An assessment of the effectiveness of the Crossways Farm Village constructed wetland in the treatment of domestic wastewater

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Abstract

A mass balance study was conducted comparing inflowing and outflowing solute loads in order to calculate the treatment efficiency of a free water surface (FWS) constructed wetland used to treat domestic wastewater following primary treatment in an anaerobic reactor and oxidation in a rotating biological contractor. Water samples were taken at six locations down the length of the treatment system and analysed for nitrate nitrogen (NO_3-N), nitrite nitrogen (NO_2-N), ammonia nitrogen (NH_4-N) and phosphorus as phosphate (PO_4-P). Flow was determined using two V notch weirs combined with pressure transducers based on an empirically derived stage-discharge relationship. The concentration of each solute ($g.m^{-3}$) multiplied by flow ($m^3.day^{-1}$) provides a measurement of the mass of each solute entering and leaving the treatment wetland such that the difference (inflow – outflow) indicates the nett storage in, or loss from, the wetland.

In order to determine the water balance, apart from measuring surface inflows and outflows, rainfall was measured using an onsite rain gauge. Evapotranspirational losses were determined using the Penman-Monteith equation based on weather data collected at an onsite weather station. Other than water that entered the wetland via the primary water treatment works, surface inflows could be ignored as the wetland was sealed with a plastic liner, which also prevented groundwater inflow and outflow. Wetland outputs via surface outflow and evapotranspiration were then subtracted from wetland inputs to determine the water balance over the study period. Approximately 10.5% of water inputs into the hydrological mass balance calculation was not accounted for, which is considered to be accounted for by inaccuracy associated with the estimation of evapotranspiration and possibly by differences in water levels in the wetland at the start and end of the experiment.

Total input, output and storage of NO_3-N, NO_2-N, NH_4-N and PO_4-P was calculated from April 2016 to September 2016 to give the treatment efficiency of the FWS wetland system. Results showed a 91.5% reduction in NO_3-N, 76.6% reduction in NH_4-N, and 88.8% reduction in PO_4-P between the inflow and outflow.

Wetland sediment and vegetation (*Typha capensis*) samples were also analysed for nitrogen and phosphorus content to give an estimate of nutrient stocks/storage accumulated in plant tissues and sediments over the lifespan of the wetland. Standing stock calculations showed that a total of 450.1kg of nitrogen is stored in the wetlands, of which 69.3kg is stored in wetland sediments. Wetland phosphorus retention was found to be significantly lower with a total of 57.1kg of phosphorus, of which 77.4% was stored in sediments, indicating that wetland sediments comprise the largest store and therefore removal pathway of nutrients in the Crossways Farm Village FWS wetland.

Keywords: Mass balance; water balance; nutrients; domestic wastewater; FWS wetland.

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Chapter 1: Introduction

1.1 Research context

In the past 50 years, treatment of domestic and municipal wastewater *in situ* has become increasingly necessary and many countries have faced pressures when releasing contaminated wastewater into natural water bodies (Loupasaki and Diamandopoulos, 2012). What has emerged as a potential solution to combat this issue is the use of constructed wetlands that are specifically designed to treat various point source wastewater discharges. These constructed wetlands for wastewater treatment, also known as treatment wetlands, can be defined as engineered systems that have been designed and constructed to utilise natural processes in order to remove pollutants from contaminated water, within an artificial environment that can be manipulated (Faulwetter *et al*., 2009; Vymazal, 2011).

Treatment wetland systems have developed rapidly over the last two decades and have been constructed worldwide as an alternative to the conventional 'centralised' municipal treatment systems, which are often unavailable in remote locations where wastewater is generated (Machado *et al*., 2016). In developing countries such as South Africa, rapid population expansion has resulted in significant domestic waste production by decentralised and rural communities that cumulatively contribute significant amounts of nutrients and organic matter to natural streams, rivers and lakes (Mara, 2004). Using constructed wetlands for the treatment of agricultural and domestic effluent has the potential to provide more water for reuse and can improve the sanitation and water quality of freshwater systems used by small rural communities (Newman *et al*., 2000). The potential for achieving improved water quality through a cost effective and environmentally friendly solution that accords with the economic, social and environmental principles of sustainability, has led to growing interest and investment in constructed wetlands for treating wastewater and agricultural effluent, such that it can be safely released to the environment or be recycled (Mara, 2004).

1.2 Introduction to wetlands

Wetlands are characterised by the prolonged saturation of soils that inhibits decomposition and results in the formation of an anaerobic environment with a high soil organic matter content (Ellery *et al.*, 2009). Due to the available water and nutrients, plant productivity in these systems is high and this creates an environment with spatially separated aerobic and anaerobic zones; conditions that are conducive to a wide range of biogeochemical transformations (Vymazal, 2007).

Wetlands therefore offer a wide range of direct and indirect ecosystem services and provide a natural mechanism that regulates flow, sediment transport and nutrient availability, and the transfer of these within and between ecosystems (Ellery *et al*., 2009). This ability to remove and transform nutrients is the driving force behind the use of constructed wetlands in the treatment of domestic wastewater (Kadlec and Wallace, 2009). A plethora of literature is available that demonstrates how effective constructed wetlands are at treating domestic and agricultural wastewater, with particular reference to nitrogen and phosphorus removal (Chung *et al*, 2007; Kadlec and Knight, 1996, Kadlec and Wallace, 2009; Lee *et al*., 2014; Tanner *et al*., 1995). Nitrogen and phosphorus are usually limiting nutrients in nature but occur at high concentrations in domestic wastewater. Excessive releases of these nutrients into natural water bodies can therefore result in eutrophication, significantly reducing the quality of available freshwater (Smith and Schindler, 2009). In addition to their ability to remove nutrients produced in anthropogenic wastes, constructed wetlands have several additional benefits and can be established on site where the wastewater is produced, with far lower energy requirements, infrastructure and set up costs. Treatment wetlands are also easily maintained by relatively untrained personnel in comparison to conventional treatment systems (Solano *et al*., 2004).

Except for a few closed system designs, constructed wetlands are not exempt from environmental and climatic factors such as temperature and rainfall that shape natural wetlands (Vymazal, 2013). These factors also affect and alter the physical structure (vegetation type and density), biogeochemical activity and microbial composition within a wetland, and indirectly determine the effectiveness of the design in treating a particular effluent (Jadhav and Buchberger, 1995). There are a wide range of designs and types of constructed wetlands that have been used to treat various types of effluent, yet the performance of current designs has varied greatly with time and locality, and is not well understood (Faulwetter *et al*., 2009; Vymazal, 2011). This variability in performance is largely due to an inadequate quantitative understanding of the relationship between environmental factors that dictate wetland hydraulics and the water balance, ecosystem structure and function and the loading rate, which ultimately determines the effectiveness of a given treatment wetland (Kadlec and Wallace, 2009). As a result, each constructed wetland system is unique and there is no ‘green box’ approach to analysing a wetland’s treatment performance. Constructed wetlands are complex systems that are nonlinear in nature and therefore they display highly variable removal efficiencies in both space and time (Kadlec and Reddy, 2001). Understanding the hydraulic connectivity between wetlands and the surrounding environment is therefore a requirement for the effective management of these systems (Breen, 1990).

Conceptually, constructed wetlands can be divided into a range of interactive components, each representing a store of nutrients (Breen, 1990). Each component represents a pathway for the physical removal, temporary storage or transfer of nutrients within components. In constructed wetlands, these components comprise incorporation of nutrients into biomass, soils and sediments, or direct removal from water into the atmosphere, when nutrients are transformed into a phase where they can be released into the environment (Breen, 1990). Analysing the efficiency of these wetland storage compartments therefore depends on influent characteristics and loading rates, the storage capacity of components, transfer rates between components, system design and hydrology, and the critical environmental factors such as light and temperature, which affect the physical storage processes and capacity of each component (Kadlec and Wallace, 2009).

1.3 Nutrients in wetlands

Nutrient transformation is recognised as an important ecosystem service provided by wetlands (Mitsch and Gosselink, 2007; Acreman and Fisher, 2004) as they are able to substantially improve the quality of wastewater. Wetland plants play a significant role in the wastewater treatment process, reducing flow and encouraging sedimentation (Vymazal, 2007). Plants have the ability to take up nutrients, which may later be deposited in the sediments as organic matter (Mitsch and Gosselink, 2007). Plant litter also provides a food source and substrate for invertebrate and microbial activity. These processes are key components that enhance nitrogen and phosphorus cycling and retention in treatment wetlands (Yao *et al.,* 2011).

As wetland treatment technologies continue to evolve, substantial effort is being applied to better understand short- and long-term treatment variability in these systems (Knight *et al.*, 2000). Since treatment variability is influenced by wetland design, understanding factors that influence wetland performance such as the water balance, hydraulic and nutrient cycling processes are increasingly important (Kadlec and Wallace, 2009). Such knowledge may then be applied to other systems in areas with similar environmental conditions and wetland characteristics, in order to make informed decisions regarding new designs or the upgrading of existing treatment systems.

1.4 Crossways Farm Village treatment wetland

This study focuses on using a mass balance approach to measure the effectiveness (treatment efficiency) of a FWS constructed wetland system in the Crossways Farm Village estate, intended to act in the secondary treatment of domestic wastewater.

The use of a mass balance approach requires quantifying the water balance of the wetland, considering all flows into, through and out of the wetland system, in combination with the measurement of nutrient concentrations. This approach allows nutrient concentrations to be converted to fluxes, which allows one to account for the pollutant outflows relative to inflows and thus determine the effectiveness of the treatment wetland (Breen, 1990).

The Crossways Farm Village wastewater treatment works (WWTW) is comprised of a series of interactive components each designed to play a specific role in the treatment process. Domestic wastewater is pumped into an anaerobic reactor that digests and filters the solid matter. It is then oxidised in a rotational bio-disc contractor and gravity fed into a clarifier before entering the FWS wetland system via an overflow weir. This constructed wetland system feeds wastewater through three artificially created wetland basins (hereafter referred to as “cells”) before being discharged via surface flow into a nearby dam.

Crossways Farm Village is an “eco-estate” that is conservation orientated, aimed at minimising the ecological footprint of the development. Domestic wastewater produced from approximately 50-60 households is therefore treated within the estate using an aesthetically pleasing FWS constructed wetland. Typical FWS constructed wetlands contain areas of open water, floating vegetation and emergent macrophytes, either by design or as an unavoidable consequence of the design configuration (Kadlec and Wallace, 2009). FWS constructed wetlands are the nearly exclusive choice for the treatment of urban, agricultural and storm water effluent discharges, due to their ability to deal with pulsed flows and changing water levels, a situation the characterises the Crossways Farm Village WWTW.

1.5 Research question, aim and objectives

Research question: Can FWS constructed wetlands that are lightly loaded with secondary domestic wastewater, effectively treat and/or store nitrogen and phosphorus in winter months?

Aim: To assess the effectiveness of the Crossways Farm Village FWS constructed wetland in the treatment of secondary domestic wastewater.

Objectives

- To quantify the water balance of the treatment wetland
- To determine the inputs, outputs and storage of nutrients, (nitrogen and phosphorus), in the FWS constructed wetland system
- To use this information in order to make recommendations regarding the use of this technology for similar and/or broader application

Chapter 2: Literature Review

2.1 The need to treat domestic wastewater given the problem of eutrophication

The need to reduce anthropogenic nutrient inputs to aquatic ecosystems in order to protect drinking-water supplies and aquatic ecosystems has been widely recognised (Smith *et al.,* 2006). Eutrophication has become the primary water quality issue for most freshwater and coastal marine ecosystems in the world (Smith and Schindler, 2009). Cultural eutrophication, defined as excessive plant growth resulting from nutrient enrichment by human activities, is a major driving force behind the severe water quality reductions in South Africa. Its affects are one of the most visible examples of human impacts on the biosphere that needs to be addressed and combatted to sustain the human population (Cronk, 1996). Eutrophication therefore has many undesirable economic and social effects, which cumulatively have major economic costs and transnational implications (Smith and Schindler, 2009).

Eutrophication of freshwater systems is largely attributed to elevated levels of both nitrogen and phosphorus in their various forms (Mitsch *et al*., 2000). Nitrogen, needed for protein synthesis, and phosphorus, needed for DNA, RNA, and energy transfer, are both required to support aquatic plant growth and are the key limiting nutrients in most aquatic and terrestrial ecosystems (Conley *et al*., 2009). These nutrients often enter fluvial systems and water resources via diffuse or non-point sources associated with surface runoff, as well as from point sources typically associated with concentrated farming activities and formal and informal settlements (Conley *et al.*, 2009). Nitrogen is abundant in nature but exists almost entirely in the form of molecular nitrogen (N_2) that is unusable by most organisms (Galloway *et al.*, 2003). Inorganic forms of nitrogen (ammonia, nitrate) that can be used and taken up by organisms, are formed by either lightning or nitrogen fixing bacteria (Knight *et al*., 2000). Humans have however, changed this dynamic and excess levels of bioavailable nitrogen have become a source for pollution through the affect it has on the foundation of the food chain. The Haber-Bosch reaction in particular made it possible to produce nitrogen compounds that enhanced agricultural production globally, but an unintended consequence was atmospheric and aquatic pollution on a scale that was globally significant (Galloway *et al*. 1995; Vitousek *et al*. 1997; Townsend *et al*. 2003). The main sources of reactive forms of nitrogen include atmospheric deposition, commercial synthetic fertilizers used in agriculture, oxidation of soil organic nitrogen, and nitrified wastewater from septic tanks, conventional wastewater treatment plants and animal farming activities (Galloway *et al*., 2003).

The impacts of such over enrichment of dissolved nutrients include excessive primary production, especially by algae and cyanobacteria, which leads to high bacterial populations and high respiration rates (Conley *et al*., 2009). This excessive biomass accumulation depletes oxygen in the water when the algae die and decompose. Reduced oxygen concentrations thus lead to hypoxia in surface waters and anoxia in poorly mixed bottom waters or surface waters at night during calm, warm conditions (Smith and Schindler, 2009). Reduced dissolved oxygen causes the death of aquatic animals, the bodies of which settle, decompose and result in the release of many materials normally bound to bottom sediments, including various forms of phosphorus (Correl, 1998). This release of phosphorus reinforces and enhances the process of eutrophication. Excessive enrichment of this nutrient is the most common cause of eutrophication in freshwater systems including rivers, streams, lakes and reservoirs, emphasising the importance of addressing such nutrient releases that are reducing the quality of this already limited resource (Knight *et al.*, 2000). Nutrient releases attributed to human activities and waste disposal have now been nationally recognised as having a direct impact on the quality of South Africa's water resources (Mara, 2004).

2.2 Legislation and regulation

The South African Water Act 54 of 1956 was promulgated in 1956. Section 21 of this Act required the permitting of all effluent dischargers, including domestic and municipal sewage works (Tewari, 2009). The General and Special Standards under the Act were subsequently published in the Government Gazette in 1984, which set effluent discharge quality limits for domestic and industrial discharges. This was known as the Uniform Effluent Standard approach (Growlland-Gualtieri, 2007). However, this approach did not take into account the assimilative capacity of the receiving water, or the limitations thereof (Tewari, 2009). The White Paper on Water Policy in South Africa was then published by the Department of Water Affairs and Forestry in 1997. The White Paper identified this concern and suggested a change in the way water quality was managed in the country. The National Water Act, 36 of 1998 was subsequently promulgated and provided the tool to effect these changes (NWA, 1998). This act adopted the Receiving Water Quality Objectives (RWQO) perspective. This "polluter pays" type of approach takes into account the impacts on the receiving water as well as the impacts on other water users and aimed to achieve the long-term sustainable use of this limited resource (Growlland-Gualtieri, 2007). Particular emphasis was also given to the availability of water in sufficient quantity and of acceptable quality for basic human needs and for the natural aquatic environment (Pienaar and van der Schyff, 2007). Monitoring wastewater releases into natural water bodies using this advanced legislation should therefore adequately regulate the cause and sources of excessive nutrient releases into the country's water resources. However, the implementation of the legislation has been weak, and financial limitations have reduced the monitoring of nutrient releases, particularly for rural and decentralised small-scale wastewater releases (Pienaar and van der Schyff, 2007). Nevertheless, the promulgation of the legislation has culminated in a list of wastewater limit values that serves to regulate the acceptable concentrations of pollutants in wastewater that can be released into a given natural water body within South African borders.

2.3 Decentralised wastewater treatment

It is common knowledge that the increased anthropogenic discharge of nutrients into aquatic ecosystems has resulted in large-scale changes in their structure and functioning (Conley *et al*, 2009; Cronk, 1996; Smith and Schindler, 2009). The use of conventional "centralised" municipal systems for wastewater management has become impractical and inefficient, predominantly due to rapid population growth in developing countries throughout the world (Mara, 2004). The required planning and infrastructure needed to deal with such large-scale wastewater production has become impossible to manage when *ad hoc* housing and informal peri-urban settlements rapidly develop (Mthembu *et al*, 2013). As a result, decentralised and onsite domestic wastewater treatment from single sources and cluster scale housing in rural and remote areas, has gained acceptance across the globe. Evidence has also accumulated to favour nutrient removal and retention technologies as the primary means of combatting excessive waste production in remote areas (Ho and Anda, 2006; Mara, 2004).

Several types of biologically orientated treatment systems (septic tanks, settling ponds, lagoons and FWS constructed wetlands), have been developed for treating rural and remote small-scale wastewater flows (Mara, 2004). These cost-effective systems that typically rely on microbiological processes of nutrient transformation, significantly reduce the input, maintenance and human resources costs that are often unavailable in such circumstances (Mthembu *et al.*, 2013). Of these small-scale systems, the use of constructed wetlands for wastewater treatment has emerged as a major solution to the problem of decentralised wastewater management and has proven to be effective at treating wastewater (Zhang *et al*., 2014). However, there is a fair amount of variability and uncertainty regarding the performance and maintenance of these systems, often relating to wetland design and environmental factors affecting system performance (Vymazal, 2007).

Continued research has shown that measuring the effectiveness and performance of such treatment wetlands requires the use of a mass balance approach (Breen, 1990; Chung *et al*., 2007; Lee *et al*., 2014; Kadlec *et al*., 2005; Kadlec and Wallace, 2009). The mass balance approach takes the relationship between seasonal changes in environmental factors and nutrient removal variability, and their effects on a given wetlands treatment performance, into consideration (Breen, 1990). Undertaking a water and nutrient mass balance is therefore a necessary approach in any attempt to accurately quantify and relate treatment performance to environmental factors that create variability in the effectiveness of constructed wetland treatment systems.

2.4 Water treatment in constructed wetlands

Nutrient transformation and removal in constructed wetlands has been well documented and successfully implemented around the world (Kadlec and Wallace, 2009). However, understanding the interactions between and within wetland plants, microorganisms, soils and substances in the wastewater is complex and there are often aspects that are unique to each treatment system (Stottmeister *et al.*, 2003). Wetlands are typically characterised by adequate light availability, water, and nutrient supply, and therefore the primary productivity in such ecosystems is high. Associated with this high productivity is a high capacity to decompose and transform organic matter and other minerals and nutrients. Macrophytes, in treatment wetlands, play a vital role through the release of oxygen from their roots and by the provision of a food source and substrate for microbial activities (Vymazal, 2007). These plants are also able to tolerate remarkably high concentrations of nutrients and heavy metals, and may accumulate them in their tissues (Vymazal, 2007).

An active transformation and reaction zone within constructed wetlands is the rhizosphere of wetland plants, where a wide range of physiochemical and biological processes are induced by the interaction of plants, microorganisms, the soil and pollutants (Stottmeister *et al*., 2003). In constructed wetlands, the main mechanism responsible for the transformation and mineralisation of nutrients and organic pollutants is however not by plants, but by microorganisms, although they are largely interdependent (Kadlec and Wallace, 2009). In essence, most wetland reactions are the result of the activity of bacteria or other microorganisms that are attached to submerged surfaces within the wetland (Vymazal, 2009). This transfer of a chemical or nutrient from water to an immersed solid surface is the first and most important step in the overall microbial removal mechanism in constructed wetlands (Kadlec and Wallace, 2009). Physiochemical processes that lead to the removal and transformation of suspended sediments, solids and other nutrients and minerals, is facilitated by the formation of areas with soils inundated to a shallow depth, maximising the extent of the sediment/water interface (Keller and Knight, 2004). Consequently, high primary productivity, the presence of aerobic and anaerobic sediments within close proximity of each other, and the accumulation of litter, creates a wide range of environmental conditions that cumulatively result in the transformation and immobilisation of nutrients (Faulwetter *et al.*, 2009). Microbially mediated nutrient transformation in combination with physicochemical processes such as the fixation of phosphates by iron and aluminium in the substrates, are thus dominant processes and mechanisms that lead to water treatment in constructed wetlands (Stottmeister *et al*., 2003).

2.5 Factors controlling nutrient transformation and removal

Microorganisms are widely considered to be the main force driving the treatment processes in treatment wetlands, as they mineralise organic matter under both aerobic and anaerobic conditions (Truu *et al*., 2009). Submerged plant tissues are colonised by dense communities of photosynthetic algae, bacteria and protozoa as biofilms (Stottmeister *et al*., 2003). Biofilms are any submerged surface area in the wetland that can support the attachment and growth of microbes, including both living and dead macrophyte tissue. Biofilms are responsible for most of the microbial processes that transform nutrients in wetlands (Stottmeister *et al.,* 2003). Plants create and ensure the availability of a substrate for biofilm growth through the creation of a litter/humus layer (Stottmeister *et al.*, 2003). As plants grow and die, leaves and stems that fall into the water column and settle on the wetland bed, creating multiple layers of organic debris (Keller and Knight, 2004). This accumulation of partially decomposed biomass creates highly porous substrate layers that provide additional attachment surfaces for microbial organisms. Partially decomposed litter material provides a stable carbon (energy/food) source that sustains and promotes the growth of the various microbial populations, and also leads to the creation of new stable sediments that are available for sorption and burial removal pathways (Keller and Knight, 2004).

Aquatic plants adapted to growing in wetland environments are typically emergent macrophytes (Campbell and Reece, 2002). Macrophytes are one of the key characteristics used to define wetlands and are an indispensable component of this ecosystem (Brix, 1997). Macrophytes, like all other photoautotrophic organisms, use solar energy to assimilate inorganic carbon from the atmosphere to produce organic matter, which then provides the main energy source to heterotrophs such as animals, bacteria and fungi (Stottmeister *et al.*, 2003). Emergent macrophytes play a crucial role in constructed wetlands directly through their physical presence and in the supply of oxygen to the root zone of anaerobic soils, which govern the transformation of nutrients and metabolism of microorganisms (Brix, 1997).

The presence of dense vegetation in wetlands also distributes and reduces the current velocity of the water (Jadhav and Buchberger, 1995). This reduced flow rate creates better conditions for sedimentation of suspended solids and reduces the risk of erosion and resuspension (Brix, 1997). The physical growth of macrophytes therefore significantly increases the contact time between the wastewater and the plant and soil surface area, increasing the effluent contact with microbes, thus significantly reducing nutrient levels in wetlands (Jadhav and Buchberger, 1995). Macrophytes are also important for stabilising the soil surface in FWS constructed wetlands, as their dense root systems prevent the formation of channels. The growth and decomposition of macrophyte roots not only provide oxygen and carbon to the root zone, but also create good conditions for the physical filtration of water through aerobic and anaerobic zones of the wetland (Brix and Schierup, 1990).

