3^- (Mean ± 95%CI) concentrations at various seasonal temperature ranges*

3.4.5.2 Nitrogen removal rate coefficients:

The kinetic rate constants were calculated from Eq. 3.2 and compared with values from the similar Brighton wetland (Kadlec et al., 2012) as well as the 50th percentile values presented in Kadlec and Wallace 2008). The average annual k-value for TN was 7.9 ± 4.7 m/yr (C*=0.6 mg/L, P=3) which is comparable to the 50th percentile k-value of 12.6 m/y and higher than the 3.9 m/y reported for the Brighton wetland. The average annual k-value for NH_4^+ reduction was 15.5 ± 7.8 m/yr (C*=0 mg/L; P=3), which is similar to the 50th percentile k-value of 14.7 m/y and higher than 3.8 m/y reported for the Brighton wetland. The average annual k-value for NO_3^- was 3.0 ± 2.3 m/yr (C*=0.05 mg/L, P=3), which is much lower than the 50th percentile k-value of 26.5 m/y and 101 m/y reported for the Brighton wetland. The very low kinetic rates for nitrate removal are likely related to a lack of available carbon for denitrification in the system and suggest that the plant litter is not a sufficient carbon source.

3.4.6 Phosphorus

Inlet average TP concentrations ranged from 0.63 to 3.27 mg/L over the study period with higher concentrations observed during the first 4 years of operation. Outlet concentrations were greatly reduced to between 0.04 and 0.19 mg/L with no apparent trend or correlation to influent (R^2=-0.01) (Figure 3.11). The average outlet concentrations consistently met the treatment objective of 0.3 mg/L TP.

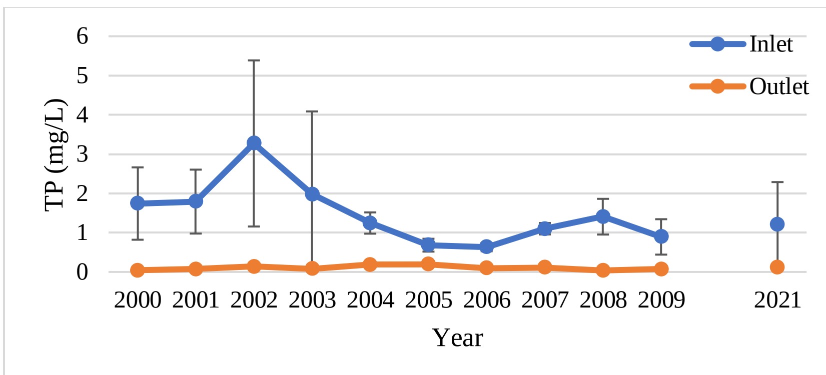


Figure 3.11: *Wetland System Inlet and Outlet TP (Mean ± 95%CI) concentrations by Year*

High removal efficiencies of 83-94% were observed as well as no significant differences in outlet concentrations over the three operating periods (P>0.05) (Table 3.5). SRP concentrations were also reduced to very low concentrations during Startup and Restart Phases of 0.05 ± 0.02 and 0.03 ± 0.02 mg/L, respectively.

Table 3.5: *TP and SRP percentage removal at different phases of wetland*

	Lagoon (Mean ± 95 CI)	W2 (Mean ± 95 CI)	Removal Efficiency (%)
TP			
Startup (2000-2002) †	1.88 ± 0.80	0.11 ± 0.07	94
Mature (2003-2009) †	1.11 ± 0.28	0.17 ± 0.04	85
Restart (2021) †	1.20 ± 1.08	0.20 ± 0.25	83
SRP			
Startup (2000-2002) †	1.18 ± 0.41	0.05 ± 0.02	96
Restart (2021) †	0.23 ± 0.42	0.03 ± 0.02	87

†Lagoon and W2 difference statistically significant ($P<0.05$) using ANOVA test

The Brighton FWS wetland study treating municipal lagoon effluent for ten years reported an average of 32.5% TP removal rate with 0.4 mg/L of inlet concentration and HLR of 19.3 $m^3/m^2/yr$ with no effect of wetland aging observed (Kadlec et al., 2012). A FWS wetland treating septic tank effluent located at Truro, Nova Scotia was studied by Boutilier et al. (2010) with a loading rate of 5.1 $m^3/m^2/yr$ and an average influent concentration of 3.5 mg/L TP and 2.2 mg/L SRP. They reported a treatment efficiency of 39% and 35% for TP and SRP, respectively. Two full-scale FWS domestic wastewater treatment wetlands removed 35% of inlet TP concentrations between 6.6 to 9.1 mg/L (Gunes et al., 2012; Tsihrintzis et al., 2007). This study observed consistent removal efficiency greater than 83% for both TP and SRP, which is better than the comparative literature.

A clear trend of increasing TP in both W1 and Pond cells was observed from the Startup to Mature Phase and from the Mature to Restart Phase ($P<0.05$) despite a decreasing trend in lagoon concentrations (Figure 3.12). This suggests that TP in the water column of W1 and Pond is increasing with accumulating sediment, which is supported by a strong correlation observed between TSS and TP of W1 and Pond cells during the Restart Phase ($R^2=0.67$). However, this effect has not been observed in the second wetland cell, where no differences in TP were observed between the three Phases (P>0.05). The two long-term mechanisms of P removal are sediment accumulation and adsorption to wetland soils (Dolan et al., 1981; Reddy et al., 1999). The very low SRP concentrations throughout the system, with no significant differences between Startup and Restart Phases (Figure 3.12), strongly suggest that the clay soils of the system have not reached saturation after 11 years of loading. The trend of TP reduction with treatment cells observed in the Restart Phase and the strong correlation to TSS suggests that as sediment levels increase, the subsequent treatment cells play an important role in sediment control and corresponding TP settling, which was not observed during the Startup or Operating Phases.

No significant differences between W2 and VFS were observed for either TP or SRP during any of the operating Phases (P>0.05).

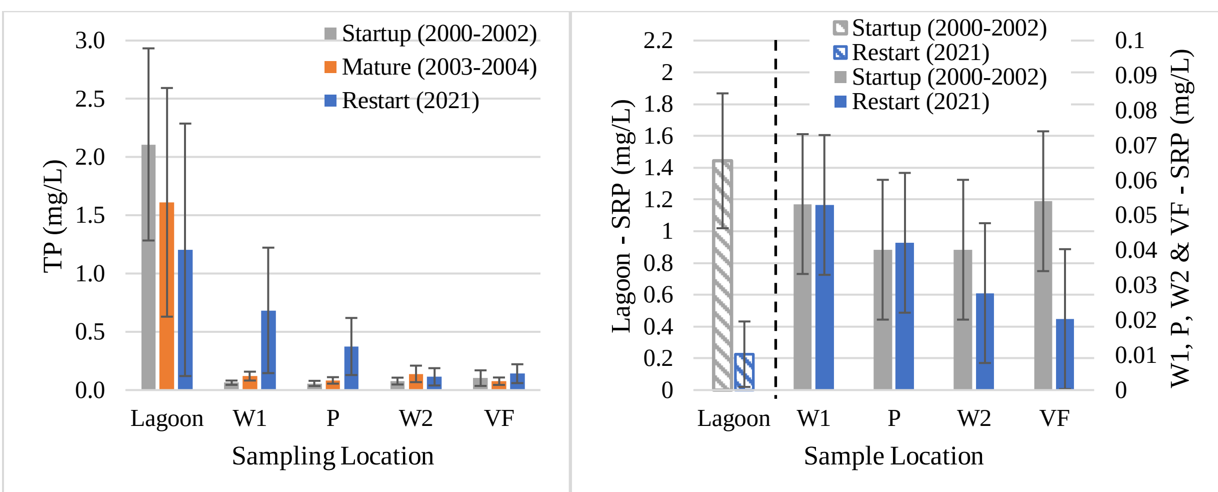


Figure 3.12: *Comparison of TP and SRP (Mean ± 95%CI) concentrations at various wetland segments*

3.4.6.1 Seasonal temperature Effects:

The lagoon concentrations were strongly related to seasonal temperature variation with TP increasing from 1.0 mg/L in early summer to a maximum of 2.2 mg/L at the height of summer and then declining to 0.7 mg/L in late fall (Figure 3.13), most likely corresponding to the production of algae in the lagoon. Outlet concentrations were stable at 0.23 ± 0.03 mg/L during the summer months, despite increasing inlet concentrations, and were stable but significantly lower during the fall at 0.10 ± 0.03 mg/L ($P<0.05$), despite declining inlet concentrations. This suggests that internal production of algae and plant detritus during the peak growing season (June-August) resulted in higher export of TP than in the fall season. An increase in outlet TP was also observed in the very late fall with the lowest temperature range of 0-9ºC, suggesting a possible release of phosphorus from decaying litter following plant senescence.

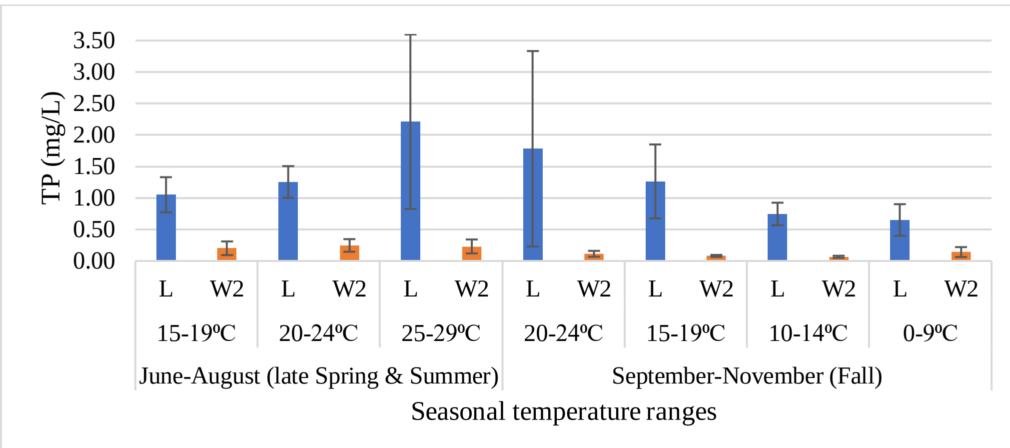


Figure 3.13: *Wetland System TP (Mean ± 95%CI) concentrations at various seasonal temperature ranges*

The lack of vegetation in colder temperatures is also one factor that decreases phosphorus removal efficiency (Kadlec, 2010). A study on various wetlands resulted in low to no effects on TP due to temperature changes (Kadlec & Reddy, 2001). The Brighton wetland operated in all seasons and resulted in higher TP removal in the winter and early spring (January to April) with temperatures below 10ºC and 0.6 mg/L inlet concentrations in that season. Also, the study observed a negative removal rate with an average inlet concentration of 0.2 mg/L in summer and early fall as wetlands released phosphorus and increased outlet concentrations (Kadlec et al., 2012). However, this study resulted in better performance in higher temperatures during the Summer (25-29^oC) and Fall (10-24^oC) seasons with more than 90% removal efficiency.

