

EVALUATION OF POND/WETLAND/VEGETATIVE FILTER SYSTEM TO TREAT BEEF
MANURE PILE AND OUTDOOR CONFINEMENT AREA RUNOFF

by

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Abstract

Regulations with respect to the storage and handling of animal waste by livestock operations state that no person shall keep animals in a permanent confinement area unless there is a nutrient management strategy. Various studies indicate that constructed wetlands and vegetative filters are effective add-on technologies to supplement the treatment provided by conventional sedimentation pond systems in livestock operations. Seven months of data from a Pond/Wetland/Vegetative Filter system receiving cattle feedlot runoff were used to i) evaluate its efficiency removing organic matter, solids and nutrients from water, (ii) quantify constructed wetland kinetic removal rates described by the P-C-k model and examine the impact of temperature, (iii) evaluate and compare the performance of a two types of vegetative filter systems at different hydraulic loading rates; and (iv) recommend an optimum management option with design loading for beef producers. Results indicate that the effect of isolated rain events on the performance of the wetland showed to be contingent on the intensity of the event, former humidity conditions of the feedlot and the hydraulic capacity of the wetland. This experiment suggests that even relatively small VFSs or short FPs can markedly improve quality of runoff from livestock operations, and that it is possible to achieve significant mass and concentration removals if they are properly operated and maintained in conjunction with a pre-treatment system. The Pond/Wetland/Vegetative Filter system was effective at reducing Chemical Oxygen Demand COD (+92%), Total Phosphorus (+93%), Orthophosphate as P (+91%), Total Inorganic Nitrogen (+96%), Ammonia as N (+97%) and Nitrate as N (+82%) from manure and exercise yard runoff, providing a cost-effective treatment option for beef producers.

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List of Abbreviations

AFO	Animal feeding operation	NMP	Nutrient Management Plan
ANOVA	Analysis of Variance	NO ₃ ⁻	Nitrate
BMP	Best management practice	N _{org}	Organic Nitrogen
BOD ₅	5 days - Biological Oxygen Demand	NVZ	Nitrate Vulnerable Zone
C*	Background concentration	OMAFRA	Ontario Ministry of Food and Rural Affairs
CGAP	Code of Good Agricultural Practice	P	Number of tanks in series
COD	Chemical Oxygen Demand	P1	Pond 1
CW	Constructed Wetland	PC	Pump Chamber
DO	Dissolved Oxygen	Piez	Piezometer
EPA / USEPA	Environmental Protection Agency (of the United States)	PO ₄ ³⁻	Orthophosphate
FAO	Food and Agricultural Organization (of the United Nations)	SS	Sum of Squares
FP	Flow Path	TIN	Total Inorganic Nitrogen
FP#	Lysimeter # (in Flow Path)	TN	Total Nitrogen
FWS	Flow Water Surface (Constructed Wetland)	TP	Total Phosphorus
HLR	Hydraulic Loading Rate	TSS	Totals Suspended Solids
HRT	Hydraulic Retention Time	USGS	United States Geological Survey
k	First-order areal rate constant	VF	Vegetative Filter
MSLF	Lack of Fit Mean Square	VFS	Vegetative Filter Strip
MSPE	Pure Error Mean Square	VFS#	Lysimeter row # (in Vegetative Filter Strip)
NH ₄ ⁺	Ammonia	W	Wetland
		WHO	World Health Organization
		θ	Arrhenius Coefficient

Chapter I.

Introduction

1.1 Background

In the last 25 years, global meat production has doubled (FAO, 2018), and with it the threats to the water, soil and air from released and leached animal waste. Many regions, especially developing countries, have seen the largest increases in livestock production. The densification of meat producing farms has resulted in an increase in the production and accumulation of animal waste that was not previously the case (FAO, 2005). In 2005, countries like the Netherlands already generated 15 million tonnes more manure than it could apply to its land each year (FAO, 2005). This accelerated accumulation of animal waste has a negative impact on the environment when it is not well managed.

The problem is not restricted to large livestock producers. In a study carried out in 2000, it was concluded that although the large-scale swine operations could represent a greater environmental risk than small-scale ones, due to their awareness, they were not necessarily any worse polluters than small operators (Hassinger II *et al.*, 2000). Many regulations governing animal feeding operations (AFOs) do not address pollutants from a vast majority of livestock producers (OMAFRA, 2002; EPA, 2003; European Council, 2008). In the USA, of the estimated 1.3 million U.S. farms with livestock, only 238,000 are considered animal feeding operations and only 15,500 (1.2%) are regulated. These remaining non-regulated operations are considered small livestock producers, which although smaller in size continue to significantly contribute to water pollution in many watersheds (EPA, 2003).

Currently, there are laws that regulate the storage and use of manure in different countries, and

propose the implementation of management practices that help reduce the impact of these wastes on natural resources. To effectively implement these practices, it is necessary to have clear design guidelines that provide livestock producers with viable treatment alternatives that adjust to the changing nature of their waste.

1.2 Impacts of livestock operations runoff

Precipitation reaching uncovered livestock operation areas such as confinement pens, feed storage manure storage and land spreading, can transport contaminants into the environment. Concentrated runoff can cause “exceedances” of water quality standards, threatening drinking water supplies and posing a risks of degradation of aquatic life (EPA, 2011). Major forms of pollution associated with manure management in livestock production include: eutrophication of surface water; leaching of nitrates and pathogens into groundwater, and the buildup of excess nutrients and heavy metal in the soil.

The accumulation of manure and urine in livestock farms generates significant impacts on the environment and society (Steinfeld, 2006; Vaillant *et al.*, 2009; Márquez Molina *et al.*, 2014). Cattle excrete approximately 5-6% of their body weight per day, excreting 70-80% of the nitrogen and 90% of the phosphorus from their diet (Vaillant *et al.*, 2009). It is estimated that in the US livestock are responsible for an estimated 33% of the load of nitrogen and phosphorus into freshwater resources in the US, and that in China this percentage reaches 72% for N and 94% for P (Steinfeld, 2006). These residues can potentially contribute to environmental pollution from nutrients (nitrogen and phosphorus), organic matter, sediments, pathogens, trace elements, hormones and antibiotics to either surface or groundwater resources.

The increasing rate at which nutrients and organic matter enter ecosystems accelerates eutrophication events in receiving environments. This can result in the development of harmful algae

blooms which can produce cyanotoxins that may pose a risk to human health. As well, a decline in dissolved oxygen levels in surface water bodies can affect the proper functioning of ecosystems and poses risks to aquatic life. All this results in increased operating expenses of public water supplies, loss of domestic and recreational use opportunities, and shifts in habitat characteristics which directly affects biodiversity (Steinfeld, 2006; Burkholder *et al.*, 2007; Wang and Wang, 2016).

The potential effects reported on the accumulation of nutrients in the groundwater by the operation of livestock farms are varied. According to the United States Geological Survey (USGS) report for the National Water-Quality Assessment Program in 2012, concentrations of dissolved phosphorus in agricultural watersheds in the states of Washington and Nebraska exceeded the recommended regional criteria for total phosphorus, established by the U.S. Environmental Protection Agency (USEPA), in over 75 percent of the stream samples (USGS, 2012). The criteria differ among the sites because of natural variations in background concentrations, but ranges between 0.022 mg/L to 0.076 mg/L in the U.S. (USGS, 2012). At sites with high concentrations in groundwater, groundwater discharge into streams located in the same watershed increases the concentrations of nutrients in the streams. Thus, nutrient losses into groundwater can also trigger noxious algal blooms or eutrophication processes.

Exposure to nitrate can occur in both private wells and community water supplies, and may present health risks. In a survey conducted between 1991 and 1992 in Ontario, Canada, out of 1292 wells that were tested where 50% of the land area was used for agricultural production, 14% contained nitrate concentrations above 10 mg/l (as N) limit (Goss, Barry and Rudolph, 1998). In Europe, the European Commission's report on Nitrates Directive implementation for the period 2004-2007 revealed that 15% of groundwater monitoring stations in the 27 European Union member states (EU-27) found nitrate levels above the limit of 50 mg NO₃⁻/L (11.3 mg/L as N) (European Commission, 2010). Also, Molina *et al.* (2014) analyzed the distribution of nitrate in groundwater near livestock feeding lots in

Argentina and found concentrations between 6 and 56 mg/L (Márquez Molina *et al.*, 2014). Likewise, a study in the US revealed that in 200 wells of mainly residential use in areas of Delaware where land use was mostly from poultry production, the average concentration of nitrate was 21.8 mg/L (Ritter and Chirnside, 1987). These values are considered highly toxic according to the World Health Organization (WHO), which provides a guideline value for nitrate concentration in drinking water of only 10 mg/L (as N), and states that higher concentrations can cause methemoglobinemia in children, or stomach cancer, reproductive problems and insulin-dependent diabetes in adults (Burkholder *et al.*, 2007; WHO, 2011).

1.3 Legislation and Regulation

There is a wide range of laws and policies at the international, and national level, which aim to control and mitigate the risks of water pollution caused by the loss of agricultural contaminants to aquatic environments. The Nitrates Directive (1991) forms an integral part of the Water Framework Directive of the European Union and represents one of the main tools to protect waters against agricultural pressures (European Commission, 2010). The Directive aims to protect water quality across Europe by preventing nitrates from agricultural sources polluting ground and surface waters. In the US, the unified National Strategy for Animal Feeding Operations developed jointly by the U.S. Department of Agriculture (USDA) and EPA, enforces the National Pollutant Discharge Elimination System by requiring the largest animal feeding operations (AFOs) to obtain permits under the authority of the Clean Water Act (USDA and EPA, 1999). In Canada, water quality is protected under the Fisheries Act at a federal level or under the provincial management acts at a provincial level. In the province of Ontario specifically, water quality is protected under the Nutrient Management Act by the Ontario Ministry of Agriculture and Rural Affairs (OMAFRA) as well as more generally through the Water Resources Act by the Ontario Ministry of Environment, Conservation and Parks (MOECP).

All these regulations recognize that the treatment objectives for contaminants coming from livestock

producers depend on the context in which the operations are developed. Removal criteria should be linked to the background concentration of each pollutant, climate conditions, soil characteristics, water body dilution factors and the assimilation capacity of the local environment. Therefore, regulatory agencies and policies promote the creation of programs based on the limitations of each operation or region. The Nitrates Directive (1991) designates Nitrate Vulnerable Zones (NVZs) to generate compulsory action programs, and promotes the establishment of a Code of Good Agricultural Practice (CGAP), defined by every State Member, to be implemented by farmers on a voluntary basis. Similarly, the US National Strategy for AFOs and the Ontario Nutrient Management Act expects AFOs owners and operators to develop and implement technically sound and economically feasible comprehensive Nutrient Management Plans (NMPs), which balance land application of manure with crop nutrient requirements.

In the case of beef operations, CGAPs and NMPs should include strategies for the management and treatment of runoff. The vast majority of these operations maintain their animals in open lots that expose large areas to precipitation (EPA, 2003). The runoff produced in feedlots and exercise yards, in conjunction with permanent nutrient storage facilities, such as open-top manure stockpiles, generate the need to collect and handle highly loaded wastewater after rain events (EPA, 2003). The inadequate collection, management and treatment of this type of wastewater can endanger water supplies. In Canada, of the 229 manure spills recorded by the Ontario Ministry of the Environment that impacted surface water bodies in the South West Region of Ontario between 1988 and 1999, 17% were attributed to problems with manure handling and storages (Goss *et al.*, 2001). Thus, regulations require that any permanent nutrient storage facility must be equipped with a runoff management system capable of handling all the liquid generated by the facility (EPA, 2003; OMAFRA, 2017). Given that manure runoff management has broad goals, the proposed treatment technologies vary among regions, depending on the specific context of each operation.

1.4 Proposed Treatment Technologies

CGAPs and NMPs identify the action required for meeting nutrient management goals at specific agricultural operations and give a set of recommended Best Management Practices (BMPs) that the operators can implement. Constructed wetlands (CWs) have been demonstrated to be effective add-on technologies to supplement the treatment provided by conventional sedimentation pond systems, and are under study as a BMP to treat manure storage runoff (Pries and McGarry, 2001; Riemersma, 2001; Kadlec and Wallace, 2009). Sedimentation ponds or basins are man-made depressions confined by earthen structures that are used to remove settleable matter from water and protect downstream treatment structures from clogging. Ponds can also be used to remove contaminants from water by means of plant-uptake or biological activity (EPA, 2009). Wetlands are ecosystems that are inundated by water either permanently or seasonally because of their location in the landscape and that, given a high rate of biological activity, can transform common water pollutants into harmless byproducts. Modern treatment wetlands, or CW, are systems that have been designed to emphasize specific characteristics for improved treatment capacity (Kadlec and Wallace, 2009).

Where further polishing of wetland effluent needs to be accomplished due to stringent discharge requirements, vegetative filters (VFs) are being considered as an alternative treatment technology to prevent contaminants to be discharged into the environment (Riemersma, 2001). VFs are vegetative areas such as pastures or grassed waterways that are used for treating runoff by settling, filtration, adsorption of pollutants and infiltration (Dickey & Vanderholm, 1980). Compared with many conventional technologies that rely on inputs of fossil fuels, natural treatment technologies, such as ponds, CWs and VFs, may provide livestock operators with viable, cost-effective solutions to manage feedlot runoff.

The treatment mechanisms of treatment ponds, CW and VFs are natural processes driven by solar radiation, kinetic energy of wind, and the storage of potential energy in biomass that make these technologies appropriate for use by livestock producers (Kadlec and Wallace, 2009). However, their design requirements and models used to design and evaluate CWs and VFs, such as kinetic constants and optimal hydraulic loading rates, have been largely based on municipal systems, with a scarcity of data from livestock wastewater treatment systems and even fewer studies focused on cold-climate applications.

In the case of CWs, there are two globally recognized models for their design: Reed *et al.* (1995) model (Equation 1.1) and Kadlec & Knight (2009) P-C*-k model (Equation 1.2), both based in first-order kinetics where the wetland is considered as an attached growth biological reactor.

$$\ln\left(\frac{C_i}{C_e}\right) = K_T \cdot \frac{V \cdot n}{Q} \quad (1.1)$$

C_i : influent constituent concentration, mg/L
 C_e : effluent constituent concentration, mg/L
 K : volumetric first-order kinetic rate constant, d⁻¹
 V : treatment volume of the wetland, m³
 n : porosity (percent, expressed as a decimal fraction)
 Q : average flow rate through the wetland, m³/d

$$\frac{C_e - C^*}{C_i - C^*} = \frac{1}{(1 + k/[P \cdot Q/A])^P} \quad (1.2)$$

C^* : constituent background concentration, mg/L
 k : areal first-order kinetic rate constant, m/yr
 P : is the apparent number of tanks in series (TIS)
 A : treatment surface area of the wetland, m²

The main differences between the models are that Reed's equations use a volumetric basis for the rate constant and a plug-flow approach, as opposed to an areal basis and a tanks-in-series approach

as used in the P-C*-k model (Reed *et al.*, 1995). Plug-flow will theoretically provide more pollutant removal than a completely mixed reactor for the same reactor volume and, intuitively, could describe better the flow in a wetland. However, ideal plug-flow can never be achieved in practice and it has been noted that observed weathering behavior in real wetland environments may be represented by a tanks-in-series model more accurately (Kadlec and Wallace, 2009). Given that some water quality parameters are in fact mixtures of different chemical compounds that are lumped as the same class of materials due to measurement procedures, they can be weathered upon exposure to outdoor environments. Weathering refers to the gradually selective stripping of individual components from a mixture of contaminants, or water constituents, such as BOD or TSS, and that can be degraded or removed at different rates (Kadlec and Wallace, 2009). The P-C*-k model includes the parameter P to account for this phenomenon, and thus, is more widely accepted than other models or design procedures for the design of constructed wetlands, mainly for the treatment of domestic and municipal wastewater.

As for vegetative filters, the literature offers a varied list of design considerations, but there is not yet a globally accepted procedure. Koelsch *et. al* (2006) present a summary of sizing recommendations for VFs based on past research and field demonstration projects. Within the listed criteria are included: minimum infiltration areas to accommodate precipitation events with specific intensities and return periods (from 1-yr, 2-hr to 25-yr, 24-h storms); minimum flow through times (from 5-min to 2-hr); specific hydraulic loading rates (from 4.5 mm/day to 5cm/week); nutrient loading rates equal to crop nutrient utilization; infiltration area to drainage area ratios (from 2:1 to 10:1); minimum filter length per percent slope unit (200ft/ 1% slope); and even minimum infiltration area per cattle head (from 1 ha/200 head to 1 ha/1000 head). Most of the design criteria reported in the literature come from experience but lack a scientific basis, which makes them difficult to be implemented under different influent concentrations or flow and soil conditions. Given the lack of consensus, high safety factors must be applied, which usually leads to over-estimation of filter areas.

In this research, data from an experimental pond-CW-VF system treating beef manure pile and exercise yard runoff in eastern Ontario, Canada, is studied. The pond, CW and VF technologies were monitored individually and simultaneously in this research, however, for analysis purposes the system was divided in two components: 1) pond-CW system, and 2) VFs infiltration systems. This segmentation in the analysis was possible given that the operation parameters of both treatment stages were independent from each other.

1.5 Research Objectives

It is hypothesized that constructed wetlands in conjunction with vegetative filter systems can achieve significant mass and concentration removals from manure pile and exercise yard runoff and that a pond-CW-VF system would be a cost-effective treatment option for beef producers in Canada.

The goal of this research is to enhance our current understanding of treatment solutions for livestock production waste through the study of a pond-CW-VF application, and to strengthen the confidence with which the CWs and VFs designs are developed. This research will provide new knowledge for the design and operation of pond-CW-VF treatment systems applied to livestock production wastes. As immediate objective of this study was to identify appropriate operating conditions for which these technologies were optimal.

Specific objectives include the following:

1. To characterize beef farm manure pile and exercise yard runoff;
2. To evaluate the pond-CW system performance in treating water quality constituents;

3. To quantify CW kinetic removal rates described by the P-C-k model and examine the impact of temperature;
4. To evaluate and compare the performance of a two types of VF systems across various loading rates in removing nutrients and organic matter from pretreated beef farm runoff;
5. To recommend an optimum management option with design loading for beef producers.

1.6 Thesis Outline

The thesis is organized in manuscript style. Chapter II presents the data and results from the full scale pond-CW system treating livestock runoff evaluated over a seven-month period. This Chapter addresses objectives 1, 2 and 3. Chapter III presents the efficiency results of two VF systems receiving wetland effluent at different hydraulic loading rates and describes the side-by-side-comparison between the them. This Chapter addresses objectives 4 and 5. Chapter IV synthesizes the data and integrates the material addressed in previous chapters. Finally, Chapter V provides a global summary and analysis of the thesis findings as well as recommendations for future research.

References

- Burkholder, J. A., Libra, B., Weyer, P., Hearhcote, S., Kolpin, D., Thorne, P. S. and Wichman, M. (2007) 'Impacts of waste from concentrated animal feeding operations on water quality', *Environmental Health Perspectives*, 115(2), pp. 308–312. doi: 10.1289/ehp.8839.
- Dickey, E. C., & Vanderholm, D. H. (1980). Performance and Design of Vegetative Filters for Feedlot Runoff Treatment. *Livestock Waste: A Renewable Resource*, 257–260.
- EPA (2003) 'National Pollutant Discharge Elimination System Permit Regulation and Effluent Limitation Guidelines and Standards for Concentrated Animal Feeding Operations (CAFOs); Final Rule, Part II', *Fed. Reg.*(2003), 68(29), pp. 7176–7274.
- EPA (2009) *Stormwater Wet Pond and Wetland Management Guidebook*. 1st edn, Center For Watershed Protection. 1st edn. Ellicott: US Environmental Protection Agency.
- EPA (2011) EPA Issues Compliance Orders to Six Concentrated Animal Feeding Operations in Iowa, Kansas and Nebraska, EPAS WEB ARCHIVE. Available at: https://archive.epa.gov/epapages/newsroom_archive/newsreleases/fea0e2129093c049852578f4005f2b5f.html (Accessed: 20 July 2018).