Wetland plants, like other photoautotrophs, require and use nutrients for growth and reproduction (Campbell and Reece, 2002). Emergent macrophytes usually take up nutrients through their root system, which are then assimilated into amino acids that make up plant tissue biomass (Vymazal, 2007). Wetland plants are highly productive and can therefore consume and remove considerable amounts of nutrients during the growing season (Crities *et al.,* 2006). However, the removal of nutrients (nitrogen and phosphorus) via harvesting of aboveground biomass is low, in comparison to the highly elevated nutrient levels found in domestic wastewater (Kadlec and Wallace, 2009). Nevertheless, in lightly loaded systems, multiple harvesting of wetland plants may substantially reduce nutrient levels (Vymazal, 2007).

It is generally accepted that low levels of soil carbon limit the amount of energy available to microorganisms (Kadlec and Wallace, 2009). However, the process of carbon input from plants into their rhizosphere, known as rhizodeposition (Campbell and Reece, 2002), is a source of energy for soil microbes and enhances their growth and reproduction (Fontainea *et al.,* 2003, Abbot and Murphy, 2007). The release of carbon is mainly derived from decomposing plant biomass that is readily available in wetland ecosystems. Rhizodeposition products such as exudates, dead cell material and mucigels (a complex material composed of root mucilage and bacterial slime, which acts to control aggregation of soil particles in the rhizosphere), promote several biological processes, releasing various sugars, amino acids and excreted vitamins, which are then used by microorganisms as a substrate (Stottmeister *et al.*, 2003). Carbon levels in the soil therefore strongly correlate to microbial populations, which ultimately controls the treatment ability of a given wetland system (Fontainea *et al.,* 2003).

2.6 Free Water Surface (FWS) constructed wetlands

FWS constructed wetlands closely resemble natural depression wetlands in appearance and function, with a combination of open water areas, emergent vegetation, varying water depths and other typical wetland features (Vymazal, 2013). Most FWS wetlands consist of one or more vegetated shallow basins (cells), ponds or channels with clay or high-density polyester plastic lining to prevent groundwater seepage, yet still containing soils or a substrate to support emergent macrophyte vegetation (Kadlec and Wallace, 2009). Appropriate inlet and outlet structures (berms, dikes, weirs) are also required to monitor and maintain appropriate flow rates and water levels.

The water depth in FWS wetlands usually ranges between 0.1m and 0.8m (Van Deun, 2015). In general, water immediately above the soil substrate of the wetland is anaerobic, above which the water column is generally aerobic. The source of oxygen in this zone is atmospheric aeration, caused by mixing at the water-air interface. FWS wetlands therefore require a large surface area relative to influent loading rates. However, they are much easier and cheaper to construct than any other treatment wetland and are particularly effective in regions with warmer climates (Machado *et al*., 2016). These wetlands also have a high aesthetic value and create valuable habitats for several organisms, potentially contributing to biodiversity conservation.

Typha spp. (cattails or bulrushes) are the most common macrophyte species used in FWS constructed wetlands and their efficiency in nutrient removal has been well documented throughout the world (Lorenzen *et al*., 2001; Miao, 2004; Vymazal, 2013). The species is an erect rhizomatous perennial plant with long broadly linear leaves and a jointless stem bearing the inflorescence. *Typha* spp. are known for their rapid biomass accumulation and therefore high nutrient uptake in FWS treatment wetlands. Asexual reproduction via a creeping rhizome allows this robust macrophyte species to completely colonise wetland habitats and form dense stands in less than a year (Crites *et al*., 2006). *Typha spp.* can thrive in permanent inundation at > 0.3m but can also tolerate periods of drought. Average plant height ranges between 2m and 4m and root penetration is relatively shallow at approximately 0.3m (Vymazal, 2013). The plant also has significant cultural and aesthetic value and provides valuable food sources and nesting sites for water birds.

2.7 Domestic Wastewater Characteristics

Wastewater is defined as water that carries wastes from homes, businesses and industries, for which disposal is more economical than use at the time and point of its occurrence (Mara, 2004). Wastewater components show different degrees of environmental nuisance and contamination hazard due to their chemical and microbiological characteristics (Bohdziewicz and Sroka, 2006). Typical domestic wastewater is comprised of 99.9% water, together with relatively small concentrations of suspended and dissolved organic and inorganic solids (Kadlec and Wallace, 2009). Among the organic substances present in domestic sewage are carbohydrates, lignin, fats, soaps, synthetic detergents, proteins and their decomposition products as well as various natural and synthetic organic chemicals from process industries (Henze *et al*., 2001). Domestic wastewater also contains a variety of inorganic substances from domestic and industrial cleaners, including a number of potentially toxic elements such as arsenic, cadmium, chromium, copper, lead, mercury and zinc (Henze *et al*., 2001). Even if toxic materials are not present in concentrations likely to affect humans, they might well be at phytotoxic levels, which would limit the wastewaters potential for reuse or release into the environment (Mara, 2004).

Domestic wastewater is comprised of a complex combination of both organic and inorganic compounds as well as bacteria and microorganisms. The major water treatment function of FWS wetlands treating secondary domestic wastewater is typically to remove suspended solids (organic matter), reduce nutrient (nitrogen and phosphorus) concentrations and decrease biological oxygen demand (BOD) (Kadlec and Wallace, 2009; Tanner *et al*., 1998).

Suspended solids entering a treatment wetland may display a wide variety of characteristics and domestic wastewater at all pre-treatment stages will contain suspended materials that are primarily organic materials (Kadlec and Wallace, 2009). A major function performed by wetlands is the removal of these suspended solids from wastewater moving through the wetland. Low flow velocities coupled with the presence of dense plant stems, litter, sand or gravel, promotes settling and interception of solid materials in FWS wetlands with irregular shapes, variable depths and flow patterns (Van Deun, 2015). This transfer of suspended solids from water to wetland sediment bed has important consequences for the endpoint water quality and entire wetland ecosystem functioning. Many pollutants such as metals and organic chemicals and nutrients are associated with the incoming suspended matter, thus the removal of suspended solids is a crucial component in the treatment process of secondary domestic wastewater (Kadlec and Wallace, 2009).

Of the vast array of constituents found in domestic wastewater, organic matter is the major pollutant due to its effect on oxygen consumption and availability (Crites *et al*, 2006). Dissolved oxygen (DO) is a key factor in treatment wetlands for two main reasons: firstly, it is a key element in several pollutant removal mechanisms and secondly it is a universally recognised regulatory parameter for monitoring discharges to surface waters (Kadlec and Wallace, 2009).

Nitrogen and phosphorus are mineral nutrients required for all biological life (Howard-Williams, 1985). In domestic wastewater, these compounds exist in very high concentrations and are often monitored as a part of water quality testing (Kadlec and Wallace, 2009). At high concentrations, these nutrients become pollutants that can cause significant water quality problems, exponentially increasing aquatic plant growth that disrupts the light, temperature and oxygen levels in the water below, leading to eutrophication and hypoxia (Smith and Schindler, 2009).

2.8 Nutrient transformation in wetlands

Nitrogen in water is measured as the common inorganic form of nitrate-nitrogen (NO_3-N) that is dissolved in water and readily absorbed by photosynthetic organisms such as algae (Kadlec and Wallace, 2009). The common form of phosphorus measured in water is phosphate (PO_4-P), which is generally attached to sediment particles (Kadlec and Wallace, 2009). Sources of nitrates and phosphates include releases from domestic and municipal wastewater treatment plants, runoff from fertilised lawns and agricultural lands, faulty septic systems, animal manure runoff, and industrial waste discharge (Van Deun, 2015).

Nitrogen compounds are among the principle constituents of concern in wastewater because of their role in eutrophication and particularly, their effect on the oxygen content of receiving waters, and therefore toxicity to aquatic invertebrate and vertebrate species (Kadlec and Wallace, 2009). The nitrogen cycle (Figure 1) is a complex biogeochemical cycle with multiple biotic and abiotic transformation phases, including a variety of inorganic and organic forms that are essential for plant growth and therefore, for all life (Galloway *et al*., 2003). The most important inorganic forms of nitrogen in wetlands are ammonium, nitrite and nitrate (Kadlec and Wallace, 2009). Gaseous nitrogen may exist as dinitrogen, nitrous oxide, nitric oxide and ammonia. Nitrogen is also invariably present in FWS wetlands in organic forms as proteins, amino acids, urea, uric acid, and purines and pyrimidines (Kadlec and Wallace, 2009).

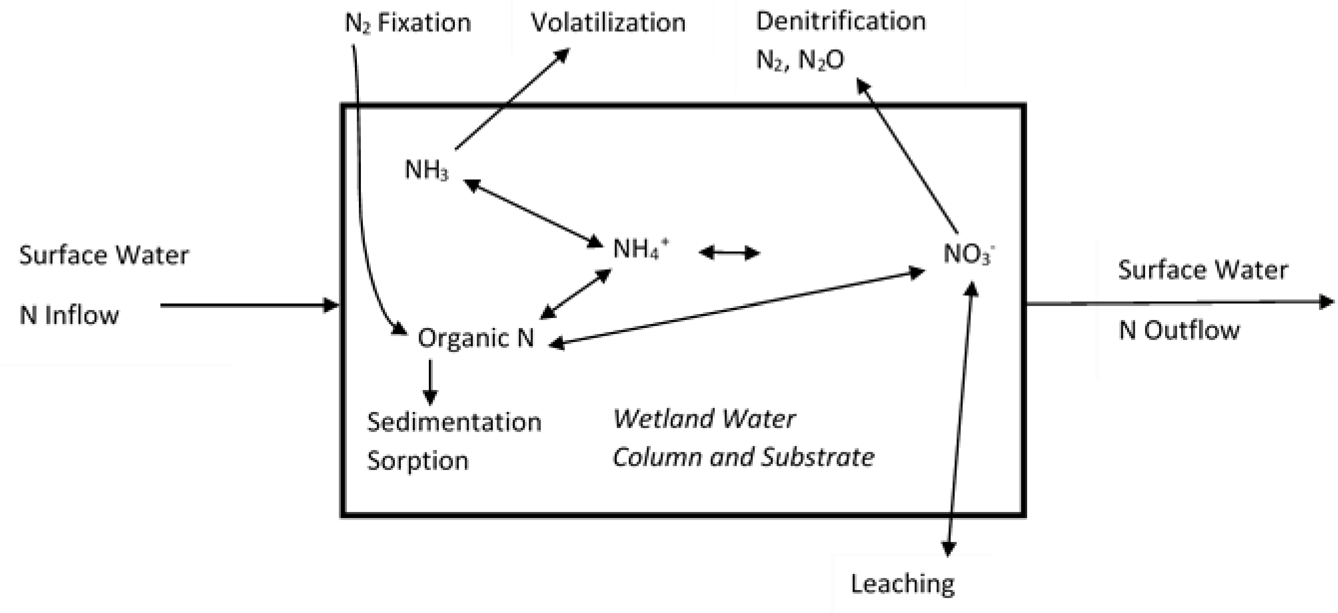

Figure 1: Conceptual model showing nitrogen cycling in FWS wetlands (after Spieles and Mitsch, 1999)

Nitrogen assimilation refers to a variety of biological processes that convert inorganic nitrogen forms into organic compounds that are required for the creation of cells and tissues (Kadlec and Wallace, 2009). The two forms of nitrogen generally used for assimilation are ammonia and nitrate nitrogen, and both compounds are readily taken up by plants and stored in organic forms in wetland vegetation (Vymazal, 2007). Macrophyte growth, however, is not the only potential biological assimilation process: microorganisms and algae also utilise nitrogen, and ammonia is incorporated into amino acids by many autotrophs and microbial heterotrophs (Vymazal, 2007).

The processes that affect the removal and retention of nitrogen during wastewater treatment in constructed wetlands are diverse (Figure 2) and include volatilisation, nitrification, denitrification, nitrogen fixation, plant and microbial uptake, mineralisation (ammonification), nitrate reduction to ammonium (nitrate-ammonification), anaerobic ammonia oxidation, sorption, desorption, burial and leaching (Stottmeister *et al.*, 2003). However, only a few processes can ultimately remove nitrogen from the wastewater while most processes just convert it from one to another of its various forms (Stottmeister *et al.*, 2003). Nitrogen removal in FWS wetlands predominantly occurs via the process of denitrification where dinitrogen gas (N_2) and nitrous oxide are released into the atmosphere and removed from the water (Lee *et al.*, 2014).

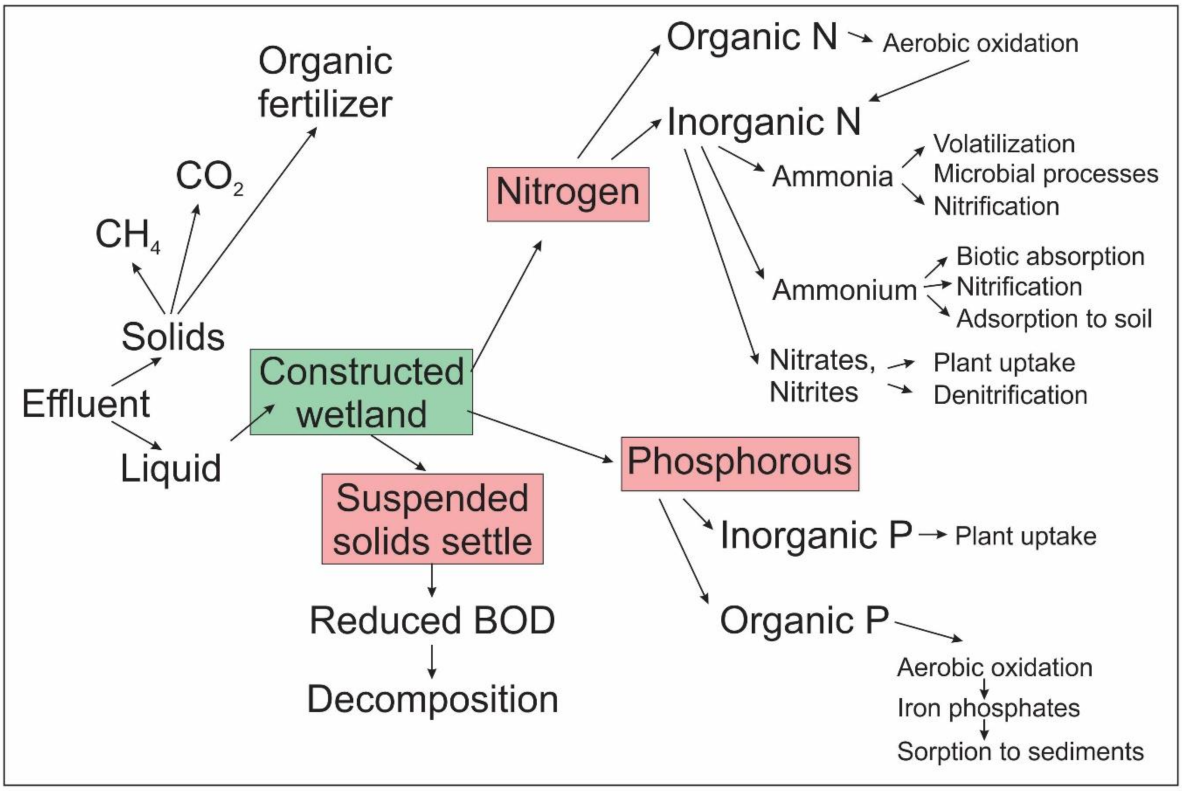

Figure 2: Typical nutrient transformations in constructed wetlands (Stottmeister *et al*., 2003)

The various forms of nitrogen are chemically transformed from inorganic to organic compounds and back from organic to inorganic (Spieles and Mitsch, 1999). Some of these processes require energy (typically derived from an organic carbon source) to proceed, and others release energy, which is used by various organisms for their growth and survival (Kadlec and Wallace, 2009). All of these transformations are necessary for wetland ecosystems to function, and most chemical changes are controlled through the production of enzymes by the living organisms they benefit (Kadlec and Wallace, 2009).

Ammonification is the microbial conversion of organic nitrogen to ammonia and plays an important role in nitrogen transformation in wetlands (Lee *et al*., 2014). The process of ammonification does not remove nitrogen from wastewater but instead converts organic nitrogen to ammonia, which is then available for other processes such as nitrification, adsorption and plant uptake (Lee *et al*., 2014). Ammonia typically comprises more than half of the total nitrogen (TN) in a variety of domestic effluents (Kadlec and Wallace, 2009), and is therefore one of the principal forms of nitrogen of concern in most wastewaters. Reducing ammonia concentrations often drives the design process for many FWS treatment wetland systems due to its role in degrading the environmental condition of receiving water bodies (Van Deun, 2015).

Nitrite (NO_2) is an intermediate oxidation state of nitrogen between ammonia and nitrate (Kadlec and Wallace, 2009). As a result of this energetic intermediate condition, nitrite is not chemically stable in most wetlands and is generally only found at low concentrations (Kadlec and Wallace, 2009).

2.9 Nitrogen storage in wetlands

Organic nitrogen compounds make up a significant proportion of the dry weight of wetland plants, microbes, detritus, fauna and soils (Kadlec and Wallace, 2009). The mass of these nitrogen storages varies significantly in various wetland types and gives a reflection of the type and character of the wetland in question (Van Deun, 2015). A general idea of the sizes of these different nitrogen storage compartments is necessary to understand nitrogen fluxes within any wetland ecosystem.

Nitrogen storage in constructed wetlands may occur as a result of the deposition of nitrogen in the sediments as organic matter (Bastviken, 2006). This sedimentation and accretion of nitrogen in constructed wetland soils occurs during settling of solids, death of below ground biota and litterfall (Lee *et al*, 2014). Solids removal is mainly related to sedimentation on the wetland bed, and the quantity of nitrogen found in suspension largely depends upon the organic matter content, type of wastewater and degree of pre-treatment (Kadlec and Knight, 1996). Some studies have shown that nitrogen removal due to sedimentation of suspended solids is the most significant nitrogen removal process in FWS wetlands (Mitsch *et al*., 2000; Schierup *et al.,* 1990; Shamir, 1998). In a typical constructed wetland for municipal wastewater treatment, 1-2mm. yr^{-1} of low bulk density solids (about $0.03 g.cm^{-3}$) are accumulated, and this accretion gradually increases toward the constructed wetland inlet (Bastviken, 2006).

Sedimentation occurs when particulate matter (inorganic and/or organic suspended sediment) entrained in the water column settles out, due to the reduced water velocity, shallow water depth and filtering action of emergent vegetation (Shamir, 1998). Vegetation density plays a crucial role in the deposition of organic matter on the bed by decreasing the flow velocity and therefore, increasing the residence time of water in the wetland (Jadhav and Buchberger, 1995). The residence time is therefore a major factor that dictates the spatial and temporal dynamics of the sedimentation process (Bastiviken, 2006). Other factors that influence these processes are operational parameters such as inflow rate and water depth (Bastviken, 2006). Literature is scarce on the deposition processes that are responsible for nitrogen removal in FWS wetlands and most of the reports are from studies done on sedimentation in natural wetlands (Kadlec and Wallace, 2009). Extensive research in these ecosystems has shown that approximately 95% of the total nitrogen is present in natural wetlands soils and sediments, which thus constitute a major storage compartment for the removal of nitrogen (Shamir, 1998). In such circumstances, microbial decomposition of organic matter is incomplete, and the remaining nutrients are buried and stored in newly accreted sediments (Keller and Knight, 2004).

The nitrogen content of the living biomass in wetlands varies considerably among species, plant parts and within various wetland sites (Kadlec and Wallace, 2009). Organisms like plants, fungi and certain bacteria that cannot fix atmospheric or dissolved nitrogen gas (N_2) depend on the ability to assimilate the available nitrate or ammonia for their growth and survival (Vymazal, 2007). Since treatment wetlands are usually nutrient enriched, total biomass is enhanced by fertilisation with effluent, which compounds the effect of increased nitrogen content such that treatment wetlands produce large biomass storages compared to unfertilised natural wetlands (Vymazal, 2013). For example, live cattail (*Typha)* leaves in a FWS wetland treating wastewater in the discharge area of the Houghton Lake, Michigan, averaged 2.0% nitrogen, compared to those in nutrient poor control areas, where similar tissue contained an average of 1.1% nitrogen. Dead leaves also had nitrogen concentrations of 1.6% in the constructed wetland versus 0.7% in a natural unfertilised wetland, and leaf litter had concentrations of 3.6 % in the constructed wetland versus 1.5% nitrogen in the unfertilised wetlands (Kadlec and Wallace, 2009).

Biomass accumulation and/or plant growth is also largely dependent on factors other than nutrient loading, including incoming solar radiation, ambient temperature, substrate type and various other factors affecting plant metabolism (Kadlec and Reddy, 2001). Nutrient uptake and assimilation therefore show a strong seasonal variability in FWS wetlands. Klopatek (1978) for example, showed that cattail (*Typha*) biomass collected at the end of each growing season in autumn, displayed much lower nitrogen content than at the start of the growing season in spring. These seasonal storages reflect the growth cycle of wetland plants and are an important consideration for harvesting biomass to enhance nutrient removal, particularly in lightly loaded systems (Spieles and Mitsch, 1999; Vymazal, 2007).

2.10 Phosphorous removal and storage in wetlands

Phosphorus is also an essential element and plant nutrient required for plant growth, and is frequently a limiting factor affecting vegetation productivity in natural wetland systems (Acreman and Fisher, 2004; Vymazal, 2007). The introduction of trace amounts of this element into receiving waters can therefore have a profound effect on the structure of aquatic ecosystems and can result in eutrophication (Cronk, 1996). Several processes ultimately result in the removal of phosphorus from wastewater in treatment wetland environments, but some only have a limited capacity for removal. Once the phosphorus assimilating capacity is reached or exceeded no further removal via this pathway will occur (Reddy *et al*., 1999).

In general, the pathways of phosphorus removal from wastewater include movement from the water column to the sediment (sorption) and storage of refractory residual phosphorus compounds by burial and uptake by macrophytes (biomass accumulation; Keller and Knight, 2004). Sorption and biomass removal mechanisms have finite phosphorus retention capacities, whereas accretion or burial processes are ongoing (Kadlec and Wallace, 2009). According to Kadlec (1994), accumulative phosphorus removal processes essentially follow first order reactions, which means that the removal of phosphorus to new soils is also proportional to the concentration of phosphorus in the surface waters, and to the surface area of the wetland. Prior to phosphorus removal from the water column into sediments through chemical saturation processes, time needs to be given to allow for the chemical equilibrium to be achieved between water and sediment (Keller and Knight, 2004).

The processes involved in phosphorus transformation and removal in constructed wetlands include adsorption, desorption, precipitation, dissolution, plant and microbial uptake, leaching, mineralisation, sedimentation (soil accretion) and subsequent burial (Stottmeister *et al.,* 2003; Vymazal, 2007). The removal of phosphorus in all types of constructed wetlands is generally relatively low unless substrates with high sorption capacity are used (Vymazal, 2007).

Phosphorus in wetlands generally occurs as phosphate, although it may be in the form of organic compounds (Kadlec and Wallace, 2009). Orthophosphate (PO_4-P) is the generic term used to describe inorganic phosphate ions, which are the only forms of phosphorus believed to be utilised directly by algae and macrophytes and thus, represents a major link between organic and inorganic phosphorus cycling in wetlands (Reddy *et al*., 1999). Organic phosphorus forms can be generally grouped into easily decomposable phosphorus and slowly decomposable organic phosphorus (Wu *et al.,* 2014).