3.5 Conclusion

The wetland system was shown to provide effective long-term treatment of municipal lagoon effluent. No aging effect was observed for BOD and TP removal, while internal TP concentrations increased with time and are likely related to increasing sediment levels. TSS removal efficiency was significantly higher during the restart phase and is likely due to the increase in operating depth from 25 to 50 cm. Higher NO_3^- and NH_4^+ removal efficiencies were observed during the start-up period and can be attributed to plant growth during wetland establishment and higher removal rates of organic N and NO_3^- were observed during the restart phase, likely due to a combination of increased water depth and sediments in the wetland system.

The wetland system was effective at removing organic matter and nutrients with average removal efficiencies of 97% TSS, 57% COD, 83% Organic N, 37% NO_3^--N, 18% NH_4^+-N, 83% TP, and 87% SRP in restart period. The vegetated filter generally did not contribute to increased water quality, likely due to the high removal efficiencies observed in the wetland system.

No seasonal temperature effect was observed for either BOD_5 or NO_3^-. However, a seasonal temperature effect was observed on TP performance as the removal efficiency was higher during the late summer and fall compared with early summer and late fall.

The kinetic removal rate (k) was calculated for the lagoon polishing wetland system using the first-order P-k-C* model. The averaged k values were: 45.8 m/yr for BOD, 7.9 m/yr for TN, 15.5 m/yr for NH_4^+, and 3.0 m/yr for NO_3^-. The BOD, TN, and NH_4^+ removal rate constants were within the ranges observed in the literature from similar applications, while the NO_3^- k-value was lower than in other similar studies and suggests limited denitrification potential from low-carbon wastewater sources.

3.6 References

APHA. (1998). Standard Methods for the Examination of Water and Wastewater (22nd edition). American Public Health Association.

APHA. (2012). Standard Methods for the Examination of Water and Wastewater (22nd edition). American Public Health Association.

Boutilier, L., Jamieson, R., Gordon, R., Lake, C., & Hart, W. (2010). Performance of surface-flow domestic wastewater treatment wetlands. Wetlands, 30(4), 795–804. https://doi.org/10.1007/S13157-010-0067-1

Brisson, J., & Chazarenc, F. (2008). Maximizing pollutant removal in constructed wetlands: Should we pay more attention to macrophyte species selection? Science of the Total Environment, The, 407, 3923–3930. https://doi.org/10.1016/j.scitotenv.2008.05.047

Brix, H., & Schierup, H. (1988). SEWAGE TREATMENT IN CONSTRUCTED REED BEDS — DANISH EXPERIENCES. Water Pollution Research and Control Brighton, 1665–1668. https://doi.org/10.1016/B978-1-4832-8439-2.50158-9

Cameron, K., Madramootoo, C., Crolla, A., & Kinsley, C. (2003). Pollutant removal from municipal sewage lagoon effluents with a free-surface wetland. Water Research, 37(12), 2803–2812. https://doi.org/10.1016/S0043-1354(03)00135-0

Doku, I. A., & Heinke, G. W. (1995). Potential for Greater Use of Wetlands for Waste Treatment in Northern Canada. Journal of Cold Regions Engineering, 9(2), 75–88. https://doi.org/10.1061/(ASCE)0887-381X(1995)9:2(75)

Dolan, T. J., Bayley, S. E., Zoltek, J., & Hermann, A. J. (1981). Phosphorus Dynamics of a Florida Freshwater Marsh Receiving Treated Wastewater. The Journal of Applied Ecology, 18(1), 205. https://doi.org/10.2307/2402490

Fraser, L. H., Carty, S. M., & Steer, D. (2004). A test of four plant species to reduce total nitrogen and total phosphorus from soil leachate in subsurface wetland microcosms. https://doi.org/10.1016/j.biortech.2003.11.023

Gearheart, R. A. (1992). Use of Constructed Wetlands to Treat Domestic Wastewater, City of Arcata, California. Water Science and Technology, 26(7–8), 1625–1637. https://doi.org/10.2166/WST.1992.0606

Greenway, M. (2005). The role of constructed wetlands in secondary effluent treatment and water reuse in subtropical and arid Australia. Ecological Engineering, 25, 501–509. https://doi.org/10.1016/j.ecoleng.2005.07.008

Gunes, K., Tuncsiper, B., Ayaz, S., & Drizo, A. (2012). The ability of free water surface constructed wetland system to treat high strength domestic wastewater: A case study for the Mediterranean. Ecological Engineering, 44, 278–284. https://doi.org/10.1016/j.ecoleng.2012.04.008

Guo, C., Cui, Y., Dong, B., Luo, Y., Liu, F., Zhao, S., & Wu, H. (2017). Test study of the optimal design for hydraulic performance and treatment performance of free water surface flow constructed wetland. Bioresource Technology, 238, 461–471. https://doi.org/10.1016/J.BIORTECH.2017.03.163

Jenssen, P. D., Maehlum, T., & Krogstad, T. (1993). Potential Use of Constructed Wetlands for Wastewater Treatment in Northern Environments. Water Science and Technology, 28(10), 149–157. https://doi.org/10.2166/WST.1993.0223

Jinadasa, K. B. S. N., Tanaka, N., Mowjood, M. I. M., & Werellagama, D. R. I. B. (2006). Chemistry and Ecology Free water surface constructed wetlands for domestic wastewater treatment: A tropical case study Free water surface constructed wetlands for domestic wastewater treatment: A tropical case study. Chemistry and Ecology, 22(3), 181–191. https://doi.org/10.1080/02757540600658849

Kadlec, R. H. (2010). Phosphorus Removal in Emergent Free Surface Wetlands. Journal of Environmental Science and Health, 40(6–7), 1293–1306. https://doi.org/10.1081/ESE-200055832

Kadlec, R. H., Pries, J., & Lee, K. (2012). The Brighton treatment wetlands. Ecological Engineering, 47, 56–70. https://doi.org/10.1016/J.ECOLENG.2012.06.042

Kadlec, R. H., & Reddy, K. R. (2001). Temperature effects in treatment wetlands. Water Environment Research: A Research Publication of the Water Environment Federation, 73(5), 543–557. https://doi.org/10.2175/106143001X139614

Kadlec, R. H., Tilton, D. L., & Ewel, K. C. (2009). The use of freshwater wetlands as a tertiary wastewater treatment alternative. https://doi.org/10.1080/10643387909381671

Kadlec, R., & Wallace, S. (2008). Treatment wetlands. https://books.google.com/books?hl=en&lr=&id=hPDqfNRMH6wC&oi=fnd&pg=PP1&ots=k8M07QbT6N&sig=sfZ5zY6zFfG83Xs3yYO1Eok28Rs

Land, M., Granéli, W., Grimvall, A., Hoffmann, C. C., Mitsch, W. J., Tonderski, K. S., & Verhoeven, J. T. A. (2016). How effective are created or restored freshwater wetlands for nitrogen and phosphorus removal? A systematic review. Environmental Evidence, 5(1), 1–26. https://doi.org/10.1186/S13750-016-0060-0/FIGURES/14

Li, X., & Jiang, C. (1995). Constructed wetland systems for water pollution control in North China. Water Science and Technology, 32(3), 349–356. https://doi.org/10.2166/WST.1995.0157

Liu, C., Xu, K., Inamori, R., Ebie, Y., Liao, J., & Inamori, Y. (2009). Pilot-scale studies of domestic wastewater treatment by typical constructed wetlands and their greenhouse gas emissions. https://doi.org/10.1007/s11783-009-0155-8

Liu, J. J., Dong, B., Guo, C. Q., Liu, F. P., Brown, L., & Li, Q. (2016). Variations of effective volume and removal rate under different water levels of constructed wetland. Ecological Engineering, 95, 652–664. https://doi.org/10.1016/J.ECOLENG.2016.06.122

Mæhlum, T., Technology, P. S.-W. S. and, & 1999, undefined. (1999). The removal efficiency of three cold-climate constructed wetlands treating domestic wastewater: effects of temperature, seasons, loading rates and input concentrations. Elsevier. https://doi.org/10.1016/S0273-1223(99)00441-2

Nilsson, J. E., Liess, A., Ehde, P. M., & Weisner, S. E. B. (2020a). Mature wetland ecosystems remove nitrogen equally well regardless of initial planting. Science of the Total Environment, 716. https://doi.org/10.1016/J.SCITOTENV.2020.137002

Nilsson, J. E., Liess, A., Ehde, P. M., & Weisner, S. E. B. (2020b). Mature wetland ecosystems remove nitrogen equally well regardless of initial planting. Science of The Total Environment, 716, 137002. https://doi.org/10.1016/J.SCITOTENV.2020.137002

Picard, C. R., Fraser, L. H., & Steer, D. (2005). The interacting effects of temperature and plant community type on nutrient removal in wetland microcosms. Bioresource Technology, 96(9), 1039–1047. https://doi.org/10.1016/J.BIORTECH.2004.09.007

Polprasert, C., Koottatep, T., & Tanner, C. C. (2005). Integrated pond/wetland systems. In: Shilton, A. N. (Andy N.), Pond Treatment Technology. IWA Publishing, London, Pp. 328–345.

Reddy, K. R., Kadlec, R. H., Flaig, E., & Gale, P. M. (1999). A Review Phosphorus Retention in Streams and Wetlands: A Review. Critical Reviews in Environmental Science and Technology, 29(1), 83–146. https://doi.org/10.1080/10643389991259182

Reed, S. C., Crites, R. W., & Middlebrooks, E. J. (2005). Natural Wastewater Treatment Systems. 576. https://books.google.com/books/about/Natural_Wastewater_Treatment_Systems.html?id=K6k8yE8rw0MC

Saeed, T., & Sun, G. (2011). A comparative study on the removal of nutrients and organic matter in wetland reactors employing organic media. Chemical Engineering Journal, 171(2), 439–447. https://doi.org/10.1016/J.CEJ.2011.03.101

Steinmann, C. R., Weinhart, S., & Melzer, A. (2003). A combined system of lagoon and constructed wetland for an effective wastewater treatment. Water Research, 37, 2035–2042.

Tanner, C. C. (2001). Growth and nutrient dynamics of soft-stem bulrush in constructed wetlands treating nutrient-rich wastewaters. In Wetlands Ecology and Management (Vol. 9).

Tsihrintzis, V. A., Akratos, C. S., Gikas, G. D., Karamouzis, D., & Angelakis, A. N. (2007). Performance and cost comparison of a FWS and a VSF constructed wetland system. Environmental Technology, 28(6), 621–628. https://doi.org/10.1080/09593332808618820

USEPA. (2022). The Universe of Lagoons: An analysis of state and tribal lagoon wastewater treatment systems and socioeconomic, environmental justice, and compliance patterns in small, rural communities in the United States.