- European Commission (2010) The EU Nitrates Directive 2010 Factsheet. European Union. Available at: <http://ec.europa.eu/environment/pubs/pdf/factsheets/nitrates.pdf>.
- European Council (2008) Directive 2008/1/EC: Integrated pollution prevention and control (IPPC).
- FAO (2005) Pollution from industrialized livestock production, Livestock Policy Brief. Rome.
- FAO (2018) World Livestock Primary Statistics. Available at: <http://www.fao.org/faostat/en/#data/QL>.
- Goss, M. J., Rollins, K. S., McEwan, K., Shaw, J. R. and Lammers-Helps, H. (2001) 'The Management of Manure in Ontario with Respect to Water Quality', Agriculture.
- Goss, M. J., Barry, D. A. J. and Rudolph, D. L. (1998) 'Contamination in Ontario farmstead domestic wells and its association with agriculture: 1. Results from drinking water wells', *Journal of Contaminant Hydrology*, 32(3-4), pp. 267-293. doi: 10.1016/S0169-7722(98)00054-0.
- Hassinger II, W. J., Monahan, K. A., Scanlon, T. L. and Parsons, T. D. (2000) 'Nutrient management practices among swine operations of various sizes', *Journal of the American Veterinary Medical Association*, 217(10), pp. 1526-1530.
- Kadlec, R. H. and Wallace, S. D. (2009) *Treatment Wetlands*. 2nd edn. Boca Raton: Taylor and Francis Group, LLC.
- Koelsch, R. K., Lorimer, J. C., & Mankin, K. R. (2006). *Vegetative Treatment Systems for Management of Open Lot Runoff: Review of Literature*. *Biological Systems Engineering: Papers and Publications*, 22 (1), 141-153
- Márquez Molina, J. J., Urricariet, A. S., Sainato, C. M., Losinno, B. N. and Heredia, O. S (2014) 'Effects of feedlot manure on soil and groundwater assessed with electrical resistivity tomography', *Environmental Earth Sciences*, 73(4), pp. 1459-1472. doi: 10.1007/s12665-014-3496-y.
- OMAFRA (2002) Regulation 267/03 - Nutrient Management Act, 2002. Available at: <https://www.ontario.ca/laws/regulation/030267#BK27>.
- OMAFRA (2017) 'Handling Runoff From Solid Agricultural Source Material Storages and Outside Livestock Areas', pp. 1-8.
- Pries, J. and McGarry, P. (2001) 'Constructed Wetlands For Feedlot Runoff Treatment in Manitoba', in *Water Environment Association of Ontario Technical Conference*. Waterloo.
- Reed, S.C., R.W. Crites, E.J. Middlebrooks (1995) *Natural Systems for Waste Management and Treatment - Second Edition*, McGraw Hill Co, New York, New York.
- Riemersma, S. L. (2001) *Appropriate Treatment and Management Options for Beef Feedlot Runoff in Alberta*. University of Calgary.
- Ritter, W. F. and Chirnside, A. E. M. (1987) 'Influence of Agricultural Practices on Nitrates in the Water Table Aquifer *', 19, pp. 165-178.
- Steinfeld, H. (2006) *Livestock Long Shadow*. Edited by FOA. Rome. Available at: <http://www.fao.org/3/a-a0701e.pdf>.
- USDA and EPA (1999) *U.S. Department of Agriculture & U.S. Environmental Protection Agency Unified National Strategy for Animal Feeding Operations*. USA.
- USEPA (2013) 'Literature Review of Contaminants in Livestock and Poultry Manure and Implications for Water Quality', *Office of Water*, 12(10), p. 943. doi: 10.1016/S1474-4422(13)70230-8.

USGS (2012) Phosphorus and Groundwater : Establishing Links Between Agricultural Use and Transport to Streams.

Vaillant, G. C., Pierzynski, G. M., Ham, J. M. and Derouchey, J. (2009) 'Nutrient Accumulation below Cattle Feedlot Pens in Kansas', *Journal of Environment Quality*, 38(3), p. 909. doi: 10.2134/jeq2008.0205.

Wang, L. K. and Wang, M. S. (2016) *Natural Resources and Control Processes*. doi: 10.1007/978-3-319-26800-2.

WHO (2011) *Nitrate and nitrite in drinking-water*. Geneva.

Chapter II.

Performance and Kinetics of a Pond-Constructed Wetland System Treating Beef Manure Pile and Exercise Yard Runoff in Eastern Ontario

2.1 INTRODUCTION

The recycling of nutrients, implicit in the use of animal manure as an agricultural fertilizer, has always been a central component to sustainable livestock production. However, beef producers often store manure in uncovered piles and keep animals in outdoor confinement areas, giving rise to the generation of manure pile and exercise yard runoff. Regulations with respect to the storage and handling of animal waste by livestock operations requires the implementation of waste storage and treatment systems to prevent discharges from production areas where animals are kept in permanent confinement (USDA and EPA, 1999; OMAFRA, 2002; European Commission, 2010). Conventional uncovered manure storage facilities, permanently exposed to rainfall, represent a challenge for farmers given the significant quantity of runoff. Therefore, innovative, energy efficient and low-cost management practices are required to meet the economic constraints of livestock operations. Generally, the availability of space is not a constraint on many farms, which is why the implementation of constructed wetland technology can provide an appropriate solution. Compared with many conventional technologies that rely on the use of fossil fuels, the treatment mechanisms of constructed wetlands are natural processes driven by solar energy, and have been demonstrated to be effective add-on technologies to supplement the treatment provided by settling pond systems (Pries and McGarry, 2001; Riemersma, 2001; Kadlec and Wallace, 2009). Several key removal mechanisms for organic matter and nutrients are dependent on the presence or absence of oxygen

in the system, especially nitrogen conversion processes. Free-water surface constructed wetlands (FWS CWs) can provide these conditions as a continuum of aerobic to anoxic phases are present in the water column and benthic layer. As well, FWS CWs can attenuate pulse flows from storm events and can tolerate variable water levels. FWS CWs meet livestock producer's preference for low cost and low labor requirements, while using the land available on farms (Bosak *et al.*, 2016).

The P-C*-k model, described by Kadlec and Wallace (2009) is a widely used model to design and characterize the performance of constructed wetlands treating domestic wastewater. Recent studies also demonstrate its applicability in the design of agricultural operation treatment FWS CWs (Jamieson *et al.*, 2007; Beutel *et al.*, 2014); however, data from livestock applications remains scarce. In this study, data from an experimental pond-FWS CW system constructed in 2014 treating beef manure pile and exercise yard runoff in eastern Ontario, Canada is presented and discussed. The objectives of this study are: (i) to characterize manure pile and exercise runoff, (ii) to evaluate the pond-FWS CW system performance in treating water quality constituents, and (iii) to determine wetland kinetic removal rates described by the P-C*-k model and to examine the effect of temperature on kinetic rates. It is intended that the results derived from this study will serve to broaden our current understanding of runoff treatment wetland systems and to strengthen the confidence with which the constants are applied in the design of future treatment systems.

2.2 METHODS

2.2.1 Site Description and Operation

The pilot scale pond-FWS CW constructed wetland system evaluated in this study is located at the Pemdale beef farm in Winchester, Ontario, Canada (approx. 60 km south-east of Ottawa, ON, Canada), which comprises an average of 30 cattle. Over the wetland operating season from May to November, 30-year climate normals vary from 1.3°C in November to 20.5 °C in July with an average of 13.4 °C

and average precipitation of 610 mm. The 2017 operating season had a somewhat above average daily temperature of 15.2°C and was an abnormally wet year, with the farm receiving 922 mm of rainfall from May to November.

The manure is piled on an uncovered 1000 m² concrete pad and spread onto the surrounding fields every three years as fertilizer. Manure and exercise yard runoff are discharged from the concrete pad into a concrete pump chamber (PC) and then pumped into a primary treatment pond (P1), through a header pipe. The pond has an area of 109 m² and a depth of 1.5 m. An overflow weir in vegetated clay soil runs the entire width of the pond with the effluent flowing into a surface flow wetland (W). Both P1 and W were constructed in the native clay soil compacted with an excavator shovel. The constructed wetland spans approximately 9.0 x 11.5 m (103.5 m²) and has an average depth of 0.3 m. The wetland contains mainly cattails (*Typha sp.*) that start re-growing during April and reach full growth by June. The wetland effluent is stored in an existing farm pond (P2) where it flows to an existing farm surface drain. The pond-FWS CW was constructed during the fall of 2014 and is depicted in Figure 2.1.

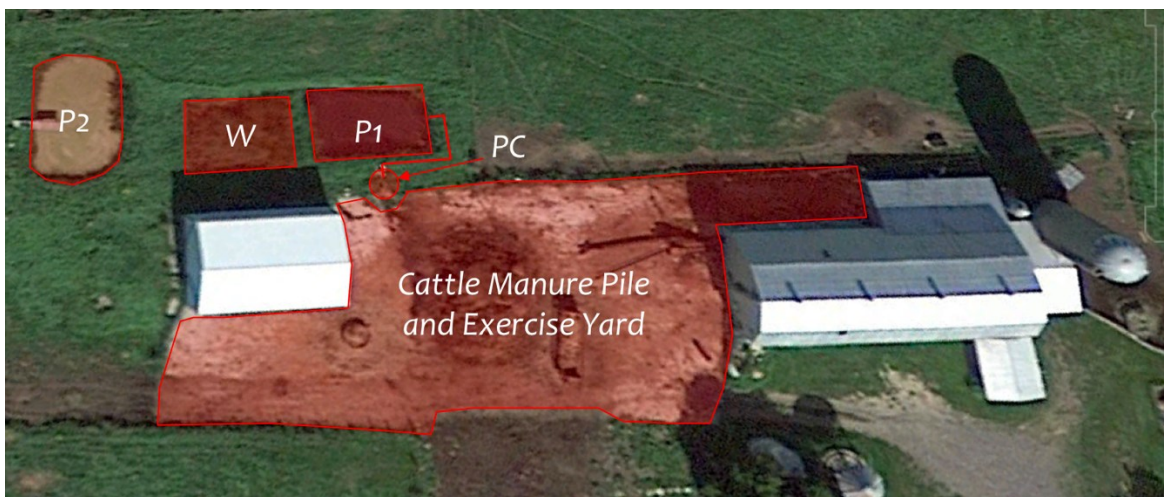


Figure 2.1. Plan view of the exercise yard and Pond/Wetland system in the Pemdale farm, Winchester, ON. Google Earth V 7.3.1.4507 imagery (September 2nd 2016, eye alt 137 m).

A total of 44 sampling visits were conducted throughout seven months, from the beginning of the growing season in May until the beginning of freeze-up in November. Cumulative pump runtime data was recorded during every visit and *in-situ* measurements of water temperature, dissolved oxygen and pH were conducted with a HACH-HQ40D multimeter (LDO101 probe for DO and pH PHC101 probe for pH) in PC, P1 and W. Grab samples were collected once or twice a week from May to November of 2017 and immediately transported to the Environmental Engineering Research Laboratory at the University of Ottawa for analysis. The following analyses were conducted using the procedures outlined in the Standard Methods for the Examination of Water and Wastewater (APHA, 2005): Chemical oxygen demand (COD) - method 5220; biochemical oxygen demand (BOD₅) - method 5210 B; total phosphorus (TP) – method 4500 B & C; phosphates (PO₄³⁻ as P) – method 4500 P; and total suspended solids (TSS) – method D & E. For nitrogen species the following analysis methods were used: total nitrogen (TN) – Hach method 10072; ammonia (NH₄⁺ as N) - Nessler method; nitrates (NO₃⁻ as N) – Hach method 10206 TNT 835. In the case of NH₄⁺ and NO₃⁻, the samples were filtered prior to analysis to avoid turbidity interference. Duplicate samples of TN and TP were also filtered prior to analysis to determine the soluble organic nutrient fractions.

Data from the meteorological station of Kemptville, ON, (25 km from the site) was used as an approximation of local precipitation, air temperature, wind speed and relative humidity during May. The Winchester Agricultural Research Station (adjacent to the Research Site) provided the meteorological data for the rest of the study period (June-November).

2.2.2 Data Analysis

The information collected was divided into six periods to study temporal variation in temperature on system performance, isolate periods of consistent precipitation and reduce variability due to short-term variations of inflow and outflow quality. A lag time, based on a three tank in series model (Kadlec, 1994), was utilized to align the W inlet and outlet samples collected for each of the 44 sample

dates (Fig. 2.2a-b). The pond was assumed to be completely mixed, so no lag correction between PC to P1 samples was applied. To calculate residence times, a volumetric efficiency coefficient of 0.82 was used for W (Kadlec and Wallace, 2009).

Given that the flow was dependent on the intensity and frequency of precipitation events, the hydraulic loading rate (HLR) and the hydraulic retention time (HRT) in each subsystem were variable. The average flow varied from 0.7 to 7.1 m³/d in P1 and from 1.2 to 7.9 m³/d in W. The slight increase in the flow values in W is due to the water mass balance that considers the volume of daily precipitation and daily evaporation of P1 in the calculation of the W influent. This mass balance was also used to correct for any effect of dilution or concentration of the constituents in W, given that its small volume was more sensitive to any change due to precipitation or evapotranspiration. To calculate the daily evaporation rates in P1, the formula developed by Hargreaves (1968) was used as it has been found to provide a reasonable estimate of evaporation in Northern Ontario regions (Rondon, 1972; Panu and Nguyen, 1994). The retention time ranged between 15.3 - 149.0 days and 3.9 - 25.0 days in the P1 and W, respectively. Table 2.1 shows these values for each of the six periods analyzed, as well as the monthly-averaged water temperature used for further correction of kinetic constants.

Table 2.1. Means of flow, HLR, HRT and water temperature for the six monitoring periods in the pond-FWS CW system.

Period	Label	n	Q PC & P1	Q W	HLR P1	HLR W	HRT P1	HRT W	Temp
	#		(m ³ /d)	(m ³ /d)	(L/m ² -d)	(L/m ² -d)	(d)	(d)	(°C)
May 11 - Jun 12	1	7	4.7	4.4	42.9	42.9	23.3	7.0	18.3
Jun 13 - Jul 17	2	8	4.5	5.2	41.4	49.9	24.2	6.0	21.0
Jul 18 - Aug 21	3	10	6.1	6.7	56.3	64.6	17.8	4.6	18.9
Aug 22 - Sep 26	4	9	0.7	1.2	6.7	12.0	149.0	25.0	15.9
Sep 27 - Oct 24	5	5	1.7	2.4	15.8	23.0	63.3	13.1	12.7
Oct 25 - Nov 15	6	5	7.1	7.9	65.5	76.6	15.3	3.9	6.9
Mean			4.1	4.6	37.4	44.1	46.1	9.6	14.8
St.Dev			2.3	2.3	20.9	22.4	48.1	7.4	5.2

To characterize the performance of W in reducing contaminants, the P-C*-k model (Kadlec and Wallace, 2009) was used. The P-C*-k model, or *relaxed Tanks in Series (TIS) concentration model* is defined to be:

$$\frac{C_e - C^*}{C_i - C^*} = \frac{1}{(1 + k/[P \cdot HLR])^P} \quad (2.1)$$

where C_i is the inlet concentration (mg/L), C_e is the outlet concentration (mg/L), C^* is the background concentration, P is the apparent number of tanks in series (TIS), and k is the first-order areal rate constant (m/yr). Considering the potential for dependence of the kinetic constants on temperature, the Arrhenius temperature coefficient θ was incorporated in the formula to normalize k for temperature effects. The k values are reported at 20°C:

$$k = k_T \cdot \theta^{T-20} \quad (2.2)$$

where θ is the Arrhenius temperature coefficient.

These two formulas were used to calibrate the parameters of the P-C*-k model that best predict the concentration values observed in the W effluent for each of the constituents for the six periods from May to November.

A routine was programmed to uniformly generate random values of the parameters C*, P, θ , and k and evaluate iteratively the function described in the P-C*-k model for each constituent. The ranges used to generate the array of values for each parameter were: [1 – 4] for P as used by Kaldec & Wallace (2009); [0 – 68 m/y (BOD₅ and COD); 0-18 m/y (TP and PO₄³⁻); 0-32 m/y (TN); 0-26 m/yr (NH₄⁺); 0-9.2 m/y (NO₃⁻)] for k from the maximum values found in literature for each constituent (LWDB, 1994; Rozema *et al*, 2016); [0 – 12 mg/L (BOD₅); 0 – 240 mg/L (COD); 0-5 mg/L (TP and PO₄³⁻); 0-18 mg/L (TN); 0-2.5 mg/L (NH₄⁺); 0-1 m/y (NO₃⁻)] for C* from the minimum concentration values observed in the study for each constituent (Figure 2.2); and [0 – 1.07] for θ as widest range observed by Kaldec & Wallace (2009). In each of the 3000 iterations performed for each constituent, the Sum of Squares (SS) between the modeled and observed effluent concentration values was assessed. These SS values were then used to rank the goodness of fit of the combinations of C*, θ , P, and k, generated with the routine. All the best fit combinations that yielded within 1.5 times the lowest SS were used to calculate average model parameters.

The F-Test for statistical lack of fit analysis was employed to determine if the model was adequate and described well the observed values (Kozak *et al*, 2008).

$$F' = \frac{MSLF}{MSPE} = \frac{\left(\frac{SSLF}{c-4}\right)}{\left(\frac{SSPE}{n-c}\right)} = \frac{\left(\frac{\sum_{i=1}^c \sum_{j=1}^{n_1} (\bar{y}_i - \hat{y}_{ij})^2}{c-4}\right)}{\left(\frac{\sum_{i=1}^c \sum_{j=1}^{n_1} (y_{ij} - \bar{y}_i)^2}{n-c}\right)} \quad (2.3)$$

Where n is the overall sample size, c is the number of fitted values (each value corresponding to a period) and the null hypothesis (H_0 : *no significant lack of fit*) is rejected if $F' > F(c - 4, n - c, 0.005)$.

2.3 RESULTS AND DISCUSSION

The concentration of each of the constituents observed in P1 and W are presented in Figures 2.2a and 2.2b.

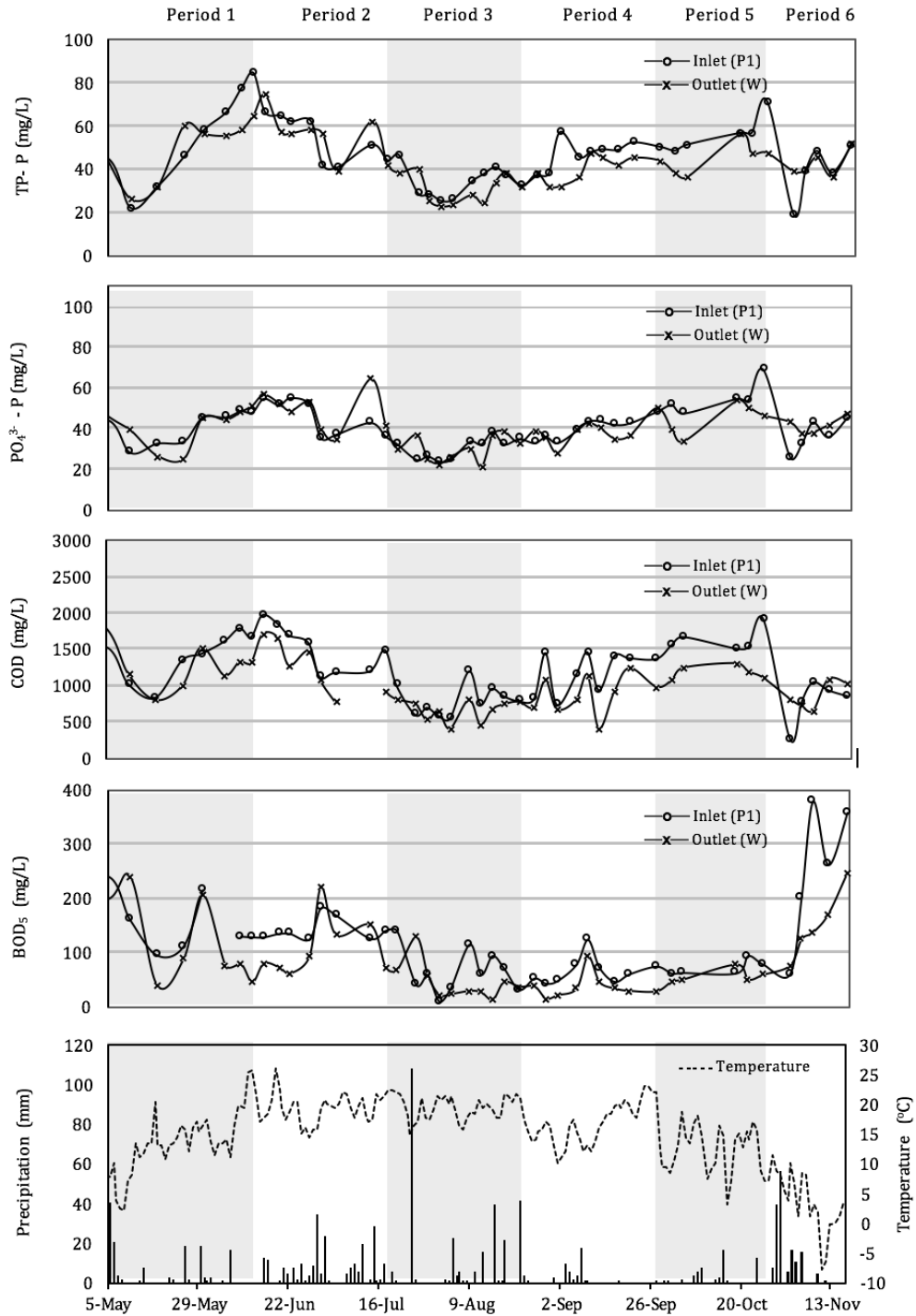


Figure 2.2a. Daily precipitation, average daily air temperature and wetland inflow (P1) and outflow (W) concentrations of the following constituents: TP, PO₄³⁻, COD and BOD₅, along the six selected periods (shaded bands) during the 2017 operating season.