Wetlands provide the environment and conditions for the conversion of all such forms of phosphorus (Wu *et al*., 2014). Soluble reactive phosphorus is taken up by plants and converted to organic phosphorus, or it may become sorbed to wetland soils and sediments (Vymazal, 2007). Organic phosphorus may also be released as soluble phosphorus if the organic matrix is oxidised. Insoluble precipitates may however form under some circumstances but can again re-dissolve in altered environmental conditions (Vymazal, 2007; Stottmeister *et al.*, 2003).

Phosphorus compounds accumulate in and comprise a significant portion of the dry weight of wetland plants and animals, detritus, microbes and soils (Kadlec and Wallace, 2009). The mass of these phosphorus storages varies in different wetland types and within different seasons of the year (Vymazal, 2007). The dominant fraction of phosphorus is however generally stored in wetland soils and sediments (Keller and Knight, 2004). Plants and litter comprise the majority of the remaining phosphorus, with very little mass contained in microbes, algae and water (Keller and Knight, 2004).

Sediment accretion in wetlands occurs through a variety of physical and chemical processes and most of the phosphorus in the soil column is structural phosphorus, both in organic and inorganic forms (Kadlec and Wallace, 2009). Water movement is typically very slow in wetlands, which facilitates the physical deposition of clastic and organic sediments that settle under conditions of low flow velocity (Keller and Knight, 2004). Physical settling can also occur through the collision of particles with plant stems, the trapping of particles in biofilms attached to macrophytes and the sediment/water interface, or any other process that moves particles to a sediment surface (Kadlec and Knight, 1996). Sediment accretion is therefore a product of the type, rate and character of external loading sources and a variety of internal wetland processes such as plant growth, decomposition and net accretion of residual organic solids or accreted sediments (Keller and Knight, 2004). During decomposition processes, the majority of assimilated phosphorus is transformed and released back into the water column, but at least 10-20% is permanently stored in soils and sediments as the residual from microbial decomposition processes (Kadlec and Wallace, 2009). These new stable biomass residuals contain phosphorus as part of their structure and hence the process of accretion represents a burial process for phosphorus that is resistant to decomposition (Keller and Knight, 2004). Microbial communities of wetland soils are therefore important for both decomposition of organic material and remobilisation and cycling of nutrients, and thus play an integral role in the metabolism of the entire ecosystem (Kadlec and Wallace, 2009).

The removal of phosphorus from water by wetland plants has been the subject of many studies (Tanner, 1995; Reddy and DeBusk, 1985; Vymazal, 2007). Research has shown that the nutrient content of many plant species including *Typha* spp. displays increasing tissue phosphorus content with increasing phosphorus availability (Keller and Knight, 2004). There is also a substantial increase in standing crop with an increase in phosphorus availability ranging from 1kg.m^2 of plant biomass at low nutrient conditions to 6kg.m^2 at high nutrient conditions (Kadlec and Wallace, 2009). Tissue nutrient concentrations in live cattail (*Typha*) plants from a nutrient rich FWS wetland in Houghton Lake, Michigan, indicated that phosphorus concentration in plant tissue averaged at 0.18%, whereas those in nutrient poor control wetlands averaged 0.09% phosphorus (Kadlec and Wallace, 2009).

The processes of growth, death, litterfall and decomposition operate year-round at variable rates depending upon seasonality, climatic conditions and habitat characteristics (Kadlec and Wallace, 2009). Therefore, phosphorus concentrations in wetland sediment and plant biomass also display seasonal variability such that for example, aboveground biomass collected at the end of the growing season displays much lower phosphorus concentrations than in spring (Klopatek, 1978). This seasonal storage variability reflects the growth and death cycle of wetland plants, affecting the storage and release of nutrients, with important consequences on treatment wetland systems where harvesting is used to limit overall nutrient accumulations (Vymazal, 2007).

2.11 Biogeochemical cycling and carbon sequestration in wetlands

Wetland ecosystems play a key role in regulating the exchange of various greenhouse gases to and from the atmosphere, including water vapour, carbon dioxide, methane, nitrous oxide and sulfur dioxide (Pritchard, 2009). Wetland environments tend to act as sinks for carbon and nitrogen and sources for methane and sulfur compounds. However, each wetland is unique and situations can vary from place to place, time to time and between the various wetland types (Pritchard, 2009). Wetland ecosystems comprise approximately 5-8% of the total land surface, yet they contain approximately 20-30% of the carbon stored in terrestrial ecosystems (Mitsch and Gosselink, 2007). This critical ecosystem service provided by wetlands contributes to regulating and buffering the effects of climate change induced by anthropogenic carbon emissions. Furthermore, the ability of wetlands to absorb bioavailable nitrogen and phosphorus into organic nitrogen and phosphorus, allows wetlands to act as biological filters, enhancing downstream water quality and preventing eutrophication (Fennessy *et al*., 2008). Thus, constructed wetlands and wetland restoration efforts have promoted extensive research into the biogeochemical processes which drive the ecosystem services provided by wetlands.

One of the least studied functions of FWS wetland ecosystems is plant litter decomposition, yet it represents the critical feedback loop that recycles and transfers nutrients and aids in mediating the sequestration of soil carbon (Fennessy *et al.,* 2008). In most ecosystems, plant primary productivity is largely dependent on the recycling of nutrients, particularly if there is limited nutrient input via other pathways. The release of nutrients through decomposition also plays a key role in sustaining the bacterial population responsible for further degradation of the organic substrates that rapidly accumulate in the soil organic matter of such highly productive wetland environments (Mustafa and Scholtz, 2010). Plants in treatment wetlands undergo a life-death-litterfall-decomposition cycle which results in the formation of new soils and sediments, immobilisation of nutrients and growth of biomass (Mitsch and Gosselink, 2007). Waterlogging of wetland soils limits oxygen diffusion into sediment profiles, creating anaerobic conditions that slow decomposition rates, leading to the buildup and storage of large amounts of organic carbon in wetland sediments (Page and Dalal, 2011). This forms a critical part of the biogeochemical cycling in wetlands and is an essential feedback loop which, through slow decomposition processes, allows wetlands to be important carbon sinks. In general, wetland plants grow at a faster rate than they decompose, contributing to net carbon accumulation and burial (Page and Dalal, 2011).

The anaerobic digestion (decomposition) process happens through the concerted action of a community of bacteria and archaea that metabolise in the absence of oxygen (Angelidaki, 2003). These organisms live in a parallel series symbiotic relationship that can be grouped into four main steps: hydrolysis, acidogenesis, acetogenesis and methanogenesis (Speece, 1983). In short, complex organic matter is hydrolysed by fermentative bacteria to simple short chain molecules that are used by methane producing archaea to produce methane and carbon dioxide (Speece, 1983). The anaerobic digestion process in wetlands produces biogas (swamp gas) that has an average content of 55-65% methane and 35-45% carbon dioxide (Chynoweth, 2001). It is through this decomposition process that the nutrients within organic biomasses are converted into a form available for uptake by vegetation, thereby exercising a critical control on vegetation productivity and cycling of nutrients (Mustafa and Scholtz, 2010). By converting and transforming inorganic forms of these elements to organic forms, wetlands can thus effectively store nutrients and reduce the risk of eutrophication of downstream environments (Mitsch and Gosselink, 2007).

On a global or broad regional scale, temperature and precipitation are largely responsible for determining the rate and extent of decomposition (Baker *et al.,* 2001). Generally, warmer temperatures and higher precipitation result in higher rates of decomposition, faster litter turnover, and less organic matter accumulation. At a local scale, the rate of decomposition and quantity of nutrients cycled through the decomposition mechanism are influenced primarily by:

- the quality of the resource being decomposed;
- local physicochemical properties (i.e. temperature and moisture regime, pH, oxygen) that affect the microorganism population composition and structure; and
- the length of time the resource is in contact with the soil microenvironment (Baker *et al.,* 2001).

Carbon sequestration in wetlands is a consequence of the carbon dioxide taken in from the atmosphere assimilated into organic carbon as biomass and then transferred or accumulated into a wetlands soil organic matter (Villa and Bernal, 2017). A wetlands ability to sequester carbon is the product of a complex balance between wetland carbon inputs and outputs (Figure 3), which are direct product of a given wetlands hydrogeomorphic type, age, landscape position, climate and environmental condition (Pester *et al.,* 2012). Carbon inputs into wetland ecosystems are predominantly derived from carbon contained in the organic matter of fallen and dead vegetation in and around the wetland ecosystem, as well as dissolved and suspended carbon received from inflowing waters and runoff. Subsequently, a portion of the carbon outputs consist of dissolved and suspended particulate organic carbon in outflowing waters, although the loss of carbon from wetlands predominantly occurs through the release of inorganic carbon emitted as carbon dioxide and methane during the mineralisation of organic matter during decomposition (Villa and Bernal, 2017).

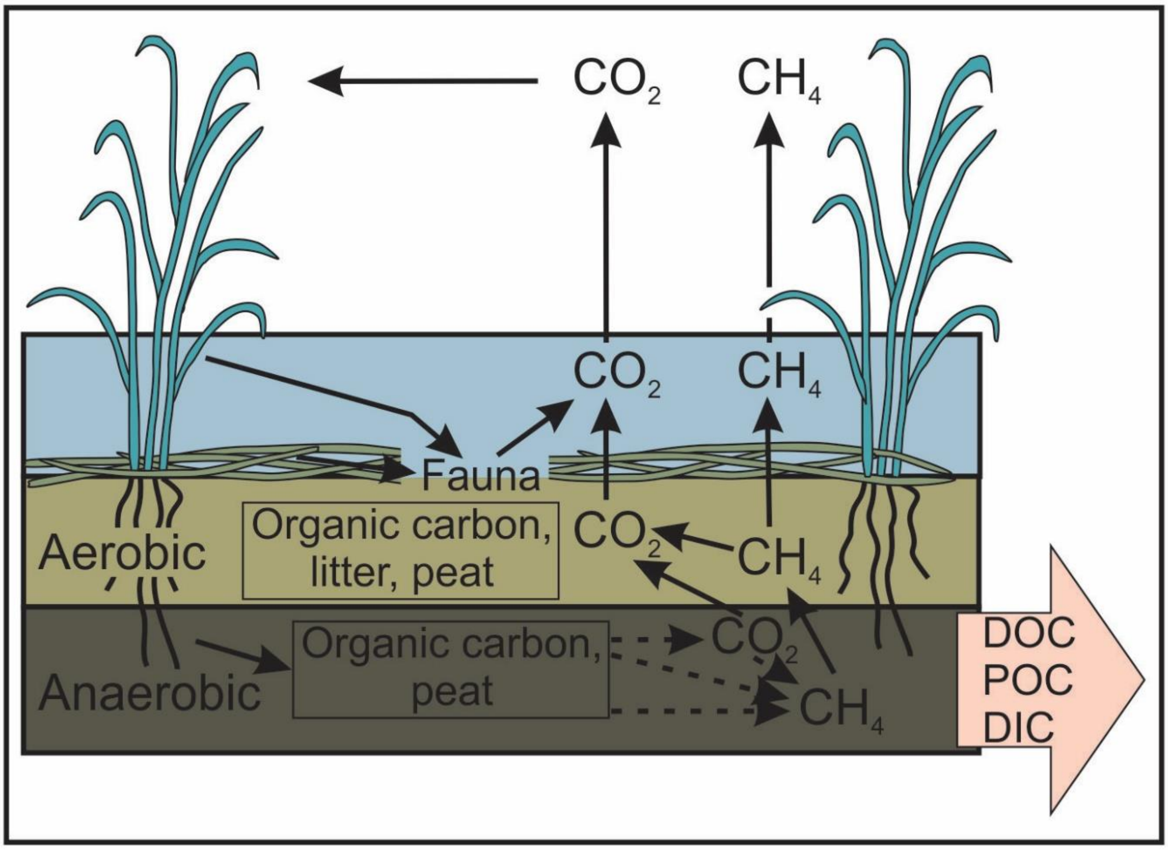

Figure 3: Carbon cycling in wetlands (adapted from Lloyd *et al*. 2013). Abbreviations: DOC = dissolved organic carbon, POC = particulate organic carbon, DIC = dissolved inorganic carbon.

The balance between carbon inputs and outputs is a complex interaction with various feedback effects (Figure 3). For example, high carbon inputs into a wetland ecosystem can lead to high carbon accumulation but, can also result in high gas emissions. As a result, the carbon cycle in wetlands cannot be understood without comprehending the factors that affect decomposition and hence mineralisation of organic carbon (Villa and Bernal, 2017). This reduced breakdown of plant material is characteristic of anaerobic conditions and is a consequence of low levels of microbial activity. However, due to the substantial differences in plant and microbial community structure as well as hydrological dynamics such as frequency and extent of flooding, generalisations or the application of models to de-compositional processes is difficult (Baker *et al*., 2001).

The process of organic matter decomposition in wetland ecosystems is carried out by microbes through microbial biomass accumulation and metabolism (Fennessy *et al.,* 2008). These microbes obtain energy from the oxidation of organic compounds using electron acceptors (Villa and Bernal, 2017). In order to oxidise such organic compounds, enzymes are produced by the microbes that aid in the degradational process. Organic matter substrates are also highly variable in their resistance to degradation by soil microbes such that certain substrates are easily degraded while others are more resistant, requiring more energy and nutrients (Villa and Bernal, 2017). Such factors result in the accumulation of more resistant organic compounds that microbes cannot easily degrade further, resulting in the accumulation of degradation resistant soil organic matter (Fennessy *et al.,* 2008). This organic matter and the carbon contained within it are buried within the wetland soils and thus represent a carbon sink.

The main factors responsible for reducing organic matter decomposition and thus promoting carbon accumulation are: limited nutrient availability for microbial growth, physical protection through the formation of aggregates and high content of organic compounds with low degradability (Villa and Bernal, 2017). However, biogeochemical cycles in wetlands are coupled. For example, sulfate reduction directly influences the carbon cycle in wetland ecosystems, limiting methane production (Pester *et al*., 2012). Organic carbon degradation in wetlands is catalyzed by various functional groups of both anaerobic and aerobic microorganisms, whose competition or co-operation for electron donors, determines if carbon is exported from a wetland as either methane or carbon dioxide (Pester *et al*.,2012). Sulfate reducing microorganisms are known to be metabolically versatile and can use several different substrates ranging from hydrogen to amino and short chained fatty acids (Nedwell and Watson, 1995). Sulfate reducing microorganisms are also energetically favored when competing for substrates involved in the methanogenic degradation pathway of anaerobic decomposition processes. As a result, a considerable portion of the carbon flow is diverted from methane to carbon dioxide production.

2.12 Factors affecting the water balance of FWS wetlands

A key to quantifying the effectiveness of the Crossways Farm Village wetland in removal of nitrogen and phosphorus required the use of a mass balance approach, which requires understanding the water balance of the wetland.

Wetland hydrology is widely recognised as the primary controlling factor affecting the formation, size and persistence of wetland environments (Maltby, 2009). The hydrological budget known as the ‘water balance’ is the term used to describe the accounting of inflowing and outflowing water sources within a wetland system (Gilvear *et al*., 1992). There are several components that comprise a wetlands water balance, which can be separated into inflows in the form of rainfall, surface water inflow and groundwater inflow, and outflows as evapotranspiration, surface outflows and groundwater outflows. The balance between these wetland inputs and outputs determines the hydroperiod or hydrological signature of a given wetland (Mitsch and Gosselink, 2007). Seasonal variability in the hydrology can limit or enhance the movement of sediments and nutrients in a given wetland system. The wetland biota may respond with significant changes in species composition, distribution and richness, which affect the overall productivity of the wetland (Mitsch and Gosselink, 2007). Small changes in the hydrology of a wetland may, therefore, have a profound effect on the wetlands hydrochemistry and physiochemical environment, influencing the overall ability of the system to deal with and treat nutrients in wastewater.

A knowledge of the hydrology of a treatment wetland and quantification of the water inputs and outputs are therefore two prerequisites to analysing the treatment efficiency of a given wetland system (Gilvear *et al*., 1992). The characteristics of individual wetlands vary considerably and are often determined by the nature of exchanges and fluxes of water within a wetland, which are related to seasonal climatic variability (Gilvear & Bradley, 2009). As a result, calculating the water balance of a wetland can be complex, and there is often considerable variability (Kadlec and Wallace, 2009). Most of the water balance studies in wetlands have thus been conducted on relatively simple systems with distinct inlet and outlet structures (Owen, 1994).

2.12.1 Dominant hydrological variables in FWS wetlands

2.12.1.1 *Rainfall*

Given that in continental areas all surface water is ultimately derived from rainfall, wetland environments are directly and indirectly sustained by the contribution of rainfall inputs (Ellery *et al*., 2009). However, for the purposes of this component of the water balance, the interest is solely in rainfall that falls on the wetland surface. Rainfall may result in significant additions of water into FWS wetlands and may dictate the effectiveness of a given treatment system. Large scale rainfall events may suddenly raise wetland water levels, resulting in significant outflow events as the wetland empties to the appropriate water level given the slope of the wetland from inlet to outlet. In such circumstances, stored nutrients and pollutants may be resuspended and transported out of the system, resulting in deceased treatment efficiency. FWS consequently need to be designed to deal with pulsed flows associated with such climatic events and atmospheric processes (Van Deun, 2015). Analysing rainfall inputs is thus a critical component of the water balance of FWS wetlands.

Rainfall and climate data can easily be acquired at a regional scale from online databases for broadscale applications. However, according to Kadlec and Wallace (2009), rainfall amounts should be measured at or near the wetland site as some rainfall events may be extremely localised. Rainfall is measured in millimetres, which is easily converted to litres per square metre, and is typically recorded using a tipping bucket rain gauge connected to an automatic weather station that is located on site (Beebe *et al*., 2014; Clulow, 2007; Owen, 1994).

2.12.1.2 *Evapotranspiration*

Evaporation is the process whereby liquid water is converted to water vapour and removed from the evaporating surface (Allen *et al*., 1998). In order to change the state of water molecules from a liquid to a vapour, latent energy is required. Solar radiation, air and water surface temperatures are the sources of this energy (Kadlec and Wallace, 2009). Air movement and/or wind enhances water vapour removal from the wetland surface into the surrounding atmosphere as it increases the atmospheric water pressure deficit (Kadlec and Wallace, 2009).

Transpiration is the process of vaporisation of liquid contained in plant leaves through small openings in the leaf called stomata (Allen *et al*, 1998). Water taken up by plant roots is transported into the leaves and transpiration serves to cool the emergent portions of the plant from continuous heat energy received in the form of solar radiation (Kadlec and Wallace, 2009). Transpiration also allows plant nutrients taken up from the soil by plant roots to reach sites where they are needed in metabolic reactions. Transpiration rates are controlled by water availability in the substrate, plant characteristics and the climatic parameters controlling evaporation (Allen *et al.*, 1998).

Total evaporation is a combination of evaporation and transpiration, collectively termed evapotranspiration (ET; Beebe *et al.,* 2014). In nature, evaporation and transpiration occur simultaneously and it is difficult to separate the two components. Evaporation from the soil or water surface dominates in the early stages of development of a wetland, while transpiration begins to dominate later when the canopy begins to shade the water surface (Kadlec and Wallace, 2009).

A common assumption in wetland science is that evapotranspiration (ET) may be represented by the reference crop (ET_o) computation (Kadlec and Wallace, 2009; Allen *et al*, 1998; Campbell Scientific, 1999). In order to produce a consistent and universally applicable method for calculating evapotranspiration, the Environmental and Water Resources Institute (EWRI) of the American Society of Civil Engineers (ASCE) established a universal evapotranspiration equation that standardises the wide variety of calculations for reference evapotranspiration (Clulow, 2007; Allen *et al*., 2000). The equation calculates reference evapotranspiration (ET_o) using the Penman-Monteith equation and uses four main meteorological inputs (Clulow, 2007) in the form of daily solar radiation ($MJ.m^2.d^{-1}$), air temperature (^oC), wind speed ($m.s^{-1}$) and relative humidity (%). Therefore, these climatological parameters are considered essential when assessing evapotranspirational water loss in wetlands.

Reference evaporation is the total evaporation from a reference surface and is related to the transpiration of various crops by means of a crop factor where the crop is not short of water or nutrients and fully covers the soil surface (Brown, 2007). The concept of reference evaporation allows the study of the evaporative demand of the atmosphere independently of the crop type, crop growth, management practices and soil factors (Brown, 2007). This allows for the comparison of evapotranspiration from different locations or different seasons as long as they refer to the same reference surface. Given that the main factors affecting *ET* are climatic parameters, it can be computed from meteorological data alone (Clulow, 2007). An open water surface was suggested in the past as a reference surface but this is very different to a typical land surface with plants. Therefore, it was felt that relating evapotranspiration to a specific crop would incorporate the biological and physical processes more realistically (Clulow, 2007). Short grass that fully covers the surface and is not short of water (or nutrients) is an extremely well-studied surface and has now been accepted as a universal reference surface (Allen *et al.*, 1998).

A number of studies show that the Penman-Monteith form of the evapotranspiration combination equation consistently outperforms others (Campbell Scientific, 1999; Clulow, 2007). This equation includes more of the factors that influence plant water loss than any of the other equations and is, therefore, assumed to provide a more accurate estimate (Clulow, 2007).

Wetland size has a significant influence on a given wetland's water temperature (Kadlec and Wallace, 2009). In large wetlands, water loss via evapotranspiration is primarily driven by the amount of solar radiation reaching the wetland surface, whereas in small wetlands, evapotranspiration is significantly augmented by heat transfer from the surrounding air (Kadlec and Wallace, 2009). Smaller wetland's, therefore, have far higher water temperatures and thus evapotranspirational water loss is often amplified in such systems.

2.12.1.3 *Surface water inflow and outflow*

Surface water may be permanently, seasonally, or temporarily present in wetlands, depending on the local climate, season, geology, topography and ecology of the system (Mitsch and Gosselink, 2007). Two major features of constructed wetlands are responsible for controlling surface water depths and flow rates, namely; the structure of the wetland's inlet/s and outlet/s and resistance to flow within the wetland (Kadlec and Wallace, 2009). Most FWS treatment wetlands are therefore designed with some type of inflow and outflow control structures in order to minimise variation in the wetlands water level, thereby enhancing the wetland’s wastewater treatment potential (Kadlec and Knight, 1996). Wastewater treatment facilities generally measure and record wastewater inputs using inline flow meters. Alternatively, in the absence of a point input source, pressure transducers combined with a confining structure or weir may be used in the same way that surface water outflow is typically measured (Kalbus *et al*., 2006).