Vymazal, J. (2010). Constructed Wetlands for Wastewater Treatment. Water 2010, Vol. 2, Pages 530-549, 2(3), 530–549. https://doi.org/10.3390/W2030530

Vymazal, J. (2011). Constructed wetlands for wastewater treatment: Five decades of experience. Environmental Science and Technology, 45(1), 61–69. https://doi.org/10.1021/ES101403Q/SUPPL_FILE/ES101403Q_SI_001.PDF

Vymazal, J. (2018). Do Laboratory Scale Experiments Improve Constructed Wetland Treatment Technology? Environmental Science and Technology, 52(22), 12956–12957. https://doi.org/10.1021/ACS.EST.8B05709/ASSET/IMAGES/LARGE/ES-2018-057098_0003.JPEG

Waara, S., Gajewska, M., Cruz Blázquez, V., Alsbro, R., & Norwald, P. (2015). Long term performance of a FWS wetland for post-tertiary treatment of the sewage-the influence of flow, temperature and age on nitrogen removal. 13–18. http://www.diva-portal.orghttp://urn.kb.se/resolve?urn=urn:nbn:se:hh:diva-30137

Wood, A., Brix, H., & Kadlec, R. H. (1995). Constructed wetlands in water pollution control: Fundamentals to their understanding. Water Science and Technology, 32(3), 21–29. https://doi.org/10.1016/0273-1223(95)00601-X

WSER. (2012). Wastewater Systems Effluent Regulations. https://laws-lois.justice.gc.ca/eng/regulations/sor-2012-139/fulltext.html

Wu, H., Zhang, J., Li, P., Zhang, J., Xie, H., & Zhang, B. (2011). Nutrient removal in constructed microcosm wetlands for treating polluted river water in northern China. Ecological Engineering, 37(4), 560–568. https://doi.org/10.1016/J.ECOLENG.2010.11.020

CHAPTER: 4

Long-Term Removal, Partitioning, And Storage of Nutrients in a Free Water Surface Constructed Wetland Polishing Municipal Lagoon Effluent

4.1 Abstract

The research was conducted on a marsh-pond-marsh free surface flow wetland system situated in Alfred, Ontario polishing municipal lagoon effluent on a seasonal basis. The wetland operated for 10 years, was dormant for 11 years, and was restarted in 2021. At the end of the 2021 operating season the wetland compartments of soil, sediment, plants, and water column were characterized for nitrogen and phosphorus species and compared with the nutrient mass fluxes over the 11 operating years. The system was loaded with 267 kg of phosphorus (P) over operating years with partitioning of: 50.8 ± 25.7% in soil, 25.1 ± 7.9% in sediment, 10.9 ± 3.8% in plants, 1.1 ± 0.4% in the water column and 12% exported in the effluent. The system was loaded with 1258 kg nitrogen (N) over operating years with partitioning of: 9.6 ± 1% in soil, 6.8 ± 1.4% in sediment, 16 ± 6% in plants, 1.6 ± 0.4% in water, 44.4% exported in effluent and 21.5 ± 12.6% lost to the atmosphere.

Keywords: Constructed wetland, Sediments, Soil, Plants, Nutrient accumulation, Mass balance

4.2 Introduction

Wetlands offer essential environmental services like nutrient management, nutrient retention, and wastewater treatment (Zedler & Kercher, 2005). Constructed wetlands (CWs) are artificial systems that follow a combination of biological, chemical, and physical processes found in natural wetlands, but in a more contained environment (Vymazal, 2010; Ezzat & Moustafa, 2020). Constructed wetlands are designed systems that are intended and built to use natural processes including wetland vegetation, soils, sediments, and associated microbial communities to help treat wastewaters (Vymazal, 2010).

Wetlands store nutrients and convert them from biologically available to non-biologically available forms and vice versa (Reddy et al., 1999). Uptake and release by plants, macrophytes, and microorganisms; sorption and exchange interactions with soils and sediments; chemical precipitation in the water column; and sedimentation and entrainment are some of the phosphorus retention mechanisms (Reddy et al., 1999; Vymazal, 2007). The soil is the most important long-term ecosystem P storage compartment, according to studies of wetland ecosystem structure and function (Dolan et al., 1981; Richardson & Marshall, 1986). The deposit of nitrogen as organic matter in the sediments may lead to nitrogen storage in constructed wetlands (Bastviken, 2006). The long-term storage of phosphorus mainly depends on soil and sediment accumulation (Dolan et al., 1981; Reddy et al., 1999). N accumulation in different wetland sediment ranged from 15.9 to 26.6 g/m^2 (Wu et al., 2012). Debusk & Reddy (2004) reported that phosphorus concentrations were 1150 mg/kg and 640 mg/kg at 0-10 cm and 10-30 cm depth of soil, respectively. The range of the P concentration of soil is 200 to 5000 mg/kg, with a mean of 600 mg/kg (Strickland et al., 2010).

Typha latifolia is an aquatic macrophyte that emerges from the water and plays an essential role in wetland transition (Weisner, 1993). Typha latifolia has been utilized to treat municipal wastewater as a secondary and/or tertiary treatment with significant performance (Chung et al., 2008). In emergent marsh wetlands, below-ground biomass can be considerably higher than above-ground biomass (Richardson & Marshall, 1986). Reddy & Smith (1987) reported that nitrogen and phosphorus standing stock for emergent species in the range of 14-156 g N/m^2 and 1.4-37.5 g P/m^2 respectively, but they also indicated that more than 50% of this could be stored below ground. Plants uptake soluble reactive phosphorus and convert it to tissue phosphorus, where a portion will ultimately be deposited into wetland soils and sediments (Vymazal, 2007).

Interconversion of all types of phosphorus occurs in wetlands. Soil accumulation, adsorption/desorption, precipitation/dissolution, plant/microbial assimilation, fragmentation and leaching, and mineralization are phosphorus conversions in wetlands (Vymazal, 2007). Braskerud (2002) defines phosphorus mass balance as phosphorus retention as the difference between phosphorus mass inflow and mass outflow. In constructed wetlands, nitrogen is removed through several processes, including substrate adsorption, plant absorption, volatilization, ammonification, nitrification, and denitrification. However, nitrification and denitrification are the main removal mechanism in most constructed wetlands (Sudarsan et al., 2021). The nitrogen accumulation and the amount loss to the atmosphere between the nitrogen mass inlet and mass outlet are described as nitrogen mass balance. This research study is focused on the nutrient accumulation in sediments, plants, and soil. The objectives of this study were: i) Investigate and compare the sediment, soil, and plant nutrient storage in a wetland system. ii) Compute phosphorus and nitrogen mass balances in the wetland system over the operating period.

4.3 Materials and methods

4.3.1 Site description and operation

The Alfred wetland system (45º32’36” N, 74º51’32” W) is situated around 70 kilometers east of Ottawa. The effluent from the village of Alfred is currently controlled in a two-cell wastewater lagoon that discharges once a year in the spring to the Azitica Brook, which eventually runs into the Ottawa River north of the village. The constructed marsh-pond-marsh wetland – vegetated filter system was designed to polish municipal lagoon effluent to meet stringent summertime discharge requirements to a dry stream.

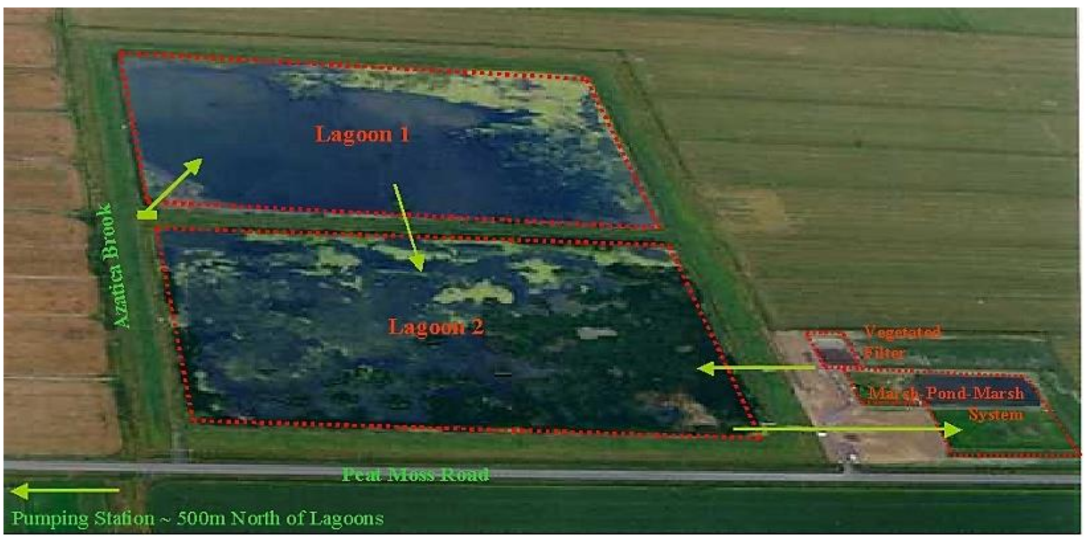


Figure 4.1: *Aerial photograph of the Alfred Wetland and the Municipal Sewage Lagoons*

The FSW I and FSW II wetland cells are shallow basin surface-flow wetlands with a water depth of 25 cm from the years 2000 to 2009. The depth of water was increased to 50 cm after a long dormant period in the restart year 2021 to account for accumulated sediment and to improve TSS effluent quality. The FSW I is 78 m long and 40 m wide, while the FSW II is 39 m long and 40 m wide. The Typha latifolia cattails were densely planted in both wetlands, with a few bulrushes (Scirpus acutus) along the edges of both wetlands. The pond measures 78 m long and 40 m wide with an average depth of 95 cm from the year 2000-2009. The average depth of the pond was 120 cm in 2021.

The system was operated from 2000 to 2009, and then it was restated in 2021 after an eleven-year dormant period. In the lagoon, a ¾ HP dosing pump (Myers WHR7-21C) was installed with a 3-inch outlet pipe. The pump was set to operate every 5 minutes and 30 seconds with intervals of 11 minutes. The lagoon wastewater was pumped to the header pipe of wetland cell 1 using a 3-inch ABS pipe. Wetland effluent was pumped back into the lagoon.

4.3.2 Sampling and analysis

4.3.2.1 Sediment sampling:

A total of 24 sediment samples were collected across the wetland and pond cells. Samples were collected using a Sludge Judge™ on October 28th, 2021. All the samples were analyzed for total solids (TS), volatile solids (VS), total phosphorus (TP), orthophosphate (PO_4^{3-}-P), total nitrogen (TN), ammonia (NH_4^+-N), nitrate (NO_3^--N) at the environmental engineering laboratory at the University of Ottawa. The samples were analyzed using the following procedure as per 'Standard Methods for the Examination of Water and Wastewater' (APHA, 2012). TS and VS were analyzed as per method 2540 G. TP and PO_4^{3-}-P were measured as per method 4500-P B & D (stannous chloride). NH_4^+ -N and NO_3^- -N were measured according to methods 4500-NH_3 D and 4500-NO_3^- D respectively. TN was measured using the HACH method 10072. Grab samples were *in-situ* analyzed for pH with a HACH-HQ40D multimeter (PHC101 probe for pH). Three sludge samples from the start, middle, and end of cell 1, pond, and cell 2 were shipped to the Agriculture & Food Laboratory at the University of Guelph for total organic carbon (TOC) analysis.