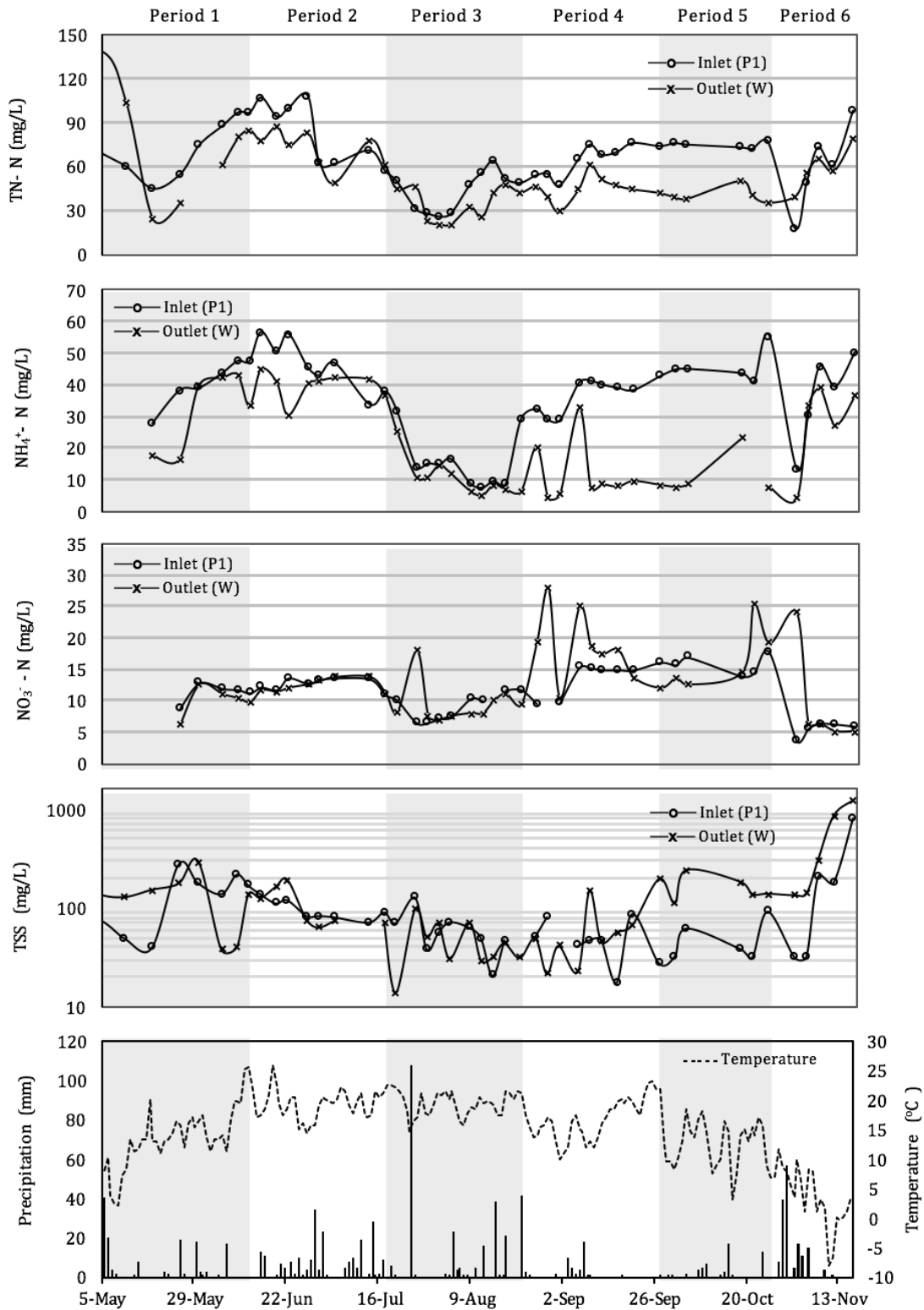


Figure 2.2b. Daily precipitation, average daily temperature and wetland inflow (P1) and outflow (W) concentrations of the following constituents: TN, NH₄⁺, NO₃⁻ and TSS, along the six selected periods (shaded bands) during the 2017 operating season

The similarity between the W inlet and outlet concentration curves (Figure 2.2), especially in the case of PO_4^{3-} and TP, could indicate that these constituents were not being removed from the system or that the HRT was short during periods of operation. Although this last parameter varied by almost one order of magnitude over the study period due to precipitation events (Table 2.1), a tracer study would have been convenient to detect short circuiting and to quantify any change in the HRT in W.

Average concentrations of the water constituents in PC, P1 (i.e. inflow of the wetland) and W (i.e. outflow of the wetland) are provided in Table 2.2.

Table 2.2. Means of concentrations and percent removal in the Pond/Wetland system (May 5 to Nov 15, 2017).

	Avg. Concentration \pm St. Dev. (mg/L)			Removal (%)		
	PC	P1	W	PC-P1	P1-W	PC-W
<i>BOD₅</i>	326 \pm 322	114 \pm 78	80 \pm 61	65	30	76
<i>COD</i>	3079 \pm 1526	1196 \pm 416	970 \pm 320	61	19	69
<i>TP</i>	83.4 \pm 39.8	47.0 \pm 14.7	43.3 \pm 12.5	44	8	48
<i>PO₄³⁻-P</i>	72.1 \pm 25.2	40.5 \pm 10.1	39.6 \pm 9.8	44	2	45
<i>P_{org}-P</i> ‡	11.3 \pm 47.1	6.5 \pm 17.9	3.7 \pm 15.9	43	43	67
<i>TN</i>	145.0 \pm 74.0	65.1 \pm 21.8	51.5 \pm 20.2	55	21	64
<i>NH₄⁺-N</i>	72.6 \pm 42.8	35.2 \pm 13.9	21.8 \pm 14.7	52	38	70
<i>NO₃⁻-N</i>	27.0 \pm 14.1	11.5 \pm 3.6	12.9 \pm 5.7	57	-12	52
<i>N_{org}-N</i> ‡	45.4 \pm 86.6	18.1 \pm 26.1	16.2 \pm 25.7	60	4	65
<i>TSS</i>	220 \pm 146	104 \pm 127	148 \pm 214	53	-43	33

‡ Calculated from mass balance

2.3.1 Organic Matter

The organic matter was analyzed as both BOD_5 and COD to reveal potential patterns between their respective removals and their ratios throughout the system. The COD/ BOD_5 ratio observed in PC averaged 7.6 with a range of 6.0-14.9; which is similar to reported values in previous cattle manure runoff analysis, which ranged from 8.8 to 24.8 (Miner *et al.*, 1966; Loehr, 1967).

The COD/BOD₅ in PC (i.e. raw manure pile runoff) throughout the year shows a high correlation with temperature ($R^2 = 0.96$) (Fig. 2.3). It has been demonstrated that with increasing temperature the degradation of organic matter in composting piles is accelerated resulting in stability being more quickly attained, a lower BOD₅ and higher COD/BOD₅ (Tian *et al.*, 2012). Stability is achieved when the biodegradability of the pile content decreases and subsequently the microbial activity is limited by access to readily biodegradable matter. Biodegradability is proportional to the fraction of BOD₅ in organic matter (Wei *et al.*, 2014), which affects the COD/BOD₅ ratio of the pile. Figure 2.3 shows that during the warmer months the manure pile resembles a compost pile with a reduced fraction of readily degradable organic matter being washed out during rain events. This suggests that the way the manure is being stored, the season and the ambient temperature could have a significant impact on the composition of the organic matter entering the system and, ultimately, in its stability.

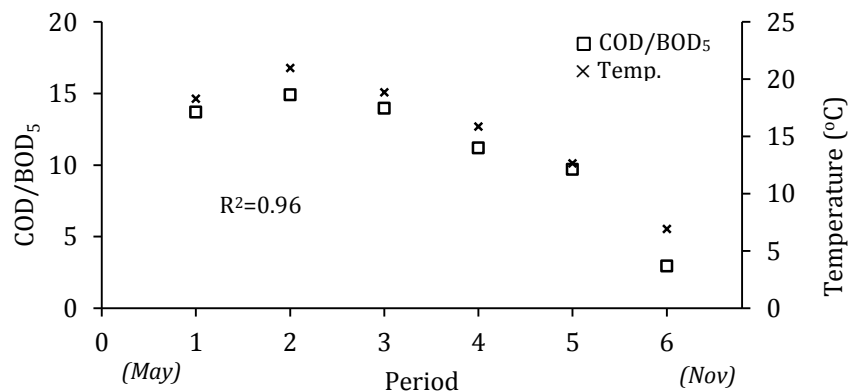


Figure 2.3. Correlation between COD/BOD₅ and temperature in PC over 6 periods.

The influent COD/BOD₅ ratios are much higher than typical municipal wastewater, which has COD/BOD₅ ratios ranging from 1.3 to 3.3 (Kadlec and Wallace, 2009; Metcalf & Eddy, 2014); possibly due to the high content of cellulose in manure, which is subjected to much slower microbial hydrolysis rate than the proteins and lipids commonly found in sewage (Chen *et al.*, 2017). The Pond/Wetland system was effective at removing 76% BOD₅ and 69% COD, with most removal

occurring in the pond (Table 2.2). This is lower than expected in a domestic wastewater surface flow CW treatment system, where efficiencies above 80% can be achieved in the wetland stage alone (Wang *et al.*, 2017). This may be due to the high COD/BOD₅ ratio, which increases throughout the system, and indicates that the organic matter becomes progressively more difficult to treat as the readily biodegradable fraction is removed (Table 2.3).

2.3.2 Phosphorus

Table 2.3 divides TN and TP into particulate and soluble components to better understand nutrient removal mechanisms. Soluble TN includes NO₃⁻, NH₄⁺ and soluble organic N, while soluble TP includes PO₄³⁻ and soluble organic P.

Table 2.3. COD/BOD₅ ratio and N, P particulate/soluble components across locations

	COD/BOD ₅ PC	TN _{particulate} (mg/L)	TN _{soluble} (mg/L)	TP _{particulate} (mg/L)	TP _{soluble} (mg/L)
PC	7.6	11.1	133.9	11.5	71.8
P1	9.8	2.7	62.4	1.9	45.1
W	11.6	2.0	49.5	0.0	43.3

The Pond/Wetland system was quite effective at reducing TP and PO₄³⁻ by 48 and 45%, respectively (Table 2.2). However, most of the removal occurred in P1, where 9.6 mg/L or 84% of the particulate P was removed, likely through sedimentation, and 26.7 mg/L or 37.2% of soluble P was removed either through sorption or plant uptake (Table 2.3). In settling or stabilization ponds, sorption processes have finite retention capacities and can remove no more phosphorus after reaching saturation of active sites. In surface flow wetlands, the median time to saturation is about one year, and it is expected to be shorter in ponds since the volume to surface area ratio is larger, and the lack of area available limits their sorption capacity (Kadlec and Wallace, 2009). It is presumed that the elevated removal of soluble phosphorus observed in P1 was due to plant or microbial uptake related

processes, given the soluble nature of PO_4^{3-} , and the lack of evidence of other significant soluble phosphorus removal mechanisms under these conditions (Kadlec and Wallace, 2009; Beutel *et al.*, 2014; Wang and Wang, 2016). This proposed pathway of removal is supported by the observation of significant growth of floating macrophytes, or duckweed (*Lemna sp.*), throughout the monitoring period in the pond.

By contrast, all the remaining 2.1 mg/L of particulate P was removed in W, while only 4% of the soluble P was removed. The apparent lack of removal in W could be due to the seasonal translocation of phosphorus from rhizomes to aboveground plant parts. During the spring growth spurt, up to 100% of the plant growth P needs may be withdrawn from storage in the belowground biomass (Bernard, 1999, as cited in Kadlec, 2009). Even in warmer months, microbial and plant uptake may be operating rapidly, but so are leaching and decomposition processes in litter, reducing the net removal rate of this nutrient (Bernard, 1999, as cited in Kadlec, 2009). Overall, the wetland performed poorly with respect to phosphorous removal.

Other researchers have reported low TP treatment efficiencies for wetlands receiving wastewater from livestock and agricultural operations, especially as the wetland ages and the sorption sites for P reach saturation (Pries and McGarry, 2001; Stone *et al.*, 2002; Jamieson *et al.*, 2007; Kadlec and Wallace, 2009; Tunçsiper, Drizo and Twohig, 2015). Stone (2002) indicates that the removal of phosphorus from swine derived wastewater can reach 85% at input concentrations of between 1 and 3 mg/L, but as the concentration increases beyond 40 mg/L, this percentage can drop to 6%, which coincides with the removal observed in W in this study.

2.3.3 Nitrogen

The TN average concentration in PC was quite elevated at 145.0 mg/L, with 50.1% as NH_4^+ , 31.3% as N_{org} and 18.6% as NO_3^- . The average concentration of 27.0 mg/L NO_3^- -N was higher than literature

values, where concentrations of 10 mg/L $-N$ were not exceeded in cattle feedlot runoff (Miner *et al.*, 1966; Jamieson *et al.*, 2007; Kadlec and Wallace, 2009). NO_3^- is largely absent in fresh manure; however, nitrification can occur in a manure pile if favorable conditions of moisture content and oxygen transfer are present. For example, peaks of up to 72 mg/L of NO_3^- -N were observed following intermittent low intensity rain (<10 mm / d), such as during period 5 (Fig. 2.2b). These observations are supported by both Tian *et al.* (2012) and Chen *et al.* (2015), who reported increases in NO_3^- concentrations in manure compost piles of more than 600% on a dry weight basis. The relatively high NO_3^- concentrations observed in this study supports the hypothesis that compositing processes are occurring in the manure pile.

The removal of TN in the system was greater in P1 as compared to the wetland. The nitrogen removal mechanisms present in a stabilization pond have been identified by previous research as follows: sedimentation of the particulate nitrogen, mainly organic nitrogen (N_{org}); plant uptake of inorganic nitrogen, mainly NH_4^+ as preferable source of N for assimilation given that it is more reduced energetically than NO_3^- ; and the ammonification-nitrification-denitrification cycle (Kadlec and Wallace, 2009). The latter depends on dissolved oxygen (DO) stratification in the pond, which promotes the conversion of N_{org} and NH_4^+ to NO_3^- on the surface and the stripping of NO_3^- to $N_{2(g)}$ in the deeper anaerobic zones. In P1 all nitrogen species were reduced by 52 to 60% (Table 2.2), keeping their percent distribution almost unchanged from PC to P1. Most of the particulate N (76%), was removed through sedimentation, while the remaining 53.4% of soluble N_{org} was converted to NH_4^+ or hydrolyzed (Tables 2.2 and 2.3). This suggests that in P1 the removal rate of NH_4^+ was greater than that of the other nitrogen species given that it had to counteract the simultaneous ammonification of N_{org} to maintain the same concentration ratio. Nitrification processes are inhibited at DO concentrations below 1 mg/L (Stenstrom and Poduska, 1980), as was the case in P1 where the average DO concentration was 0.52 ± 0.52 mg/L. Given the absence of optimal conditions for

nitrification, it is hypothesised that the active net removal of NH_4^+ and soluble N_{org} observed in P1 was through plant uptake.

Regarding NO_3^- removal, although plant uptake can also play a role, it is more likely that the anaerobic conditions of P1 promoted the removal of this form of nitrogen through denitrification. Cedergreen and Madsen (2002) found that duckweed has between 2 to 30 times higher affinity and 3 to 11 times higher capacity for NH_4^+ uptake than for NO_3^- , depending on the plant structure and the availability of each nitrogen species. Although NO_3^- was also subject to plant uptake in P1, the higher availability of NH_4^+ , and the higher affinity of the plant for this species suggests that this mechanism had a marginal role in NO_3^- removal.

A close correlation between the concentration removal of PO_4^{3-} and the sum of NH_4^+ and soluble N_{org} in P1 is observed ($R^2= 0.97$) in Figure 2.4, while this relationship is not evident in W. These observations support the hypothesis that the most important sink of nutrients in P1, and in the overall system, is linked to plant uptake. Many authors have reported an active role of duckweed in the removal of nutrients in stabilization ponds. The N:P ratio in the harvested biomass varies from 2.55:1 to 3.57:1 in duckweed based treatment applications with livestock wastewater (Cheng *et al.*, 2002; Mohedano *et al.*, 2012), or up to 4.31:1 in municipal wastewater applications (Zhao *et al.*, 2014). This relationship has been shown to vary depending on the concentration of nutrients in the lagoon (Adhikari, Harrigan and Reinhold, 2015), and therefore, it is related to the type of water that is being treated. The same authors report an observed duckweed biomass production of between 18 and 29 $\text{g}/\text{m}^2/\text{day}$ on a dry weight basis. In the present case study, the N:P ratio in the removal of PO_4^{3-} and the sum of NH_4^+ and soluble N_{org} in P1 was 2.65:1 mg / L (Fig. 2.4) and an estimated biomass yield of 33 $\text{g}/\text{m}^2/\text{day}$ on a dry weight basis based on a duckweed N content of 6.6% (Mohedano *et al.*, 2012). Considering that the N: P ratio observed in the removal of NH_4^+ and soluble N_{org} and PO_4^{3-} falls within the reported N:P range for duckweed and that the biomass yield estimated in P1 is comparable

to values reported in literature, it is likely that the observed nutrient removal in P1 was largely due to plant biomass growth. The correlation supports this hypothesis since it indicates that the removal of both nutrients is subject to a common process, and that the performance in the treatment of both P and N responds to the same external factors.

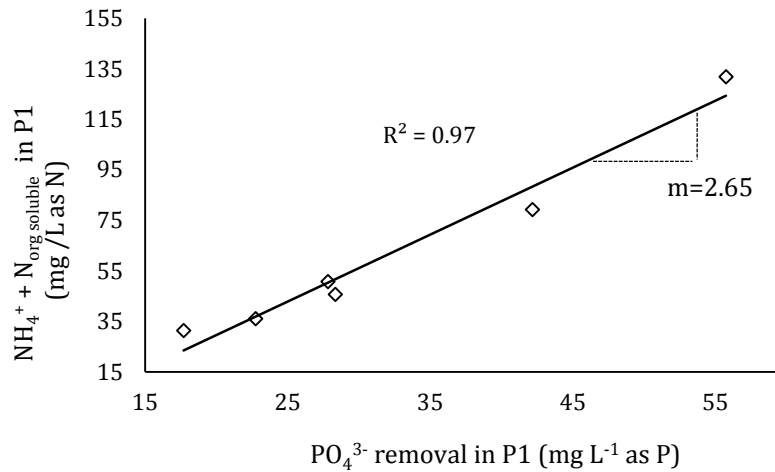


Figure 2.4. Correlation between concentration removal of PO_4^{3-} and $NH_4^+ + N_{org}$ soluble in P1 for the 6 study periods during the 2017 operating season

TN removal was lower in W (21%) compared to P1 (55%). Ammonification and nitrification appear to be the dominant processes taking place, with 1.1 mg/L reduction in soluble N_{org} , a 13.4 mg/L (38%) reduction in NH_4^+ -N, and a 1.4 mg/L increase in NO_3^- -N. Average DO concentrations in W varied monthly from 0.8 to 2.1 mg/L with an overall average of 1.4 mg/L, which is sufficient for nitrification (Stenstrom and Poduska, 1980), while anoxic zones are likely present in the sediment. Although there was evidence of nitrification, it is likely that this process could be inhibited by the high concentration of organic material in W influent. The high C/N ratio favours the conditions for heterotrophs (C-removers) to outcompete autotroph nitrifiers, inducing low nutrients removal. The lower TN removal rates of W as compared to P1 could also be linked to the fact that plant uptake did not appear to play a significant role in W due to a lack of commensurate PO_4^{3-} removal.

2.3.4 Suspended Solids

The system performed poorly with respect to TSS removal. P1 represented the largest net removal of this constituent in the system, which may be attributed to longer retention times and a layer of duckweed that protected the water from wind action. Generally, wetlands are expected to achieve net solids removal higher than 50% in agricultural contexts (Cronk, 1996; Knight *et al.*, 2000), however, this case study showed the opposite. Resuspension of solids in the water column of W was observed during the last 3 study periods from August 22 - November 15 (Figure 2.2b). Period 5, which corresponds to the period with the lowest precipitation coincided with the period of greatest removal of solids in P1 (72.3%) as well as with the greater resuspension of TSS in W (most negative removal, -198%). Disturbing bottom sediments in W during sampling may have been the cause of the elevated TSS concentrations observed. Also, Kadlec and Wallace (2009) argue that in most cases the concentration of TSS in the effluent of a wetland is attributed to biological processes, rather than by the concentration of TSS in the incoming flow. In particular, the solids leaving the system are often generated debris fragments from the accretion by decomposition of leaves and plant material and the resuspension of already settled solids in wetland as opposed to the solids entering the system (Kadlec and Wallace, 2009).

2.3.5 First Order Kinetic Rate Constants

The parameters C^* , k and θ recommended in the literature for agricultural runoff are calculated from the average values of wetland applications that treat wastewaters from livestock operations of different types (swine, cattle, dairy and poultry) and show a high level of variability, likely based on wastewater source and climate (LWDB, 1997; Stone *et al.*, 2002; Jamieson *et al.*, 2007). P-C*-k model parameters were optimized in this study using sum of squares as a lack of fit minimization function with results presented in Table 2.4.

Table 2.4. Optimized P - C^* - k model parameters and θ coefficients for each water constituent

		BOD ₅	COD	TP	PO ₄ ³⁻ -P	TN	NH ₄ ⁺ -N	NO ₃ ⁻ -N
n		24	363	126	117	334	186	176
C^* (mg/L)	Mean	8.8	125.9	2.5	2.5	10.1	1.7	0.5
	± 95%CI	1.1	7.3	0.3	0.3	0.6	0.1	0.0
k_{20} (m/yr)	Mean	8.73	2.42	0.79	0.45	4.07	7.28	-1.00
	± 95%CI	0.48	0.08	0.04	0.04	0.12	0.14	0.05
P	Mean	2.0	2.4	2.6	2.5	2.3	2.9	2.6
	± 95%CI	0.3	0.1	0.2	0.2	0.1	0.1	0.1
θ	Mean	1.061	1.034	1.032	1.035	1.034	1.050	1.035
	± 95%CI	0.003	0.002	0.004	0.004	0.002	0.002	0.003
R ²	Mean	0.93	0.60	0.91	0.88	0.69	0.91	0.81
	Std. Dev	0.01	0.02	0.02	0.01	0.02	0.01	0.02
F .statistic [†]	=MSLF/MSPE	0.34 _c	1.57 _a	0.18 _c	0.33 _c	0.65 _a	0.51 _b	0.47 _b

† Null hypothesis (H_0 :no significant lack of fit) rejected if ratio between Lack of Fit Mean Square (MSLF) and Pure Error Mean Square (MSPE) is higher than: _a $F(2,38,0.05)=3.245$; _b $F(2,37,0.05)=3.252$; _c $F(2, 39, 0.05)=3.238$.