Treatment wetlands are usually fed by a fairly constant supply of wastewater that sustains surface water flows during dry periods. During wet periods, rainfall may contribute significant amounts of surface water into the wetland system through direct rainfall on the wetland, overland flow (non-channelised sheet flow) and through interactions between groundwater and surface water, given that ground water discharged into wetlands becomes surface water (Kalbus *et al*., 2006). Surface water may flow in channels or across the surface of a wetland. Flow paths and the velocity of water movement over the surface of a wetland are affected by the gradient between the inlet and outlet structure and resistance to flow (wetland design and vegetation height and density) within the wetland (Van Deun, 2015). Several mathematical descriptions based on utilisation of mass, energy and momentum conservation equations, combined with frictional resistance equations such as Manning's equation, can be used to calculate surface water flow in wetlands (see Kalbus *et al*., 2006). However, recent advances have rendered much of the earlier work on FWS constructed wetland surface water flow rates, based on Manning's formula as incorrect, as the equation is intended to calculate turbulent flows, whereas FWS wetlands almost always have a laminar flow regime (Mencio *et al*., 2014). Recent research has shown that the “laminar flow friction”, or Kadlec's equation, is most accurate when calculating surface water flows or hydraulic retention times of FWS wetlands (Van Deun 2015).

2.12.1.4 *Groundwater inflow and outflow*

Ground water originates as rainfall and through seepage from surface-water bodies (Maltby, 2009). Surface water slowly filters downward through unsaturated soils and rocks until it reaches the saturated zone. This process is known as ground water recharge (Gilvear *et al*., 1992). Ground water flows through permeable rocks and other earth materials in response to the hydraulic head or pressure difference created by the movement of water within the bedrock pores (Kadlec and Wallace, 2009). The hydraulic head of the groundwater can also cause water to move back to the land surface and into surface water bodies in a process known as groundwater discharge (Gilvear *et al*., 1992). Groundwater discharge occurs through wells, seepage, springs and directly through evapotranspiration where plant roots reach the water table, or where the water table is near the land surface (Mitsch and Gosselink, 2007).

If a given wetland's water level is higher than the water table of its surroundings, water will flow out of the wetland and into the water table (known as a recharge wetland). Conversely, groundwater inflow into wetlands occurs when the surface water level in the wetland is lower than the water table of the surrounding land (known as a discharge wetland; Mitsch and Gosselink, 2007). The elevation of the water table and hydraulic head ultimately dictates the exchange between surface and groundwater in wetlands (Van Deun, 2015).

Most natural wetlands occur in areas where soils have low permeability, allowing for the persistence of surface water (Maltby, 2009). Typical FWS constructed wetlands are often designed and built with an impermeable plastic liner or clay bottom, in order to minimise such surface water loss and prevent contamination of groundwater (Kadlec and Wallace, 2009; Van Deun, 2015). FWS wetlands therefore closely resemble surface water depression wetlands, which are dominated by surface runoff, precipitation and evapotranspiration, with little or no groundwater outflow (Mitsch and Gosselink, 2007). Groundwater recharge and discharge may however occur following heavy rainfall events that may significantly raise the water table and hydraulic head of the groundwater surrounding a wetland.

Given the complexity involved in accurately quantifying groundwater loss in FWS wetlands, most studies typically account for groundwater losses using the water balance or water budget approach (Van Deun, 2015). The underlying framework of this approach is that any gain or loss of surface water can be related to the water source. If evapotranspiration is calculated and rainfall and surface inflows are measured, the groundwater component can be calculated by difference (Kalbus *et al*., 2006).

Many different methods can, however, be used to calculate surface and groundwater interactions (Kalbus *et al*., 2006). Each method differs in resolution and in timescales they represent, with varying levels of accuracy created by the spatial and temporal variability in ground and surface water exchanges, and because individual measurements need to be integrated at various scales (Mencio *et al*., 2014). The most accurate representations therefore, need to include an analysis of water exchanges at multiple scales within a single study site (Mencio *et al*., 2014).

Chapter 3: Study Area

3.1 Location and topography

The Crossways Farm Village Estate is situated in the Eastern Cape province of South Africa on the border between the Kouga and Uitenhage District Municipalities. Located 43km west of Port Elizabeth (33° 55’ 45” S; 25° 10’ 54” E; Figure 4), the study area comprises the eastern most extension of the Cape Floristic Region, near the small town of Thornhill. The Kouga district covers an area of approximately 2400km^2 and is characterised by a diverse combination of geology, soils, topography and climate which varies substantially over short geographical distances (Haigh *et al.,* 2002). The study area has physiographic elements that are more typical of the southern Cape coastal region, namely the quartzite mountains of the Cape Folded Belt and an undulating coastal foreland that stretches from the mountains to the coast (Cowling, 1984). The wide alluvial valley and braided course of the Gamtoos River is another distinctive topographical feature of the Kouga area, and comprises a fertile floodplain that supports intensive agricultural activity (Lewis, 2008).

The most prevalent topographical feature of the region is however the Great Winterhoek mountain range, which reaches an elevation of 1 750 meters above sea level (m.a.s.l), of which the Cockscomb peak is a significant feature (Cowling, 1984). This range comprises the eastern end of the inland Swartberg-Baviaanskloof mountain range axis, which runs parallel to the Kouga mountain range in an east-west orientation. East of the Gamtoos River lies a smaller mountain range with a more south-easterly trend, known as the Elandsberg Mountains. The other mountain range in the region is the Van Staden’s Mountains to the north of the study site, with an elevation in the region of 300 – 600 m.a.s.l.

The coastal foreland is comprised of a plain that cuts across all geological formations (Cowling, 1984), which occurs within 5km of the coastline and extends to the foot of the Van Staden’s Mountains. Towards the mountain footslopes the plain is characterised by gravel-covered interfluves above deep and narrow V-shaped incised valleys. Away from the mountains, the plain is more extensive with large areas covered by a layer of limestone and marine gravels. Most of the drainage networks in the study area display a trellis drainage pattern due to the precipitous topography and orientation of the various mountain ranges (Haigh *et al*., 2002; Lewis, 2008).

The coastal region of the Kouga district stretches from the Van Staden's River in the east to the Tsitsikamma River in the west and is characterised by a series of half-heart shaped bays, formed due to the seaward protrusion of erosion resistant quartzite lithologies and the greater susceptibility to erosion of softer sedimentary rocks (Cowling 1984). Four major tidal rivers drain the landscape and enter the Indian Ocean within the Kouga region. The Kromme, Kabeljous and Seekoei Rivers, which, in contrast to the Gamtoos River, flow in strike valleys that are deeply incised into the coastal plain. The Van Staden's River drains the catchment in which Crossways Farm Village is situated.

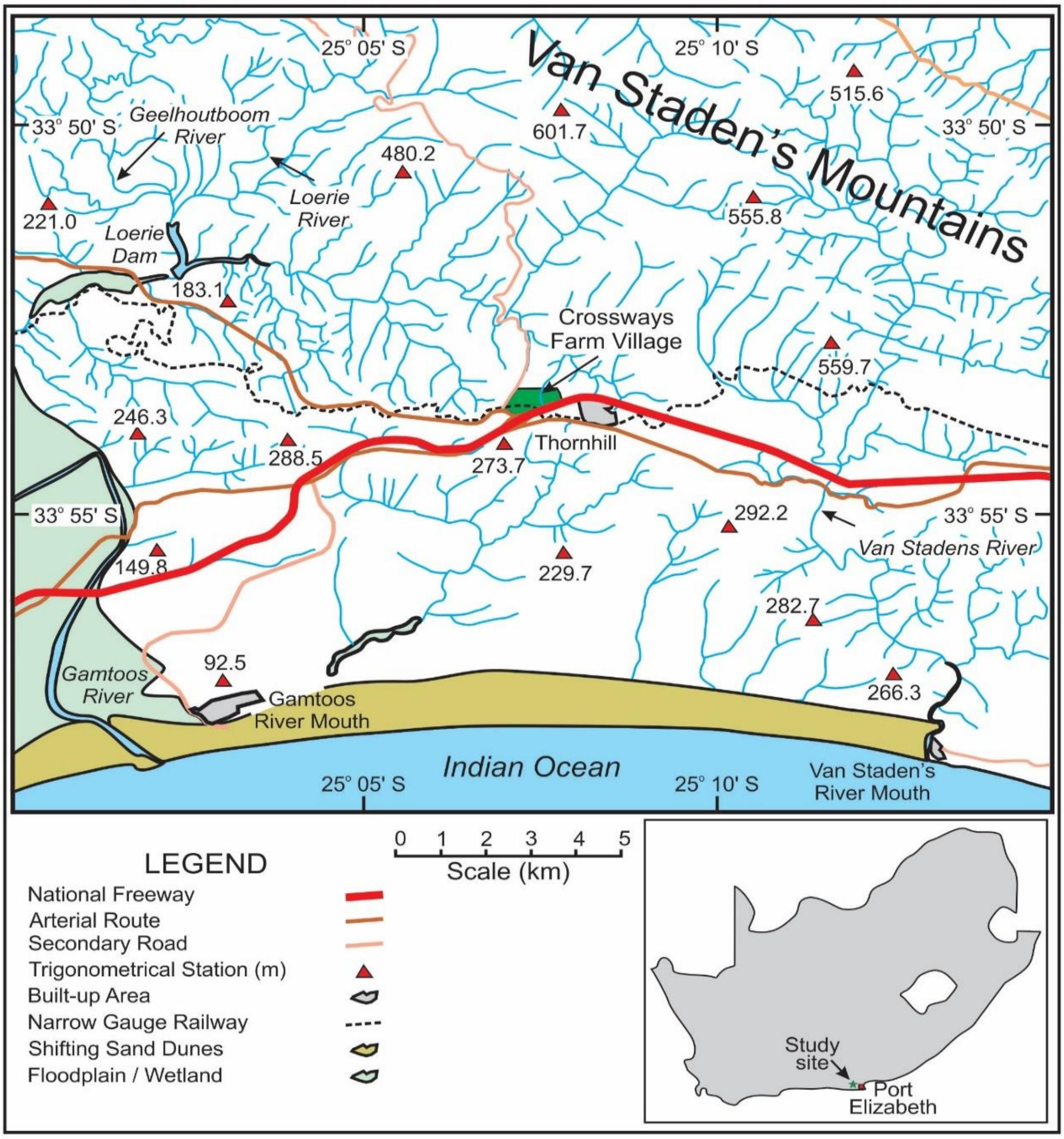

Figure 4: Map showing location of study site, including topographical and landscape features of the surrounding area. Modified from Chief Directorate National Geo-spatial Information sheets 3324 DD and 3325 CC (2011).

3.2 Climate

The study area lies at the junction between the temperate and subtropical climatic regions, with the average daily temperatures ranging from 25°C in summer to 12°C in winter (Mucina and Rutherford, 2006). The Kouga region tends to display a bimodal rainfall pattern, although rainfall occurs throughout the year. Annual average rainfall ranges between 500mm and 800mm (Figure 5) with low intensity orographic rain predominating year-round (Cowling, 1982; Cowling and Holmes, 1992). Pre-frontal thunderstorms are occasional in summer although the spring and summer months (September – February) are usually the driest. On average, October has the most rainfall days with January having the least. January has the highest mean temperature (23°C), with June and July having the lowest (11°C) overnight average temperature (Cowling and Holmes, 1992). Mean annual temperatures are mild (17-18°C) and relatively uniform on the coastal plain, and cooler in the high-lying mountain areas (15-16°C). Highest absolute maxima (40°C) are recorded in autumn and spring and are associated with hot berg wind conditions.

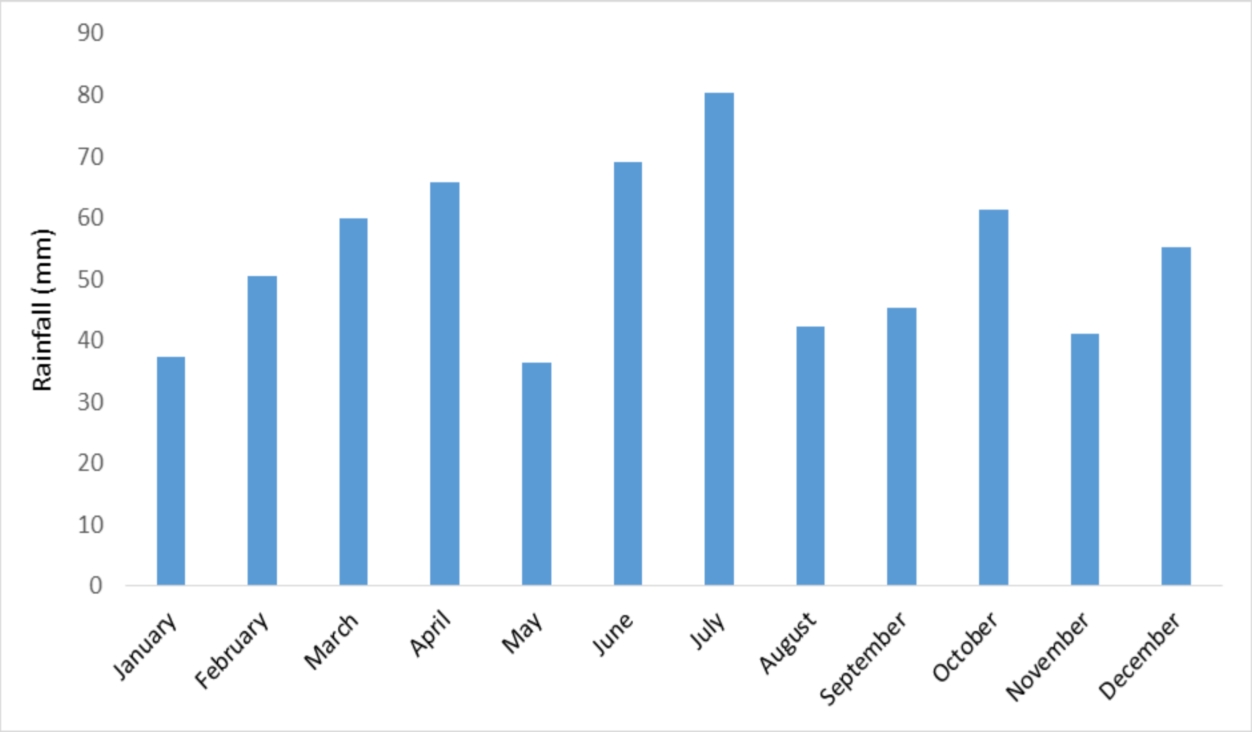

Figure 5: Mean monthly rainfall for Humansdorp from January 2009 to December 2016 (S.A.W.S, 2017)

The area's complex topography however, results in highly variable rainfall patterns within the Kouga region. For example, on the Gamtoos valley floor, annual average rainfall is approximately 400mm, which is in close proximity to the upper slopes of the Elandsberg mountains west of the Van Staden's Mountains, which receives an average annual rainfall more than 1000mm (Cowling, 1984).

3.3 Geology

The geology of the Kouga region is dominated by rocks of the Cape Supergroup and predominantly comprises Table Mountain and Bokkeveld Group sediments of shale, siltstone and quarzitic sandstones (Mucina and Rutherford, 2006). Table Mountain Group rocks comprise the mountain ranges, but they also underlie considerable areas of the coastal plain. The high lying ridges of the various mountain ranges comprise massive, well jointed quarzitic sandstones, that have been folded and metamorphosed such that they are highly resistant to weathering and erosion (Cowling, 1984). In contrast, the shales and siltstones are more easily weathered and eroded, and therefore predominate the valley floors, floodplains and coastal foreland (Cowling, 1984). Bokkeveld beds composed of soft yellow, grey and greenish shales, can also be found in a narrow wedge on the coastal plain. The Gamtoos River valley comprises a combination of Bokkeveld, Table Mountain and Uitenhage Group rocks.

3.4 Vegetation

The Kouga district is characterised by a unique suite of environmental conditions that has created a remarkably high diversity of vegetation types within the region (Mucina and Rutherford, 2006; Cowling and Potts, 2015). The study area comprises the eastern most extension of the Cape Floristic Region, which is internationally recognised as one of the six floral kingdoms of the world and is one of South Africa's eight World Heritage Sites. The Cape Floristic Region (which has its epicentre in the Western Cape), is known for the extreme floral diversity and endemism, comprising over 9000 plant species of which 70% are categorised as endemic (Cowling and Potts, 2015).

According to Cowling and Potts (2015), the Kouga portion of the Cape floristic region is predominantly comprised of patches of the Fynbos, Renosterveld, Thicket, Forest and Grassland biomes, which are largely controlled in their distribution by soil fertility and moisture regime of the various topographical areas. The Cape Floristic Region is however, largely dominated by the fynbos biome (80.5%). The beta and gamma diversity of the Fynbos Biome decreases noticeably from west to east, such that there is relatively low species richness in the fynbos occupying the Eastern Cape (Cowling and Heinjnis, 2000; Mucina and Rutherford, 2006). Nevertheless, there are several vegetation types within the Kouga region that are characterised as severely threatened and vulnerable due to the widespread disturbances produced by overgrazing and frequent clearing and burning of natural vegetation for agricultural purposes, in combination with the widespread infestation of alien invasive species such as black wattle (*Acacia mearnsii*; Mucina and Rutherford, 2006).

3.5 Land-use and Socio-Economic Characteristics

The Kouga Municipality covers the towns of Jeffreys Bay, Humansdorp, St Francis Bay, Cape St Francis, Oyster Bay, Patensie, Hankey, Loerie and Thornhill. According to the 2011 Census, the region has a total population of 98 559 with an estimated growth rate of 2.4% per annum (DEDEA, 2011). Approximately 70% of the total population lives in urban areas with Humansdorp representing the regional service centre as it supplies the surrounding coastal towns and agricultural communities with services and commodities.

The Kouga district is an important agricultural region in the Eastern Cape and has a long history of intensive agriculture, pasture and dairy farming dating back to the times of European settlement (Rebelo, 2013). The fertile floodplain of the Gamtoos River is one of the major agricultural areas in the region, with the surrounding towns of Loerie, Hankey, Patensie and Thornhill serving as the focal points of this highly intensive agricultural region (DEDEA, 2011). The region surrounding the town of Humansdorp is also an iconic dairy and beef region in the Kouga district and is the largest supplier of milk in South Africa, providing an important income for the Kouga region and Eastern Cape as a whole (DEDEA, 2011). While the study area has traditionally been characterised by economic activities largely focused agriculture, tourism has become an increasing source of income in the study area.

The coastal towns of Jeffreys Bay, St Francis Bay, Cape St Francis and Oyster Bay, are well established tourist destinations, largely based on water related recreational activities such as surfing and fishing. In addition to the world renowned surfing events such as the Billabong Pro, which attracts thousands of tourists, the coastal towns of the Kouga region are extremely popular holiday destinations, particularly over the festive season. The Integrated

Development Plan (IDP) for the region has thus placed tourism at the forefront of local economic development strategies that aim to capitalise on the massive influx of tourists to the coastal region of the Kouga district (DEDEA, 2011).

The Kouga region has also positioned itself as one of the major energy hubs in southern Africa. Two cluster wind farms are in operation on the outskirts of Jeffreys Bay/Humansdorp and Oyster Bay, which are among the largest wind-energy producing areas in South Africa, providing the Eskom gridline with over 460 000MWh per annum (Hosking *et al*., 2015). Furthermore, Thuyspunt (between the coastal cities of Oyster Bay and Cape St Francis) has been selected by Eskom as a possible site for the construction of a thermal nuclear power station, and the Environmental Impact Assessment and planning process is currently underway.

3.6 The Crossways Farm Village – Research study site

The Crossways Farm Village Estate covers an area of approximately 560ha and land use on the property is mainly dairy farming, pasture production and residential housing (currently 50-60 households). The property has a long history of intensive dairy farming and irrigated pasture production. Large portions of the farm have historically been cleared for agricultural purposes and the area is severely infested with dense stands of eucalyptus and black wattle trees, with very little of the natural vegetation intact.

The research study site (Crossways Farm Village wastewater treatment works), comprises a combination of generic wastewater treatment techniques with a FWS constructed wetland acting as a tertiary wastewater treatment system. The treatment system relies on a primary anaerobic treatment phase followed by an aerobic phase. The secondary phase involves reducing suspended solids in a clarifier. Wastewater treatment ends in three interconnected wetland cells (Figure 6) that form the FWS constructed wetland examined in this study. Each component of the treatment system is designed to complete a specific role in the treatment process to ensure that National Water Quality Guidelines and legal discharge requirements are met by the three-phase wastewater treatment system. Domestic wastewater from approximately 50-60 households is pumped into an anaerobic reactor, which digests and filters the solid matter. It is then oxidised using a rotating biological contractor (RBC) and gravity- fed into a clarifier before entering the FWS wetland.

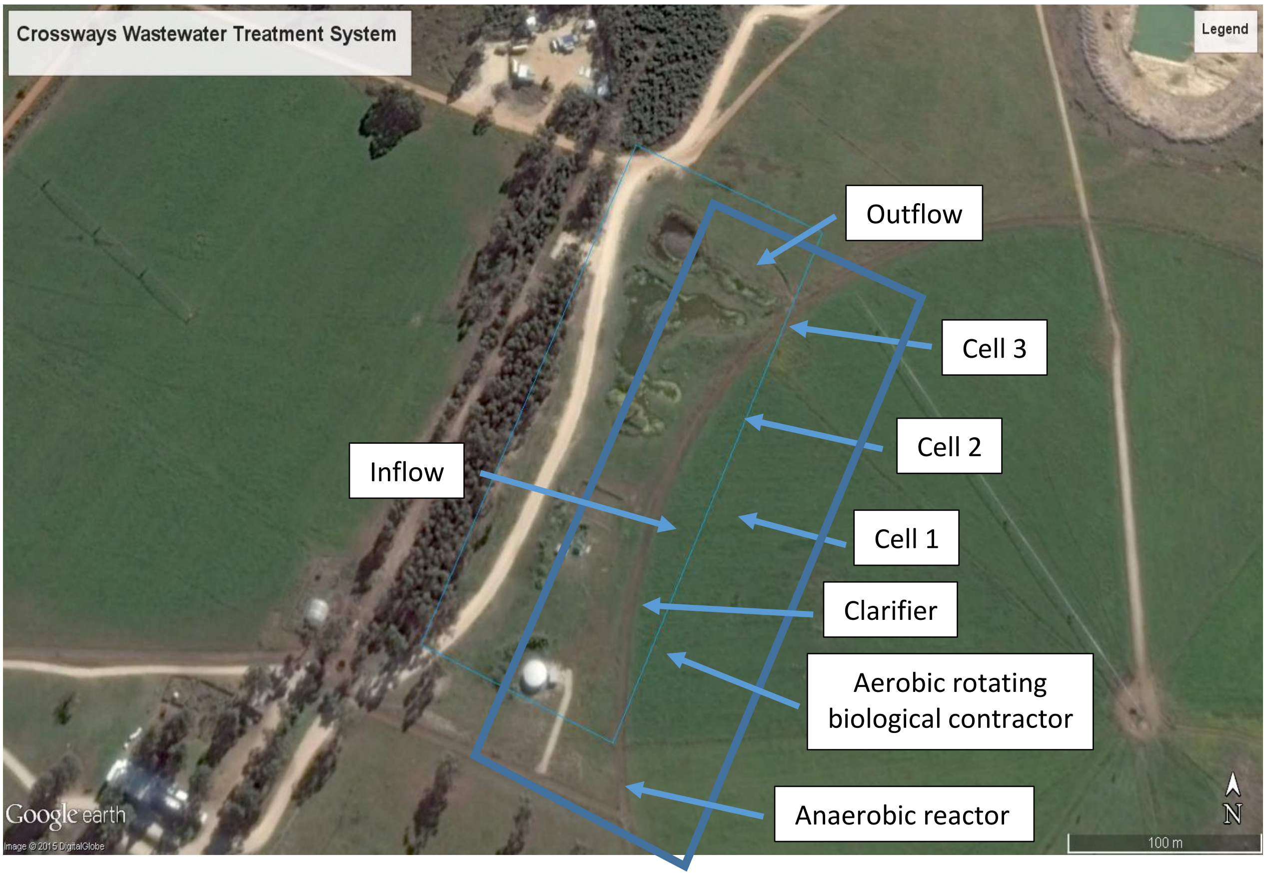

Figure 6: Google Earth image showing research study site and surrounding features

The research study site exists on a remarkably flat plain that stretches for approximately 2km^2 south and 1.5km^2 north of the treatment wetland system. The FWS constructed wetland system is flanked on either side by artificial features that isolate the wetland and prevent surface runoff entering the treatment system. To the west of the wetland system is a raised road with drainage furrows on either side that stretch the length of the study site. To the east lies a deep drainage furrow that surrounds the pivot circle irrigation scheme and runs along the length of the wetland. Throughout the duration of the study period, no crop production or irrigation took place, with all irrigation infrastructure removed and/or dismantled. In order to increase the volume of water that can be stored in the treatment wetlands, the edges of each oft the wetland cells have been raised such that a plastic liner could be lifted to increase the maximum retention level of the wetland system in the event of flooding or increases in wastewater inputs as the development expands.