4.3.2.2 Plant sampling:

Three wetland plant samples (above-ground biomass) were collected from each wetland cell on October 13th, 2021. Wetland plant density was measured in 1 m x 1 m squares, in addition to plant height and stem diameter using a digital caliper. The wet weight for each sample was measured in the University of Ottawa environment engineering laboratory and then sent to the Agriculture & Food laboratory at the University of Guelph for plant tissue TP and TN analysis.

4.3.2.3 Soil sampling:

Soil samples were collected from the inlet, middle, and outlet of both wetland cells and the pond cell. Each sample was subdivided into three depths for analysis (0-5, 5-10, and 10-15 cm). All soil samples were collected on October 27th, 2021, and then sent to the Agriculture & Food laboratory at the University of Guelph for Extractable-P analysis. Soil inorganic nitrogen was extracted by mixing for 90 minutes 6 grams of air-dried soil with 2M KCl (Keeney & Bremner, 1966) with the extractant analyzed for NH_4^+ and NO_3^- as the sediment samples described above.

4.3.2.4 Water Quality Sampling and Analysis:

Wetland water quality monitoring was conducted from 2000 to 2009, the system was dormant from 2010 to 2020 and resumed operation in 2021. Water quality samples for nutrients were collected from the influent (lagoon) and wetland system outlet (W2) on a weekly to bi-weekly basis during overall operating periods. Complete data sets for TP were available, while data for SRP was only available from 2000 to 2002, and 2021. Complete datasets for NO_3^--N and NH_4^+-N were available, while TN data was available for 2000-2003 as well as in 2021.

4.3.3 Wetland water balance

A water balance was conducted on the wetland system for each season of operation. Inflow was measured while outflow was calculated using a water balance approach as described in Equation

4.1. The inflow of the wetland system was controlled by the inlet pump rate and varied somewhat by year, with an average of 141 m^3/day and a range of 134 – 152 m^3/day. Daily precipitation and temperature data were obtained from Environment Canada weather stations at the Ottawa and Montreal Intl. Airports were averaged as the research site is roughly equidistant between the two weather stations. To calculate the daily evapotranspiration rate in the wetland system, the formula described in Equations 4.2(a) and 4.2(b) by Hamon (1963) was used as it required fewer data inputs and performed better compared to other methods (Lu et al., 2005). Exfiltration from the wetland system was negligible due to the compacted clay base of the system.

Outflow = Inflow + Precipitation – Evapotranspiration **Equation 4.1**

$$\text{Evapotranspiration (ET in mm)} = 0.63 * D^2 * 10^{\frac{7.5\,T_a}{T_a+273}} \qquad \textbf{Equation 4.2 (a)}$$

$$D = \frac{1}{90} arccos\left\{- tan(\emptyset) \cdot tan\left[23.45^0\, sin\left(\frac{J-80}{365} \cdot 360^0\right)\right]\right\} \qquad \textbf{Equation 4.2 (b)}$$

Where; D = ratio of maximum sunshine duration (in an hour)
T_a = Daily temperature (ºC)
$\emptyset$ = Latitude of site location
J = Julian day of any date of interest

4.3.4 Nutrient Storage Reservoirs and Mass Balances

The wetland system was divided into four nutrient storage compartments: soil, sediment, macrophytes, and water column. As phosphorus is conservative, a complete mass balance in the system was determined as described in Equation 4.3 and compared with storage compartments as described in Equation 4.4.

$$P_{In} - P_{Out} = P_{Storage} \qquad \textbf{Equation 4.3}$$

$$P_{storage} = P_{Plants} + P_{Sediments} + P_{Soil} + P_{water} \qquad \textbf{Equation 4.4}$$

Nitrogen storage can be calculated from Equations 4.5 and 4.6; however, a complete mass balance is not possible due to losses to the atmosphere (N_2 and N_2O), additions through biological nitrogen fixation, and incomplete TN data series. However, storage components can be compared with the annual N loading during 2021.

$N_{In} - N_{Out} = N_{Storage}$ **Equation 4.5**

$N_{storage} = N_{Plants} + N_{Sediments} + N_{Soil} + N_{water}$ **Equation 4.6**

4.3.5 Statistical analysis

The sediment and plant data were analyzed using a one-way analysis of variance (ANOVA) test to determine if there was any significant difference in the concentration of constituents between wetlands cells. The soil data was reviewed using a one-way analysis of variance (ANOVA) test to find any significant difference between different depths for each cell as well as between cells at equivalent depths. The statistical significance for the ANOVA test is considered as $P<0.05$. The statistics were performed using the Data Analysis Tool in MS Excel.

4.4 Results and Discussion

4.4.1 Water Balance

The average monthly precipitation (PPT) and ET rate over the operating period are presented in Figure 4.2. The average monthly water balance over the operating years was calculated from the percentage rate of outflow to inflow. The monthly outflow was increased on average by 3.6% over the study period with a maximum monthly increase of 19.3% and a decrease of 14.3%. Generally, higher ET led to slightly lower outflows in June, July, and August of 0.9, 2.5, and 2.0%, respectively, while average outflows increased by 4.6% in September, 11.4% in October, and 11.1% in November due to higher precipitation and lower ET.

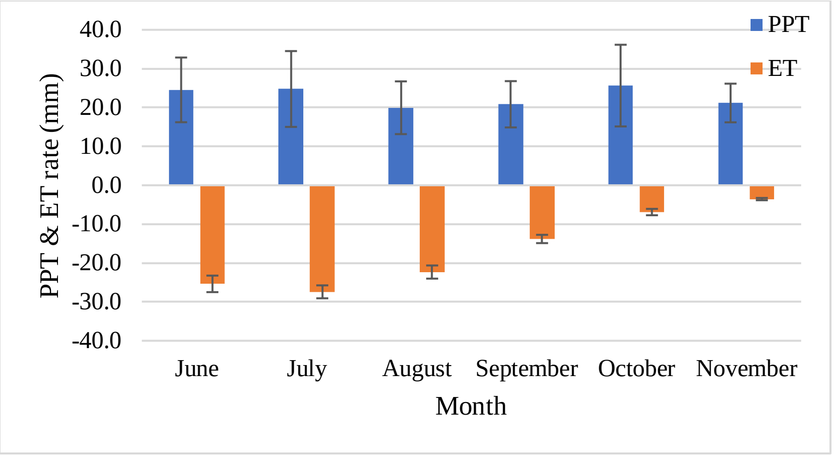


Figure 4.2: *Average ± STDEV monthly PPT & ET rate (mm) over the operating period*

4.4.2 Sediment Characterization

The wetland and pond sediments will comprise settled particulates from the influent, decomposing plant litter, microbial biomass, and possibly precipitates. Wetland systems function as sediment traps, storing more suspended particles than are usually discharged (Arp & Cooper, 2004). The sediment layer is the only sustainable storage compartment for phosphorus and other non-volatile compounds in the wetland system as soil adsorption sites will eventually become saturated and plant storage reaches a steady state after the initial establishment period.

4.4.2.1 Sediment Characteristics:

The average sediment characteristics for the wetland and pond cells are described in Table 4.1. The average pH value for sediment in both the wetlands and pond was 7.5. The pH was between 6.5 to 8.5, which is the optimal range for ammonification (Patrick & Wyatt, 1964), nitrification (Amatya et al., 1970), and organic matter oxidation.

Table 4.1: *Sediment Characteristics*

Parameter	W1 (Mean ± Stdev)	Pond (Mean ± Stdev)	W2 (Mean ± Stdev)
Sediment Depth (cm)	19 ± 6	15 ± 5	20 ± 6
TS (mg/L)	7568 ± 3104 (7665)	44294 ± 52367 (33980)	8144 ± 1495 (7950)
VS (mg/L)	1779 ± 310 (1825)	3661 ± 3723 (2900)	1438 ± 374 (1520)
TOC* (mg/L)	575 ± 311 (705)	596 ± 194 (499)	604 ± 204 (552)
TN (mg/L)	43 ± 11 (43)	128 ± 114 (119)	46 ± 18 (40)
NO_3^--N (mg/L)	7.1 ± 4.4 (6.3)	55.2 ± 45.5 (40.8)	8.7 ± 3.3 (7.9)
NH_4^+-N (mg/L)	3.5 ± 3.8 (1.7)	23.8 ± 41.1 (7.1)	12.2 ± 4.2 (12.3)
TP (mg/L)	24.8 ± 7.2 (26.1)	78.0 ± 52.4 (84.4)	28.7 ± 2.9 (28.6)
PO_4^{3-}-P (mg/L)	0.9 ± 1.1 (0.3)	15.2 ± 20.4 (7.5)	1.3 ± 0.5 (1.3)
C:N:P*	19.6:1.5:1	15.1:1.4:1	20.5:2.0:1
TN/VS (%)	2.4 ± 0.7	3.5 ± 1.2	3.7 ± 2.9
TP/VS (%)	1.4 ± 0.5	2.8 ± 1.7	2.2 ± 0.9
pH	7.5 ± 0.1	7.5 ± 0.2	7.5 ± 0.2

*For TOC: the value of n=3 for all three locations.
Note: The value in the bracket shows the median value for that parameter.

High TS concentrations, as well as high TS to VS ratios, were generally observed in the sediment samples across the system. This is most likely from clay soil becoming entrained with some sediment samples. High average TS and VS values were observed in the pond as compared to wetland cells 1 and 2, however, no significant differences were observed due to the high variance in pond sediment concentrations ($P>0.05$). TOC concentrations were very similar between the three cells and ranged from 575-604 mg/L ($P>0.05$). As with VS, the two wetland cells had similar average concentrations while the pond cell had higher average concentrations for the nitrogen and phosphorus species, with TP being significantly higher ($P<0.05$). The organic P fraction was predominant in all three cells and organic N predominated in the wetland cells, while surprisingly high nitrate concentrations were observed in the Pond sediment. Lower NH_4^+ observed in the wetland cells compared with the pond sediment could relate to a combination of plant uptake and greater nitrification in the shallower water column. Plants have been shown to significantly affect total N reduction in constructed wetlands (Chung et al., 2008; Fraser et al., 2004b).

The C:N:P ratios were very similar between the two wetland cells, while there was proportionally less carbon in the pond sediment. The N:P ratio in wetland sediments ranged from 1.5 to 2.0 in both wetlands. This ratio was lower than the plant's N:P ratio of 10:1 to 12:1. The C and N loss to the atmosphere also resulted in a higher proportion of P in the wetland sediment system.