The values of C^* recommended by Knight (2000), are within the range found in the optimization. However, the data show that this parameter is not significant within the ranges stipulated. Although the 95% confidence intervals of C^* were narrow for most constituents, representing between 5 and 12% of the mean value, big changes in this parameter did not affect the goodness of fit between the observed values and the modelled ones. This indicates that the model is not very sensitive to changes in background concentration.

The incorporation of the parameter P accounts for non-ideal flow patterns typical of treatment wetlands. Previous reported k values were calculated assuming a hydraulic efficiency intermediate between completely mixed and plug flow, approximated by three completely mixed tanks in series for all constituents (Kadlec and Wallace, 2009). However, some authors suggest different P values

after studies with tracers or in configurations that minimize short circuiting and promote plug-flow (Kadlec, 2009; Beutel et al., 2014). In this study, the optimized P value varied from 2.0 to 2.9 tanks in series. As with C^* , variation in P showed to be insignificant for the model outcome given that it was not able to identify any specific value of P that could minimize the SS of the modeled data.

The calculated θ coefficients indicates that there is a significant effect of temperature on the kinetic constants in W . The value of $\theta = 1.06$ found for the correction of the k_{BOD} contradicts Kadlec and Wallace (2009), who recommend a value of $\theta = 1.00$ for the design of domestic wastewater wetlands, which suggests k is not temperature dependent. However, Knight *et al.* (2000) calculated an average value of $\theta = 1.03$ from the livestock wastewater database, supporting this study's findings that temperature has an effect on k_{BOD} . The value for $k_{NH_4^+}$ of $\theta = 1.05$ is consistent with θ coefficients found in the literature of approximately 1.05 (Knight *et al.*, 2000; Kadlec and Wallace, 2009).

The calculated, optimized values of θ for k_{TP} , k_{PO_4} and k_{NO_3} do not have much physical sense, since the kinetic rates for these three parameters are all very low or negative, and were not applied to the data. The small (<2% of mean value) standard deviations of R^2 in Table 2.4 indicate that all parameter solutions within the 95% CI ranges fit the data very closely, therefore parameters with narrow confidence intervals are more sensitive in the model. Given that, the θ coefficients found in this study confirm the dependence of the performance of W on temperature and the high sensitivity of the model to changes of this variable.

For every constituent, the F- Test for statistical lack of fit analysis accepted the null hypothesis, which states that there is no significant lack of fit between the modelled and the observed data. This indicates that the model inadequacy is not significantly greater than the experimental error and therefore the model describes the observed data.

Table 2.5 shows a summary of the k values obtained after optimization compared to average values reported in the literature for livestock operations and their corresponding ranges. It should be noted that, as indicated in the table, some of the values reported are based on the C*-k model (Kadlec and Knight, 1996), which assumes a P value close to 3 in the calculations. Although all the constants fall within the wide range of results found in the literature, the TN and TP k_{20} values are lower than the ones reported by different authors. This is reflected in very similar W inflow and outflow concentrations shown in Figure 2.2, especially for TP. Part of the differences between observed values and literature may be due to factors such as the origin of the wastewater to be treated, the age of the wetlands included in each study, the time horizon used in the calculation of the constants, and operational parameters such as the configuration of each system and the level of DO in the water. This last factor is critical when predicting the performance of a wetland in the treatment of different forms of nitrogen, given that their removal is contingent on chain processes that are very sensitive to changes in DO concentration. Constants describing the conversion or removal of different forms of nitrogen between different studies should not be comparable unless there is a correction factor for DO concentration in each system.

Table 2.5. Summary of average and range of first order areal kinetic constants, k , (m/yr) for sizing livestock wastewater treatment wetlands.

	Type	BOD ₅	TP	TN	NH ₄ ⁺ -N	NO ₃ -N
<i>This Study</i> ¹	Cattle	8.7 (2.3 - 37.5) ^a	0.8 (-3.6 - 2.75) ^a	4.1 (0.5 - 8.9) ^a	7.3 (2.7 - 17.0) ^a	-1.0 (-22.3 - 1.5) ^a
(LWDB, 1997) ²	All livestock	22 (7 - 68)	8 (2 - 18)	14 (5 - 32)	10 (-1 - 26)	-
(Stone <i>et al.</i> , 2002) ²	Swine	-	1.79	8.76	9.19	-
(Jamieson <i>et al.</i> , 2007)	Dairy	7.0 (0.64 - 38.7)	1.6 (-10.2 - 5.8)	-	4.3 (-0.3 - 17.2)	-0.2 (-15.1 - 4.3)
(Rozema <i>et al.</i> , 2016) ²	Dairy, abattoir	6 (0.2 - 24.1)	2.9 (-1.3 - 15.2)	-	3.3 (-1.1 - 9.1)	2.5 (-0.2 - 9.2)

a. Range from individual period k calculated using optimized C^* - P and θ

1. There is no reference information for k_{COD} and k_{PO4} , whose averages and ranges in this study were 2.4 (-3.7 - 9.4) and 0.5 (-6.3 - 1.31), respectively.

2. Based on the C*-k model (Kadlec and Knight, 1996).

The k_{20} value should not change with time in a given wetland since it is characteristic of each system. However, by using the C^* , P and θ optimized values to recalculate the individual k values from each of the 6 defined periods, a high variation was observed, reported as the range of variation of the optimized k_{20} constants in Table 2.5. This shows that although a unique optimized k_{20} value could adjust to some extent the observed and predicted values of concentration of the W effluent on an annual scale, the k values varied significantly on a monthly scale (approximate length of each period).

No clear evidence of correlation between k_{20} and precipitation, flow, retention times or with input concentrations was found, but similar patterns of change were observed in the concentration of the constituents in P1 and W after rain events with similar intensity; which could explain the variability of k_{20} observed on a monthly scale.

Figures 2.5 and 2.6 show the reduction in the concentration of TN, TP, COD and BOD₅ in the wetland influent (i.e. P1) and in the wetland effluent (i.e. W) with respect to antecedent rainfall and grouped into thresholds 50 and 100% of the system being flushed. Low cumulative rainfall in the preceding days since the second-last sampling resulted in increased influent concentrations of all constituents with respect to past samples (shaded region A, Fig. 2.5). In contrast, after intense precipitation events or after conditions of high cumulative precipitation, the influent concentrations decreased significantly with respect to past samples (shaded regions B and C, Fig. 2.5). These patterns of concentration change were similar for the wetland effluent (Fig. 2.6). When the volume of recent precipitation events approaches or exceeds the volumetric capacity of P1 and W, a displacement of untreated but strongly diluted water towards the end of the system is generated. This results in noise in the calculation of W efficiency and kinetics in the treatment of pollutants, given that the k_{20} values calculated during periods where this occurs are affected by a dilution factor. For a better understanding of the role of the wetland in the net removal of pollutants during these events it would be necessary to implement sampling frequencies that are linked to the intensity of precipitation events and conduct a separate analysis for each flow event with detailed time series data.

In a study of the characterization of livestock feedlots Miner *et al.* (1996) concludes that runoff from low-intensity rainfall is generally more heavily polluted than that from an intense rain given that in these events the nutrients and organic matter in manure has more time to solubilize in the incoming water (Miner *et al.*, 1966). This could explain the large increase in the concentration of all constituents even two sampling dates after moderate intensity rainfall events. After W inlet concentrations increase, the treatment performance may also be compromised given that there is greater oxygen demand in the system that can affect sensitive processes such as nitrogen removal. A significant difference between values in regions A, B and C was observed in both Figures 2.5 and 2.6 ($P < 0.05$).

Although it is not ideal to use contemporaneous inflow and outflow values to apply the P-C*-k model, the k_{20} values of all the constituents during each of the rainfall events were calculated to better understand how the dilution or concentration of the influent affected the efficiency of the wetland. Figure 2.6 shows the average change in the k_{20} value of the plotted constituents during the rain events grouped in each shaded zone. In zone A, which corresponds to a more concentrated influent, there is an average increase of 76% in all the constituents k_{20} value. On the other hand, when the influent is more diluted, the change in the kinetic rate of the wetland is negative, showing an average reduction of 129% in zone B and 561% in zone C with respect to optimized year-based k_{20} values reported in Table 2.4. Changes in k_{20} values are more pronounced for BOD, COD and TN; whose removals are dependant to common variables, such as DO.

These changes in k_{20} values indicate that the kinetics of the wetland are significantly affected by rainfall events which affect both constituent input concentration and retention time. Therefore, while the P-C*-k model can effectively be used to design event-based systems for average annual performance, the model does not adequately account for individual events. As explained by Kadlec & Wallace (2009), as loadings increase, pollutant removal mechanisms could change: aerobic processes become less of a probable factor and are replaced by anoxic processes.

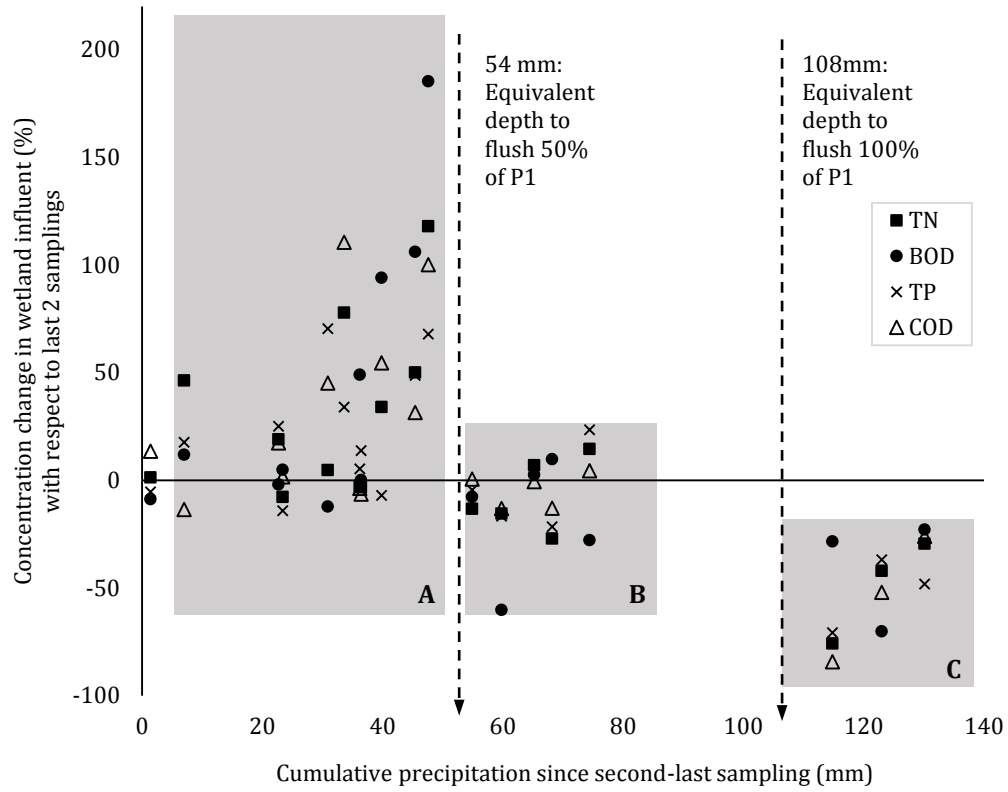


Figure 2.5. Change in wetland influent concentration under different cumulative precipitation scenarios.

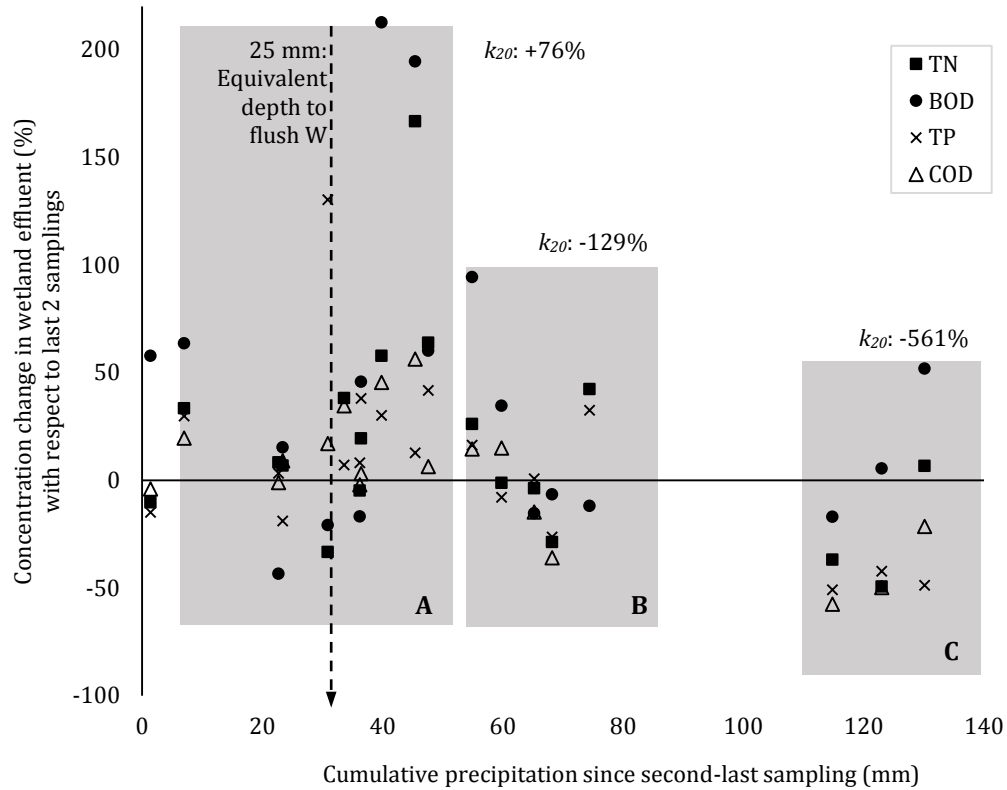


Figure 2.6. Change in wetland effluent concentration under different cumulative precipitation scenarios. k_{20} percent changes are with respect to optimized year-based k_{20} values reported in Table 2.4.

The number of factors that can affect the performance of a wetland are too many to be described accurately by only one constant (k). The high range of reported k values in the limited literature reflects the number of operating and external variables, which are not considered in the parameter calculation. It is important to note that values for k , C^* , P , and θ optimized from insufficient data sets may lead to unrealistic parameter estimates. Ideal data sets would include a broad range of inlet concentrations and hydraulic loading rates over long enough periods to allow consistent wetland performance as indicated by stable outflow concentrations (LWDB, 1997).

The event-driven nature of the wetland system makes it difficult to determine rate coefficients with precision based on a kinetic model for continuous-flow systems. In periods where precipitation events are far apart the system may have been functioning as a batch reactor instead of a tank-in-

series configuration. This is evident in Figures 2.2.a-b between the months of September and October, where there was almost no precipitation and effluent concentrations of NH_4^+ , TN, TP and COD decreased consistently until in the case of NH_4^+ reaching background concentrations. This phenomenon can give rise to problems with the interpretation of the k_{20} values obtained, given that rate coefficients in batch systems will differ considerably from those determined under continuous flow conditions. In spite of the above, first order models are still recommended to describe nutrients and organic matter removal in runoff treatment wetlands over averaged periods of operation (Kadlec and Wallace, 2009).

2.4 CONCLUSIONS

A Pond/Wetland system treating cattle manure pile and exercise yard runoff was evaluated over a 7-month operating period from May to November to determine kinetic removal rates and to study the impact of precipitation events and temperature on treatment kinetics.

The runoff had a high average COD/BOD₅ ratio as well as high NO_3^- concentrations suggesting that the manure pile is acting as a compost system, which plays an important role in the strength of the runoff constituents entering the system.

The pond provided almost all of the removal of dissolved phosphorus in the system. This, in conjunction with a close correlation with removal of dissolved NH_4^+ and soluble N_{org} combined ($R^2 > 0.97$) suggests that plant uptake in the pond, likely through the observed prevalence of duckweed, was the predominant mechanism in nutrient removal.

DO levels favored denitrification in the pond and nitrification in the wetland. The latter suggests that although having only one pond in the system could bring important organic matter and nutrient

removal, the wetland is key to increasing the oxygen levels in the water and to promoting the transformation of nitrogen species, which improves the overall treatment.

The Pond/Wetland system was effective at removing organic matter and nutrients with average removal efficiencies of 76% BOD₅, 69% COD, 64% TN and 48% TP. The Pond/Wetland system was shown to be a viable management practice for cattle manure pile and exercise yard runoff treatment in a temperate climate.

From the optimization of the parameters of the model P-C*-k and θ , it was observed that the number of tanks in series and the background concentration are not very sensitive in the model, while most of the kinetic rates of removal have a strong dependence on temperature. The k_{20} values normalized to 20°C were 8.8 m/y for BOD₅, 2.4 m/y for COD, 0.8 m/y for PO₄³⁻, 0.5 m/y for TP, 4.1 m/y for TN, and 7.3 m/y for NH₄⁺. The computed Arrhenius temperature correction coefficient applied to the k values were 1.061 for BOD₅, 1.034 for COD and TN, 1.032 for TP, 1.035 for PO₄³⁻ and NO₃⁻, and 1.050 for NH₄⁺. The model was effective (and adequate) at representing average system performance on a seasonal to yearly time scale, with all the optimized k constants falling within the ranges reported in literature from similar applications. However, high variations were observed when the model was applied at a daily to monthly time scale due to the event-based nature of the system.

The study shows that short-term k_{20} values increase with low intensity storm events and decrease with medium to high intensity storm events as precipitation affects both the hydraulics and the input concentrations of the system. The effect of isolated rain events on the performance of the wetland showed to be contingent on the intensity of the event, former humidity conditions of the feedlot and the hydraulic capacity of the wetland.

More complex multi-parameter models should be implemented to better represent event-driven systems and to better represent the linkages between DO and nitrogen speciation. Tracer studies are

recommended to calculate hydraulic retention times and to detect possible short circuiting within the wetland that may affect the treatment capacity of the system.

REFERENCES

- Adhikari, U., Harrigan, T. and Reinhold, D. M. (2015) 'Use of duckweed-based constructed wetlands for nutrient recovery and pollutant reduction from dairy wastewater', *Ecological Engineering*. Elsevier B.V., 78, pp. 6–14. doi: 10.1016/j.ecoleng.2014.05.024.
- APHA (2005) *Standard Methods for the Examination of Water and Wastewater*. 21th edn. Washington DC.: American Public Health Association.
- Beutel, M. W., Morgan, M. R., Erlenmeyer, J. J. and Brouillard, E. S. (2014) 'Phosphorus Removal in a Surface-Flow Constructed Wetland Treating Agricultural Runoff', *Journal of Environment Quality*, 43(3), p. 1071. doi: 10.2134/jeq2013.11.0463.
- Bosak, V., VanderZaag, A., Crolla, A., Kinsley, C. and Gordon, R. (2016) 'Performance of a constructed wetland and pretreatment system receiving potato farm wash water', *Water (Switzerland)*, 8(5), pp. 1–14. doi: 10.3390/w8050183.
- Cedergreen, N. and Madsen, T. V. (2002) 'Nitrogen uptake by the floating macrophyte *Lemna minor*', *New Phytologist*, 155(2), pp. 285–292. doi: 10.1046/j.1469-8137.2002.00463.x.
- Chen, R., Wang, Y., Wang, W., Wei, S., Jing, Z. and Lin, X. (2015) 'N₂O emissions and nitrogen transformation during windrow composting of dairy manure', *Journal of Environmental Management*. Elsevier Ltd, 160, pp. 121–127. doi: 10.1016/j.jenvman.2015.06.021.
- Chen, R., Nie, Y., Kato, H., Wu, J., Utashiro, T., Lu, J., Yue, S., Jiang, H., Zhang, L. and Li, Y. (2017) 'Methanogenic degradation of toilet-paper cellulose upon sewage treatment in an anaerobic membrane bioreactor at room temperature', *Bioresource Technology*. Elsevier Ltd, 228, pp. 69–76. doi: 10.1016/j.biortech.2016.12.089.
- Cheng, J., Landesman, L., Bergmann, B. A. and Classen, J., J. (2002) 'Nutrient Removal from Swine Lagoon Liquid by *Lemna minor* 8627', 45(4), pp. 1003–1010.
- Cronk, J. K. (1996) 'Constructed wetlands to treat wastewater from dairy and swine operations: A review', *Agriculture, Ecosystems and Environment*, 58(2–3), pp. 97–114. doi: 10.1016/0167-8809(96)01024-9.
- European Commission (2010) *The EU Nitrates Directive 2010 Factsheet*. European Union. Available at: <http://ec.europa.eu/environment/pubs/pdf/factsheets/nitrates.pdf>.
- Jamieson, R., Gordon, R., Wheeler, N., Smith, E., Stratton, G. and Madani, A. (2007) 'Determination of first order rate constants for wetlands treating livestock wastewater in cold climates', *Journal of Environmental Engineering and Science*, 6(6), pp. 65–72. doi: 10.1139/S06-028.
- Kadlec, R. H. and Wallace, S. D. (2009) *Treatment Wetlands*. 2nd edn. Boca Raton: Taylor and Francis Group, LLC.