Primary treatment – Anaerobic reactor and rotating biological contractor (RBC)

RBC’s are widely used in conjunction with anaerobic reactors (Kadlec and Wallace, 2009). The anaerobic reactor precedes the RBC and serves to remove, retain and partially stabilise floatable and settle-able solids introduced with the wastewater and sludge from the sedimentation tank (Van Deun, 2015).

The RBC unit is a biological treatment system, which consists of a large number of circular discs spaced uniformly along the length of a horizontal shaft. The shaft is mounted across a tank above the water level such that approximately 40% of the disc area is submerged in the wastewater. A biomass film develops on the discs as the contractor slowly rotates. A film of wastewater attaches to the biofilm and is carried through the air resulting in the aeration of the anaerobic wastewater exiting the anaerobic reactor.

Secondary treatment – sedimentation tank

After being aerated by the RBC, wastewater is gravity-fed into a clarifier (Dortmund type sedimentation or humus tank), where it is screened to produce a clear effluent that feeds via an overflow weir into the FWS wetland.

Free Water Surface wetland – free water surface constructed wetland

The Crossways Farm Village FWS wetland is designed with a highly irregular shape and it serves as the tertiary and post wastewater treatment phase for the domestic wastewater before it is released to the environment. The design ensures maximum residence time of the water entering the wetland due to the sinuous nature of flows. The low flow velocity is further maintained by dense stands of *Typha capensis* that dominate each cell. Several other aquatic plants such as oxygen weed, water lilies and various macrophyte species such as papyrus are present within the treatment wetland, although *Typha* stands comprise approximately 80% cover, particularly in the final compartment of the treatment wetland. The FWS wetland covers an area of approximately 0.4ha. Water depth across the bed of the wetland is generally less than 1m deep, although the wetland beds are markedly undulating with one island present in each wetland compartment. Furthermore, the Crossways Farm Village wetland system exists above a clay layer and has been compacted with clay and lined with plastic sheeting in order to prevent wastewater from contaminating groundwater resources. Clay and gravel were then inserted into the wetland in order to allow for macrophyte establishment and to stabilise the plastic liner.

The FWS treatment wetland system was constructed at the start of 2013 and began receiving wastewater inputs in March 2013 following the completion of the housing estate. Therefore, the treatment wetlands had been in operation for approximately 157 weeks at the time of completion of this study. Over this period the Crossways Farm Village have received only a minor increase in the number of houses contributing wastewater to the treatment system. According to the landowner, no maintenance or alterations have been conducted in the treatment wetlands during this period. The wastewater treatment system is currently designed to treat up to 240kl. d^{-1}.

Chapter 4: Methods

Understanding a wetlands water balance and quantifying its effluent mass balance are two necessary prerequisites to analyse any treatment wetland's performance (Kadlec and Wallace, 2009). Several studies have used a mass balance approach within closed systems to demonstrate how effective constructed wetlands are at treating secondary wastewater (Acreman and Fisher, 2004; Breen, 1990; Chung *et al*, 2007; Lee *et al*, 2014; Samie *et al*, 2004), yet very few studies have attempted a mass balance on FWS wetlands due to the operational complexity. Calculating the effectiveness of the Crossways Farm Village FWS treatment wetland therefore necessitated the measurement of all water inflows and outflows, as well as inflowing and outflowing effluent nutrient concentrations and an estimation of wetland nutrient stores. Data collection for this study took place between April and September 2016.

4.1 Determining the water balance

The water balance of the Crossways Farm Village constructed wetland was determined as follows (Mitsch and Gosselink 2007):

$$\underline{\mathbf{\Delta S = P + S_i + G_i - ET - S_o - G_o}}$$

Where: S = storage,

P = precipitation

S_i = surface water inflows

G_i = groundwater inflows

ET = evapotranspiration

S_o = surface water outflows

G_o = ground water outflows

Given that the wetland cells were sealed with a plastic liner, groundwater inflows and outflows were absent. Diffuse surface inflows were also absent for the same reason. Therefore, the only surface inflow to the wetland was wastewater from the secondary treatment clarifier. All surface water outflows occurred northwards from the wetland via an artificial drain.

4.1.1 Surface water inflow and outflow

Installation of weirs at the inflow and outflow

V Notch weirs are simply a ‘V’ notch cut into a plate which is placed vertically so that it obstructs open channel flow, causing the water to flow over the V notch (Kadlec and Wallace, 2009). It is used to measure flow of water in the channel, by measuring the head of water above the base of the V notch, which is referred to as the hydraulic head or stage height. A formula can be applied to give a value for discharge from the stage height, with units expressed in volume per unit time. The relationship between stage height and discharge is known as the stage-discharge relationship.

The installation of two V notch weirs (one at the inflow and one at the outflow) was required in order to quantify surface water inflows and outflows to and from the FWS constructed wetland system respectively. The weir at the inlet to the constructed wetland needed to be cut and designed to fit inside a manhole. The dimensions of the V notch also considered the installation of a pressure transducer upstream of the weir, with the base of the notch at a height of 22cm above the bed of the inlet furrow. The inflow to the drain that was accessed via the manhole received water via an overflow weir from the clarifier, which is gravity-fed into the inlet furrow.

The outflow weir structure was designed after analysing the depth and extent of the clay layer underlying the wetland. This was done by augering on either side of the furrow, measuring the width and depth of the furrow and determining the height of the notch relative to the flow bed of the drain plus transducer height. The outflow weir was placed into position using a Top Lift Bulldozer (TLB) and sealing the weir with clay (Figure 7). The weir was then sealed using the remaining clay and the furrows where reopened, allowing surface water to exit the final cell of the treatment wetland. Both the inflow and outflow weirs were installed on the 13th of March 2016. The weirs were then monitored and allowed to stabilize for a period of 18 days, before each pressure transducer was installed. A further two days were required in order to calibrate each pressure transducer before the study commenced in April 2016.

Figure 7: Installation of the V notch weir at the wetland outflow, with the steel sheet v-notch weir in position before opening the channel (a) and after opening the channel (b).

Each weir was fitted with a pressure transducer in order to measure changes in the water level, giving of the hydraulic head of water used in developing a stage discharge relationship for each weir. The pressure transducer outputs indicate the observed pressure which is then converted into the head of water and flow (Kalbus *et al*, 2006). Each sensor was calibrated by taking repeated manual measurements of the stage height of the water from the base of the V notch and comparing these results to measurements recorded on the datalogger.

Calculating surface water flows

Influent and effluent volumes were calculated by developing a stage discharge relationship. The dimensions of the influent weir V Notch (8cm width x 16cm height) were applied to a formula (Figure 8), using stage height, slope and angle of the 'V' notch, to give the total influent volume. Surface water outflow in the FWS wetland was calculated using a 15cm equilateral V Notch weir and a pressure transducer. The same formula as the influent calculations was applied to the wetlands outflow, allowing for the calculation of both surface water inflow and outflow.

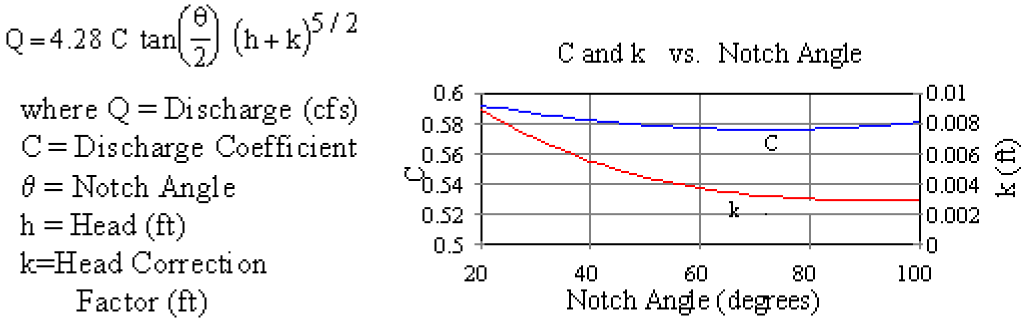

Figure 8: Formula used to calculate flow using V notch weirs (Van Deun, 2015)

4.1.2 Rainfall input and evapotranspiration output

An automatic weather station was installed on site with the primary purpose of providing an accurate estimate of evapotranspiration losses from the wetland and the input of water by rainfall (Figure 9). Weather (micrometeorological) data can be used to calculate a short grass reference evaporation (*ETo*) according to the American Society of Civil Engineers (ASCE)-Environmental and Water Resources Institute method, based on the original Penman-Montieth combination equation (Campbell Scientific, 1999). The Penman-Montieth method is internationally recognised and popular for a number of reasons, including the relatively low data requirements and the relationship established between the reference and the total evaporation, known as the crop factor (Clulow, 2007). This allows agronomists and hydrologists to estimate total evaporation from standard weather data (Clulow, 2007). Campbell Scientific has further incorporated this standardised and approved version of the Penman-Monteith equation into their calculations. The equation uses solar irradiance, relative humidity, wind speed and rainfall from micrometeorological data in order to give an estimate of actual evapotranspiration (Campbell Scientific, 1999).

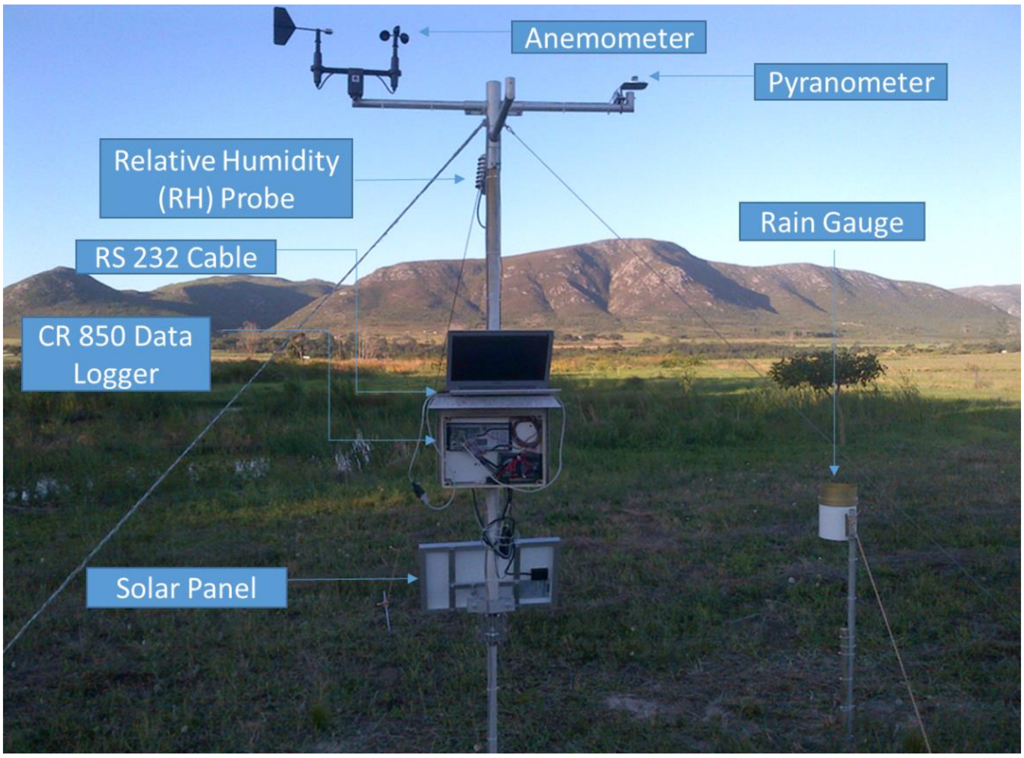

Figure 9: Components and sensors of the Automatic Weather Station

The Campbell Scientific CR850 Datalogger was programmed to include a calculation for evapotranspiration using the above-mentioned four climatic parameters (Table 1). The weather station included a wind sentry to measure wind speed, a sensor to measure wet and dry bulb temperatures to determine air temperature and relative humidity, a sensor to measure solar radiation and a tipping-bucket rain gauge to measure rainfall. A single value for evapotranspiration is expressed in hourly and daily time steps and stored on the datalogger. The Penman Monteith-based ASCE Standardised Reference Evapotranspiration (ASCE RET) Equation for the short reference surface (e.g. grasses) was employed to predict reference evapotranspiration (RET) on a daily time step. The ASCE RET equation is a form of the Penman-Monteith equation, developed to predict ET from standardised vegetative surfaces, based on available weather data and specified constants for the surface and aerodynamic resistance (Clulow, 2007). The ASCE suggests that this equation should be used as the standard method for obtaining ET results (Campbell Scientific, 1999). The equation for standardised daily reference evapotranspiration, ET_{sz} (mm d^{-1}), from Marasco *et* al. (2014) is given by:

$$ET_{sz} = \frac{0.408\Delta(R_n-G)+\gamma(C_n/T+273)u_2(e_s-e_a)}{\Delta+\gamma(1+C_d u_2)}$$

where:

Δ is the slope of the saturation vapor pressure-temperature curve (kPa $°C^{-1}$);

R_n is the calculated net radiation (MJ m^{-2} d^{-1});

G is the soil heat flux density at the soil surface (MJ m^{-2} d^{-1});

γ is the psychrometric constant (kPa $°C^{-1}$);

C_n is the numerator constant for the short reference type (900 K mms^3 Mg^{-1} d^{-1} for the daily time step);

T is the mean daily air temperature at 1.5 to 2.5 m height (°C);

u_2 is the mean daily wind speed at 2 m height (m s^{-1});

e_s is the mean saturation vapor pressure at 1 to 2 m height (kPa);

e_a is the mean actual vapor pressure at 1.5 to 2.5 m height (kPa); and

C_d is the denominator constant for short reference type (0.34 s m^{-1})

Table 1: Summary of the sensors and units used in calculating water inputs and outputs

Measurement	Sensors	Scan Rate (s)	Units and output interval
Wind speed	RM Young wind sentry		hourly and daily (m/s^{-1})
Relative humidity and air temperature	Vaisala HMP50	10	hourly and daily %; °C
Solar irradiance	Li 200SZ	10	hourly (W m^{-2}) and daily (MJ m^{-2})
Rainfall	TE 25	10	hourly and daily (mm)
Water level (Stage)	CS451Pressure Transducer/ Weir	10	hourly and daily (cm)

According to Marasco *et al.* (2014), the inverse of latent heat of vaporisation (2.45MJ kg^{-1}) multiplied by the density of water (1.0Mg.m^{-3}) is 0.408, the coefficient in the numerator of the above equation (Marasco *et al.*, 2014). Variables for the equation were defined from solar radiation, wind speed, air temperature, relative humidity, date, site elevation, and latitude according to the methodology required for daily time steps specified for the ET_{sz} calculation (Campbell Scientific, 1999). All environmental measurements were measured at the ASCE specified elevations above the ground surface. Net shortwave radiation or R_n was estimated from albedo and solar radiation based on the conversion table specified in the ASCE RET method. In accordance with this method, mean daily air temperatures (T), and saturation vapor pressure (e_s), were defined as the mean of the minimum and maximum hourly values (Campbell Scientific, 1999). The actual vapor pressure (e_a) was calculated hourly and averaged daily, as preferred by the ASCE methodology. This technology allows one to generate an accurate estimate of the amount of water lost from the wetland as a result of evapotranspiration with minimal computational difficulties (Campbell Scientific, 1999).

A TE52 Texas electronic rain gauge was installed and rainfall data was captured for hourly and daily rainfall data (Table 1). The rainfall volume measured by the gauge in mm was multiplied by the surface area of treatment wetlands to give the total input volume in (m^3/per unit time). Three rainfall samples were also collected during August and September 2016 and analysed for nitrates phosphates and ammonium.

4.2 Determining the nutrient mass balance

4.2.1 Calculating nutrient mass balance

A nutrient mass balance based on the concentration of nutrients in the inflowing water (surface inflow of effluent and depth and volume of rainfall) and of surface outflows was calculated. Rainwater samples were collected on three occasions and tested for both nitrogen and phosphorus. In each instance nitrogen and phosphorus values were undetectable in rainwater and thus the contribution of nitrogen and phosphorus via rainfall was not considered in the nutrient mass balance. Groundwater inflows and outflows were also ignored as they were also considered negligible. Water loss via evapotranspiration was established but it does not constitute a pathway of nutrient removal. However, water loss via evapotranspiration should lead to increased nutrient concentrations proportional to the water loss via evapotranspiration. Six sample sites along the length of the treatment system were identified for nutrient analysis (Figure 11). Samples were collected weekly and analysed for NO_3-N, NO_2-N, NH_4-N and PO_4-P (Table 2), using the Palintest photometric method. The difference between nutrient concentrations at the inflow in $mg.l^{-1}$ and at the outflow ($mg.l^{-1}$) multiplied by the volume of wastewater entering the wetland in cubic meters per second, gives the difference in nutrient mass between the inflow and outflow and therefore effectiveness of wetland in storing or treating wastewater.

The calculation of nitrogen and phosphorus removal by the treatment wetland was calculated as follows for a daily time step:

$$\mathbf{Gi = (Ai \times A) - (Ao \times D)}$$

Where: Gi = removal or storage efficiency

Ai = influent nutrient concentration (N and P) in $g.m^{-3}$

A = influent discharge in $m^3.s^{-1}$

Ao = effluent nutrient concentration (N and P) in $g.m^{-3}$

D = effluent discharge in $m^3.s^{-1}$

Efficiency is calculated as $g.s^{-1}$ given that concentration is expressed as $g.m^{-3}$ and discharge as $m^3.s^{-1}$.

4.2.2 Measuring water nutrient concentrations

Water samples were collected weekly at each weir location along the treatment system. Each sample was collected in a 300ml plastic sample bottle, by placing the sample bottle below the water surface and allowing it to fill completely before being sealed underwater to prevent air bubbles from being trapped in the sample bottle. Sample bottles were then labelled and placed in a cooler box before being transported to the laboratory. All water samples were filtered through a filter with a pore size of 45µm using a vacuum filter, and phosphate samples were further filtered using Whatman no. 42 filter paper. All samples were then tested immediately after collection and preparation in order to preserve the integrity of the sample. A Palintest 7100 spectrophotometer was used to determine nutrient levels at various sites along the treatment system by following test instructions given in the Palintest 7100 operations manual. Each nutrient is exposed to a specific reagent which produces variable colourmetric reactions that are proportional to the nutrient concentration in the sample (Palintest, 2012). The intensity of the colour produced by the reactions is determined by the spectrophotometer which displays a value for the concentration of each specified nutrient per sample.

Nitrates

In the Palintest Nitratest method nitrate is first reduced to nitrite and the resulting nitrite concentration is then determined by a diazonium reaction to form a reddish solution (Palintest, 2012). This reduction stage is carried out using a zinc-based Nitratest powder and Nitratest tablet which aids rapid flocculation after a one-minute contact period (Palintest, 2012). The nitrite resulting from the reduction stage reacts with sulphanilic acid in the presence of N-(-1-Naphthyl)-ethylene diamine to form a reddish solution (Palintest, 2012).

Nitrites

The method is based on a colourmetric procedure using an iodide containing reagent system (Palintest, 2012). Nitrites catalyse the oxidation of the iodide to iodine under mildly acid conditions to produce a brown colouration (Palintest, 2012). Over the range of the test, a series of colours from colourless through yellow to brown are produced.

Phosphates

The test is based on the vanadomolybdate method where phosphates react with ammonium molybdate in the presence of ammonium vanadate to form the yellow phosphovanadomolybdate (Palintest, 2012).

Ammonium

The Palintest Ammonia test is based on an indophenol method (Palintest, 2012). Ammonia reacts with alkaline salicylate in the presence of chlorine to form a green-blue indophenol complex with catalysts included to ensure complete and rapid colour development (Palintest, 2012).

Table 2: Sensor, units and output intervals used in recording nutrient concentrations

Measurement	Sensor	Scan Rate	Units
NO_3-N	Photometer	Weekly	mg/l
NO_2-N	Photometer	Weekly	mg/l
NH_4-N	Photometer	Weekly	mg/l
PO_4-P	Photometer	Weekly	mg/l

4.3 Analysing nutrient storage in wetland plants and sediments

The water status and internal water movement of a wetland defines its extent and influences the soils and nutrients, thus controlling the character of biota and therefore pollutant reduction potential of the wetland (Reddy *et* al., 1999). A given wetland systems extent, depth, flow rate (related largely to slope), vegetation density and overall storage volume, directly affects its effectiveness in water treatment (Vymazal, 2012). It also determines the length of time that water spends in the wetland, and thus the opportunity for interactions and chemical transformations between waterborne substances and the wetland ecosystem (Vymazal, 2012).

The dominant macrophyte was *Typha capensis*, comprising an estimated 80% of plant biomass within the wetland. Plant density and height of each ramet (individual in a clone) was measured in four 1 m^2 quadrats in each wetland cell (12 quadrats in total; Figure 11). Two *Typha* clones per wetland cell, each containing at least five connected ramets, were removed for analysis. One ramet from each clone was selected for chemical analysis, ensuring that individuals of different ages were selected. The six individual plants were then packaged, labelled and transported back to the water quality laboratory for cleaning and processing.

Individual plants were thoroughly cleaned and wiped before being weighed in order to avoid inclusion of sediments and organic films on the plant stems and roots. Plant height was measured. Wet plants were then individually separated by plant organ into roots, stems and leaves, and then finely chopped and weighed to give the total weight of each plant organ and overall "wet" weight for each ramet. Wet plant samples were dried in an oven at 80°C until no further loss of weight was observed. Dried samples where then re-weighed to give total dry weight of each organ and for each individual ramet. These samples were milled to 5um and 10g portions were sent to B.E.M Laboratories to be analysed for total nitrogen (TN) and total phosphorus (TP). At B.E.M Laboritories, milled samples were then ashed at 480^{o}C and then shaken up in a 50:50 HCL (32%) solution for extraction through filter paper. The total phosphorus content of the extracts were then measured with a Varian ICP-OES optical emission spectrophometer. Total nitrogen content of the milled samples was determined through total combustion in a Leco N-analyser, these are the standard ISO accredited analytical methods used to calculate TN and TP content in plant tissue.