4.4.2.2 Sediment Accumulation:

The sediment accumulation rate is a critical design and operating factor for constructed wetland systems and is required for the design of operating depth, sludge storage depth as well as desludging frequency. It was hypothesized that sediment accumulation would be greatest in W1 compared with P and W2 and that sediment accumulation would be higher at the beginning of each cell.

Sediment depth with distance from the inlet is presented in Figure 4.3. No significant differences between the average depths in each cell were observed with an average of 18.2 ± 3.0 cm at 95% C.I.; however, the average sediment depth was lower in the pond cell, possibly reflecting a more compressed sediment layer in the deeper pond cell. Johannesson et al. (2011) investigated a constructed wetland for 4 years treating sanitary wastewater and the depth of accumulated sediment near the inlet zone ranges from 10-30 cm with a thicker layer closest to the inlet pipe. A clear trend of increasing sediment depth with distance was observed in the first cell, which is contrary to the hypothesis and suggests a migration of solids over time toward the end of the first wetland cell. A trend in both the Pond and W2 cells of declining sediment depth with distance was as expected, although no significant differences were observed ($P>0.05$). The similar levels of solids in the three cells suggest that the solids accumulation relates more to the natural production of solids from decaying vegetation than to the influent TSS captured in the system.

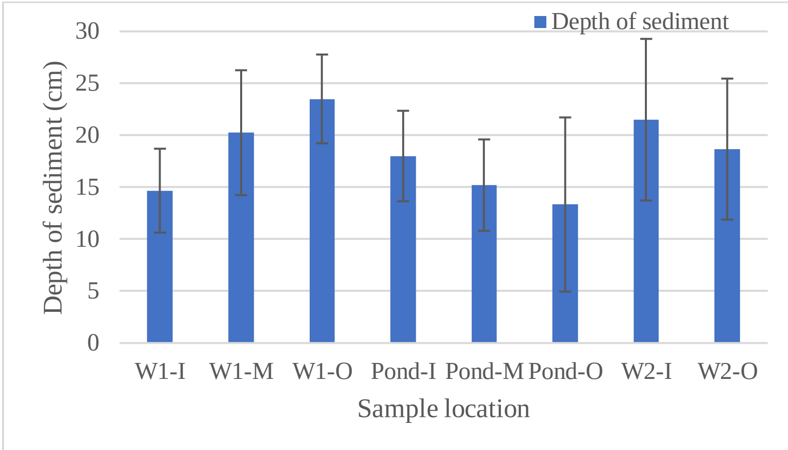


Figure 4.3: *Depth of sediment (Average ± STDEV) at the inlet (I), middle (M), and outlet (O) locations of Wetland 1, Pond, and Wetland 2 Cells*

The average rate of sediment accumulation is 0.86 cm/yr over the 21 years since the system was constructed or 1.65 cm/yr if sediment accumulation over the 10 operating years. A constructed wetland treating agricultural and urban drains resulted in 0.5 to 1.0 cm/yr of sediment accumulation (Siobhan Fennessy et al., 1994). A natural wetland in Nebraska, U.S. observed that sediment depth ranged between 23 to 38 cm and sediment accumulation rate was between 0.18 cm/yr to 0.29 cm/yr (Tang et al., 2015). This study resulted in a higher sediment accumulation rate as compared to the cited literature values.

4.4.2.3 Sediment Areal Mass:

The areal mass of VS, N, and P in the sediment is presented in Figure 4.4. The two wetland cells had a similar average areal mass for VS, N, and P, while the Pond cell contained a significantly higher areal mass than either wetland cell ($P<0.05$). This indicates that the Pond was acting as an important nutrient sink in the overall system. The higher areal mass in the Pond cell relates to higher concentrations and somewhat lower depths indicating a higher density of sediment in the

Pond. No significant differences were observed for VS and N from the inlet to the outlet zone of either wetland ($P>0.05$), while significant differences were observed between the inlet and outlet zone of the Pond cell ($P<0.05$), suggesting that solids migrating from Wetland 1 settled in the Pond cell. Plant nutrient uptake in the wetland cells could also be a factor in the lower N and P mass observed in the wetland sediments.

The total areal mass for TN was 8.4 ± 3.3 g/m^2 in W1 and 9.7 ± 6.5 g/m^2 in W2, while the total areal mass for TP was 5.5 ± 2.3 g/m^2 in W1 and 6.7 ± 2.2 g/m^2 in W2. The pond retained high areal mass for both TN and TP with 14.4 ± 8.4 g/m^2 and 11.8 ± 7.0 g/m^2, respectively. A short-term study by Li et al. (2020) on a surface flow CW treating rural domestic wastewater resulted in a yearly accumulation of 71.7 g/m^2 of nitrogen and 18.2 g/m^2 of phosphorus. Wu et al. (2012) reported nitrogen accumulation in different wetland sediments ranging from 15.9 to 26.6 g/m^2. This study observed lower nitrogen and phosphorus areal mass accumulation in wetland sediment compared to other studies and reflects the relatively low input loading and suggesting significant potential storage capacity in the sediment compartment.

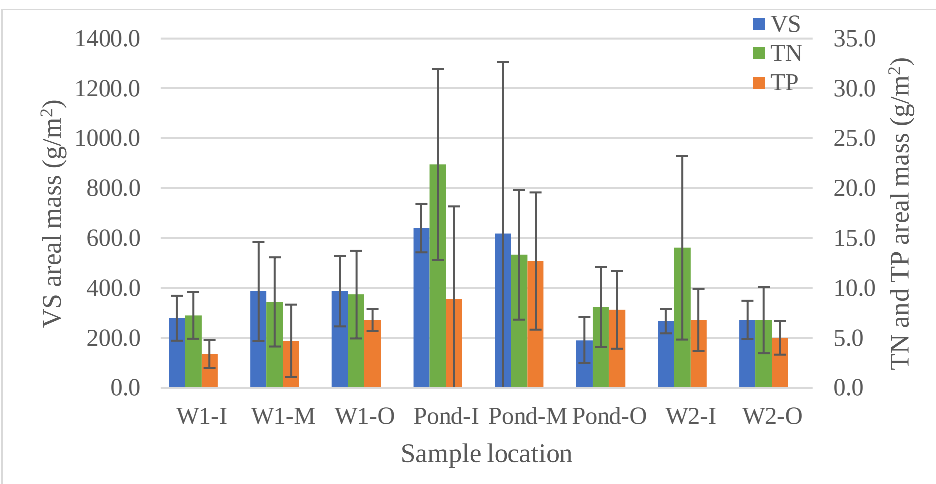


Figure 4.4: *VS, TN, and TP areal biomass (Average ± STDEV) at the inlet (I), middle (M), and outlet (O) of Wetland 1, Pond, and Wetland 2 Cells*

4.4.3 Nitrogen and phosphorus in soil

Soil samples from wetland and pond cells were collected at three depths between 0 to 15 cm at the inlet, middle, and outlet of each cell. As no differences in NO_3^-, NH_4^+ and P content within cells at a given depth were observed ($P>0.05$), the data for each cell at each depth were grouped together (Table 4.2). No significant differences in nitrate or ammonium concentration with depth or between cells were observed ($P>0.05$). A significant decrease in soil P was observed in W1 between the 0-5 and 5-10 cm, declining from 8.0 ± 2.7 to 3.1 ± 1.2 mg/kg ($P<0.05$), while no significant differences were observed between 5-10 and 10-15 cm ($P>0.05$). Also, no significant differences with depth were observed in W2 ($P>0.05$), although a trend of decreasing P content with depth was evident. A significant decline in P content was observed at 0-5 cm between W1 and W2 ($P<0.05$), although no significant differences were observed between the two wetland cells at 5-10 and 10-15 cm depths. This indicates that W1, which experienced the highest P loading, had the highest P adsorption. However, this was only significant in the first 5 cm of the compacted clay liner, suggesting that P adsorption is mostly limited to the surface of the clay due to its very low hydraulic conductivity.

Table 4.2: *NO_3^-, NH_4^+, and Extracted P (Average ± STDEV) content from the inlet, middle, and outlet of W1, P, and W2 at three different depths (0-5 cm, 5-10 cm, 10-15 cm)*

Location	**NO_3^-**					
	0-5 cm		**5-10 cm**		**10-15 cm**	
	(mg/kg dry)	(g/m^2)	(mg/kg dry)	(g/m^2)	(mg/kg dry)	(g/m^2)
W1	35.6 ± 18.5	2.5 ± 1.3	44.1 ± 30.9	3.1 ± 2.2	51.6 ± 34.7	3.6 ± 2.4
Pond	81.0 ± 65.8	5.7 ± 4.6	69.4 ± 23.2	4.9 ± 1.6	66.0 ± 19.1	4.6 ± 1.3
W2	50.9 ± 28.8	3.6 ± 2.0	45.9 ± 31.7	3.2 ± 2.2	39.3 ± 17.1	2.8 ± 1.2
Location	**NH_4^+**					
	0-5 cm		**5-10 cm**		**10-15 cm**	
	(mg/kg dry)	(g/m^2)	(mg/kg dry)	(g/m^2)	(mg/kg dry)	(g/m^2)
W1	24.4 ± 15.0	1.7 ± 1.1	16.5 ± 8.0	1.2 ± 0.6	17.6 ± 7.4	1.2 ± 0.5
Pond	18.7 ± 9.5	1.3 ± 0.7	13.6 ± 5.5	1.3 ± 0.4	14.4 ± 7.2	1.0 ± 0.5
W2	29.6 ± 17.0	2.1 ± 1.2	18.0 ± 5.8	1.0 ± 0.4	25.2 ± 18.3	1.8 ± 1.3
Location	**Extracted-P**					
	0-5 cm		**5-10 cm**		**10-15 cm**	
	(mg/kg dry)	(g/m^2)	(mg/kg dry)	(g/m^2)	(mg/kg dry)	(g/m^2)
W1	8.0 ± 2.7	0.56 ± 0.21	3.1 ± 1.2	0.21 ± 0.08	1.9 ± 0.7	0.13 ± 0.05
Pond	4.6 ± 2.2	0.32 ± 0.16	2.7 ± 0.6	0.19 ± 0.04	2.7 ± 0.5	0.19 ± 0.03
W2	2.0 ± 0.4	0.14 ± 0.03	1.7 ± 0.3	0.12 ± 0.02	1.3 ± 0.3	0.09 ± 0.02

A short-term study by Eylon (1998) on FWS wetland treating dairy wastewater in Glendale, Arizona found <2 mg/kg of NO_3^- and 101-149 mg/kg of NH_4^+ in the first 15 cm of clay soils with average concentrations of <2 mg/L of NO_3^- and 196 mg/L of NH_4^+ respectively. However, this study with an average NO_3^- of 1.03 mg/L and NH_4^+ of 0.42 mg/L over the operating period resulted in high nitrate and low ammonia accumulation in the soil as compared to literature values. The soil absorbed a higher amount of NO_3^- as compared to NH_4^+ and is free to move to the roots by diffusion and mass flow (Schjørring, 1986). Phosphate ions have a strong affinity for clay minerals with a high anion exchange capacity. The amount of clay in soil has a significant impact on phosphate adsorption and can improve phosphorus retention (Mwende Muindi, 2019; Davis, 1994). A study by Reddy et al. (1996) reported that TP in the top 30 cm of wetland soil was 232 mg/kg and 75 g/m^2. A study of the Olsen-P adsorption capacity of Ontario soils reported a Qmax of 225±35 mg/kg from clay soil (Wang et al., 2015). The soil P measured in this study is much lower than the reported adsorption maximum for clay soils, suggesting that adsorption is limited by low-concentration gradients and very low hydraulic conductivity. However, this also implies that the system is nowhere close to reaching P saturation. On the other hand, Ockenden et al. (2014) found that the P accumulation in clayey soil was very low as compared to other types of soil and ranged between 0.0006-0.01 g/m^2, which is much lower than what was observed in this study.