- Knight, R. L., Payne Jr, V. W. E., Borer, R. E., Clarke Jr, R., and Pries, J. H. (2000) 'Constructed wetlands for livestock wastewater management', *Ecological Engineering*, 15(1-2), pp. 41-55. doi: 10.1016/S0925-8574(99)00034-8.
- Kozak, A., Kozak, R., Staudhammer, C., and Watts, S. (2008) *Introductory Probability and Statistics: Applications for Forestry and Natural Sciences*, 1st Ed. London, UK: CAB International.
- Loehr, R. C. (1967) 'Effluent Quality from Anaerobic Lagoons Treating Feedlot Wastes', *Journal of the Water Pollution Control Federation*, 30(3), pp. 384-391.
- LWDB (1997) *Constructed Wetlands for Livestock Wastewater Management*.
- Metcalf & Eddy (2014) *Wastewater Engineering: Treatment and Reuse*. 5th edn. USA: McGraw-Hill.
- Miner, J. R., Lipper, R. I., Fina, L. R. and Funk, J. W. (1966) 'Cattle Feedlot Runoff: Its Nature and Variation', *Water Pollution Control Federation*, 38(10), pp. 1582-1591.
- Mohedano, R. A., Costa, R., Tavares, F., and Belli Filho, P. (2012) 'High nutrient removal rate from swine wastes and protein biomass production by full-scale duckweed ponds', 112, pp. 98-104. doi: 10.1016/j.biortech.2012.02.083.
- OMAFRA (2002) Regulation 267/03 - Nutrient Management Act, 2002. Available at: <https://www.ontario.ca/laws/regulation/030267#BK27>.
- Panu, U. S. and Nguyen, T. (1994) 'Estimation of Mean Areal Evaporation in Northwestern Ontario', *Canadian Water Resources Journal*, 19(1), pp. 69-82. doi: 10.4296/cwrj1901069.
- Pries, J. and McGarry, P. (2001) 'Constructed Wetlands For Feedlot Runoff Treatment in Manitoba', in *Water Environment Association of Ontario Technical Conference*. Waterloo.
- Riemersma, S. L. (2001) *Appropriate Treatment and Management Options for Beef Feedlot Runoff in Alberta*. University of Calgary.
- Rondon, F. F. (1972) *Estimation of irrigation requirement for Venezuela*. Utah State University.
- Rozema, E., VanderZaag, A., Wood, J., Drizo, A., Zheng, Y., Madani, A. and Gordon, R. (2016) 'Constructed Wetlands for Agricultural Wastewater Treatment in Northeastern North America: A Review', *Water*, 8(5), p. 173. doi: 10.3390/w8050173.
- Stenstrom, M. K. and Poduska, R. A. (1980) 'The effect of dissolved oxygen concentration on nitrification', *Water Research*, 14(6), pp. 643-649. doi: 10.1016/0043-1354(80)90122-0.
- Stone, K. C., Hunt, P. G., Szogi, A. A., Humenik, F. J. and Rice, J. M. (2002) 'Constructed Wetland Design and Performance for Swine Lagoon Wastewater Treatment', *Society*, 45(3), pp. 723-730.
- Tian, W., Li, L., Liu, F., Zhang, Z., Yu, G., Shen, Q. and Shen, B. (2012) 'Assessment of the maturity and biological parameters of compost produced from dairy manure and rice chaff by excitation - emission matrix fluorescence spectroscopy', *Bioresource Technology*. Elsevier Ltd, 110, pp. 330-337. doi: 10.1016/j.biortech.2012.01.067.
- Tunçsiper, B., Drizo, A. and Twohig, E. (2015) 'Constructed wetlands as a potential management practice for cold climate dairy effluent treatment - VT, USA', *Catena*. Elsevier B.V., 135, pp. 184-192. doi: 10.1016/j.catena.2015.07.028.
- USDA and EPA (1999) *U.S. Department of Agriculture & U.S. Environmental Protection Agency Unified National Strategy for Animal Feeding Operations*. USA.
- Wang, L. K. and Wang, M. S. (2016) *Natural Resources and Control Processes*. doi: 10.1007/978-3-319-26800-2.

- Wang, M., Zhang, D. Q., Dong, J. W. and Tan, S. K. (2017) 'Constructed wetlands for wastewater treatment in cold climate — A review', *Journal of Environmental Sciences (China)*. Elsevier B.V., 57, pp. 293–311. doi: 10.1016/j.jes.2016.12.019.
- Wei, Z. Zhang, X., Wei, Y., Wen, X., Shi, J., Wu, J., Zhao, Y. and Xi, B. (2014) 'Fractions and biodegradability of dissolved organic matter derived from different composts', *Bioresource Technology*. Elsevier Ltd, 161, pp. 179–185. doi: 10.1016/j.biortech.2014.03.032.
- Zhao, Y., Fang, Y., Jin, Y., Huang, J., Bao, H., Fu, T., He, Z., Wang, F. and Zhao, H. (2014) 'Potential of duckweed in the conversion of wastewater nutrients to valuable biomass : A pilot-scale comparison with water hyacinth', *Bioresource Technology*. Elsevier Ltd, 163, pp. 82–91. doi: 10.1016/j.biortech.2014.04.018.

Chapter III.

Side by Side Comparison of a Vegetated Filter and a Flow Path Treating Beef Farm Constructed Wetland Effluent

3.1 INTRODUCTION

Many livestock operations employ outdoor confinement areas and manure storages that, when exposed to rain and snow, create runoff that can present a significant risk to receiving water quality. Preceded by a well-designed settling basin or constructed wetland, vegetative filter (VF) systems are an alternative zero (surface) discharge technology that may address surface and groundwater impairment from open livestock operations. VF systems promote infiltration of runoff and pollutants into the soil profile, enhance filtration of suspended sediment by vegetation, favor soil adsorption and enhance nutrient uptake by plants (Fajardo, Bauder, & Cash, 2001).

Various studies indicate that VFs are effective and potentially acceptable treatment alternatives for livestock operations (Koelsch *et al.*, 2006). However, the wide range of conditions under which such systems have been tested, and the wide variability in the effectiveness to remove contaminants have made it difficult to determine a uniform criterion to evaluate their performance. Generally, VFs can be designed as channelized flow, where flow is concentrated in a relatively narrow channel or flow path (FP), or as overland flow, where effluent is dosed through a stationary perforated pipe, allowing sheet flow through a 2-5% sloped area, or a vegetated filter strip (VFS), with widths ranging from 5 to 30 m (Dickey & Vanderholm, 1980). Despite their potential for improving water quality, VF performance is contingent on many variables including: flow regime, slope, sizing, soil type and

condition, influent concentrations, hydraulic loading and vegetation (Koelsch *et al.*, 2006; Lim, Edwards, Workman, Larson, & Dunn, 1998; Rahman, Rahman, & Wiederholt, 2011). Dickey & Vanderholm (1980) show that channelized flow systems require over five times greater flow lengths for a similar degree of treatment than overland flow systems treating feedlot runoff. Also, Dillaha *et al.* (1988), demonstrated that VF performance is critically dependent on the flow regime, with a VF system removing 70-95%, and 61-70% less P, and N for concentrated flow than for diffuse flow (Dillaha, Sherrard, Lee, Mostaghimi, & Shanholtz, 1988). The treatment mechanisms and the design parameters for optimal performance have been difficult to define, creating a restriction on livestock producer's ability to implement their use (Dickey & Vanderholm, 1980).

Currently the literature provides recommendations for the design of this type of vegetative system. Based on a summary of design criteria used over the past 40 years, Koelsh *et al.* (2006) suggests that a vegetative filter area to drainage area ratio of 2 is necessary to achieve removals of more than 50% in any constituent concentration (Koelsch *et al.*, 2006). Some other criteria are based on the area needed to infiltrate runoff from rainfall events of different intensities and return periods, while others use minimum contact times and desired treatment levels (Koelsch *et al.*, 2006). In Ontario, the main VF design guide restricts the sizing to a maximum hydraulic loading rate (HLR) of 5 cm/week or to the infiltration rate of the saturated soil (OMAFRA, 2006). The uncertainty associated with the wide variety of design procedures generally leads to oversizing of the filters. Since VF systems typically replace land otherwise used for crop or forage production, there is a tendency to minimize filter areas. Hence the need to generate science-based knowledge to optimize design criteria for the benefit of producers and water quality in areas with open livestock operations.

In this study, data from an experimental vegetated filter strip (VFS) constructed in 2014 and a channelized vegetated filter, or flow path (FP), receiving beef manure pile and exercise yard runoff pretreated in a constructed wetland in eastern Ontario is presented and discussed. The study

objectives were to: i) evaluate and compare the performance of a VFS and a FP in removing nutrients and organic matter from pretreated beef farm runoff and ii) recommend an optimum management option with design loading for beef producers.

3.2 METHODS

3.2.1 Site Description and Operation

The VFS and FP compared in this study are located at the Pemdale beef farm in Winchester, Ontario, Canada (60 km south-east of Ottawa), which comprises an average of 30 cattle. This area has 30-year-average temperature normals of 13.4°C between May and November (with a maximum average monthly temperature of 20.5°C in July and a minimum of 1.3°C in November) and a 30-year-average precipitation of 610 mm over the 7 month period (Government of Canada, 2018). In 2017, the average daily temperature was slightly above average at 15.2°C while the site received significantly higher rainfall of 922 mm. As described in Chapter II, runoff from a 1000 m² outdoor confinement area and manure pile is treated by a Pond/Wetland system and then stored in a second treatment pond (P2). Influent from P2 was pumped to the VFS and FP simultaneously via two 0.3HP Goulds model PS41 centrifugal sewage pumps. Pump runtime data was collected by means of a programmable logic controller.

The VFS was constructed during the fall of 2014 and is depicted in Figure 3.1. The VFS has two pairs of 30 cm zero-tension lysimeters, VFS1_{A-B} and VFS2_{A-B}, installed along its 15.6 m length, and was dosed to a header pipe that spans 6 m, defining an infiltration area of approximately 93.6 m², at a 2-3% slope. Each pair of lysimeters were aligned and installed at a distance of 2.3 and 11.1 m from the header, respectively, and were arranged in two parallel rows (labeled as A and B in VFS, Fig. 3.1). The lysimeters are composed of a 30 cm dia. plastic collection funnel installed at a depth of 30 cm and

draining through a 3.8 cm braided plastic hose to a 10 L plastic container with a standpipe to the surface to allow for sample collection. The collection funnel is filled with 2 cm washed gravel, 0.5 cm aquarium stone, silica sand and a geotextile layer to prevent soil migration into the sample container.

The FP follows the natural drainage path for overland flow from the barnyard and adjacent fields. Five lysimeters were installed in a single row at 2.5 m, 9.0 m, 18.2 m, 28.7 m, and 38.7 m from the dosing pipe (Fig. 3.1). The path had an approximate width of 2 m, defining an infiltration area of 87.4 m².

In addition to the monitoring lysimeters in each infiltration system, three piezometers were installed in the study area. The piezometers consist of 5 cm dia. perforated PVC pipes installed 2.0 m in the ground. The first piezometer (Piez1) was located 13 meters away from the last lysimeter of the FP and upslope from both infiltration systems and serves to monitor background groundwater quality. The other two piezometers (Piez2 and Piez3) were located in the middle of the VFS at 6.6 m from the header pipe and in the middle of the FP at 13.6 m from the inlet pipe, respectively. The depth of the water table at each of the piezometers was measured during each visit to the farm.

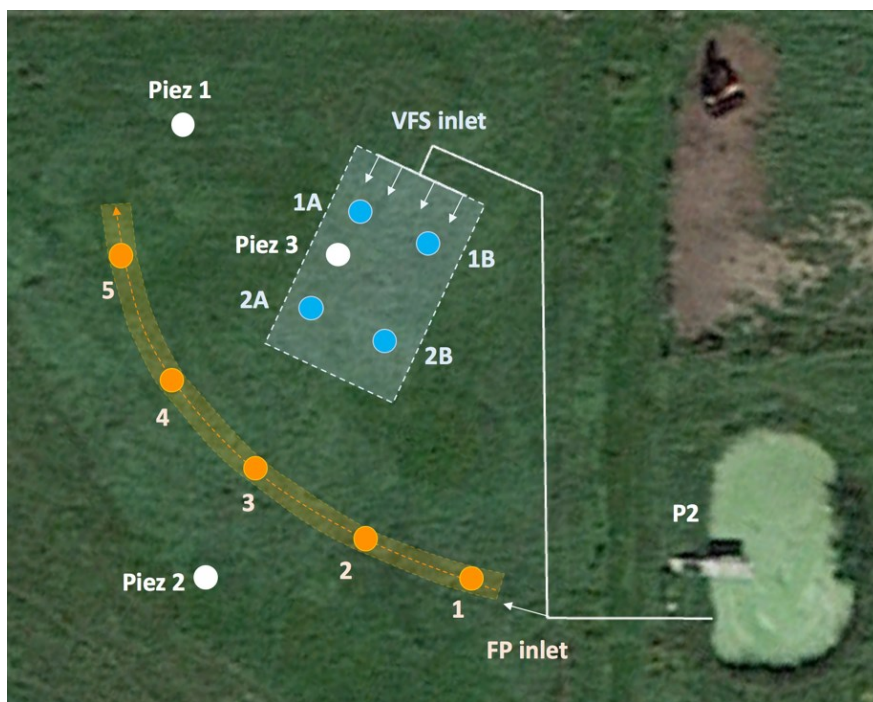


Figure 3.1 Plan view of the Pond 2 and the two infiltration systems (Vegetative Filter Strip and Flow Path) in the Pemdale farm, Winchester, ON. Google Earth V 7.3.1.4507 imagery (September 2nd 2016, eye alt 94 m).

The effluent from the post wetland storage pond (P2) was dosed to both FP and VFS at six different loading rates (1.7, 2.5, 3.0, 3.4, 5.0, 6.6 cm/d). Each dosing program operated for approximately one month. A total of 40 sampling visits were conducted throughout the 7-month operating period, from May until the beginning of freeze-up in November. The dosing schedule is summarized in Table 3.1. Lysimeter and piezometer samples were collected by means of a peristaltic pump and immediately transported to the Environmental Engineering Research Laboratory at the University of Ottawa for analysis. Cumulative pump runtime data was also recorded during each visit. The following analyses were conducted using the procedures outlined in the Standard Methods for the Examination of Water and Wastewater (APHA, 2005): chemical oxygen demand (COD) - method 5220 and phosphates ($\text{PO}_4^{3-} - \text{P}$) - method 4500 P. For nitrogen species, the following analysis methods were used: ammonia ($\text{NH}_4^+ - \text{N}$) - Nessler method; and nitrates ($\text{NO}_3^- - \text{N}$) - Hach method 10206 TNT 835.

Table 3.1. Dosing schedule for FP and VFS during 2017 operating season

Period	Pump program	Flow (m ³ /d)	HLR (cm/d)
Aug 9 - Aug 25	15 sec/ 1 hr	1.6	1.7
Aug 26 - Sep 19	22 sec/1 hr	2.3	2.5
Sep 27 - Nov 10	30 sec/1 hr	2.7	3.0
May 26 - Jun 19	1 min/2hrs ^a	3.0 ^a	3.4 ^a
Jul 14 - Aug 8	45 sec/1 hr	4.5	5.0
Jun 20 - Jul 13	1 min/1 hr ^a	6.0 ^a	6.6 ^a

^a FP was not dosed during this period

The soil at the Pemdale farm is a North Gower clay loam and was rototilled to ensure uniformity and promote growth of the field grasses. Orchardgrass (*Dactylis glomerata*) was the predominant grass species, however, the presence of dandelion (*Taraxacum sp.*) and thistle (*Cirsium arvense*) was also observed throughout the field. Grass and soil samples were collected in the vegetated filter area for nutrient analysis. Three samples of grass, each representing an area of 1 m², were taken twice during the experiment after reaching full growth and before cutting on May 25th and July 20th. The quadrants were selected randomly and were cut by hand. All samples were weighted and dried to calculate their water content, and immediately shipped to the University of Guelph Agriculture and Food laboratory for nutrient content analysis. Soil samples were collected at four locations in the VFS at the end of the experiment: at 3 m behind the header pipe (background), and at 2.3 m, 6.6 m and 11.1 m after the header pipe adjacent to VFS1_{A-B}, Piez3 and VFS2_{A-B}, respectively. Also, only for bioavailable-PO₄³⁻ content analysis, soil samples were collected at 5 locations in the FP: at 2.5 m, 9 m, 18.2 m, 28.7 m and 38.7 m from the inlet pipe, at the same distances as FP1-5, respectively. Each 45 cm soil core was divided into three 15 cm segments. Nitrogen extraction was achieved by mixing for 90 minutes 6 grams of air-dried soil with 2M KCl (Bremner & Keeney, 1966). Phosphorus extraction was conducted using two methods: Mehlich-2 extraction was achieved by mixing for 5 minutes 2 grams of air-dried soil with 20 ml of Mehlich-2 solution (Mehlich, 1984); and Olsen extraction was achieved by mixing for 30 minutes 2 grams of air-dried soil with 40 ml of NaHCO₃ 0.5M (Olsen et al., 1954).

The extracts were filtered, and laboratory analyses were conducted to measure the concentration of ammonium, nitrate and orthophosphate following the methods described above for these constituents.

3.2.2 Data analysis

A volume correction was applied to the VFS and FP efficiency calculations to eliminate the dilution effect caused by direct rain in the system. Data from the meteorological station of Kemptville, ON, (25 km from the site) was used as an approximation of local precipitation, air temperature, wind speed and relative humidity during May. The Winchester Agricultural Research Station (adjacent to the research site) provided the data for the remainder of the study period (June-November). To calculate the daily evapotranspiration rates in the field, the modified version (Rondon, 1972) of the formula developed by Hargreaves (1968) was used based on the fact that it has been found to provide a reasonable estimates of evaporation in Ontario (Panu & Nguyen, 1994; Rondon, 1972).

Unexpectedly high concentrations of many constituents were measured in FP3, suggesting an alternative source of contamination at this sample location. Although fencing was installed around the perimeter of the FP and VFS to keep the cattle from disturbing the experiment, the faeces or decaying body of a small animal could explain the anomalous results observed. Additionally, sample volumes collected were much lower than at adjacent lysimeters. For these reasons, FP3 data was omitted from the mass balance analysis.

The efficiencies of VFS and FP were compared independently under the different dosing programs tested. Data was analyzed using the one-way analysis of variance (ANOVA) test to determine if there were significant differences in the concentration of constituents during each dosing program throughout each system. In addition, this statistical test was also used to determine significant differences between final concentrations under different HLRs.

The comparison of the efficiency between both systems was carried out by means of the same statistical test taking into account two criteria: linear distance from the inlet and infiltration surface area. At equivalent distances or equivalent areas, the average efficiencies achieved during the entire study period were compared. In this case, the percent removal with respect to the initial concentration of each constituent was used as a comparison parameter since the inlet concentrations were different between dosing programs.

To calculate the degree of correlation between variables and to describe trends during the comparison of the efficiency of the systems, the Pearson correlation coefficient was also used in some cases. Under this test, highly correlated variables have coefficients above 0.5, where the maximum is 1.0.

Finally, mass removal was calculated from the difference of the total mass dosed to the systems and the mass captured in the lysimeters. The former value was obtained by multiplying the measured (and emptied) water volume captured in the lysimeters by the concentration of each constituent.

3.3 RESULTS AND DISCUSSION

3.3.1 Constituents concentration

The concentration data for each constituent are presented in Figures 3.2-5, while removal efficiencies are presented in Tables 3.2-4. The values from FP3 (18.2m) should be applied with caution due to high standard deviations and low sample volumes.

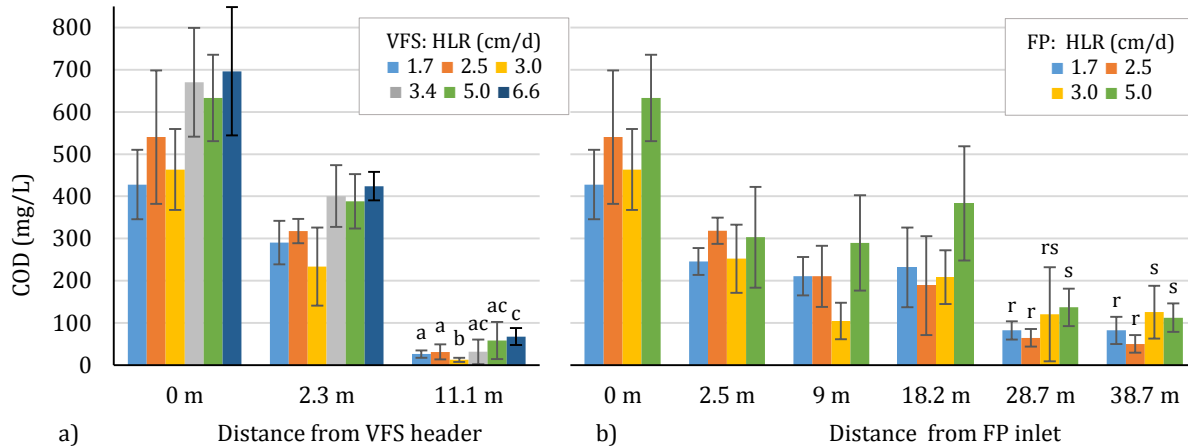


Figure 3.2. COD mean[‡] concentrations at different HLRs along: a) VFS; and b) FP

[‡]The labels shown on the average concentration bars corresponding to the VFS2 lysimeter (11.1 m from header) indicate whether the final concentrations achieved in VFS at different HLRs are significantly different. The labels corresponding to FP4 and FP5 lysimeters (27.8 and 37.8 m from inlet) in addition to comparing the final efficiency of FP at different HLRs, also allow to determine whether there are significant differences between the concentrations reached in these last two lysimeters. In each system, the average concentrations labeled with the same letter are not statistically different under an ANOVA test with a significance level of 95% ($p < 0.05$).