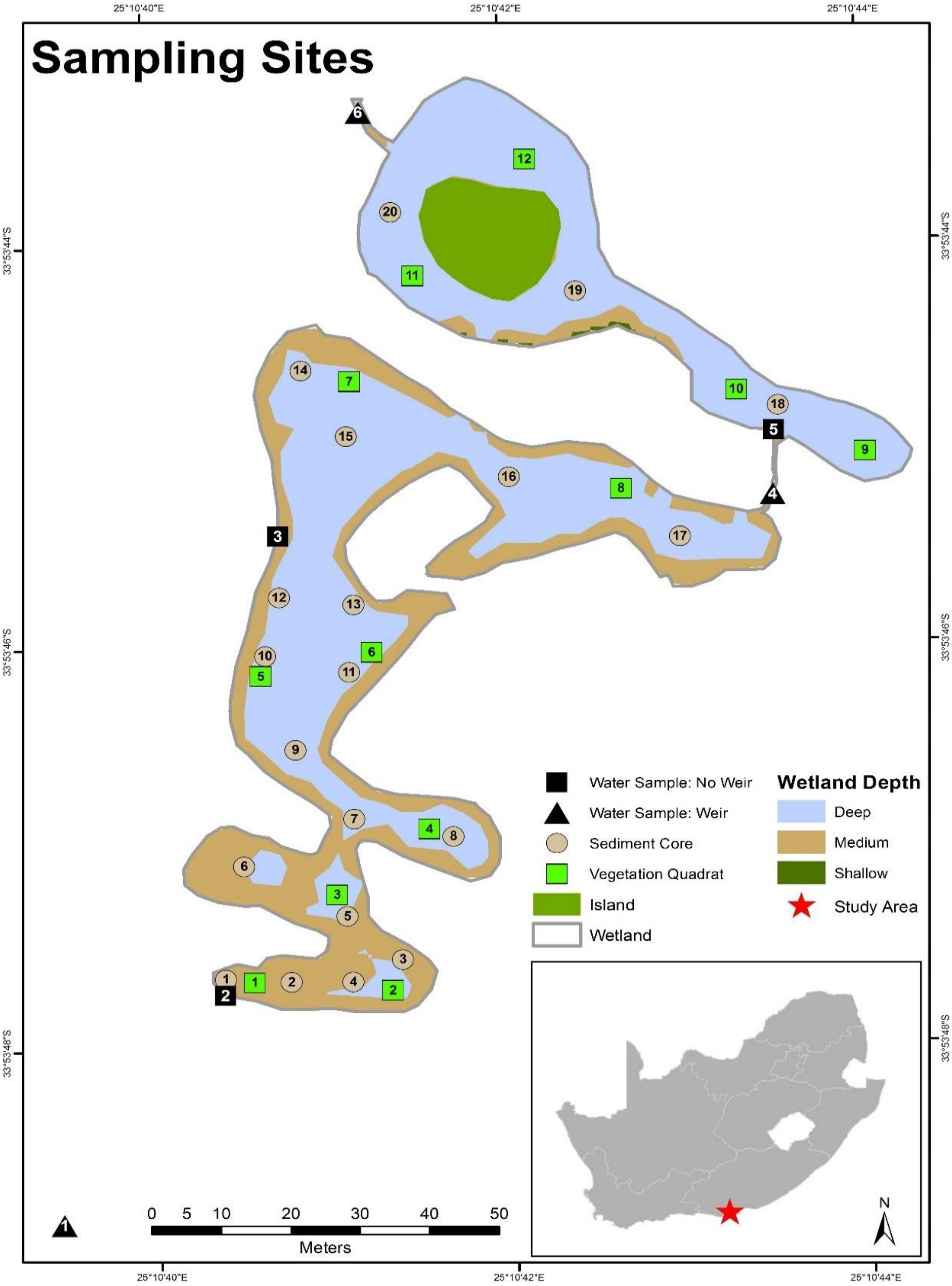

Figure 10: Map showing the location of sediment cores and sampled vegetation, (the numbering indicates the sequence and total number of samples taken).

Wetland sediment mass was calculated from 20 cores of known diameter and depth taken at various points throughout the wetland (Figure 10). Cores where taken using 50mm clear PVC pipes which were pressed into the sediments until the clay layer was reached, plugging the bottom section of the PVC. The pipes where then carefully removed and transported to the soils laboratory. The 20 cores where then extruded from the pipes and placed into glass beakers. Clay plugs were carefully cut out and removed in order to avoid inclusion of the basal clay layer in the sediment sample. Wetland sediment samples where then individually weighed to give the wet weight. Using the weight of each sample relative to the diameter of the PVC tube and the depth of soil sampled, the total mass of sediment in the wetland could be calculated.

Four samples per wetland cell where then selected for further analysis (12 samples in total). Samples were placed in an oven at 60°C until no further loss in weight was observed. Dry samples where then weighed and passed through a 2mm sieve. Sieved samples were weighed into 20 g portions and sent to B.E.M Laboratories for analysis of total nitrogen and phosphorus. Total nitrogen in sediment samples was determined by B.E.M laboratory through the total combustion procedure using a Leco Truspec CN N elemental analyser. Total phosphorus analysis was determined by extracting total phosphorus from sediment samples through acid digestion using a 1:1 mixture of 1 N nitric acid determined with a Varian ICP-OES optical emission spectrophotometer.

The total amount of nitrogen and phosphorus stored in the wetland's sediments could then be calculated by multiplying the average nutrient mass in each core sample by the total volume of sediment within each wetland cell.

4.4 Topographic survey

A total station was used to develop a DEM required in calculating the extent, slope, depth and volume of water in the treatment wetlands. A total of 1000 points were taken throughout the wetland and transferred into excel for analysis. Excel data was then uploaded into ARCMap and given appropriate projection so that an accurate and detailed three-dimensional model of the treatment wetlands could be developed. Using ARCMap the extent of the wetland could be calculated. The slope of the wetland was calculated by selecting a point at the inflow and one at the outflow, and subtracting their relative elevations divided by the distance between each point.

Chapter 5: Results

5.1 The water balance

A total of 906.45m^3 of water entered the FWS wetland system either as rainfall or pre-treated domestic wastewater (Table 3). Surface wastewater inflows reflect the amount of water used by households and amounts to a total of 263.62m^3 of wastewater entering the FWS wetlands, with peak flows occurring from 7am to 9am and from 4pm to 6 pm daily. On average, approximately 43.94m^3 of domestic wastewater was released into the FWS wetland system each month, with the highest monthly wastewater inflow rate being August, accounting for 25.8% of the total amount of wastewater released during the study period (Table 3). The lowest monthly wastewater inflow rate occurred in April 2016, in which only 3.2% of the total wastewater inflow was released into the treatment wetlands.

Figure 12 shows the pulsed nature of rainfall and wastewater inputs into the wetlands. A total of 642.82m^3 of rainwater entered the treatment wetlands during the study period, giving a monthly rainfall input average of 107.14m^3. Rainfall occurred sporadically yet consistently throughout the study, with no prolonged period without rainfall. Rainfall generally occurred as low intensity orographic rain with occasional prefrontal thunderstorms resulting in short yet high intensity rainfall events such as on the 6th of May 2016 (Figure 11A), where 45.98m^3 of rain entered the wetland in a two-hour period. July had the highest monthly rainfall average comprising 28.8% of the total amount of rainfall, while August had the lowest monthly average with only 8.1% of the total rainfall inputs recorded during the study period (Table 3). Rainfall accounted for 70.9% of all water inflows into the treatment wetlands, indicating that it is the dominant water source.

Table 3: Total monthly wetland inputs and outputs over the duration of the study period

Month	Rainfall		Wastewater inputs		Evapotranspiration		Surface water outflows		Inflow total (l)	Outflow total (l)	Deficits (l)
	Total (l)	% of total	Total (l)	% of total	Total (l)	% of total	Total (l)	% of total			
April	70440.6	11.0	8525.2	3.2	47627.9	16.3	71697.6	13.9	78965.8	119325.5	-40359.7
May	99544.8	15.5	32633.8	12.4	50590.3	17.3	189639.9	36.7	132179	240230.2	-108051.2
June	78876.6	12.3	37013.4	14.0	37716.3	12.9	46648.2	9.0	115890	84364.5	31525.5
July	185170.2	28.8	57219.2	21.7	39012.6	13.3	140780.7	27.2	242389.4	140780.7	101608.7
August	52303.2	8.1	68103.7	25.8	57830.9	19.7	27136.0	5.3	120406.9	84966.9	35440
September	156487.8	24.3	60128.1	22.8	60292.8	20.6	41213.9	8.0	216615.9	101506.7	115109.2
Total (l)	**642823.2**		**263623.4**		**293070.9**		**517116.3**		**906447**	**771174.5**	**135272.5**

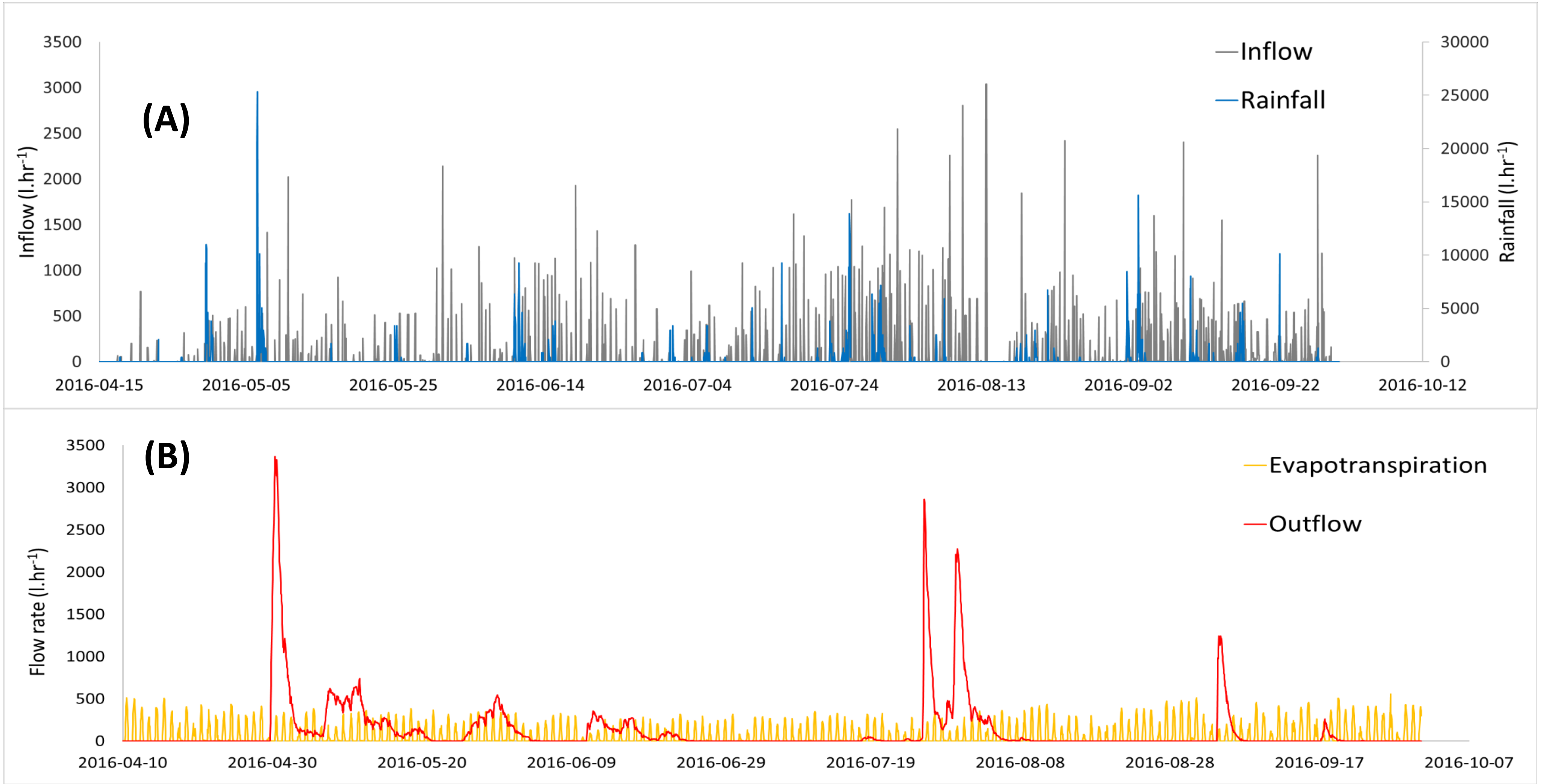

Figure 11: Wastewater and rainfall inflows (A) and evapotranspiration losses and surface outflows (B) from April to September 2016.

During the six-month study period, 810.19m^3 of water was lost from the treatment wetlands, either through evapotranspiration or via surface outflow (Table 3). The observed surface outflow events occurred following large rainfall events such that variation in wastewater inflow rates had a small effect on variation in outflow (Figure 12). Overall, a total of 517.12m^3 of water was lost from the treatment wetlands via surface water outflow, accounting for the majority (63.8%) of the total amount of water lost from the treatment wetlands during the study period. On average, 135.03m^3 of water was lost from the wetland system each month, with the highest recorded total in May, comprising 36.7% of the total amount of water lost via surface outflow (Table 3). The lowest monthly surface water outflow was in August, accounting for only 5.3% of the total. Four major peaks in outflow were observed during the study (30/04/16; 26/07/16; 30/07/16 and 03/09/16), which constitutes the majority of the wetlands outflows, which were a direct result of large rainfall events (Figure 13).

An estimated total of 293.07m^3 of water was lost from the treatment wetlands via evapotranspiration, amounting to 36.2% of the total water loss during the study period. Water loss through evapotranspiration occurred consistently throughout the study period, with an average monthly loss of 48.85m^3 (Figure 12). September had the highest recorded water loss via evapotranspiration, accounting for 20.6% of the evapotranspiration total, while June had the lowest recorded water loss at 12.9% of the evapotranspiration total.

During periods of low rainfall when surface water outflows were low, evapotranspiration was the main factor controlling water loss from the wetlands. The end of winter and start of spring were the warmest months (August and September 2016) with the highest evapotranspiration rates and thus lowest surface water outflow totals (See Table 3), even though these two months had the highest wastewater inflow totals (25.8% and 22.8% respectively). High evapotranspiration rates in August and September resulted in the lowest combined monthly surface water outflows, highlighting the dominant effect of evapotranspiration within the FWS wetlands in the absence of large rainfall events. The winter months of May and July had the highest surface water outflow totals with the first and third highest rainfall averages and, relatively low evapotranspiration rates. Periods of low rainfall (which co-incided with high evapotranspiration losses that dominated the water balance), were associated with a decrease in water volume in the wetlands. Given this there is likely to be a strong seasonal effect on the wetland system's hydrological processes.

Figure 13 shows the relationship between rainfall, surface water outflow and wetland water level. The results suggest that each large rainfall event resulted in a sudden rise in the wetland water level, followed by high surface water outflows. In contrast, low intensity rainfall periods such as those observed at the beginning of June (Figure 12A), had a limited effect on the wetlands surface water outflow. These low intensity rainfall periods either resulted in a gradual outflow or simply recharged the wetlands water volume previously lost via evapotranspiration, thus raising the water level. The four major outflow events have been labelled 1 to 4 in Figure 12A. These were produced by high intensity rainfall events or when consistent and prolonged rainfall and associated high water levels resulted in large outflow spikes, as there was less storage volume available to buffer the volume of water entering the wetland system during such rainfall events. These large spikes in outflow amounted to 58.6% of the total surface water outflow recorded during the study period (Table 4).

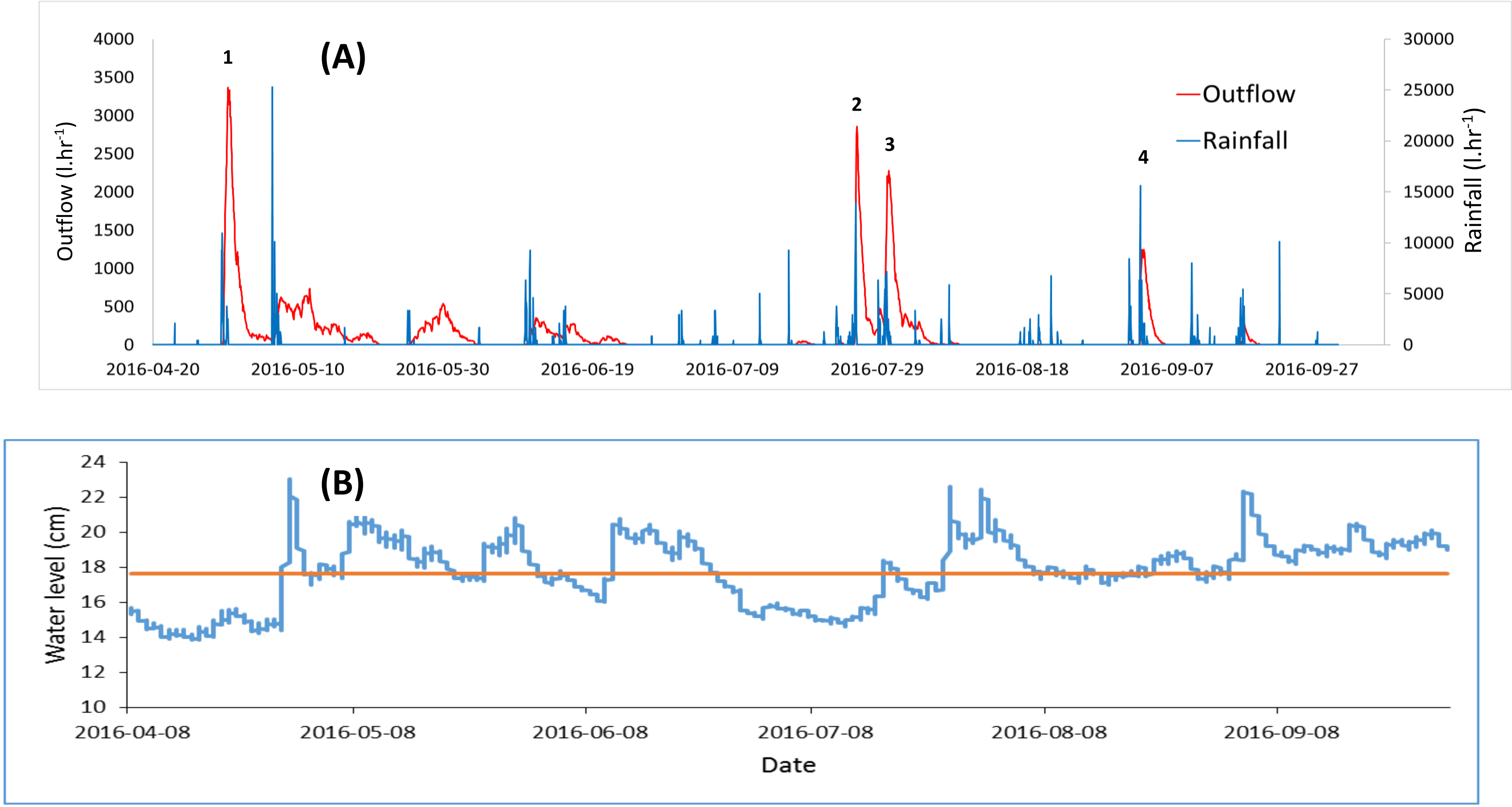

Figure 12: Outflows in relation to rainfall events (A) and wetland water level and outflows (B). Note that in (B), the horizontal line represents the water level at which surface water outflow started to occur (the maximum retention level).

Rainfall events that added more than 1 000 litres per hour to the wetlands or that added more than 60m^3 in total over less than three days, resulted in large spikes in outflow from the treatment wetlands (Table 4). The total and hourly outflows associated with such rainfall events is related to the amount and intensity (mean hourly rainfall) of rain that entered the treatment wetlands, showing that rainfall was a key factor controlling outflow from the treatment wetland system.

Table 4: High rainfall events that resulted in high outflow events

Event	Date	Hours	Rainfall (m^3)	Hourly mean (l.h^{-1})	Date	Hours	Outflow (m^3)	Hourly mean (l.hr^{-1})
1	29/04/2016 →30/04/2016	24	66.64	2776.9	29/04/2016 →03/05/2016	96	118.48	1234.1
2	23/07/2016 →26/07/2016	69	78.45	1137.1	25/07/2016 →28/07/2016	68	61.97	911.3
3	29/07/2016 →30/07/2016	43	61.16	1422.4	28/07/2016 →02/08/2016	110	88.18	501.1
4	01/09/2016 →03/09/2016	53	85.63	1615.6	03/09/2016 →03/09/2016	51	34.34	673.3

No relationship between domestic wastewater inflows and surface water outflows was observed during this study period (Figure 13). Wastewater inputs only accounted for 29.1% of all water inputs into the treatment wetlands, with the remainder being contributed by rainfall. Therefore, inflow from domestic wastewater had little effect on outflow.

The total inflows (906.45m^3) subtracted from the total outflows (810.19m^3) gave a deficit of 96.26m^3 that was unaccounted for by the water mass balance. This value amounts to 10.6% of all the water entering the wetland system that was not accounted for, indicating that water may have been lost via another pathway or, potential limitations of the monitoring methods used. Figures 11 and 12B however, illustrate that water levels were low at the start of the study, suggesting that a portion of this deficit may be accounted for by the filling of the wetland during the initial two weeks of the study period.