4.4.4 Nitrogen and phosphorus uptake by plants

The essential nutrients in the life cycles of wetland plants are nitrogen and phosphorus (Lee et al., 2012). The above-ground N and P in plant biomass for W1 and W2 were measured. Typha latifolia dominated both wetland cells with similar plant densities of 40 ± 9 and 41 ± 22 plants/m^2 (Table 4.3). No significant differences were observed in N and P plant content nor in above-ground plant nutrient content between W1 and W2 ($P>0.05$). According to Johnston (1991), the aboveground plant uptake 6 to 12 g/m^2 of nitrogen and 0.7 to 1.7 g/m^2 of phosphorus. The plant species have an aboveground nitrogen standing stock ranging from 1.4 to 37.5 g/m^2 (Reddy & DeLaune, 2008) and aboveground phosphorus standing stock values ranging from 3 to 15 g/m^2 (Brix & Schierup, 1988). Also, Gottschall et al. (2007) found that above-ground N and P plant content in October ranged from 4 to 16 g N/m^2 and 0.4 to 2 g P/m^2 respectively in a similarly constructed wetland in Eastern Ontario treating much higher strength dairy farm wastewater. The N and P plant biomass accumulation measured in this study is consistent with literature values.

Table 4.3: *Above Ground Plant Nutrients (Average ± STDEV) in October from W1 and W2*

Location	Density (Plants/m^2)	**N (%dry)**	**N (g/m^2)**	**P (%dry)**	**P (g/m^2)**
W1	40 ± 9	0.94 ± 0.13	12.0 ± 8.2	0.09 ± 0.03	1.15 ± 0.91
W2	41 ± 22	0.94 ± 0.07	18.6 ± 10.5	0.08 ± 0.04	1.49 ± 0.75

The N:P ratio has been used as a sign of N or P deficiency (Suttle & Harrison, 1988). When N: P<14, N appears to be the limiting factor, however when N: P>16, P appears to be the limiting factor (Koerselman & Meuleman, 1996). This study shows N:P ratio in plants between 10:1 to 12:1 and implies that N became a limiting factor for plant growth. The study by Svengsouk & Mitsch (2001) on Typha plants also resulted in N:P<14:1 with N limitation and suggested that nitrogen limitation might be due to high phosphorus uptake, considerable nitrogen losses due to denitrification, and nutrient translocation. Shaver & Melillo (1984) studied the efficiency of nutrient uptake in *T. latifolia* and it was discovered that as nutrient availability increased, nutrient uptake efficiency declined, while total uptake increased. Furthermore, as nutrient availability increased, nutrient absorption from vegetative parts decreased. Most of the phosphorus is absorbed by plant roots; absorption through leaves and shoots is limited to submerged species but the amount is typically low (Vymazal, 2007). Plants usually store a small portion of the total P found in wetlands, hence macrophytes in wetlands have a limited uptake capability (Brix, 1994).

4.4.5 Water Column Nutrient Storage and Seasonal Mass Removal

Nutrient concentrations and storage in the water column of the constructed wetland system during the 2021 operating season are presented in Table 4.4. The system was effective at removing 51% of TN and 83% of TP. Similar nutrient masses were stored in W1 and P due to higher concentrations in W1 and higher water column depth in P, with lower masses reported for W2.

Table 4.4: *Water column nutrient concentration and storage (Average ± 95 CI) for TN and TP in W1, pond, and W2*

Location	TN (mg/L)	TN (g/m^2)	TP (mg/L)	TP (g/m^2)
Lagoon	6.6 ± 2.5	4.4 ± 1.7	1.2 ± 1.1	0.8 ± 0.6
W1	3.6 ± 0.5	1.8 ± 0.3	0.7 ± 0.5	0.4 ± 0.3
Pond	3.5 ± 0.4	4.2 ± 0.5	0.4 ± 0.2	0.5 ± 0.2
W2	3.2 ± 0.2	1.4 ± 0.1	0.2 ± 0.3	0.1 ± 0.1

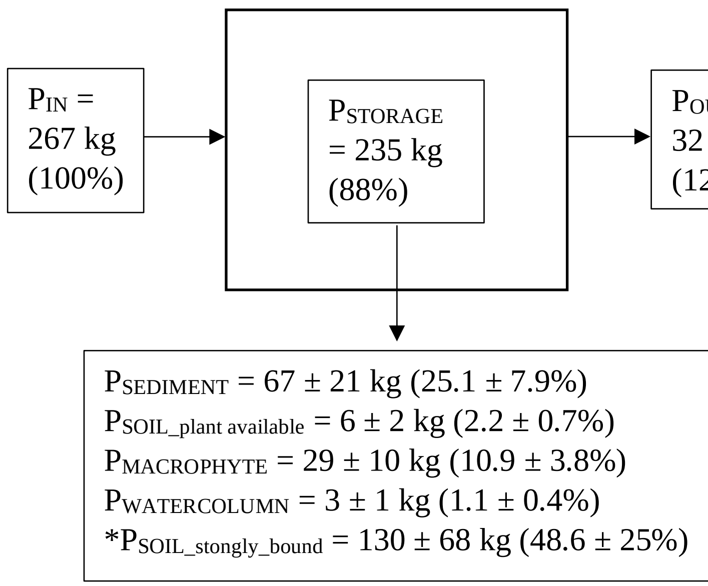
4.4.6 Phosphorus Mass Balance

There is a lack of consensus as to the relative importance and capacity of P storage compartments within constructed wetland systems, which is critical to understanding the long-term viability and performance of these natural systems for P treatment. Several studies determined the soil to be the most important compartment for long-term phosphorus accumulation, followed by plant roots and rhizomes, above-ground biomass, and plant litter (Dolan et al., 1981; Reddy et al., 1999; Dunne et al., 2007). However, the distinction between sediment and underlying soil in a constructed wetland is not clear. P adsorption to the base soil will be limited by the soil's adsorption capacity. Seasonal plant uptake of soluble phosphorus will convert SRP to plant tissue, a portion of which will become plant litter, along with settling and filtration of influent TSS will form the accumulating sediment layer. The sediment accretion is accepted as the only long-term storage compartment for P removal.

A total mass balance for P was conducted on the constructed wetland with Equation 4.7 considering the 11 operating seasons and Equation 4.8 considering the P stored in the different wetland compartments measured or estimated at the end of the 2021 operating season. Values for P storage in macrophyte rhizomes and submerged vegetation were estimated based on relevant literature. The unknown compartment of bound soil P was calculated by substituting Eq. 4.7 into Eq. 4.8. The mass balance is summarized in Figure 4.5.

$$\text{Accumulated P} = \text{Sum}\,(\text{Qin} \times \text{Cin} - \text{Qout} \times \text{Cout}) \qquad \textbf{Equation 4.7}$$

$$\text{Accumulated P} = P_{SEDIMENT} + P_{SOIL\_plant\_available} + P_{SOIL\_strongly\_bound} + P_{MACROPHYTE\_\ (rhizomes\ +\ above\ ground)} + P_{WATER\ COLUMN} \qquad \textbf{Equation 4.8}$$



* Calculated by the difference

Figure 4.5: *Phosphorus mass balance in the wetland system*

The wetland system was very effective at P mass removal, with 267 kg loaded into the system and only 32 kg discharged, resulting in an 88% removal efficiency and total mass storage of 235 kg P (Eq. 4.7).

The largest P storage compartment in the system appears to be soil, with P strongly bound to soil representing 48.6 ± 25% of total P, which was calculated by difference. Plant extractable P, measured in the top 15 cm of the system's clay soil base, represented only 2.2 ± 0.7% of total P and suggests that plant roots were effective at taking up most P in this form. W1 contained the largest proportion of soil plant extractible P, followed by Pond then W2.

The second largest P storage compartment was the sediment with 25.1 ± 7.9% of total P. The Pond contained the largest proportion of sediment P, followed by W1 than W2.

The third largest P storage compartment was the wetland plants with 10.9 ± 3.8% of total P. The above-ground phosphorus storage in macrophytes was 6 ± 3 kg representing 2.2% of the total P. W1 contained the largest proportion of plant P, followed by W2. Phosphorus contained in the plant rhizomes is greater than that of the above-ground biomass (Dunne et al., 2007) and was estimated by applying a 3.8 ratio of rhizome P: above-ground P reported in a similar constructed wetland system in Eastern Ontario treating agricultural wastewater (Gottschall et al., 2007). This provided approximate phosphorus storage in the plant rhizomes of 23 ± 11 kg for the system.

The final P storage compartment was the water column with 1.1 ± 0.4% of total P.

Soil and sediment accumulation is the most important long-term phosphorus sink in wetlands, although it may only be efficient in treatment wetlands with strong biomass production and water covering the sediment, as is the case in free water surface constructed wetlands with emergent plants (Vymazal, 2007). Wetland soils and sediments are often associated with the highest P accumulation in a wetland (Kadlec, 2006). Vegetation stores a very small proportion of nutrients when compared to sediments. Also, it indicates that the soil portion of a matured wetland system is a significant and long-lasting nutrient storage component (Mustafa & Scholz, 2011). A mass balance study by Silbernagl (2017) on a three-cell surface flow wetland following primary treatment and treating domestic wastewater located in the Eastern Cape province of South Africa found 57.1 kg of phosphorus stored in the wetland with 44.2 kg in sediments and 12.9 kg in plants.

This study supports previous findings on the importance of soil adsorption and sediment accretion as the primary P removal mechanisms in FWS wetland systems.