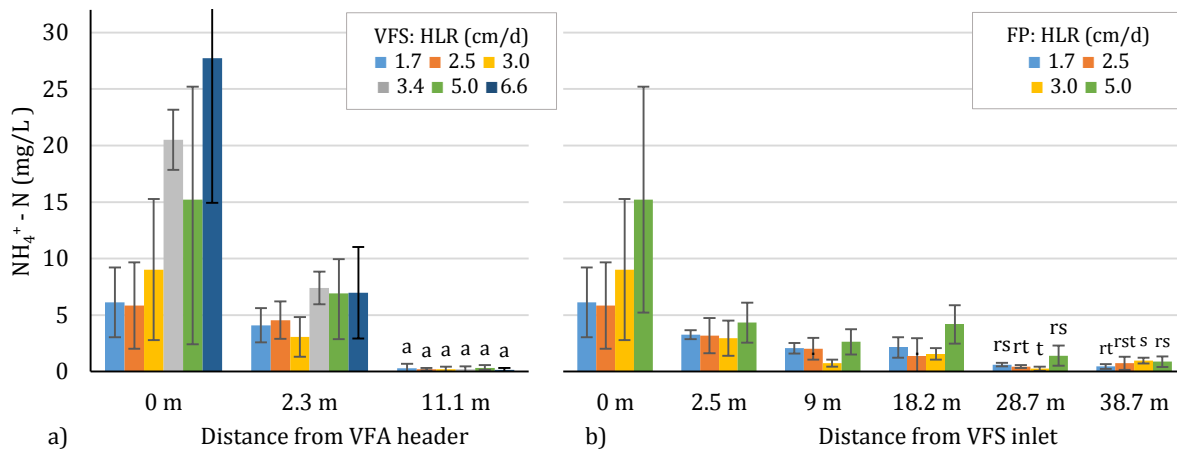


Figure 3.3. NH_4^+ mean[‡] concentrations at different HLRs along: a) VFS; and b) FP

[‡]Means labeled with the same letter are not significantly different by ANOVA test ($p > 0.05$)

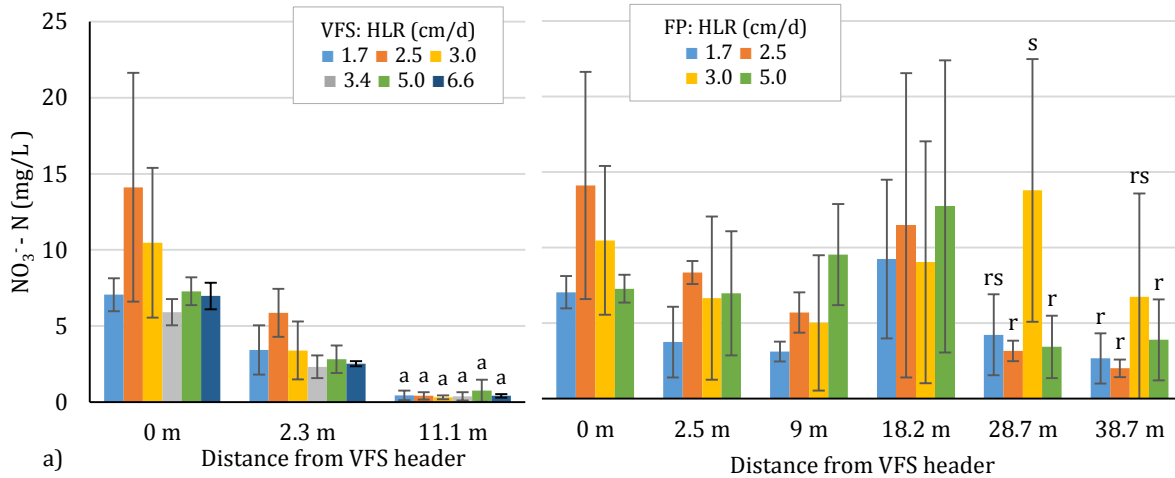


Figure 3.4. NO_3^- mean[‡] concentrations at different HLRs along: a) VFS; and b) FP

[‡] Means labeled with same letter are not significantly different by ANOVA test ($p > 0.05$)

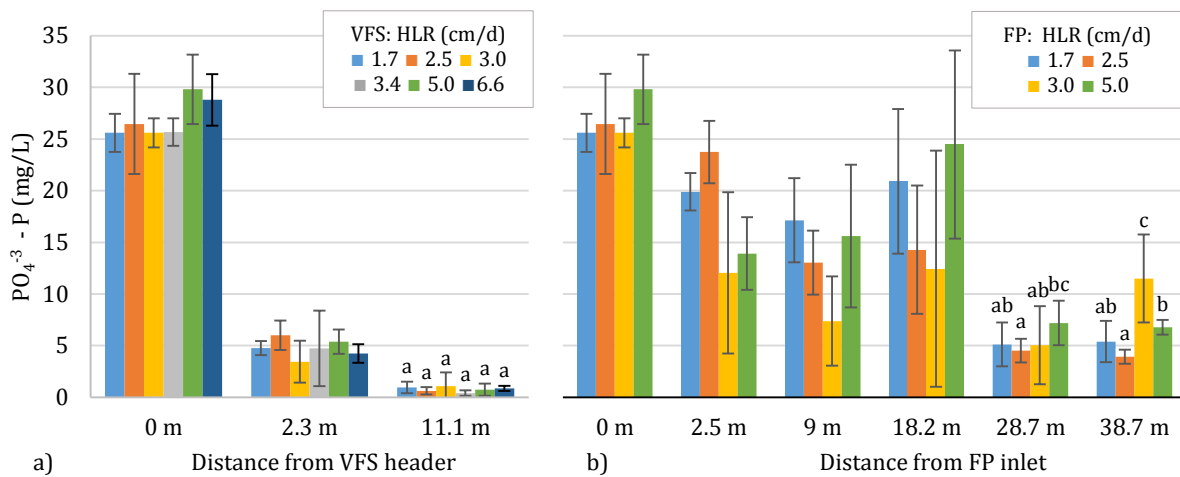


Figure 3.5. PO_4^{3-} mean[‡] concentrations at different HLRs along: a) VFS; and b) FP

[‡] Means labeled with same letter are not significantly different by ANOVA test ($p > 0.05$)

3.3.2 Organic Matter

Vegetated Filter

The HLR was varied by a factor of 3.9 from 1.7 to 6.6 cm/d for the VFS and by a factor of 2.9, from 1.7 to 5.0 cm/d for the FP. COD average inlet concentrations generally increased with loading rate from

428 ± 82 at 1.7 cm/d to 696 ± 152 mg/L at 6.0 cm/d.

Significant reduction in COD concentration was observed between Influent and VFS1 and between VFS1 and VFS2 under all loading conditions ($P < 0.05$) (Fig. 3.2a). The removal of COD in VFS1 (middle of the filter) and VFS2 (end of the filter) averaged $38 \pm 5\%$ and $93 \pm 2\%$, respectively, with no statistical difference in removal efficiency with loading rate (Table 3.2). This indicates that even a 4-fold increase in the HRL and a 60% increase in influent concentration did not affect the VFS efficiency in removing COD. In all dosing programs, the concentration of COD at the end of the filter was less than 100 mg/L with no significant differences observed with the exception of the 3.0 cm/d loading rate, which was lower than the others (Fig. 3.2a). Furthermore, average concentrations at VFS2 were not statistically different ($P > 0.05$) from background concentrations in Piez 1 at HLRs between 2.5 and 5.0 cm/d.

Table 3.2. COD concentration percent removal in both infiltration systems at different HLR.

HLR (cm/d)	Influent (mg/L)	VFS1‡	VFS2	FP1	FP2	FP3	FP4	FP5
		2.3 m	11.1 m	2.5 m	9.0 m	18.2 m	28.7 m	38.7 m
1.7	428 ± 82	29 _a	94 _b	41±13	50±12	45±24	80±8	79±12
2.5	540 ± 158	39 _a	94 _b	39±16	60±15	69±18	88±2	90±4
3.0	464 ± 96	44 _a	97 _b	38±40	76±7	58±10	75±21	73±14
3.4	670 ± 129	36 _a	94 _b	-	-	-	-	-
5.0	633 ± 102	41 _a	91 _b	54±21	55±22	41±26	80±4	83±4
6.6	696 ± 152	38 _a	90 _b	-	-	-	-	-
Average		38±5	93±2	43±7	60±11	53±13	81±5	81±7

‡ VFS1 and VFS2 means labeled with same letter are not significantly different by ANOVA test ($p > 0.05$)

Flow Path

In the FP, a general trend of reduced concentration with distance was observed from FP1 to FP5 across all loading conditions, with the exception of FP3 (Fig. 3.2b). A statistical difference was observed between the concentration of the influent and FP1 for all HLRs ($p < 0.05$); however, when

significance was tested between FP1 and other sampling locations, differences were found in two of four loading rates at FP2, in only one loading rate at FP3 and in all cases at FP4. No significant differences were observed between FP4 and FP5 for any of the loading rates, suggesting that maximum removal had been achieved by FP4 (29m).

No clear trend of loading rate on treatment efficiency was observed at any of the sampling locations, while average percentage removal increased with distance from $43 \pm 7\%$ at FP1 to a maximum of $81 \pm 5\%$ at FP4 (Table 3.2). However, a statistical difference was observed between the mean COD concentrations measured at HLRs greater than 3 cm/d in FP4 and FP5 with respect to lower HLRs (Fig. 3.2b). This suggests that the variation in dosing frequency and intensity has some effect on the efficiency of the FP removing organic matter from wastewater. The final COD concentration reached in the flow path was 70.2 mg/L on average at HLRs lower than 3 cm/d, and 124.1 mg/L at higher HLRs. These results suggest that maximum treatment is achieved at a distance of 29 m, regardless of loading rate, and that a treatment effect is observed at loading rates greater than 3 cm/d.

Comparison between VFS and FP

The two filters were compared at similar linear distances of 2.3 m (VSP1) and 2.5 m (FP1), with average removal rates observed of $38 \pm 5\%$ and $43 \pm 7\%$, respectively, and no significant differences in concentrations observed at each loading rate ($P < 0.05$). Given the width difference between filters, and the insignificant differences in removal rates, the results indicate that the FP was more efficient than the VFS on an area basis (normalized per width of the filter) in the first 2.5 meters, due to supersaturated soils around the inlet. On the other hand, when comparing the two filters on an equivalent area basis at VFS2 (94 m²) and FP5 (87 m²), the VFS showed better performance at all loading rates with $93 \pm 2\%$ COD removal compared to $81 \pm 7\%$ for the FP [$F(1, 8) = 13.9, p = 0.01$]. These results indicate that VFS provides better COD reduction than FP; possibly due to better effluent distribution in the VFS system.

Similar percentage removals in COD concentration compared to the results of this study were observed in two different experiments with VFSs at cattle feedlots. Andersen *et al.* (2009), reported COD removals ranging from 47 to 93% in six VFS systems receiving feedlot runoff with concentrations between 1,609 and 34,933 mg/L COD. The systems with the highest percent removals had drainage to filter area ratios of up to 10:1, a similar value to that used in this study, of 11:1. The authors state that beyond the changes in filter design, system management is one of the keys to maximizing system performance (Andersen *et al.*, 2009). Similarly, Dickey and Vanderholm (1981), reported COD removal rates of more than 80% in both channelized and overland flow systems. They found that, as observed in the present study, channelized systems showed lower final removal rates than VFS on a linear distance basis from the pipe inlet. The channelized flow system required a flow length over 5 times longer than the overland flow systems to achieve a similar concentration reduction (Dickey & Vanderholm, 1981).

3.3.3 Inorganic Nitrogen

Vegetated Filter

NH₄⁺ influent concentrations generally increased with loading rate and ranged from 5.8 ± 4.6 to 27.7 ± 12.8 mg N/L, while NO₃⁻ influent concentrations showed no trend and less variability with values ranging from 5.9 ± 0.9 to 14.1 ± 7.5 mg N/L. This resulted in total inorganic N (TIN) influent concentrations increasing with loading rate and ranging from 13.2 ± 4.1 to 34.7 ± 13.6 mg N/L.

Significant reductions in NH₄⁺ concentrations were observed between the influent and VSF1 at only two of six loading rates (3.4 and 6.0 cm/d), while significant reductions were observed between VFS1 and VFS2 under all loading rates (p<0.05) (Fig. 3.3a). A trend of increasing removal efficiency with loading rate was observed between the influent and VFS1, from 19% at 1.8 cm/d to 63% at 6.6 cm/d (Table 3.3), which coincides with higher influent concentrations. Almost complete removal of NH₄⁺

was observed at VFS2 under all loading rates with $96 \pm 4\%$ removal and final average concentrations of 0.23 mg/L .

NO_3^- concentrations were significantly reduced for all loading rates between the influent and VSF1 and between VSF1 and VFS2 (Fig. 3.4a). No statistical difference in NO_3^- removal efficiency was observed between loading rates at either VSF1 or VSF2; however, NO_3^- removal increased from $58 \pm 6\%$ at VSF1 to $94 \pm 3\%$ at VSF2 (Table 3.3). Average NO_3^- concentrations at VFS2 ($0.44 \pm 0.15 \text{ mg/L}$) reached background concentrations observed at Piez 1.

The two likely mechanisms for inorganic nitrogen removal are nitrification/denitrification in the 30 cm soil profile and plant uptake. The ammonia and nitrate data at VFS1 suggests limited nitrification in the beginning of the VFS with anoxic conditions supporting denitrification, possibly due to saturated soils, with removal rates of $50 \pm 13\%$ TIN observed (Table 3.3). However, almost complete removal of inorganic nitrogen was observed at VFS2 with average TIN reduction of $96 \pm 2\%$ (Table 3.3). The VFS was shown to be highly effective at reducing TIN under variable loading rates of 1.7 to 6.6 cm/d.

Table 3.3. NH_4^+ , NO_3^- and TIN concentration percent removal in both infiltration systems at different HLR.

HLR (cm/d)	Influent (mg/L)	VFS1‡	VFS2	FP1	FP2	FP3	FP4	FP5
		2.3 m	11.1 m	2.5 m	9.0 m	18.2 m	28.7 m	38.7 m
Ammonia								
1.7	6.1 ± 3.1	19 _a	93 _b	25±23	53±17	53±25	86±5	89±6
2.5	5.8 ± 4.6	29 _a	96 _b	48±34	63±27	67±45	94±4	85±16
3.0	9.0 ± 6.2	45 _c	90 _b	42±33	90±10	89±3	98±1	90±6
3.4	20.5 ± 2.7	63 _c	98 _b	-	-	-	-	-
5.0	15.2 ± 10.0	60 _c	99 _b	75±17	81±21	72±23	93±4	95±5
6.6	27.7 ± 12.8	63 _c	99 _b	-	-	-	-	-
Average		46±19	96±4	48±21	72±17	70±15	93±5	90±4
Nitrate								
1.7	7.1 ± 1.1	48 _e	93 _f	43±30	51±14	-34±63	33±47	58±58
2.5	14.1 ± 7.5	53 _e	97 _f	32±25	54±20	20±59	75±7	84±84
3.0	10.5 ± 4.9	64 _e	97 _f	47±23	35±50	-7±95	-69±99	18±18
3.4	5.9 ± 0.9	60 _e	94 _f	-	-	-	-	-
5.0	7.3 ± 0.9	62 _e	90 _f	5±60	-29±50	-78±138	56±22	48±48
6.6	7.0 ± 0.9	61 _e	94 _f	-	-	-	-	-
Average		58±6	94±3	32±19	28±39	-25±42	24±64	52±52
TIN †								
1.7	13.2 ± 4.1	32 _p	93 _q	37±15	52±14	33±34	54±31	71±18
2.5	20.0 ± 7.3	48 _p	97 _q	44±10	63±10	43±48	83±3	86±7
3.0	21.0 ± 9.5	56 _r	97 _q	48±29	67±22	49±40	35±32	61±32
3.4	26.4 ± 2.9	63 _r	98 _q					
5.0	22.5 ± 10.6	62 _r	96 _q	59±17	55±15	48±39	84±10	85±9
6.6	34.7 ± 13.6	63 _r	98 _q					
Average		50±13	96±2	47±9	59±7	43±7	64±24	76±12

† Calculated from mass balance: $NH_4^+ + NO_3^-$

‡ VFS1 and VFS2 means labeled with same letter are not significantly different by ANOVA test ($p > 0.05$)

Flow Path

No significant change in either NH_4^+ or NO_3^- concentrations were observed between the influent and FP1 ($p > 0.05$) in three out of four loading rates (Fig 3.4a-b). NH_4^+ removal increased with distance,

with concentrations declining from FP1 to FP4, with little to no differences observed between FP4 and FP5. However, NO_3^- concentrations showed little to no decline with distance. Given the conversion of NH_4^+ to NO_3^- , an increase in NO_3^- levels would not necessarily mean poor filter performance in the removal of this constituent, but perhaps a higher nitrification rate compared to the denitrification one in the FP.

NH_4^+ removal efficiency was higher at higher loading rates in the FP, possibly due to higher initial NH_4^+ concentrations, while NO_3^- removal efficiency was more variable, reflecting the variable flow and redox conditions in the soil profile of the FP as compared with the VFS (Table 3.3). No significant reduction in NH_4^+ or NO_3^- concentration between FP4 and FP5 was observed at any HLR suggesting that the FP reaches its maximum treatment capacity before a length of 30 m. An average TIN reduction of $76 \pm 12\%$ was achieved in FP5, with no evident impact of loading rate on final removal efficiency of the FP.

Comparison between VFS and FP

At VFS1, NH_4^+ concentration percent removals ranged from 19 to 63%, showing a strong relationship between the mass loading rate and the efficiency of the filter (Pearson correlation coefficient = 0.71). This tendency to have higher removal rates at higher influent concentrations was also observed in FP1 at a similar distance (Pearson correlation coefficient = 0.85). This indicates that both systems could be subject to the same mechanisms that inhibited or enhanced ammonia removal, such as nitrification or plant uptake. At higher ammonia concentrations, C/N loading ratio is lower and hence enhanced rates of nitrification could take place. As in the case of COD, although both systems had statistically the same percentage removal of NH_4^+ concentration in the first lysimeters, a significantly higher final removal in VFS was observed, making this system more effective on a distance and area basis. This could be linked to the superior performance of the VFS removing COD compared to the FP: at higher COD removal rates in VFS, the C/N ratio decreases and favours the conditions for

autotroph nitrifiers to outcompete heterotrophs (C-removers), inducing higher nitrification rates in the VFS than in the FP. In general, both systems achieved removals of more than 85% of NH_4^+ over the range of loading rates studied.

A negative relationship was found between NH_4^+ levels in the influent and NO_3^- removal in the FP (Pearson correlation coefficient at FP1 = -0.97, FP2 = -0.82, FP3 = -0.79, FP4 = -0.47; FP5 = -0.64). This suggests that during conditions of high NH_4^+ concentration, the nitrifying bacteria activity in the filter was stimulated. Such process yielded negative removals (i.e. net production) of NO_3^- in some of the FP lysimeters during some dosing programs given that at high NH_4^+ influent concentration the NO_3^- removal rate could not overcome nitrification. This is in contrast to the VFS, where consistent nitrate removal was observed at all loading rates.

Many previous studies also report increases in the initial NO_3^- concentration, and indicate that in spite of it, they did not generate an effluent with concentrations that would violate discharge standards (generally 10 mg/L-N) (Dillaha et al., 1988; Koelsch *et al.*, 2006; Rahman *et al.*, 2011). The same result was observed in the FP, where the final NO_3^- concentrations were always below 7 mg /L-N. Even in cases where negative NO_3^- removal was observed in lysimeters, the results were accompanied by a net TIN reduction in the system. There was no distinguishable pattern of NH_4^+ and NO_3^- concentration that could suggest there was one specific mechanism that governed the removal of TIN in the FP system, which had percent removal ranging from 61 to 86% (Table 3.3). Comparing the two filters on a linear distance basis (VF1 vs FP1) shows comparable TIN removal at 50 versus 47 percent removal, respectively; however, comparing on an equivalent area basis (VF2 vs FP5) shows superior performance from the VFS at 96 percent removal versus 76 percent removal in the FP.

It is necessary to mention that all measured concentrations were based on samples collected at 30 cm depth in the soil, where conditions conducive to both nitrification and denitrification reactions

could occur based on changing soil redox conditions with temporal changes in soil saturation. It is expected that denitrification would increase at greater depths, decreasing both NO_3^- and TIN concentrations.

The low concentrations of both NO_3^- and NH_4^+ in the closest piezometers to the filters confirm that these systems are efficient in mitigating the threat posed by open livestock operations runoff (Table 3.5). No differences in TIN concentrations were observed at Piez2, which is adjacent to the FP, compared to Piez1 (i.e. background), while concentrations at Piez3, which is directly below the VFS, were significantly higher than Piez1, with TIN concentrations declining from 6.57 to 1.64 mg/L as loading rates increased. These values indicate that there was a further reduction in N concentration after the first 30 cm in the soil profile, due in part to dilution with groundwater. Despite the change in the initial concentration of inorganic nitrogen infiltrated into the soil or the different HLRs used during the experiment over the course of 7 months, the quality of the groundwater directly under the VFS did not exceed drinking water guidelines and groundwater adjacent to FP showed no impact from FP loading.

3.3.4 Inorganic Phosphorus

Most of the phosphorus in the system is in the form of PO_4^{3-} , given the sedimentation of organic phosphorus during the pre-treatment of water in the Pond/Wetland system prior to Pond 2 (Mejia-Franco *et al.*, 2018). The influent was consistent across loading conditions with an average concentration of 27.0 ± 1.8 mg/L (as P).