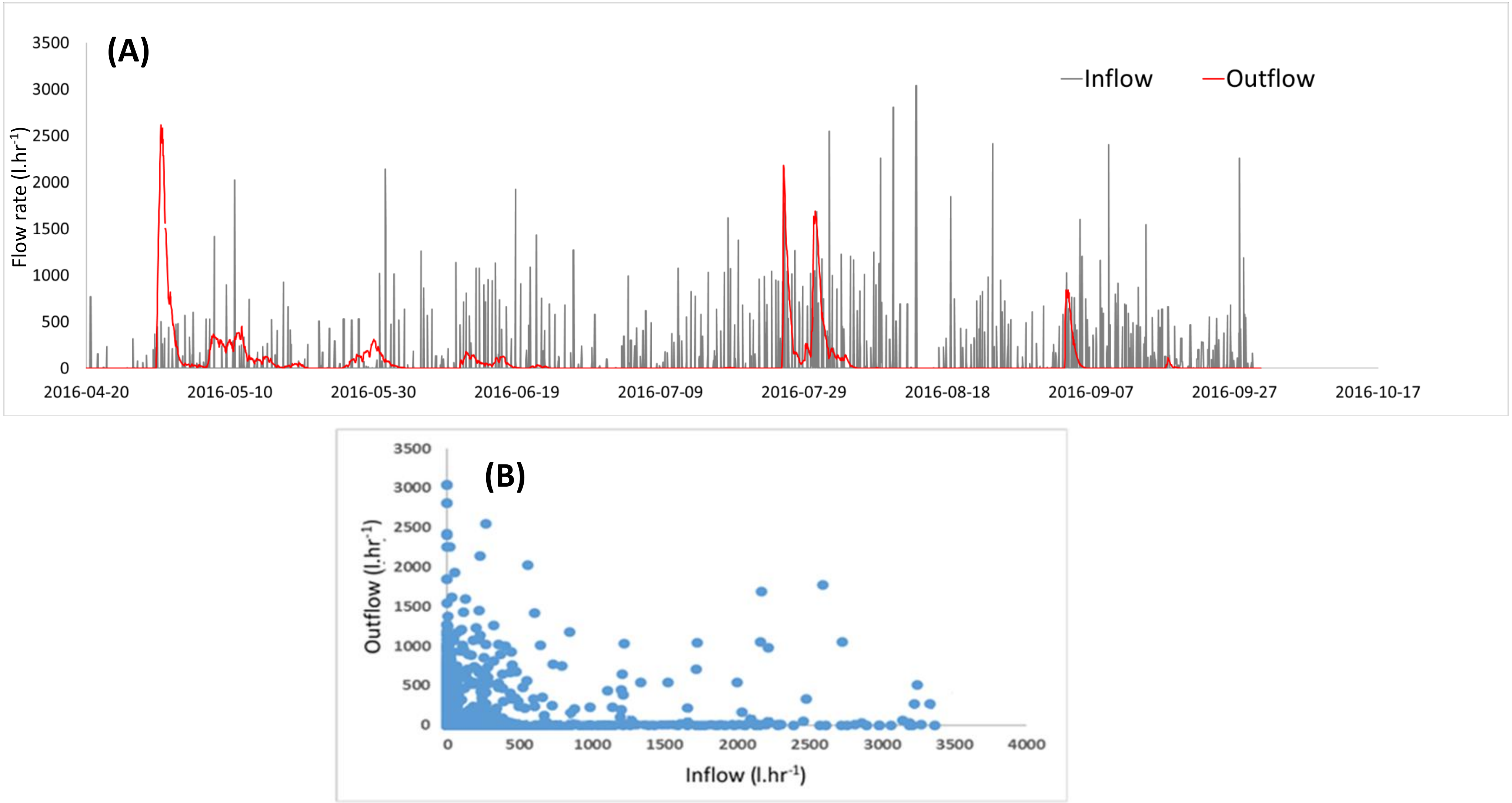

Figure 13: Surface water outflows and wastewater inflows over the study period (A) and a scatter plot showing the relationship between daily outflows and inflows (B).

5.2 The nutrient balance

The results of this study showed that PO_4-P had the highest average daily loading rate of 8.12mg.m^{-2}.day^{-1}, followed by NH_4-N (6.07mg.m^{-2}.day^{-1}) and NO_3-N (2.87mg.m^{-2}.day^{-1}), over the duration of the study period (Table 5). The range between maximum and minimum loading rate values is a consequence of the pulsed flows and inconsistent nature of waste water inputs to the wetland system.

It was calculated that a total of 6kg of PO_4-P entered the wetland system and only 0.67kg was lost from the wetland system via surface outflow (Table 6). Therefore 88.8% of all inflowng PO_4-P was removed from the wastewater by the wetland system, which accumulated approximately 7.23g.m^{-2}.day^{-1} A further total of 2.12kg of NO_3-N entered the wetland with only 0.18kg accounted for at the outflow. This gave NO_3-N the highest overall treatment removal efficiency of 91.4%. Ammonia nitrogen had the lowest overall treatment efficiency of 76.6% with 3.25kg entering the wetland system, and only 0.76kg of NH^4-N recorded leaving the outflow (Table 6). Nitrite nitrogen (NO_2-N) was not detected in this study as the concentrations were too low for the detection method used (1 – 1500mg.l^{-1}), such that it will not be reported further.

Table 5: Average, maximum and minimum PO_4-P, NH_4-N and NO_3-N inflow loading rates over the duration of the study period

Nutrient	**Average** mg.m^{-2}.day^{-1}	**Maximum** mg.m^{-2}.day^{-1}	**Minimum** mg.m^{-2}.day^{-1}
PO_4-P	8.12	393.61	0.04
NH_4-N	6.07	168.34	0.02
NO_3-N	2.87	98.10	0.02

Table 6: Total PO_4-P, NH_4-N and NO_3-N inflow and outflow totals and nutrient removal rates over the duration of the study period

Nutrient	Total inflow Kg	Total outflow kg	Removal rates $g.m^{-2}.day^{-1}$	%
PO_4-P	6.00	0.67	7.23	88.8
NH_4-N	3.25	0.76	3.37	76.6
NO_3-N	2.12	0.18	2.62	91.5

The average concentrations of PO_4-P and NO_3-N showed a consistent decrease down the length of the wetland system (Table 7). In contrast, the average NH_4-N concentration displayed a consistent decrease from sites 1-5, followed by a slight increase in the concentration at the outflow (site 6). By far the greatest reduction in the concentration of all recorded nutrients occured between sites 2 and 3, where average nutrient concentrations were reduced to greater than 80% of their original inflow readings. The average reduction between sites 2 and 3 for PO_4-P was 86.1%, 90.6% for NH_4-N and 84.3% for NO_3-N. Thereafter (sites 3-6), concentrations between sites remained fairly consistent with a gradual decrease towards the outflow. The ANOVA analysis revealed that plant tissue in wetland cell 1 had a significantly higher nitrogen and phosphorus concentrations compared to wetland cell 2 and 3 (ANOVA, $P < 0.05$) Table 7. Wetland cells 2 and 3 had similar nitrogen concentrations whereas plant tissue in cell 2 had a higher phosphorus concentration when compared to wetland cell 3 (Table 7). The analysis also showed that the reduction in nitrogen and phosphorus concentrations of sediments is similar across all 3 wetland cells (ANOVA $P> 0.11$, Table 7).

Table 7: Mean (± standard error) nutrient concentrations of effluent at the 6 sites along the treatment wetlands. Values in the same row represented by a different superscript symbol represent significantly different means (ANOVA, $P<0.05$)

Nutrient	Site 1	Site 2	Site 3	Site 4	Site 5	Site 6	$F_{(5,120)}$ value	P Value
NO_3-N (mg.l^{-1})	8.73 ± 0.66 ^{a}	6.65 ± 0.62 ^{b}	1.04 ± 0.11 ^{c}	0.62 ± 0.08 ^{c}	0.51 ± 0.08 ^{c}	0.32 ± 0.05 ^{c}	97.1	<0.000
PO_4-P (mg.l^{-1})	23.52 ± 1.0 ^{a}	17.66 ± 1.25 ^{b}	2.46 ± 0.22 ^{c}	1.72 ± 0.19 ^{c}	1.76 ± 0.17 ^{c}	1.37 ± 0.16 ^{c}	216.1	<0.000
NH_4-N (mg.l^{-1})	12.32 ± 2.09 ^{a}	11.98 ± 2.35 ^{a}	1.13 ± 0.17 ^{b}	0.55 ± 0.09 ^{b}	0.46 ± 0.05 ^{b}	1.46 ± 0.14 ^{b}	20.4	<0.000

No clear relationship between flow and nutrient concentration variability was observed during the study (Figure 14). However, the concentrations of both PO_4-P and NO_3-N were lower in the vicinity of the inflow during high flow events (greater than 800l.hr^{-1}) than during low flow events (less than 0.6l.hr^{-1}, Figure 15). Nevertheless, nutrient concentrations were reduced as water flowed through the wetlands during both high and low flow events. Mean water quality parameters for high flow events was only calculated from two sampling occasions. Therefore, standard errors were not displayed in Figure 15 and ANOVA analysis could not be used to compare nutrient concentrations during high and low flow events.

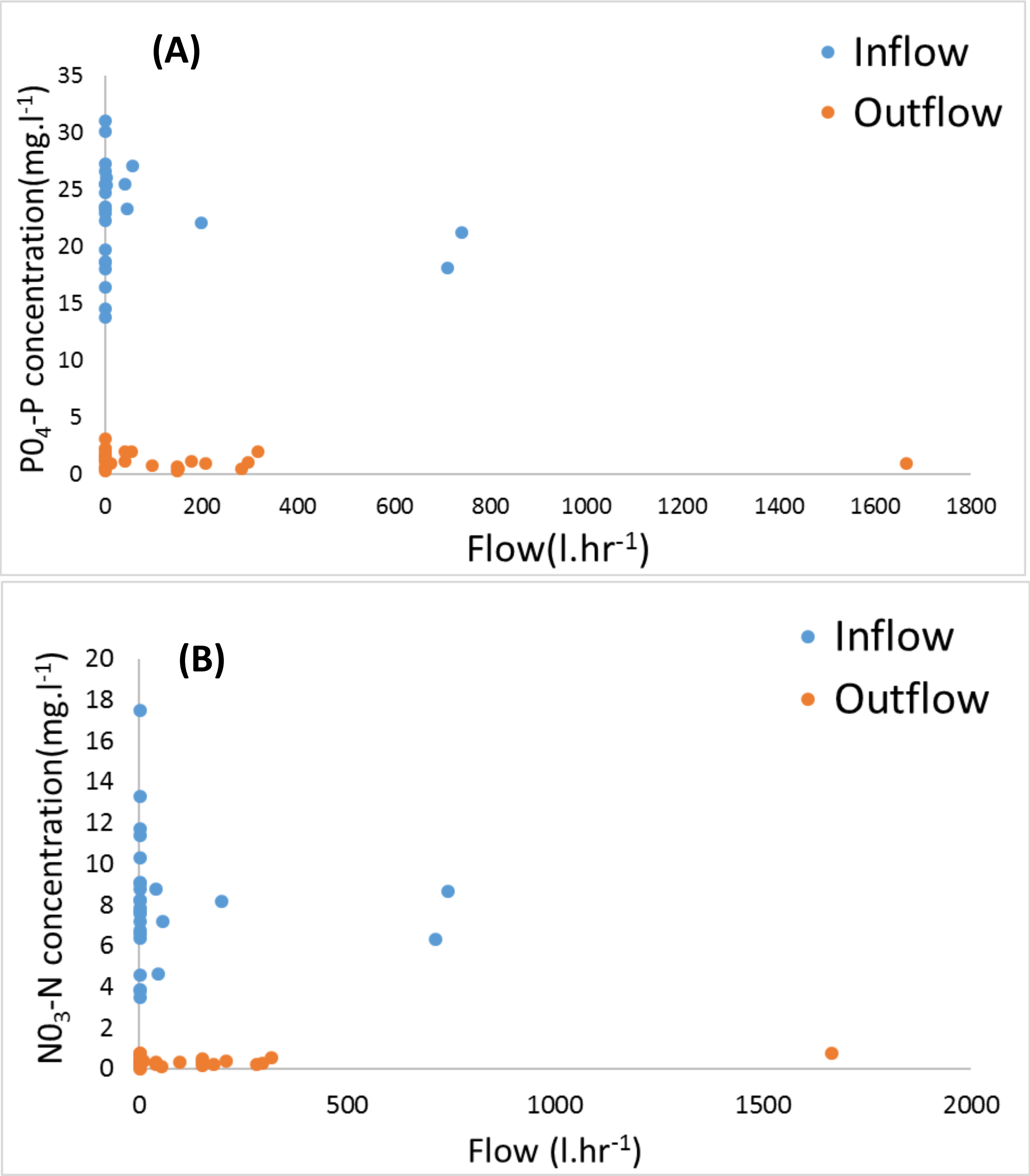

Figure 14: Rating curve showing the relationship between PO_4-P (A) and NO_3-N (B) concentrations in wastewater inflow (blue dots) and outflow (red dots).

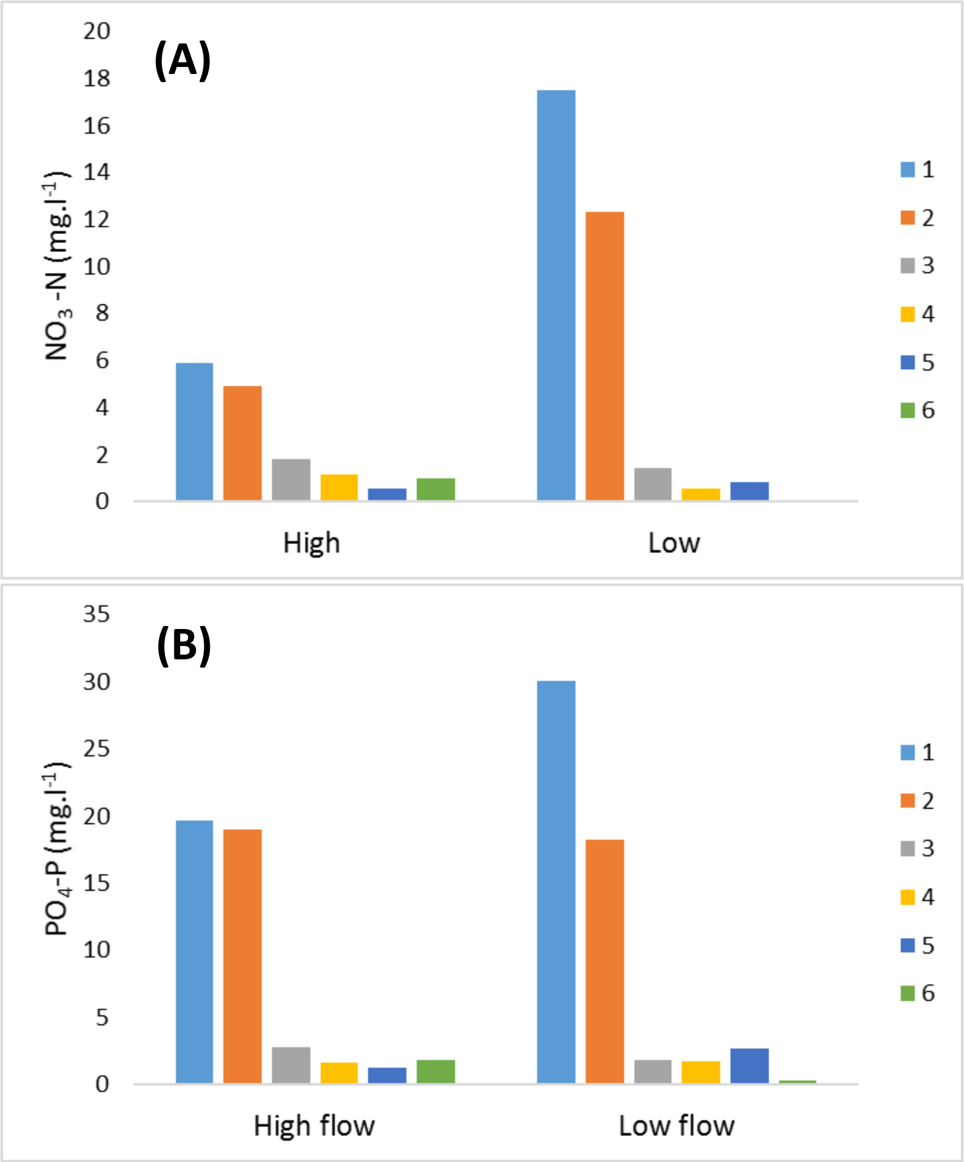

Figure 15: PO_4-P (A) and NO_4-N (B) concentrations throughout the wetland during high and low flow events.

5.3 Nutrient storages

The results showed that both nitrogen and phosphorus concentrations in *Typha capensis* decreased significantly from wetland cell 1 (closest to the wastewater inflow) to wetland cell 2 (between the inflow and the outflow; Table 8). Between wetland cells 2 and 3 the nutrient concentrations in plant tissue remained relatively constant. Nitrogen concentrations in wetland sediments remained uniform across all three wetland cells, whereas the concentration of phosphorus in wetland plants and sediments was significantly higher in the first wetland cell with decreasing concentrations in each consecutive cell down the length of the wetland system. The data clearly demonstrates that the majority of the nutrient load entering the wetlands is removed within the first cell of the wetland system.

Table 8: Mean (± standard error) nutrient concentrations of plants and sediments in the three cells making up the treatment wetland. Values in the same row represented by a different superscript symbol represent significantly different means (ANOVA, $P<0.05$)

Nutrient	Cell 1	Cell 2	Cell 3	$F_{(2,10)}$ value	P Value
N Plants (g.kg^{-1})	24.47 ± 0.26^{a}	13.68 ± 0.17^{b}	12.88 ± 0.74^{b}	194.7	<0.000
P Plants (g.kg-1)	3.66 ± 0.03^{a}	1.18 ± 0.05^{b}	1.01 ± 0.03^{c}	162.1	<0.000
N sediments (g.kg^{-1})	0.42 ± 0.02	0.43 ± 0.02	0.47 ± 0.03	1.2	0.354
P sediments (g.kg^{-1})	80.36 ± 15.39	43.93 ± 7.44	41.90 ± 10.31	2.68	0.117
Total N (kg)	160.71 ± 3.23^{a}	121.28 ± 3.94^{b}	168.15 ± 6.09^{a}	181.1	<0.000
Total P (kg)	30.01 ± 5.26^{a}	12.94 ± 2.07^{b}	14.2 ± 1.63^{b}	6.1	0.019

An estimated total of 450.1kg of nitrogen is stored within the plants and sediments in the three wetland cells. The results showed that sediments contain the largest proportion (69.3%) of this total. Nitrogen storage within wetland vegetation was however not insignificant (Table 9), and uptake by *Typha capensis* in the wetland system evidently served as an important store of nutrients. A total of 57.1kg of phosphorus is stored within the wetland system and the results again showed that sediments serve as the largest and most important sink of phosphorus, comprising 77.4% of the total amount of phosphorus stored in the wetland system.

According to the amount of nitrogen and phosphorus loaded into the wetland system over the study period, a total amount of 15.9kg of nitrogen and 44.9kg of phosphorus is expected to have been added to the wetland over its 157-week lifespan prior to sampling. Therefore approximately 3.5% of the 450.1kg of nitrogen stored within the wetlands was derived from waste water inputs, whereas 78.6% of the total amount of phosphorus stored in wetland plants and sediments was derived from waste water inputs.

Chapter 6: Discussion

6.1 Introduction

The use of FWS constructed wetlands at the Crossways Farm Village WWTW led to a significant reduction in effluent nutrient concentrations and therefore, a substantial improvement in the quality of water at the outflow of the wetland system compared to the inflow. A hydrological mass balance indicated that the Crossways Farm Village FWS wetland's hydrodynamics are largely controlled by climatic variables, which are likely to seasonally alter the treatment efficiency and nutrient removal processes in these FWS wetland systems. The study shows that FWS constructed wetlands that are lightly loaded with secondary domestic wastewater can significantly reduce nutrient concentrations by greater than 70% during winter months and improve water quality to within the appropriate limit of South Africa's National Standard's (SANS 241), for both nitrogen and phosphorus concentrations.

Nutrient concentrations in water samples showed a substantial and consistent decrease in both nitrogen and phosphorus concentrations down the length of the FWS treatment wetland. Nutrients were also stored within plants and sediments of the three wetland cells. Sediments comprised the largest sink for both nitrogen and phosphorus storage in all wetland cells. The remarkably high treatment efficiency of the FWS wetland system can be attributed to the low wastewater loading rate in relation to the size, design and age of the treatment wetland system.

6.2 Hydrological characteristics of the Crossways FWS wetland

Hydraulics, which describes the mechanics of water flow in both time and space, is recognised as one of the key factors regulating water quality improvements in constructed wetlands (Headley and Kadlec 2007; Kadlec and Wallace 2009; Kusin *et al*., 2010). Rainfall was shown to be the dominant water source recharging the wetland system and contributing to observed large outflow events. Wetland surface outflows were, therefore, the dominant output of water from the wetland system during winter when rainfall was abundant. However, evapotranspiration played a key role in controlling and mediating the wetland's water balance at the end of the study, with an increasingly dominant effect in the warmer and drier spring and summer months (August and September 2016) in the absence of large rainfall events.

The results of the hydrological mass balance showed that the majority of wetland flows could be accounted for, although inflow was greater than outflow by about 10.6% over the study period. This may be a consequence of the presence of very low water levels at the start of the study as the water entering the wetland system during this period resulted in raising the water level of the wetland to its maximum retention level. At this time, the wetland was not in a steady state (inflow = outflow), because inflow greatly exceeded outflow. Alternatively, more water than was estimated may have been lost via another pathway. According to Kadlec and Wallace (2009), it is critical to recognise that small wetlands will have significantly greater convective heat transfer and consequently evapotranspiration is amplified. Therefore, evapotranspiration estimates based on the Penman Monteith equation are often underestimated in such circumstances.

6.3 Nutrient dynamics and storage

A total of 906.45m^3 of water entered the Crossways Farm Village wetland during the six-month study period. The average influent contained approximately 23.5mg. l^{-1} of PO_4-P, 8.7mg.l^{-1} of NO_3-N and 12.3mg.l^{-1} of NH_4-N. Nitrogen and phosphorus were successfully stored in wetland plants and sediments. The first wetland cell, from which samples were collected, received the influent water and had the highest concentration of nutrients in both plants and sediments relative to the rest of the wetland.

Over the six-month study period, approximately 6.00kg of $P0_4$-P entered the wetland system with a daily loading rate of 7.23g/m^2/d^{-1}. A total of 0.67kg was exported. Approximately 57kg of phosphorus was stored in the plants and sediments of the FWS wetland system. Sediments comprise a large proportion of this total, storing 77.4% of the phosphorus currently within the wetland system. Dolan *et al*. (1981) found in a pilot-scale treatment wetland that soil and sediment was the most significant compartment for phosphorus storage, followed by plants, highlighting the importance of sediments as a nutrient sink in constructed wetlands.

A further total of 5.37kg of nitrogen entered the treatment wetland system from April to September 2016, of which only approximately 0.63kg were exported via the outflow. Approximately 450.1kg of nitrogen is currently stored in wetland plants and sediments. The analysis revealed that sediments, once again, comprised the largest storage of nitrogen in the wetland system, retaining 69.3% of the stored nitrogen in the treatment system with vegetation comprising 30.7% of the stored nitrogen. Borin and Tocchetto (2007) evaluated the performance of a constructed surface flow wetland treating diffuse nitrogen pollution from croplands and estimated the five-year nitrogen and phosphorus balances. They found that the wetland sediments accumulated more than half of the incoming nitrogen load, and that nitrogen storage was at least six times higher than for phosphorus. These findings are consistent with the results obtained in this study. According to Mustafa and Scholz (2011), treatment wetlands that are less than five years old with low loading rates, can have significant uptake and storage of both nitrogen and phosphorus, as vegetation and sediments are still being established and thus have a much higher capacity to absorb and store incoming nutrients compared to well established systems. This supports Brix's (1997) argument that plant uptake is only significant in newly constructed wetlands or under low nutrient loading conditions.

The amount of nitrogen and phosphorus that is stored in wetland plants and sediments is dependent upon several interacting physical and biological variables, although nitrogen storage processes are controlled more by environmental variables such as temperature, which affects biological activity. However, phosphorus removal processes depend largely on physical settling and sediment adsorption (Kadlec and Reddy, 2001). Therefore, research shows that phosphorus retention in sediments usually displays an exponentially declining concentration gradient with distance (and time) through a treatment wetland, due to the processes that control its retention and storage (Keller and Knight, 2004). Nitrogen on the other hand can be derived from atmospheric sources and incorporated into sediments and biomass. It may be released during the decay of living organisms, and there are a greater number of processes that can contribute to the transformation and availability of nitrogen in wetlands, as well as its spatial distribution within these systems. The distribution of nitrogen in these systems is therefore generally more uniform than for phosphorus (Keller and Knight, 2004). Given that the majority of the phosphorus within the Crossways Farm Village wetland is derived from wastewater inputs, the wetland phosphorus content is likely to be limited, particularly in the final two wetland cells.