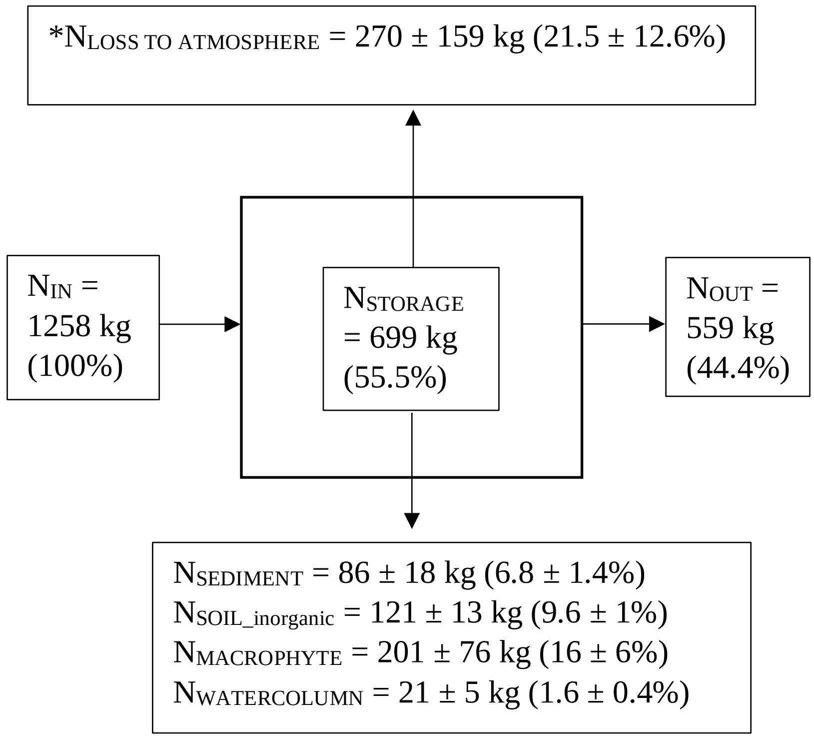
4.4.7 Nitrogen Mass Balance

The denitrification or N loss to the atmosphere accounted for major N removal from the wetland system, followed by soil plus sediments and plants (Lee et al., 2014). The difference in the accumulation rate of N for the soil and sediment was not clearly defined in the previous studies.

A total mass balance for N was conducted on the constructed wetland with Equation 4.9 considering the 11 operating seasons and Equation 4.10 considering the N stored in the different wetland compartments was calculated or estimated at the end of the 2021 operating season. Due to the lack of N data series, the monitoring years 2004 to 2009 organic N fraction was approximated from the 2000-2003 data series. The N storage in macrophyte rhizomes was estimated based on relevant literature. The unknown data of N loss to the atmosphere was calculated by substituting Eq. 4.9 into Eq. 4.10. The mass balance is summarized in Figure 4.6.

$$\text{Accumulated N} = \text{Sum}(Q_{in} \times C_{in} - Q_{out} \times C_{out}) \qquad \textbf{Equation 4.9}$$

$$\text{Accumulated N} = N_{SEDIMENT} + N_{SOIL\_inorganic} + N_{MACROPHYTE\_(rhizomes + above\ ground)} + N_{WATER\ COLUMN} + N_{LOSS\ TO\ ATMOSPHERE} \qquad \textbf{Equation 4.10}$$



* Calculated by the difference

Figure 4.6: *Nitrogen mass balance in the wetland system*

The wetland system was loaded with 1258 kg N and 559 kg N discharged, resulting in a 55.5% removal efficiency and total mass storage of 699 kg N (Eq. 4.9).

The highest N was loss to the atmosphere in the form of denitrification which was 21.5 ± 12.6% of total N.

The largest N storage compartment in the system appears to be plants, with 16 ± 6% of total N. The above-ground nitrogen in plants was 67 ± 48 kg representing 5.3% of the total N. W1 contained the largest amount of plant N, followed by W2. The N assimilation in rhizomes was estimated by applying a 2.0 ratio of above-ground N: rhizome N reported in a similar constructed wetland system in Eastern Ontario treating agricultural wastewater (Gottschall et al., 2007). This provides estimated nitrogen storage in the plant rhizomes of 134 ± 96 kg for the wetland system.

The second largest N storage compartment was the soil with 9.6 ± 1% of total N. The soil compartment stored 91 kg in the form of NO_3^- and 30 kg in the form of NH_4^+.

The third largest N storage compartment was the sediment with 6.8 ± 1.4% of total N. The Pond contained the largest proportion of sediment P, followed by W1 than W2.

The final N storage compartment was the water column with 1.6 ± 0.4% of total N.

A few processes ultimately remove total nitrogen (TN) from the wastewater while most processes just convert nitrogen to its various forms such as ammonium (NH_4^+), nitrite (NO_2^-), and nitrate (NO_3^-) (Vymazal, 2007). The inorganic nitrogen NH_4^+ and NO_3^- from the soil is taken up by plants and converted into organic compounds (Hu et al., 2021). The nitrogen storage in a three-cell surface flow wetland following primary treatment and treating domestic wastewater was 450.1 kg with 138.4 kg in plants and 311.7 kg in sediments (Silbernagl, 2017). However, this study observed high N retention in plants and low N accumulation in sediments.

4.5 Conclusion

Sediment accumulation in the system can be conservatively estimated at 1.65 cm/year and therefore designing berms and outlet structures with an additional 50 cm of freeboard can accommodate 30 years of wetland operation before sediment removal. The pond cell accumulated more organic matter and nutrients on an areal basis than the wetland cells, suggesting a migration of sediment from the first wetland cell to the pond.

The system was loaded with 267 kg of P over monitoring years with partitioning of: 50.8 ± 25.7% in soil, 25.1 ± 7.9% in sediment, 10.9 ± 3.8% in plants, 1.1 ± 0.4% in the water column, and 12% exported in the effluent. The soil was calculated to be the largest P sink with 57% of stored P and almost all attributed to strongly bound soil P. The inlet zone of the wetland had the highest accumulation of extractable phosphorus which decreased significantly from the inlet to the outlet of the wetland system and with depth from 5 to 15 cm. Sediment was the second largest P sink with 28% of stored P. The system was loaded with 1258 kg N over operating years with partitioning of: 9.6 ± 1% in soil, 6.8 ± 1.4% in sediment, 16 ± 6% in plants, 1.6 ± 0.4% in water, 44.4% exported in effluent and 21.5 ± 12.6% lost to the atmosphere, thus denitrification was the most important N removal mechanism in the system followed by plant uptake.

4.6 References

Amatya, I. M., Kansakar, B. R., Tare, V., & Fiksdal, L. (1970). Role of pH on biological Nitrification Process. Journal of the Institute of Engineering, 8(1–2), 119–125. https://doi.org/10.3126/JIE.V8I1-2.5102

Anderson, M. G. (1978). Distribution and Production of Sago Pondweed (Potamogeton Pectinatus L.) on a Northern Prairie Marsh. Ecology, 59(1), 154–160. https://doi.org/10.2307/1936642

APHA. (2012). Standard Methods for the Examination of Water and Wastewater (22nd edition). American Public Health Association.

Arp, C. D., & Cooper, D. J. (2004). Analysis of Sediment Retention in Western Riverine Wetlands: The Yampa River Watershed, Colorado, USA. https://doi.org/10.1007/s00267-004-0027-8

Bastviken, S. (2006). Nitrogen removal in treatment wetlands-Factors influencing spatial and temporal variations.

Brix, H. (1994). Functions of Macrophytes in Constructed Wetlands. Water Science and Technology, 29(4), 71–78. https://doi.org/10.2166/WST.1994.0160

Brix, H., & Schierup, H.-H. (1988). SEWAGE TREATMENT IN CONSTRUCTED REED BEDS — DANISH EXPERIENCES. Water Pollution Research and Control Brighton, 1665–1668. https://doi.org/10.1016/B978-1-4832-8439-2.50158-9

Chung, A. K. C., Wu, Y., Tam, N. F. Y., & Wong, M. H. (2008). Nitrogen and phosphate mass balance in a sub-surface flow constructed wetland for treating municipal wastewater. Ecological Engineering, 32(1), 81–89. https://doi.org/10.1016/J.ECOLENG.2007.09.007

Debusk, W. F., & Reddy, K. R. (2004). Litter decomposition and nutrient dynamics in a phosphorus enriched everglades marsh. https://doi.org/10.1007/s10533-004-7113-0

Dolan, T. J., Bayley, S. E., Zoltek, J., & Hermann, A. J. (1981). Phosphorus Dynamics of a Florida Freshwater Marsh Receiving Treated Wastewater. The Journal of Applied Ecology, 18(1), 205. https://doi.org/10.2307/2402490

Dunne, E. J., Smith, J., Perkins, D. B., Clark, M. W., Jawitz, J. W., & Reddy, K. R. (2007). Phosphorus storages in historically isolated wetland ecosystems and surrounding pasture uplands. Ecological Engineering, 31(1), 16–28. https://doi.org/10.1016/J.ECOLENG.2007.05.004

Eylon, S. (1998). Nitrogen Accumulation in A Constructed Wetland For Dairy Wastewater Treatment. http://hdl.handle.net/10150/191289

Ezzat, S. M., & Moustafa, M. T. (2020). Treating wastewater under zero waste principle using wetland mesocosms. https://doi.org/10.1007/s11783-020-1351-9

Fraser, L. H., Carty, S. M., & Steer, D. (2004). A test of four plant species to reduce total nitrogen and total phosphorus from soil leachate in subsurface wetland microcosms. Bioresource Technology, 94(2), 185–192. https://doi.org/10.1016/J.BIORTECH.2003.11.023

Gottschall, N., Boutin, C., Crolla, A., Kinsley, C., & Champagne, P. (2007). The role of plants in the removal of nutrients at a constructed wetland treating agricultural (dairy) wastewater, Ontario, Canada. Ecological Engineering, 29(2), 154–163. https://doi.org/10.1016/J.ECOLENG.2006.06.004

Hu, Y. H., Zhang, X. Y., Zhang, K., Song, M. H., Gao, J. Q., Dorodnikov, M., Soromotin, A., & Kuzyakov, Y. (2021). Tussock microhabitats increase nitrogen uptake by plants in an alpine wetland. Plant and Soil, 466(1–2), 569–580. https://doi.org/10.1007/S11104-021-05056-Y/FIGURES/7

Johannesson, K. M., Andersson, J. L., & Tonderski, K. S. (2011). The efficiency of a constructed wetland for retention of sediment-associated phosphorus. https://doi.org/10.1007/s10750-011-0728-y

Johnston, C. A. (1991). Sediment and nutrient retention by freshwater wetlands: Effects on surface water quality. Critical Reviews in Environmental Science and Technology, 21(5), 491–565. https://doi.org/10.1080/10643389109388425

Kadlec, R. H. (2006). Free surface wetlands for phosphorus removal: The position of the Everglades Nutrient Removal Project. https://doi.org/10.1016/j.ecoleng.2006.05.019

Keeney, D. R., & Bremner, J. M. (1966). Determination and Isotope-Ratio Analysis of Different Forms of Nitrogen in Soils: 4. Exchangeable Ammonium, Nitrate, and Nitrite by Direct-Distillation Methods. Soil Science Society of America Journal, 30(5), 583–587. https://doi.org/10.2136/SSSAJ1966.03615995003000050016X