Vegetated Filter

The VFS was effective at reducing PO_4^{3-} with an $82 \pm 3\%$ removal efficiency and an average concentration of 4.8 ± 0.9 mg/L at VFS1, and a $96 \pm 1\%$ removal efficiency and an average concentration of 0.78 ± 0.2 mg/L at VFS2 (Fig. 3.5a, Table 3.4). No statistical difference in PO_4^{3-}

removal efficiency was observed with loading rate at any point of the VFS. Moreover, no significant differences were observed between concentrations at VFS1 or VFS2 indicating that there was no effect of loading rate on concentration [F (5,32) = 1.52, p = 0.21 and F (5,31) = 0.9, p = 0.48, respectively]. Also, average concentrations at VFS2 were not statistically different from background concentrations in Piez 1 during the entire study period [F (6,36) = 0.87, p = 0.53], indicating that loading to the VFS did not impact groundwater quality with respect to phosphorus.

Table 3.4. PO_4^{3-} concentration percent removal in both infiltration systems at different HLRs.

HLR (cm/d)	Influent (mg/L)	VFS1 [‡]	VFS2	FP1	FP2	FP3	FP4	FP5
		2.3 m	11.1 m	2.5 m	9.0 m	18.2 m	28.7 m	38.7 m
1.7	25.6 ± 1.8	81 _a	96 _b	20±10	31±20	15±34	80±8	78±9
2.5	26.5 ± 4.8	77 _a	98 _b	9±21	50±16	47±30	83±4	85±3
3.0	25.6 ± 1.4	86 _a	96 _b	47±24	72±16	31±14	81±13	57±14
3.4	25.7 ± 1.3	81 _a	95 _b	-	-	-	-	-
5.0	29.8 ± 3.4	82 _a	98 _b	54±11	48±23	20±29	76±8	78±2
6.6	28.8 ± 2.5	85 _a	97 _b	-	-	-	-	-
Average		82±3	96±1	33±22	50±17	28±14	80±3	74±12

[‡] VFS1 and VFS2 means labeled with same letter are not significantly different by ANOVA test (p>0.05)

Flow Path

Similar to COD and NH_4^+ , a trend of declining concentration with distance was observed between Influent and FP4, with the exception of FP3, and a leveling off of treatment between FP4 and FP5, where no significant differences were observed between PO_4^{3-} concentrations at any loading rate (Fig. 3.5b). This is observed in Table 3.4, with treatment increasing from 33 ± 22% at FP1 to 80 ± 3% at FP4, with no further improvement observed at FP5. There does not appear to be a clear effect of loading rate on percentage removal, with variable response across all the lysimeters in FP. These observations are in line with the other constituents as well as with past studies. In a similar

experiment with a narrow vegetative filter (similar to FP) receiving poultry litter runoff, it was observed that the effectiveness of TKN and TP removal did not improve after 9.2 m, and NH_4^+ and PO_4^{3-} removal increased up to a filter length of 15.2 m (Chaubey, Edwards, Daniel, & Nichols, 1995).

Comparison between VFS and FP

The two systems were compared at similar linear distances (VSF1 vs FP1) as well as at similar loading areas (VSF2 vs FP5). In both cases the VFS outperformed the FP, with 82 ± 3 versus 33 ± 22 percent removal, and with 96 ± 1 versus 74 ± 12 percent removal ($P < 0.05$) (Table 3.4). However, FP had been receiving barnyard runoff for many years, which could explain the reduced treatment efficiency due to saturation of soil adsorption sites in the first 30 cm of soil along the FP.

Although in the present experiment VFS is evidently more effective than FP in removing phosphorus from water, the superiority in performance of overland flow filters over channelized filters cannot be generalized. According to Rahman et. al. (2011), removal efficiency within the first few meters of a filter decreases with time as nutrients build up and saturate the soil. The difference in operating time between VFS and FP may have generated significant differences in the performance of the systems in removing nutrients, as evidenced in the case of NO_3^- and PO_4^{3-} . Despite these differences, in comparison with previous studies both VFS and FP provided excellent removal of all constituents analyzed.

3.3.5 Groundwater

Groundwater samples were collected in three piezometers, with Piez 1 representing background concentrations, Piez 2 representing FP impacted groundwater and Piez 3 representing VFS impacted groundwater (Fig. 3.1); with comparisons of constituent concentrations presented in Table 3.5. Groundwater level in Piez 1, 2 and 3 varied between 1.0 and 1.3 m, 0.8 and 1.0 m, and 0.5 and 1.0 m, respectively. The groundwater level never approached the lysimeters in FP, while it may have

impacted the VFS lysimeters on two occasions during rain events.

COD concentrations in Piez 3 (VFS) were significantly higher than background in four of six loading conditions, while in only one of four conditions in Piez 2 (FP). Average COD concentrations were similar between Piez 1 and 2 at 22 ± 12 and 24 ± 10 mg/L, respectively, while increasing to 97 ± 26 mg/L in Piez 3. The higher concentrations observed in Piez 3 likely resulted from the piezometer's location directly below the VFS and its proximity to the VFS header. A pattern of increasing COD concentration was observed with loading rate, except at 3.4 cm/d, in all piezometers. However, the relatively small increases in COD concentration with loading rate observed at both Piez 1 and 2 suggest only a minor impact on groundwater quality within a 10-15 m radius of the VFS or FP.

A similar pattern to COD was observed with NH_4^+ , where concentrations were significantly higher in VFS than background in five of six loading conditions, while NH_4^+ concentrations were only significantly higher in FP than background in one of four loading conditions. Average NH_4^+ concentrations were similar between Piez 1 and 2 at 0.06 ± 0.06 and 0.11 ± 0.12 mg/L, respectively, while increasing to 2.02 ± 1.83 mg/L in Piez 3. Contrary to the COD observations, NH_4^+ concentrations declined with loading rate in Piez 3, with no trend observed for Piez 1 and 2. NO_3^- concentrations were generally low across the three piezometers; with significant differences in only two of six loading conditions between VFS and background and no differences between FP and background observed. Average NO_3^- concentrations were similar between the three piezometers at 0.53 ± 0.16 , 0.38 ± 0.20 , and 1.11 ± 0.49 mg/L in Piez 1, 2 and 3, respectively, with no effect of loading rate on NO_3^- concentration observed.

No significant differences in PO_4^{3-} concentrations were observed between VFS or FP piezometers and background at any loading rate, with average concentrations of 0.37 ± 0.15 , 0.47 ± 0.08 , and 0.72 ± 0.31 mg/L observed in Piez 1, 2 and 3, respectively. No trend of PO_4^{3-} concentration with loading was observed.

Table 3.5. Average concentrations of COD, NH₄⁺, NO₃⁻ and PO₄³⁻ and depth of groundwater in Piezometers 1, 2 and 3 during different HLRs. All values of concentration are in mg/L. Depth values are in m.

HLR (cm/d)		1.7	2.5	3	3.4	5	6.6	Avg
COD	Piez 1 (Background)	5 _a	14	16	37	32	27	22 ± 12
	Piez 2 (FP)	16 [†]	20	21	-	38	-	24 ± 10
	Piez 3 (VFS)	91 [†]	101 [†]	100 [†]	50	115 [†]	124	97 ± 26
NH ₄ ⁺ -N	Piez 1 (Background)	0.04	0.03	0.18	0.03	0.00	0.05	0.06 ± 0.06
	Piez 2 (FP)	0.04	0.05	0.06	-	0.28 _a [†]	-	0.11 ± 0.12
	Piez 3 (VFS)	4.23 [†]	4.47 _a [†]	1.28 [†]	0.55 _a [†]	0.87 [†]	0.69	2.02 ± 1.83
NO ₃ ⁻ -N	Piez 1 (Background)	0.47	0.46	0.44	0.84 _a	0.40	0.54	0.53 ± 0.16
	Piez 2 (FP)	0.39	0.40	0.35	-	0.36	-	0.38 ± 0.02
	Piez 3 (VFS)	0.84	2.10 [†]	0.99	0.81	0.96 [†]	0.95	1.11 ± 0.49
PO ₄ ³⁻ -P	Piez 1 (Background)	0.38	0.31	0.29	0.39	0.21	0.65	0.37 ± 0.15
	Piez 2 (FP)	0.54	0.54	0.37	-	0.43	-	0.47 ± 0.08
	Piez 3 (VFS)	0.67	0.86	0.47	0.66	0.39	1.25	0.72 ± 0.31
Depth	Piez 1 (Background)	1.30	1.13	1.06	1.07	1.00	1.05	1.10 ± 0.11
	Piez 2 (FP)	1.00	0.87	0.90	-	0.87	-	0.89 ± 0.06
	Piez 3 (VFS)	0.97	0.80 [†]	0.88 [†]	0.82 [†]	0.72 [†]	0.50 [†]	0.79 ± 0.17

^a Statistically different to concentrations in the same piezometer (same row) at other HLRs

[†] Statistically different to background value at the same HLR.

The recommended regional criteria for phosphorus concentration in groundwater concentrations differ among the sites because of natural variations in background concentrations, but for the US it ranges between 0.02 to 0.08 mg/L (USGS, 2012) and in the UK is 0.035 mg/L (Mellander *et al.*, 2013). Higher concentrations in groundwater can influence concentrations in streams, and thus, trigger noxious algal blooms or eutrophication processes. Groundwater quality in this study was an order of

magnitude higher than USGS ranges, which is to be expected from an agricultural soil (Domagalski & Johnson, 2011). With respect to NO_3^- , the global guideline value for nitrate concentration in drinking water is 10 mg N/L, where exposure to NO_3^- through contaminated wells may present health risks (WHO, 2011). In this study, NO_3^- concentrations were well below this threshold.

Márquez Molina *et al.* (2014) reported high NO_3^- and PO_4^{3-} concentrations in groundwater under two uncovered feedlot pens rural Argentina. In their analysis of the spatial distribution of nitrates in soils and groundwater at pens with different times of animal confinement, it was found that NO_3^- and PO_4^{3-} concentration averaged 52.4 mg/L-N and 1.25 mg/l-P, respectively, in pens with longer times of confinement (Márquez Molina, Urricariet, Sainato, Losinno, & Heredia, 2014). Likewise, a study in the US revealed that in 200 drinking water wells in areas of Delaware where land use was mostly from livestock production, the average concentration of NO_3^- was 21.8 mg/L - N (Ritter & Chirnside, 1987). These values show that the system of a constructed wetland followed by vegetative filters analyzed in this study represents a significant reduction in the impact that the confinement of livestock could generate on the quality of surrounding groundwater.

3.3.6 Mass Reduction

The mass removal of each constituent in each filter, concentration by flow rate normalized per surface area ($\text{g}/\text{m}^2/\text{d}$), compared to the initial mass applied during each loading condition is presented in Figure 3.6.

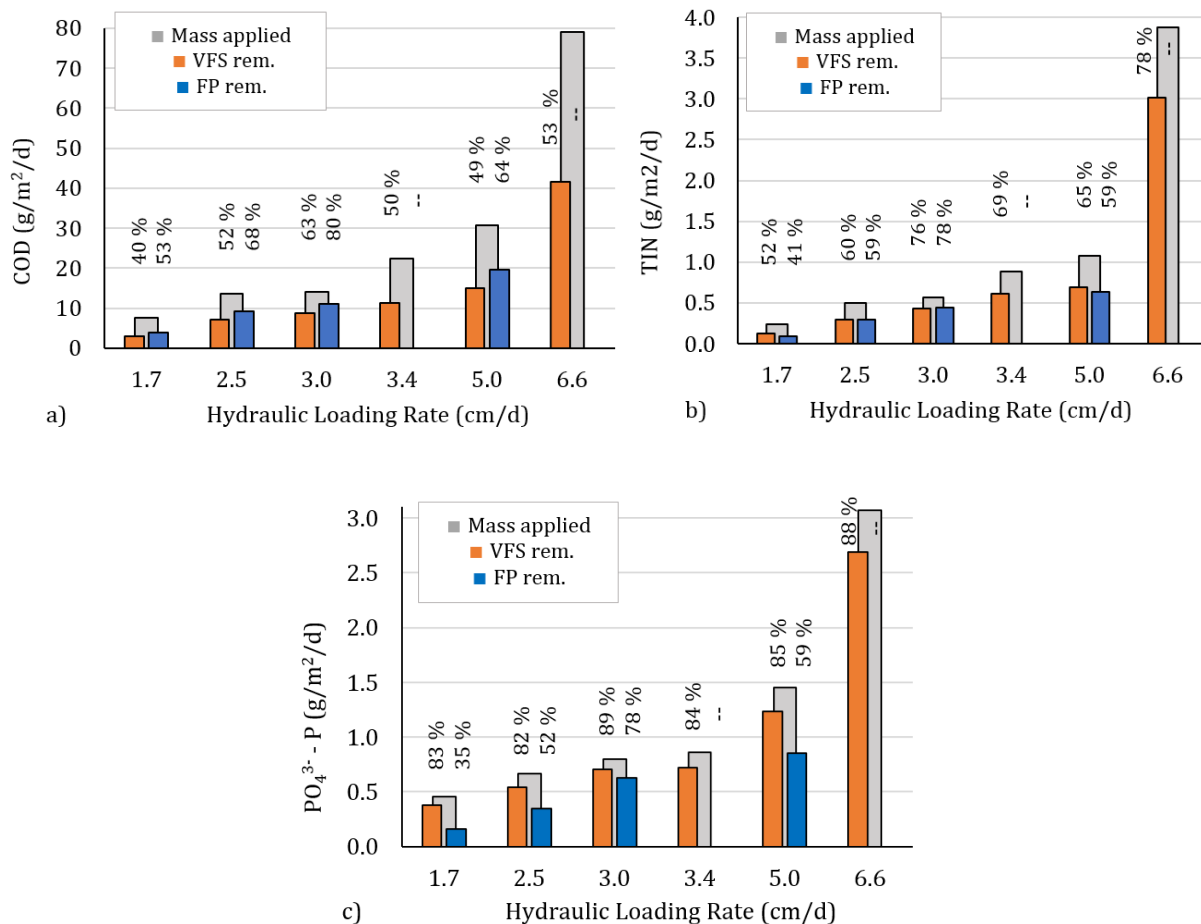


Figure 3.6. Mass reduction in FP and VFS at different HLRs for: a) COD; b) TIN[†]; and c) PO₄³⁻

† Calculated from mass balance: NH₄⁺+NO₃⁻

The FP outperformed the VFS in the mass removal of COD with 66±11% and 51±7% removal, respectively [F(1,8)=6.91, p=0.03]. No clear relationship was found between the filters mass removal performance and the HLR employed (Fig. 3.6a). The lower COD reduction observed in VFS as compared to FP relates to higher relative volumes of sample collected from the first line of lysimeters (VFS1), which were closer to the header, had higher concentrations, and thus contributed more to the accumulated mass calculated.

Average TIN mass percent removals were similar between VFS and FP at $69\pm 10\%$ and $65\pm 15\%$, respectively, despite differences between both systems removing NH_4^+ and NO_3^- separately. In the case of NH_4^+ , the removal was $61\pm 17\%$ in VFS and $78\pm 9\%$ in FP, while for NO_3^- the removal was $70\pm 6\%$ in VFS and $37\pm 30\%$ in FP. As with other constituents, no clear relationship was found between the filters mass removal performance and the HLR (Fig. 3.6b).

In contrast to COD removal, the VFS showed significantly higher performance in the removal of PO_4^{3-} than the FP, probably because it is a system with much less time in operation. An average mass removal of $85\pm 3\%$ of PO_4^{3-} was observed in VFS, compared to $56\pm 18\%$ in FP. No clear relationship was found between the filters mass removal performance and the HLR employed (Fig. 3.6c).

The percentage mass removals presented above coincide with the wide ranges reported in other studies, where different vegetation types, slopes and application rates were also evaluated (Chaubey *et al.*, 1995; Dickey & Vanderholm, 1981; Koelsch *et al.*, 2006). However, most of these studies report mass removal efficiencies based on vegetative filters superficial effluent measurements and not on infiltrated water, as in the present experiment. Under this criterion, both the VFS and the PF would have 100% removal of all constituents by being zero surface discharge systems.

3.3.7 Soil and Grass Analysis

Nutrient soil analysis from the VFS is presented in Table 3.6. No accumulation of NH_4^+ was observed in the soil, with no significant differences compared with background concentrations. This is expected as aerobic conditions in the surface soil layer supports nitrification, and it is expected that most of the inorganic nitrogen will be converted to NO_3^- . A significant accumulation of NO_3^- and PO_4^{3-} was observed in the VFS soils as compared to background concentrations. Variability in soil NO_3^- could result from variability in leaching, plant uptake or denitrification in presence of anoxic zones. PO_4^{3-} levels in the control soil section, used as a reference for nutrient background concentration,

showed that the farm soil was already optimal for crop growth (Beegle & Durst, 2002). The application of the nutrient-rich water generated 2 to 4 times higher phosphorus soil content in the VFS field.

Table 3.6. Nutrients concentration (ppm) in first 30 cm of soil at three different distances from VFS header at the end of the experiment.

Constituent	Background	2.3 m	6.6 m	11.1 m
NH ₄ - N	0.4 ± 0.1	0.2 ± 0.1	0.4 ± 0.2	0.6 ± 0.4
NO ₃ - N	1.4 ± 0.4	9.3 ± 1.6	3.9 ± 3.3	12.3 ± 14.5
Mehlich-2 PO ₄ ³⁻ - P	40.7 ± 12.8	113.1 ± 4.6	113.1 ± 62.3	159.6 ± 8.1
Olsen PO ₄ ³⁻ - P	12.6 ± 2.7	48.1 ± 3.0	45.3 ± 20.3	65.2 ± 71.2

The values in Table 3.6 correspond only to the first 30 cm of soil, but results of soil extraction show that the deepest portion (30 to 45 cm deep) still showed significant nutrients increases. Average concentrations at a 0.3-0.45 m depth along the filter were 0.3 ± 0.1 ppm for NH₄⁺, 3.3 ± 2.7 ppm for NO₃⁻ and 132.3 ± 17.9 ppm for PO₄³⁻. This indicates that the removal of N and P in the filter was not restricted in the first 30 cm, and that the above-mentioned filter effectiveness results are partial. It is expected that, just as after 30 cm depth in the soil there is still nutrient accumulation, the final concentration of the constituents analysed will be even lower at greater depths.

Bioavailable - PO₄³⁻ concentrations in the VFS and FP are presented in Tables 3.6 and 3.7, respectively, as Olsen PO₄³⁻ - P. It was expected that P concentrations obtained under the Mehlich-2 method were higher than those obtained using Olsen method since the former combines several extraction mechanisms that increase the yield given that is used for multiple nutrients extraction (Mehlich, 1984; Wuenscher et al., 2016). Bioavailable - PO₄³⁻ concentrations in the VFS are between two to three times lower than those observed in FP. This supports the hypothesis that the inferior PO₄³⁻ removal in FP compared to the VFS is due to saturation of soil adsorption sites in the first 30 cm of soil.

Table 3.7. Bioavailable- PO_4^{3-} (ppm) in first 30 cm of soil at five different distances from FP inlet at the end of the experiment.

Constituent	2.5 m	9.0 m	18.2 m	28.7 m	38.7 m
Olsen PO_4^{3-} - P	196.3 ± 27.1	145.8 ± 95.6	197.7 ± 2.52	96.9 ± 28.7	158.3 ± 18.5

The amount of TN and TP removed by plants for biomass growth was calculated from grass sample nutrient content (Table 3.8). It should be noted that the total mass of grass generated at the project site was tripled as there were two cuts for cattle feed and three growth of grass during the May-November growing season.

Table 3.8. Nutrients content in grass in the infiltration systems fields.

Constituent	% weight _{dry}	g/m ²
Total N	3.05 ± 0.44	13.3 ± 2.8
Total P	0.39 ± 0.03	1.7 ± 0.2

A nutrient mass balance on the VFS was performed with the results presented in Table 3.9. Plant uptake represented an important pathway for nitrogen removal in the experiment with 25% of applied N harvested as cattle feed, while only 2.4% was adsorbed to soil. With 27% N leached, the remaining 44.9% N can be attributed to denitrification along with experimental error.

In the case of phosphorus, the most important removal pathway was adsorption to the soil matrix of the filter, representing 27.0% of the total mass applied to the filter, with only 3.1% attributed to plant uptake. It is expected that over time the filter will lose its ability to remove this nutrient due to saturation of active absorption sites. With 13.9% leached, 56% of applied P is not accounted for in the mass balance, which could be attributed to experimental error in either the soil or leachate data.

Although more frequent harvesting of grass could improve the removal of soluble nutrients as well as increase filtration capacity (Smith, Melvin, Pope, Miller, & Cruse, 2000), these results and other studies suggest that plant uptake is not an important removal route for phosphorus (Fajardo *et al.*, 2001).

Table 3.9. Nutrient mass accumulation in first 30 cm of soil profile, in grass and in leachate collected in lysimeters, and percentage corresponding to TN and TP mass applied during the entire experiment.

	Total Nitrogen		Total Phosphorus	
	g	%	g	%
Applied	17207	100	18122	100
In grass	4345 ± 925	25.3	555 ± 66	3.1
In soil†	408 ± 418	2.4	4884 ± 1181	27.0
Leachate	2522	27.4	4707	13.9

† Mehlich 2

3.4 CONCLUSIONS

The effectiveness of two types of vegetative infiltration systems in treating effluent from a constructed wetland was compared. HLR and influent concentration did not appear to be significant factors affecting final concentration reductions in the VFS for any of the constituents analysed. An average TIN (calculated as $\text{NH}_4^+ + \text{NO}_3^-$), PO_4^{3-} and COD final concentration reduction of $96 \pm 2 \%$, $96 \pm 1 \%$, and $93 \pm 2 \%$, respectively, indicate that the VFS under variable loading conditions is highly effective at removing nutrients and organic matter from water.