Koerselman, W., & Meuleman, A. F. M. (1996). The Vegetation N:P Ratio: a New Tool to Detect the Nature of Nutrient Limitation. The Journal of Applied Ecology, 33(6), 1441. https://doi.org/10.2307/2404783

Lee, S., Maniquiz-Redillas, M. C., Choi, J., & Kim, L. H. (2014). Nitrogen mass balance in a constructed wetland treating piggery wastewater effluent. Journal of Environmental Sciences (China), 26(6), 1260–1266. https://doi.org/10.1016/S1001-0742(13)60597-5

Lee, S. Y., Maniquiz, M. C., Choi, J. Y., Kang, J. H., & Kim, L. H. (2012). Phosphorus mass balance in a surface flow constructed wetland receiving piggery wastewater effluent. Water Science and Technology, 66(4), 712–718. https://doi.org/10.2166/WST.2012.231

Li, X., Li, Y., Lv, D., Li, Y., & Wu, J. (2020). Nitrogen and phosphorus removal performance and bacterial communities in a multi-stage surface flow constructed wetland treating rural domestic sewage. The Science of the Total Environment, 709. https://doi.org/10.1016/J.SCITOTENV.2019.136235

Lu, J., Sun, G., McNulty, S. G., & Amatya, D. M. (2005). A comparison of six potential evapotranspiration methods for regional use in the southeastern United States. Journal of the American Water Resources Association, 41(3), 621–633. https://doi.org/10.1111/J.1752-1688.2005.TB03759.X

Mustafa, A., & Scholz, M. (2011). Nutrient accumulation in Typha latifolia L. and sediment of a representative integrated constructed wetland. Water, Air, and Soil Pollution, 219(1–4), 329–341. https://doi.org/10.1007/S11270-010-0710-8

Mwende Muindi, E. (2019). Understanding Soil Phosphorus. International Journal of Plant & Soil Science, 1–18. https://doi.org/10.9734/IJPSS/2019/V31I230208

Ockenden, M. C., Deasy, C., Quinton, J. N., Surridge, B., & Stoate, C. (2014). Keeping agricultural soil out of rivers: Evidence of sediment and nutrient accumulation within field wetlands in the UK. Journal of Environmental Management, 135, 54–62. https://doi.org/10.1016/J.JENVMAN.2014.01.015

Patrick, Wm. H., & Wyatt, R. (1964). Soil Nitrogen Loss as a Result of Alternate Submergence and Drying. Soil Science Society of America Journal, 28(5), 647–653. https://doi.org/10.2136/SSSAJ1964.03615995002800050021X

Peverly, J. H. (1985). Element Accumulation and Release by Macrophytes in a Wetland Stream. Journal of Environmental Quality, 14(1), 137–143. https://doi.org/10.2134/JEQ1985.00472425001400010028X

Reddy, K. R., & DeLaune, R. D. (2008). Biogeochemistry of Wetlands. https://doi.org/10.1201/9780203491454

Reddy, K. R., Flaig, E. G., & Graetz, D. A. (1996). Phosphorus storage capacity of uplands, wetlands and streams of the Lake Okeechobee Watershed, Florida. Agriculture, Ecosystems & Environment, 59(3), 203–216. https://doi.org/10.1016/0167-8809(96)01039-0

Reddy, K. R., Kadlec, R. H., Flaig, E., & Gale, P. M. (1999). A Review Phosphorus Retention in Streams and Wetlands: A Review. Critical Reviews in Environmental Science and Technology, 29(1), 83–146. https://doi.org/10.1080/10643389991259182

Reddy, K. R., & Smith, W. H. (1987). Aquatic plants for water treatment and resource recovery. 1032.

https://books.google.com/books/about/Aquatic_Plants_for_Water_Treatment_and_R.html?id=WIXb8wGdHegC

Richardson, C. J., & Marshall, P. E. (1986). Processes Controlling Movement, Storage, and Export of Phosphorus in a Fen Peatland. Ecological Monographs, 56(4), 279–302. https://doi.org/10.2307/1942548

Schjørring, J. K. (1986). Nitrate and ammonium absorption by plants growing at a sufficient or insufficient level of phosphorus in nutrient solutions. Plant and Soil 1986 91:3, 91(3), 313–318. https://doi.org/10.1007/BF02198114

Shaver, G. R., & Melillo, J. M. (1984). NUTRIENT BUDGETS OF MARSH PLANTS: EFFICIENCY CONCEPTS AND RELATION TO A V AVAILABILITY 1. Ecology, 65(5), 1491–1510.

Silbernagl, R. (2017). An assessment of the effectiveness of the Crossways Farm Village constructed wetland in the treatment of domestic wastewater.

Siobhan Fennessy, M., Brueske, C. C., & Mitsch, W. J. (1994). Sediment deposition patterns in restored freshwater wetlands using sediment traps. Ecological Engineering, 3(4), 409–428. https://doi.org/10.1016/0925-8574(94)00010-7

Strickland, T., Fisher, L., & Korleski, C. (2010). Ohio Lake Erie Phosphorus Task Force Final Report.

Sudarsan, J. S., Roy, R. L., & Nithiyanantham, S. (2021). Mass Balance Study on Domestic Wastewater Treatment Using Constructed Wetlands. Journal of Water Chemistry and Technology, 43(6), 497–502. https://doi.org/10.3103/S1063455X21060096

Suttle, C. A., & Harrison, P. J. (1988). Ammonium and phosphate uptake rates, N: P supply ratios, and evidence for N and P limitation in some oligotrophic lakes. Limnology and Oceanography, 33(2), 186–202. https://doi.org/10.4319/LO.1988.33.2.0186

Svengsouk, L. J., & Mitsch, W. J. (2001). Dynamics of Mixtures of Typha Latifolia and Schoenoplectus tabernaemontani in Nutrient-Enrichment Wetland Experiments. The American Midland Naturalist, 145(2). https://www.jstor.org/stable/3083109?seq=1

Tang, Z., Gu, Y., Drahota, J., Lagrange, T., Bishop, A., & Kuzila, M. S. (2015). Using Fly Ash as a Marker to Quantify Culturally Accelerated Sediment Accumulation in Playa Wetlands. JAWRA Journal of the American Water Resources Association, 51(6), 1643–1655. https://doi.org/10.1111/1752-1688.12347

Vymazal, J. (2007). Removal of nutrients in various types of constructed wetlands. Science of the Total Environment, 380(1–3), 48–65. https://doi.org/10.1016/J.SCITOTENV.2006.09.014

Vymazal, J. (2010). Constructed Wetlands for Wastewater Treatment. Water 2010, Vol. 2, Pages 530-549, 2(3), 530–549. https://doi.org/10.3390/W2030530

Wang, Y. T., O'Halloran, I. P., Zhang, T. Q., Hu, Q. C., & Tan, C. S. (2015). Phosphorus Sorption Parameters of Soils and Their Relationships with Soil Test Phosphorus. Soil Science Society of America Journal, 79(2), 672–680. https://doi.org/10.2136/SSSAJ2014.07.0307

Weisner, S. (1993). Long-term competitive displacement of Typha latifolia by Typha angustifolia in a eutrophic lake. Oecologia, 94(3), 451–456. https://doi.org/10.1007/BF00317123

Wu, H., Zhang, J., Wei, R., Liang, S., Li, C., & Xie, H. (2012). Nitrogen transformations and balance in constructed wetlands for slightly polluted river water treatment using different macrophytes. https://doi.org/10.1007/s11356-012-0996-8

Zedler, J. B., & Kercher, S. (2005). WETLAND RESOURCES: Status, Trends, Ecosystem Services, and Restorability. https://doi.org/10.1146/annurev.energy.30.050504.144248

CHAPTER: 5

Conclusions And Recommendations

5.1 Conclusions

The FWS constructed wetland treating municipal lagoon effluent wastewater was thoroughly investigated to develop a significant understanding of long-term system performance and nutrient retention in soil, sediments, and plants. The specific conclusions include:

- The wetland with an increased depth from 25 cm to 50 cm in the restart phase performed well at removing TSS. The removal efficiency was observed at 60% and 41% in the Startup and Mature phases, which was increased to 97% TSS removal in the restart phase.
- The BOD removal efficiency was observed as 44% and 50% in the Startup and Mature phases, respectively with effluent concentrations below the objection limit of 10 mg/L. During the restart Phase, BOD was below the detection limit with a COD removal efficiency of 57%.
- Nitrate and ammonium were largely removed during the startup phase, likely due to plant uptake during establishment. Almost no nitrate reduction was observed during the mature operating phase, likely due to a lack of carbon for denitrification in the wetland system, while high ammonium removal efficiencies were observed. However, after the dormant period, the nitrate removal efficiency increased to 37% and organic nitrogen to 83%, likely due to a combination of increased operating depth and increased sediment accumulation, while only 18% removal efficiency was achieved for ammonium.
- The wetland system performed well for TP and SRP by achieving more than 83% removal efficiency in all three phases with no reduction in SRP removal throughout the treatment

cells, while TP concentrations increased in the first wetland and pond cells, likely due to the accumulating sediment layer related TSS resuspension.

- No effect of aging on the wetland system performance was observed for BOD and TP removal.
- No seasonal temperature effect was observed for BOD_5 and NO_3^-, however, the TP removal efficiency was higher during the late-summer and fall periods compared with the early summer and late fall.
- The kinetic removal rate (k) was calculated for the FWS wetland polishing municipal lagoon effluent using the first-order P-k-C* model. The averaged k values were: 45.8 m/yr for BOD, 7.9 m/yr for TN, 15.5 m/yr for NH_4^+, and 3.0 m/yr for NO_3^-. BOD, TN, and NH_4^+ k values were consistent with the literature, while the NO_3^- k value was considerably lower suggesting that denitrification in a FWS wetland is limited with low-carbon wastewater.
- Sediment accumulation in the system can be conservatively estimated at 1.65 cm/year and therefore designing berms and outlet structures with an additional 50 cm of freeboard can accommodate 30 years of wetland operation before sediment removal.
- The system was loaded with 267 kg of P over operating years with partitioning of: 50.8 ± 25.7% in soil, 25.1 ± 7.9% in sediment, 10.9 ± 3.8% in plants, 1.1 ± 0.4% in the water column and 12% exported in the effluent.
- The system was loaded with 1258 kg N over operating years with partitioning of: 9.6 ± 1% in soil, 6.8 ± 1.4% in sediment, 16 ± 6% in plants, 1.6 ± 0.4% in water, 44% exported in effluent and 21.5 ± 12.6% lost to the atmosphere, thus denitrification was the most important N removal mechanism in the system followed by plant uptake.

5.2 Future recommendations

The following recommendations are aimed to encourage research focused on the marsh-pond-marsh FWS wetland system treating municipal wastewater lagoon:

- Conduct a C mass balance on the system and determine CO_2 emissions versus C sequestration in the sediment.
- Further analyze the soil P to elucidate the P-bound fractions and confirm the P mass balance.
- Continue to evaluate the system restart to confirm that the performance continues beyond the first season of the restart period.