However, the COD and PO_4^{3-} final concentration percent removal showed to be statistically different in FP at HLRs higher than 3 cm/d. The higher sensitivity of the FP performance in the removal of PO_4^{3-} to influent changes could be due to the fact that it had been receiving runoff for many years. Approximately 93%, 96% and 96% of incoming COD, TIN and PO_4^{3-} concentrations was removed within 11.1 m of the VFS, whereas it took more than three times that length to achieve 81, 76, 74 %

reductions in FP. It was found that there was no significant improvement in reducing contaminant concentrations in the FP after the fourth lysimeter (at 28.7 m).

Final percent removals in VFS (at VFS2) were not significantly different at any HLR for any constituent. Moreover, there was no observed impact of loading rate on percentage removal, with variable response across all the lysimeters in FP.

At comparable distances from the inlet (in the first 2.5 m), there was no statistical difference between the concentration percent removals of COD and NH_4^+ in both systems. In the case of PO_4^{3-} and NO_3^- , the VFS had statistically higher removals, and showed to be more efficient than the FP on a linear distance basis from the inlet.

The VFS was less efficient than FP on an area basis in the first portion of the filter for COD and NH_4^+ removal, likely due to supersaturation of the soil. On the other hand, if the two systems are compared considering the removal achieved in their last lysimeters, at comparable areas, percent removals of all constituents were always significantly higher in VFS than in FP.

The PO_4^{3-} and NO_3^- concentrations in groundwater did not change significantly over the entire period of the experiment, despite the changes in HLR. Also, the concentration of these two constituents were not statistically different to background levels over the entire study period in the proximities of FP (Piez 2). Groundwater COD and NH_4^+ values were affected by increases in HLR of the systems, and were significantly higher than background concentrations, mostly in Piez3.

The FP showed better performance in the mass removal of NH_4^+ and COD than the VFS by a slight difference. In contrast, VFS had a higher removal of PO_4^{3-} and NO_3^- due to the increased availability of active sites for phosphorus adsorption and oversaturated conditions in the soil that could enhance denitrification processes. Average TN mass percent removals were similar between VFS and FP in spite of differences in NH_4^+ and NO_3^- removal.

FP average mass removal of TN, PO_4^{3-} , and COD were $69\pm 10\%$, $85\pm 3\%$, and $51\pm 7\%$, respectively, while VFS mass removal of the same constituents were $65\pm 15\%$, $56\pm 18\%$, and $66\pm 11\%$, respectively. These percentages are based in the remaining volume and concentration of water collected after having passed through the first 30 cm of soil profile. Compared with other studies which only consider runoff quality, mass reductions could also be considered as 100% for all constituents since in this experiment soil infiltration of runoff prevented any effluent from leaving the vegetative area.

The observed nutrient content in grass suggests that plant uptake can be an important route for nitrogen removal in vegetative filters. A quarter of all nitrogen mass applied to the VFS was used by vegetation for biomass growth. In contrast, this was not a major route of phosphorus removal. Adsorption to soil represented a better removal mechanism for phosphorus, with about 27% of the total phosphorus mass applied to the system being removed through this mechanism. Soil and sediment microbiological tests from all subsystems involved in the treatment of this runoff would help to better understand nutrient and organic matter removal mechanisms.

The VFS showed an outstanding performance while occupying a much smaller area than recommended by different design procedures currently in use. If the design criteria suggested by Koelsch *et al.* (2006) was used, a VFS area of more than 2000 m^2 would be necessary to achieve similar removals than those observed in the present study, as opposed to the 93 m^2 that was employed. Similarly, if the Vegetated Filter Strip System Design Manual, recommended by OMAFRA, was used to size a filter capable to infiltrate the average HLR used in this study, the area of the VFS would have to be 11 times larger. Even for the program with the smallest HLR used, the VFS area should be 5 times larger according to this guide. This indicates that the role of a pre-treatment system, such as the Pond/Wetland system prior to the VFS and FP, can significantly reduce the area needed for infiltration of runoff from open livestock activities.

The present study indicates that the currently recommended design procedures can lead to oversizing of vegetative filters, and that it is possible to achieve significant mass and concentration removals if the systems are properly operated and maintained.

The absence of significant differences between the efficiencies reached in the VFS during the different dosing programs tested in this study does not allow to recommend the use of any particular HLR that optimizes the results. However, data from VFS suggest that a HLR of 6 cm/d could attain reasonable pollutants removal under the weather and influent conditions reported in this study. In terms of FP sizing, the data suggest that a filter length of 30 m is appropriate for HLRs up to 3 cm/d.

In clay soils under similar climate conditions, steeper VFS slopes and longer filter lengths could be tested to avoid supersaturation of the soil in the vicinity of the head pipe and improve the flow regime in the filter.

The study of the effect of the dosing frequency on the removal rates of the constituents was beyond the scope of this study. However, it is presumed that changes in this variable can cause variations in the nitrification-denitrification cycles in VFs and therefore merits a more profound analysis in future research.

References

- Andersen, D., Burns, R., Moody, L., Khanijo, I., Helmers, M., Pederson, C., & Lawrence, J. (2009). Performance of six vegetative treatment systems for controlling runoff from open beef feedlots in Iowa. American Society of Agricultural and Biological Engineers Annual International Meeting 2009, ASABE 2009, 9.
- APHA. (2005). Standard Methods for the Examination of Water and Wastewater (21th ed.). Washington DC.: American Public Health Association.
- Beegle, D. B., & Durst, P. T. (2002). Managing Phosphorus for Crop Production. Pennsylvania State University Extension.
- Bremner, J., & Keeney, D. (1966). Determination and Isotope-Ratio Analysis of Different Forms of Nitrogen in Soils : 3. Exchangeable Ammonium, Nitrate, and Nitrite by Extraction-Distillation Methods 1. Soil Science Society of America, 30(5), 577–582.

- Chaubey, I., Edwards, D. R., Daniel, T. C., & Nichols, D. J. (1995). Losses of Surface-Applied Poultry Litter Constituents. *American Society of Agricultural Engineers*, 38(6), 1687–1692.
- Dickey, E. C., & Vanderholm, D. H. (1980). Performance and Design of Vegetative Filters for Feedlot Runoff Treatment. *Livestock Waste: A Renewable Resource*, 257–260.
- Dickey, E. C., & Vanderholm, D. H. (1981). Vegetative filter treatment of livestock feedlot runoff. *Journal of Environment Quality*, 10(3).
<https://doi.org/10.2134/jeq1981.00472425001000030005x>
- Dillaha, T., Sherrard, J., Lee, D., Mostaghimi, V., & Shanholtz, V. (1988). Evaluation of Vegetative Filter Strips as a Best Management Practice for Feed Lots. *Water Pollution Control Federation*, 60(7), 1231–1238. <https://doi.org/10.2307/25043629>
- Domagalski, J. L., & Johnson, H. M. (2011). Subsurface transport of orthophosphate in five agricultural watersheds, USA. *Journal of Hydrology*, 409(1–2), 157–171.
<https://doi.org/10.1016/j.jhydrol.2011.08.014>
- Fajardo, J. J., Bauder, J. W., & Cash, S. D. (2001). Managing nitrate and bacteria in runoff from livestock confinement areas with vegetative filter strips. *Journal of Soil and Water Conservation*, 56(3), 185–191.
- Government of Canada. (2018). Canadian Climate Normals 1981-2010 Station Data and Daily Data Report for 2018. Retrieved May 5, 2018, from
http://climate.weather.gc.ca/climate_normals/results_1981_2010_e.html?searchType=stnName&txtStationName=kemptville&searchMethod=contains&txtCentralLatMin=0&txtCentralLatSec=0&txtCentralLongMin=0&txtCentralLongSec=0&stnID=4291&dispBack=1
- Koelsch, R. K., Lorimer, J. C., & Mankin, K. R. (2006). Vegetative Treatment Systems for Management of Open Lot Runoff: Review of Literature. *Biological Systems Engineering: Papers and Publications*, 22 (1), 141-153
- Lim, T. T., Edwards, D. R., Workman, S. R., Larson, B. T., & Dunn, L. (1998). Vegetated filter strip removal of cattle manure constituents in runoff. *Transactions of ASAE*, 41(5), 1375–1381.
<https://doi.org/10.13031/2013.17311>
- Márquez Molina, J. J., Urricariet, A. S., Sainato, C. M., Losinno, B. N., & Heredia, O. S. (2014). Effects of feedlot manure on soil and groundwater assessed with electrical resistivity tomography. *Environmental Earth Sciences*, 73(4), 1459–1472. <https://doi.org/10.1007/s12665-014-3496-y>
- Mehlich, A. (1984). Mehlich 3 soil test extractant: A modification of Mehlich 2 extractant. *Communications in Soil Science and Plant Analysis*, 15(12), 1409–1416.
<https://doi.org/10.1080/00103628409367568>
- Mejia-Franco, J., Kinsley, C., & Delatolla, R. (2018). Performance and Kinetics of a Pond/Wetland System Treating Beef Manure Pile and Exercise Yard Runoff in Eastern Ontario. Unpublished.
- Mellander, P. E., Jordan, P., Melland, A. R., Murphy, P. N. C., Wall, D. P., Mehan, S., ... Shortle, G. (2013). Quantification of phosphorus transport from a karstic agricultural watershed to emerging spring water. *Environmental Science and Technology*, 47(12), 6111–6119.
<https://doi.org/10.1021/es304909y>
- Olsen, S., Cole, C. V., Watanabe, F. S. and Dean, L. A. (1954) Estimation of available phosphorus in soils by extraction with sodium bicarbonate, Circular No.939. Washington DC., USA.

- OMAFRA. (2006). *Vegetated Filter Strip System Design Manual*. Toronto: Ministry of Agriculture, Food and Rural Affairs of Canada.
- Panu, U. S., & Nguyen, T. (1994). Estimation of Mean Areal Evaporation in Northwestern Ontario. *Canadian Water Resources Journal*, 19(1), 69–82. <https://doi.org/10.4296/cwrj1901069>
- Rahman, S., Rahman, A., & Wiederholt, R. (2011). *Vegetative Reduce Feedlot Runoff Pollutants*. North Dakota State University Extension, (NM1591), 1–4.
- Ritter, W. F., & Chirnside, A. E. M. (1987). Influence of Agricultural Practices on Nitrates in the Water Table Aquifer *, 19, 165–178.
- Rondon, F. F. (1972). *Estimation of irrigation requirement for Venezuela*. Utah State University.
- Smith, M., Melvin, S., Pope, R., Miller, G., & Cruse, R. (2000). *Vegetative Filter Strips for Improved Surface Water Quality What are vegetative filter strips?* Iowa State University Extension.
- USGS. (2012). *Phosphorus and Groundwater: Establishing Links Between Agricultural Use and Transport to Streams*.
- WHO. (2011). *Nitrate and nitrite in drinking-water*. Geneva.
- Wuenschel, R., Unterfrauner, H., Peticzka, R. and Zehetner, F. (2016) 'A comparison of 14 soil phosphorus extraction methods applied to 50 agricultural soils from Central Europe', *Plant, Soil and Environment*, 61(No. 2), pp. 86–96. doi: 10.17221/932/2014-PSE

CHAPTER IV

Synthesis

Precipitation reaching uncovered feedlots can transport contaminants into the environment. The most visible consequence of unconfined feedlot runoff is the contamination of water bodies. Currently, there are laws that regulate the storage and use of manure in different countries, and propose the implementation of management practices that help reduce the impact of these wastes on natural resources. Constructed wetlands (CW) and Vegetative Filter (VF) systems are under study as Best Management Practices (BMP) to treat manure storage runoff.

In this study, data from an experimental pond-CW-VF system treating beef manure pile and exercise yard runoff in eastern Ontario, Canada, is presented and discussed. All the treatment stages were monitored simultaneously over a seven month operating season. For analysis purposes, the system was divided in two main stages: 1) pond-CW treatment stage, analyzed in Chapter II; and 2) VFs infiltration stage, analyzed in Chapter III.

In the first stage, manure pile and exercise yard runoff was collected in a pump chamber (PC) and then pumped into a primary settling pond (P1). The effluent from P1 flowed by gravity into a surface flow CW (W) and then was stored in a secondary settling pond (P2). Water samples were collected from each point in several sampling visits, and variables such as air and water temperature, system inflow (measured in PC), pH and water dissolved oxygen were measured *in-situ*. Biological Oxygen Demand (BOD₅), Chemical Oxygen Demand (COD), Total Phosphorus (TP), Orthophosphate (PO₄³⁻ as P), Total Nitrogen (TN), Ammonia (NH₄⁺ as N), Nitrate (NO₃⁻ as N), and Total Suspended Solids (TSS) concentration was measured to: 1) characterize manure pile and exercise runoff, 2) determine removal efficiencies in P1 and W, and 3) determine wetland kinetic removal rates described by the P-C*-k model. The parameters for the P-C*-k model, used to characterized the wetland treatment,

were optimized. All rate constants quantified in this study were within the range reported in the limited literature for livestock wastewater treatment wetlands. Also, the wetland removal rates appeared to correlate with temperature for all the constituents analyzed.

The intermittent flow caused by isolated rainfall events made this wetland event-driven in nature. Thus, a change in the wetland's kinetic constants was observed due to variations in the input concentration. This effect of isolated rain events on the performance kinetics of the wetland showed to be contingent on the intensity of the event, former humidity conditions of the feedlot and the hydraulic capacity of the wetland.

The pond-CW system was found to be effective at reducing in BOD₅ (76%), COD (69%), TP (48%), PO₄³⁻ (45%), TN (64%), NH₄⁺ (70%), NO₃⁻ (52%), and TSS (33%).

Further polishing of the wetland effluent, stored in P2, was accomplished by an infiltration stage. In this second stage, water from P2 was dosed to two different types of VF, a vegetative filter strip (VFS) and a flow path (FP), at six different hydraulic loading rates to compare their performance removing organic matter (COD) and nutrients (NO₃⁻, NH₄⁺ and PO₄³⁻) from water. The VFS was dosed by a 6 m-wide header and monitored through two parallel rows of zero-tension lysimeters (which collected water at 30 cm from the ground surface) installed at two different distances from the header pipe. The FP was dosed at a single inlet point and monitored through 5 lysimeters installed along its length. The side-by-side comparison of these overland flow and channelized flow systems was carried out considering two criteria: linear distance to the inlet and infiltration surface area. At equivalent distances or equivalent areas, the average efficiencies achieved during the entire study period were compared. Also, the efficiency of VFS and PF was compared independently under the different dosing programs tested.

It was found that HLR and influent concentration did not appear to be significant factors affecting final concentration reductions in the VFS for any of the constituents analysed. However, the COD and

PO_4^{3-} final concentration percent removal were shown to be statistically different in FP at HLRs higher than 3 cm/d. Average final removal rates in the VFS were always greater than 90% for COD, 95% for NH_4^+ , 90% for NO_3^- , and 94% for PO_4^{3-} , whereas in the FP the minimum average removal rates were 73% for COD, 87% for NH_4^+ , 36% for NO_3^- , and 55% for PO_4^{3-} . It was found that there was no significant improvement in treatment of contaminants in the FP after a length of 28.7 m. Mass removal, groundwater constituent's concentration, grass nutrient content and soil nutrients concentration were also analysed in this study.

This experiment suggests that even relatively small VFSs or short FPs can markedly improve quality of runoff from livestock operations, and that it is possible to achieve significant mass and concentration removals if they are properly operated and maintained in conjunction with a pre-treatment system.

The results of this study indicate that the pond-CF-VF system can be a cost-effective treatment option for beef producers. The remaining challenge of this study is to understand how to mitigate the masking effects of precipitation events from CW rate constant calculations in order to best model their treatment efficiency.

CHAPTER V

Conclusion

The storage of manure gives rise to generation of diffuse sources of contamination. CW and VF are considered as good practices to treat livestock operations runoff; however, CW design requirements have been based mainly on municipal systems and VF design guideline are empirical in nature and vary widely between jurisdictions. Seven-month performance data from an intensively monitored Pond-CW-VF system receiving livestock runoff permitted the determination of kinetic removal rates of a FWS-CW, describe the impact of precipitation events and temperature on CW performance, and compare the performance of two different types of VFs removing contaminants from water.

The first stage of the system, comprising a pond and constructed wetland, showed a high potential as a viable management practice for cattle manure pile and exercise yard runoff treatment in a temperate climate in Ontario, Canada. The major findings include:

1. High influent concentrations of NO_3^- -N in the runoff (27.0 mg/L on average, with peaks of up to 72 mg/L) and a high average COD/BOD₅ ratio of 7.6 suggest that the manure pile acts as a compost pile and plays an important role in the strength and stability of the runoff constituents entering the system.
2. The majority of TN removal occurred in the pond given the anaerobic conditions that allowed denitrification while in the wetland a greater conversion of NH_4 to NO_3^- was observed due to greater DO availability.
3. The pond provided almost all of the removal of dissolved phosphorus, which in conjunction with the close correlation ($R^2 > 0.97$) with removal of the combined dissolved NH_4^+ and soluble N_{org} suggests that plant uptake in the pond, likely through the observed prevalence of Duckweed, played a key role in nutrient removal.

4. The Pond-CW system performed well in the removal of organic matter, NH_4^+ and TN from the wastewater, reaching average removals of 76% BOD₅, 69% COD, 70% NH_4^+ and 64% TN. In general, the pond provided the most important portion of the treatment in the overall system.
5. From the optimization of the parameters of the model P-C*-k and θ , it was observed that the number of tanks in series (P) and the background concentration (C^*) are not very sensitive in the model, and that most of the kinetic rates of removal have a strong dependence on temperature. The k_{20} values normalized to 20°C were 8.8 m/y for BOD₅, 2.4 m/y for COD, 0.8 m/y for PO_4^{3-} , 0.5 m/y for TP, 4.1 m/y for TN, and 7.3 m/y for NH_4^+ . The computed Arrhenius temperature correction coefficient applied to the k values ranged from 1.032 to 1.061.
6. It appears that k increases with low intensity storm events and decreases with medium to high intensity storm events as precipitation affect both the hydraulics and the input concentrations of the system. The effect of isolated rain events on the performance of the wetland showed to be contingent on the intensity of the event, former humidity conditions of the feedlot and the hydraulic capacity of the wetland.
7. The model was effective (and adequate) at representing average system performance at a seasonal to yearly time scale, with all of the optimized k constants falling within the ranges reported in different studies based on data from similar applications to that of the present study. High variations were observed when the model was applied at a daily to monthly time scale due to the event-driven nature of the system

Soil infiltration as a tertiary treatment method was studied by means of a side-by-side comparison of two types of VF. This second stage of the system, composed by a VFS and a FP operating in parallel, proved that it is possible to achieve significant mass and concentration removals if the systems are preceded by a well designed settling basin, and are properly operated and maintained. Specific findings include:

1. HLR and influent concentration did not appear to be significant factors affecting final concentration reductions in the VFS for any of the constituents analysed. However, the COD and PO_4^{3-} final concentration percent removal showed to be statistically different in FP at HLRs higher than 3 cm/d. The higher sensitivity of the FP performance in the removal of PO_4^{3-} to influent changes could be due to the fact that the FP had been receiving runoff for many years.
2. It was found that there was no significant improvement in reducing contaminants concentration in the FP after the fourth lysimeter (at 28.7 m).
3. An average TIN (calculated as $\text{NH}_4^+ + \text{NO}_3^-$), PO_4^{3-} and COD final concentration reduction of $96\pm 2\%$, $96\pm 1\%$, and $93\pm 2\%$, was observed, respectively, indicate that the VFS under variable loading conditions is highly effective at removing nutrients and organic matter from runoff water. It took more than three times that length to achieve similar (but never better) reductions in FP.
4. The VFS was shown to be more efficient at removing PO_4^{3-} and NO_3^- than the FP on a linear distance basis from the inlet (in the first 2.5 m). At comparable areas, final percent removals of all constituents were always significantly higher in VFS than in FP.
5. The concentration of PO_4^{3-} and NO_3^- were not statistically different to background levels over the entire study period in the proximities of FP (Piez 2). Groundwater COD and NH_4^+ values were affected by increases in HLR of the systems, and were significantly higher than background concentrations, mostly in Piez3 (close to VFS header).
6. FP average mass removal of TN, PO_4^{3-} , and COD were shown to attain $69\pm 10\%$, $85\pm 3\%$, and $51\pm 7\%$, respectively. VFS mass removal of the same constituents was $65\pm 15\%$, $56\pm 18\%$, and $66\pm 11\%$, respectively. These removal percentages are based in the remaining volume and concentration of water collected after having passed through the first 30 cm of soil profile.

7. A quarter of all nitrogen mass applied to the VFS was used by vegetation for biomass growth. Adsorption to soil represented a better removal mechanism for phosphorus, with approximately 27% of the total phosphorus mass applied to the system being removed through this mechanism.
8. Finally, the VFS showed an outstanding performance even while occupying a much smaller area than that recommended by different design procedures currently in use.

The present study indicates that the currently recommended design procedures can lead to oversizing of vegetative filters, and that it is possible to achieve significant mass and concentration removals if the systems are properly operated and maintained.

Future research avenues will include the implementation of more complex multi-parameter models to better represent event-based systems and to better represent the linkages between dissolved oxygen and nitrogen speciation in CWs. The prospect of nutrients plant uptake, specifically biological phosphorus removal, in captured runoff warrants further examination. The study of the effect of the dosing frequency on pollutant removal in VFs merits a more profound analysis. Finally, soil and sediment microbiological tests from all subsystems involved in the treatment of this runoff would help to better understand nutrient and organic matter removal mechanisms governing each stage.