Debusk and Reddy (2005) studied nutrient dynamics in the Everglades marsh and reported mean total nitrogen concentrations in soil of 28,000 mg.kg^{-1} and 29,000mg.kg^{-1} for the depth intervals 0–10cm and 10–30cm, while phosphorus concentrations were 1,150kg^{-1} and

640mg.kg^{-1} for the same depth intervals respectively. The results reported here show that nitrogen storage in sediments is far higher than for phosphorus as was observed in this study. However, in the long-term, wetland sediments do evidently serve as an important component for storage of phosphorus. Therefore, under periods of limited inflow, it is expected that the distribution of phosphorus will be reduced due to the removal of phosphorus through physical settling under low flows, in combination with enhanced uptake of phosphorus by plants and microbes due to its limited availability. In contrast, nitrogen is far more abundant in nature than phosphorus. therefore, under low flows it is expected that the concentrations of these nutrients will be highly variable due to the processes that control their availability and distribution throughout the wetland.

A study conducted by Pulley *et al.* (2015) looking at the dynamics of sediment associated contaminants during drought and flood periods in a lowland UK catchment, showed that highly polluted sediments derived from domestic wastewater releases resulted in sediment phosphorus concentrations in excess of 9000mg.kg^{-1}. The highest recorded phosphorus concentration in the Crossways Farm Village wetland was only 80mg.kg^{-1}. This is reflective of the low loading rate and fairly young age of the wetland system.

The decrease in nitrogen and phosphorus concentrations in both wetland plants and sediments from wetland cell 1 to wetland cell 2, suggests that the first FWS wetland cell is sufficient to treat the amount of nutrients entering the wetland system at present given that the nutrient content in the final wetland cell is considerably lower and far more representative of a natural wetland system.

Based on the amount of nitrogen and phosphorus loaded into the wetland over the study period, a total amount of 15.87kg of nitrogen and 44.9kg of phosphorus is expected to have been added to the wetland over its 157-week lifespan prior to sampling nutrient stocks (Table 9). Therefore, approximately 3.5% of the 450.1kg of nitrogen stored within the wetland was derived from waste water inputs, whereas 78.6% of the total amount of phosphorus stored in wetland plants and sediments was derived from waste water inputs.

Table 9: Total nitrogen and phosphorus storage and sources

	Storage			**Sources**	
	Plants (kg)	**Sediments** (kg)	**Total in wetland** (kg)	**Effluent input** (kg)	**Accounted from effluent** (%)
Nitrogen	138.4	311.7	450.1	15.9	3.5
Phosphorous	12.9	44.2	57.1	44.9	78.6

Several studies have demonstrated that phosphorus retention in constructed wetland plants and sediments generally follows a first order model, due to the factors that control its availability, retention and storage (Kadlec and Wallace, 2009; Keller and Knight, 2004; Tanner *et al.*, 1995). Phosphorus retention in wetlands is therefore strongly linked to the wastewater phosphorus concentration and loading rate, resulting in a declining concentration gradient with increasing distance from the wastewater inlet location (Keller and Knight, 2004), as was observed in this study.

6.4 Wetland flows and nutrient dynamics

Rainfall within the Humansdorp region is known to be sporadic, although the area predominantly receives winter rainfall with occasional high intensity pre-frontal thunderstorms (Cowling, 1992). High intensity rainfall events increase water levels in the wetland resulting in large outflow peaks as the rainwater flushes from the wetland system. A study conducted by Kadlec and Wallace (2009) in the Benton Kentucky wetland in September 1990, highlights the dominant effect of rainfall in such FWS wetlands. In this study, a sudden rainfall event subjected the wetland system to a surplus daily inflow load of 100% (278m^3), leading to a sudden outflow increase of approximately 300%. During each of the four large rainfall events that occurred in the study period, similar increases in outflow were observed, suggesting that seasonal rainfall inputs create and control outflow events in the FWS wetland system, particularly for treatment systems accustomed to low loading rates such as the Crossways Farm Village treatment wetland.

The long-term consequences of these events on the wetland system's nutrient dynamics is however, not insignificant and periods of high and low flows largely control the distribution and concentration of nutrients found throughout the wetland system. A study conducted in

Country Waterford by Mustafa and Scholz (2011) on nutrient accumulation in wetland plants and sediments, found that higher nutrient concentrations were noted in summer compared to winter, due to the change in the hydrological regime during the two seasons, with summer being the warmer drier months. The study also showed that there is highly reduced or no outflow during summer and much higher flows in winter compared to summer. A similar hydrological regime exists at the Crossways Farm Village wetland system and it controls the treatment efficiency and structure of the wetlands. High rainfall events were observed to enhance the movement of nutrients further downstream in the wetland system, with higher outflow, and site 4 and 5, nutrient concentrations during these periods. These events temporarily reduce the wetland systems treatment efficiency and may play a role in flushing nutrients from the first into the second and third wetland cells. In contrast, periods of low flows increase the treatment efficiency of the Crossways Farm Village treatment wetland. This means that outflow concentrations are significantly reduced during low flow periods due to the increased residence time of water within the wetland and particularly during periods of zero outflow. Given that low flows predominate, more nutrients are deposited within the first wetland cell leading to elevated plant and sediment nutrient concentrations in the first wetland cell, particularly during summer.

During winter rainfall periods, nutrients are readily supplied to the final wetland cells which promotes the growth of *Typha* during winter and at the start of spring. However, as conditions begin to get warmer and drier at the start of summer, reduced rainfall results in lower flows of water and therefore, less nutrients moving through the wetland system. Thus, substantial dieback occurs as the excessive *Typha* stands become nutrient-limited. At the same time, nutrient return from deeper sediments to the water and sediment surfaces is reduced, slowing the decomposition rate. As a result of these processes, significant dieback was observed at the end of the study in the lowermost wetland cell, while plant density, productivity and biomass within the first wetland cell increased during this period, further perpetuating the observed dieback within the final two wetland cells.

A study conducted by Steinbachova *et al*. (2006) on the influence of nutrient supply on the growth of *Typha angustifolia*, showed that under nutrient poor conditions, biomass growth in *Typha* is largely allocated to rhizome and root production. In contrast, their results showed that under eutrophic or nutrient rich conditions, biomass growth is predominantly allocated to shoot and leaf production. The results of the Crossways Farm Village wetland study support these findings, as during winter under high flow conditions, reduced nutrient availability in the first wetland cell may have resulted in the growth of new rhizomes/ramets during this period. In summer, reduced flows and increased evaporation resulted in elevated nutrient concentrations within the first wetland cell, which may have then initiated the observed growth of shoots, stems and leaves, leading to the increase in *Typha* biomass and density within the first wetland cell, as was observed in the final months of the study period.

A visit to Crossways Farm Village in January 2017 substantiated the predicted dominant effect of season on the hydrology of the wetland system. In peak summer (January 30th, 2017), no water was present within the second and third wetland cells, which had completely dried up due to elevated temperatures and a concurrent lack of rainfall, thus enhancing evapotranspiration. This seasonal drying process may ultimately result in much of the stored nitrogen and carbon being exported from the final two wetland cellss through the oxidation of sediments and soil organic nitrogen. However, during the wet winter season, nutrient storage plays a crucial role in limiting the flushing of nutrients via surface water outflows.

Furthermore, during the January 2017 revisit to Crossways Farm Village, it was observed that all aboveground plant parts within the lower two wetland cells had died. Concurrently, the vegetation biomass within the first wetland cell had increased, to the extent that no areas of open water were observed within it. This supports the hypothesis that seasonal climatic changes dictate the wetland systems hydrology and therefore, directly determine nutrient dynamics, physical structure and effectiveness of the treatment wetlands. Seasonal processes also have such a dominant effect on the Crossways Farm Village FWS wetlands due to the low loading rate, which fails to sustain water and nutrient flows during summer when evaporation, as opposed to rainfall, dominates the hydrological dynamics.

This seasonal process may also have led to the poorer performance of the Crossways Farm Village FWS wetlands in ammonia removal during the final months of the study period, which is most likely due to the return of soluble nitrogen to the water column due to the limited supply of nutrients (Mustafa and Scholz, 2011). Debusk and Reddy (2005) reported that the high sink strength of litter observed in the Everglades marsh was due to the combination of an extremely low nutrient content and relatively high carbon availability. Similar conditions seem to occur at Crossways Farm Village FWS wetland. In winter, extensive biomass growth occurs under nutrient rich conditions and in summer, under nutrient poor conditions, extensive dieback occurs resulting in partially decomposed litter being added to the wetland's soil surface. The second and third wetland cells thus act as a strong nutrient sink during this seasonal transition due to the relatively high carbon content as compared to the carbon content of the soil layer below. This process also results in the storage of both nitrogen and phosphorus within the Crossways Farm Village FWS wetland and may also have contributed to the fairly uniform distribution of nitrogen observed throughout wetland sediments.

The effect of low and high flow events on the treatment efficiency of the wetland system seems small at present, due to the young age of the wetland and therefore, enhanced capacity of the second and third wetland cells to buffer and absorb increased flows and excess nutrients that are re-suspended and transported during flood events. As the wetland ages, it is expected that these events will result in a significant decline in the systems treatment efficiency as the nutrient concentrations gradually shift towards the outflow.

Chapter 7: Recommendations and Conclusions

As wetland plants mature and increase their biomass, their role in and contribution to wastewater treatment shifts from positive to negative (Van Deun, 2015). Extensive stands of vegetation in FWS constructed wetlands can limit hydraulic connectivity, accelerate sediment creation and accumulation rates, shade autotrophic microbial communities, increase biological oxygen demand as well as elevate effluent nutrient concentrations during decay of litter at the end of the growing season (Alvarez and Becares, 2008). Harvesting and maintenance in FWS constructed wetlands can, therefore, enhance the efficiency and lifespan of a given wetland system, particularly in warm areas where biomass production is high, or in systems where vegetation has reached peak standing biomass (Alvarez and Becares, 2008). The need for harvesting and maintenance within FWS wetlands needs to be timed according to the seasonal variability of wetland vegetation and flows. This requires an understanding of standing stocks and accretion rates (Kadlec and Wallace, 2009), which can aid wetland scientists and maintenance personnel to predict when removal pathways may become saturated resulting in the need for such intervention measures.

7.1 Sediments

Several studies have shown that typical FWS constructed wetlands display a decreasing sediment accumulation rate with increasing distance from the wastewater inlet point (Keller and Knight, 2004; Mustafa and Scholz, 2010, Van Deun, 2015). The three wetland cells at Crossways Farm Village appear to have accumulated sediment at varied rates as a consequence of proximity to the inlet point but also due to the influence of seasonal hydrological changes in wetland flows. These seasonal processes result in an uneven distribution of nutrients within the wetland system, which dictates the physical structure of vegetation and therefore, the distribution of suspended solids and sediments within the various wetland cells.

Sedimentation rates often vary considerably within and between individual FWS wetland systems (Kadlec and Wallace, 2009), which necessitates desludging in order to remove accumulated sediments. Such measures are often completed in stages, which also helps to minimise the re-suspension and release of nutrients through the disturbance these intervention measures create. Periods of low or no flows provide the best opportunity for such maintenance procedures (Van Deun, 2015).

The hydrological data and observations made during this study show that during summer or periods of drought, highly reduced flows temporarily result in all incoming water, sediments, suspended solids and nutrients settling within the first wetland cell, and that during winter, dense vegetation and proximity to the inlet results in further deposition of sediments. Measurement of accumulated sediment on the bed of each wetland cell suggests that sediment accumulation rates vary substantially in different parts of the Crossways Farm Village wetland system (Table 10). The first wetland cell accumulated 19.2cm of sediment during the three years of operation compared to the third wetland cell, which has only accumulated an average of 3.8cm over three years.

These data support the findings of Mustafa and Scholz, (2011), which had high sediment accumulation rates in the first wetland cell of an integrated constructed wetland system compared to a third wetland cell. Their results for a FWS constructed wetland that had been in place for seven years showed that from the start of the study, there was an average increase in depth of 45cm of post-construction sediments in the first wetland cell of their wetland system compared to 12.6cm in the third cell. In the wetland studied by Mustafa and Scholz (2011) an annual accretion rate of approximately 6.4cm.yr^{-1} was determined for the first cell compared to the third wetland cell in their system, which accumulated 1.8cm.yr^{-1} of sediment.

Table 10: Average ± standard deviation water depth, sediment accumulation rate and predicted sediment storage capacity in the three cells of the Crossways FWS wetland

	Average water depth (cm)	Accumulated sediment (cm)	Accumulation rate (cm.a^{-1})	No of years to fill (required maintenance)
Cell 1	32 ± 3.7	19.2 ± 2.9	6.4 ± 3.3	5.0
Cell 2	39 ± 3.2	7.5 ± 1.1	2.5 ± 3.8	20.6
Cell 3	48 ± 4.1	3.8 ± 0.3	1.3 ± 1.9	57.5

The observed difference in sedimentation rates between the first and final wetland cells can be attributed to each cells proximity to the inlet, which is directly affected by the seasonal hydrological changes in the Crossways Farm Village wetland. Wetland cell one received far more water and thus suspended solids throughout the year and is more permanently flooded than the lower two wetland cells. Consequently, the final wetland cell only receives sufficient flow and therefore sediments during winter when high rainfall events are able to transport suspended solids and other sediments further along the length of the wetland. The sediment accretion rates therefore show a fairly systematic decrease with distance from the inlet location.

As the wetland ages it is expected that this accumulation gradient will likely spread towards the outflow, resulting in higher sediment and nutrient concentrations further down the length of the treatment wetland as each wetland cell reaches its maximum capacity to store sediments. This is likely to result in a decrease in the systems treatment efficiency in the absence of maintenance, particularly during high flow events where high rainfall periods will likely flush more nutrients from the wetland sediments towards the outflow. Assuming that the loading rate will remain constant, it is estimated that at a sediment accumulation rate of 6.4cm.yr-1 the first wetland cell will be filled with sediment in approximately five years' time, resulting in channelling in the first wetland cell as the system works to maintain the appropriate slope given the discharge of the wetland system. Incoming nutrients and sediments will then be less effectively treated in the first wetland cell and transported more directly to the second wetland cell, particularly during periods of high flows that may also lead to re-suspension and transportation of stored sediments from the first wetland cell to the second. These conditions will ultimately result in an increase in the sediment accumulation rate in the second wetland, reducing the residence time and distance for nutrients to travel towards the outflow during flood events. The data further suggests that the entire wetland may become filled with sediments in 57.5 years if no intervention measures are undertaken. However, it is likely that the wetlands will become inefficient long before this time due to the pulsed nature of flows produced by seasonal changes in the wetland systems hydrology.

Furthermore, the addition of an outflow weir during this study, has raised the maximum retention level of the system, resulting in an increase in the storage volume of the wetlands to hold water before it is released through the outflow. It is, therefore, likely that the addition of an outflow weir has increased the residence time of water within the final wetland cell, which will further increase the treatment efficiency of the system, particularly during high flow events. However, the reduction in flow associated with insertion of the outflow weir may also lead to an increase in the deposition of sediment, increasing the sediment accumulation rate within the wetlands. This increased accumulation enhances the treatment efficiency of the wetlands but will ultimately reduce its lifespan and increase the need for maintenance. Desludging of wetlands sediments may thus be required in order to extend the lifespan of the wetland. These sediments may be utilised as a fertiliser on the crop fields at Crossways Farm Village, allowing for the reuse of nutrients sequestered by the treatment wetland system.

7.2 Vegetation

It is well established that harvesting of vegetation in lightly loaded FWS wetland systems can result in significant improvements in removal efficiency of both nitrogen and phosphorus from wastewater (Brix, 1997; Mustafa and Scholz, 2011; Vymazal, 2007). Approximately 138.7kg of nitrogen and 21.3kg of phosphorus is currently stored within the vegetation of the Crossways Farm Village FWS wetland. Given the low wastewater loading rate, plant uptake represents a significant portion of the nutrients stored within this system. Greenway and Woolley (2001) conducted a study on a constructed wetland treating municipal wastewater in Australia, and found similar results, where between 24% and 47% of total nitrogen and between 47% and 56% of dissolved phosphorus removal was due to plant uptake. The study by Greenway and Woolley (2001) was conducted on a wetland system that was under five years old and emphasised that nutrient uptake by plants is far higher in younger systems where vegetation is still establishing itself. Crossways Farm Village wetland is a young system and harvesting may therefore have several beneficial effects that may extend the lifespan of the wetland system. Seasonal harvesting may further improve the overall treatment efficiency of the system by enhancing plant growth and standing crop, and therefore nutrient uptake, which (if harvested) directly removes nutrients from the wetland system and limits the return of nutrients to the wetlands during dieback and decomposition processes. Furthermore, removal of nutrient starved vegetation prior to complete dieback will decrease sediment accumulation through limiting the creation of new sediments as a product of organic sedimentation.

As a result of the seasonal growth and death cycle of *Typha* within the Crossways Farm Village system, it is suggested that above ground vegetation be removed from the first wetland cell at the start of winter, allowing more nutrients to be supplied to and taken up by the vegetation in the final two wetland cells during this period. Therefore, harvesting of vegetation in the final two wetland cells should be undertaken at the end of winter, in order to avoid the return of nutrients to the water and sediments during the dieback and decay of plants in summer.

7.3 Future research needs and limitations of the study

Given that constructed wetlands are complex biological systems that are largely controlled by external variables such as climate and hydrology, any generalisations or models are difficult to apply when analysing the treatment efficiency of these systems. Thus, each site has highly variable hydrological dynamics and this in turn, dictates the ability of the treatment wetland to treat incoming wastewater. Further research needs to be undertaken during summer and over a complete seasonal cycle or over multiple cycles, and particularly during extreme climatic events.

Extreme climatic events play a key role in the distribution of sediments and transportation of nutrients further along the length of the wetland system, and although they temporarily decrease the treatment efficiency of the wetlands, they distribute nutrients and enhance the lifespan of the wetland. In order to fully comprehend the influence these events have on longer term nutrient removal, storage and distribution patterns, frequent sampling prior to, during and after such events is required and will provide a more comprehensive perspective of the effects these flood events have on the treatment system. Unfortunately, this was not possible during the study period due to financial and logistical constraints, which dictated the frequency and timing of site visits. As a result, only two extreme climatic events could be analysed for mean water quality parameters. Additional analysis of sediment accumulation rates after such events would have given further depth into the importance of such events for the longer-term vegetative and sediment distribution patterns in the Crossways Farm Village wetland system.

In addition, the biogeochemical cycling and decompositional processes that contribute to the treatment efficiency and sequestration of carbon also need to be analysed to better understand how removal and storage is taking place. Understanding the dynamics of feedback loops between carbon , nitrogen , phosphorus and sulphur in the Crossways wetland system would have also given further insight into what effect these seasonal hydrological changes have on the nutrient dynamics of the wetland system, and whether they encourage the return and/or exportation of nutrients from the wetland system into the atmosphere or as surface water outflow under variable physical and climatic conditions.

The need for further research in the Crossways Farm Village FWS wetland system is evident, as seasonal changes have shown to have a profound effect on the dynamics of the treatment wetland system. Therefore it must be noted that this study focused on and is representative of the water and nutrient dynamics during the winter rainfall period and has given us a rather incomplete understanding of the overall seasonal and longer-term dynamics of the treatment wetland. Table 11 below aids in contextualising the rainfall dynamics during the study period, relative to the average winter rainfall dynamics over the 6-year period prior to the study. What is evident is that this study was conducted during a fairly dry winter rainfall season and although significant flood events did occur, the monthly average rainfall from April to September 2016 was significantly lower than previously recorded for the majority of the study period. Thus, it is likely that this may have enhance the treatment performance of the wetland system, increasing the buffering capacity of the wetlands through positive and negative feedback loops between hydrological dynamics, nutrient distribution, vegetative growth and sediment accumulation. As a result , the data obtained during the study period may slightly exaggerate the wetlands treatment performance relative to what may be considered as a fairly wet winter rainfall period where treatment efficiency may be greatly reduced due to decreased effluent residency time and limited nutrient availability required for vegetative growth as well as the exportation of accumulated sediments from the wetlands. In addition, very dry or wet summer hydrological conditions may also lead to a highly variable treatment efficiency during winter rainfall periods and thus a more accurate representation of the wetlands performance would need to include a full seasonal cycle which may then give insight into the relationship between summer hydrological characteristics and treatment wetland performance during winter.

Table 11: Average monthly rainfall volume in (mm) from April 2010 to September 2015 and April to September 2016 at Thornhill weather station (S.A.W.S, 2017)

Year	April	May	June	July	August	September
2010-2015	68.7	50.8	91.3	102.5	55.0	63.0
2016	52.0	24.8	31.6	82.0	12.4	68.6

The limitations of the study are evident, and are an unavoidable consequence of practical, financial and logistical constraints, illustrating that the quest for knowledge does not always align with research budgets and timeframes. Thus, the winter rainfall period was selected as the more dynamic and influential season within the treatment system, because during summer, flows are lower and treatment capacity is therefore expected to be higher.

Nevertheless, the implications of this study are vast, and the intention to generate new knowledge and create awareness around importance and effectiveness of such treatment wetland systems has been created. The results have proven that effective low cost alternatives can be used to reduce anthropogenic nutrient inputs into our aquatic ecosystems and has directly contributed to improving our understanding of the removal efficiencies, storage, distribution dynamics, potential lifespan and treatment capabilities of FWS wetlands within the Eastern Cape during winter rainfall periods. The results of this study have clearly demonstrated that these wetlands are effective nutrient sinks, however more questions than answers have been raised and our understanding is still rather incomplete.

7.4 Conclusions and implications of the study

Freshwater is an increasingly limited resource and our dependence on its availability has become an imminent reality throughout South Africa. This has sparked a conscious drive towards enhancing its protection and reuse. Our improved understanding of excessive nutrient pollution and the use of treatment wetlands has emphasised the importance of combatting domestic wastewater releases into natural water bodies using this technology. Nutrient removal systems are now a legal requirement in many countries in order to avoid adverse health, economic and environmental effects on downstream users. The effectiveness of these low cost FWS constructed wetland systems at performing this function, is evident in this research conducted at Crossways Farm Village Estate and can be applied to decentralised communities and estates where untreated wastewater is currently being released directly into river systems and other natural water bodies or linked to municipal systems with vast expanses of pipework.

Alternatively, a combination of central and decentralised systems may be used where FWS constructed wetlands may be coupled to the municipal wastewater treatment system as a tertiary treatment option. The initiatives of the Crossways Farm Village Estate in terms of limiting their ecological footprint through eco-friendly solutions, needs to be recognised and applied to similar situations across the country, wherever possible.

With the current national water and energy crisis, the centralised /decentralised wastewater treatment dichotomy has been brought under the spotlight and several questions have been posed as to the economic, social and environmental sustainability of our current centralised waste water treatment systems. What has become clear is that decentralised treatment technologies offer a far more environmentally friendly, resilient and sustainable option when compared to centralised wastewater treatment systems. This is largely due to the excessive amounts of water, energy and infrastructure that is required to sanitise and maintain centralised wastewater treatment systems, often in circumstances where decentralised alternatives may use far less of these resources and provide the potential for water and nutrient reuse and recycling.

Furthermore, the transportation of potable and waste water across large distances requires massive amounts of unsustainably produced energy. Centralised wastewater treatment systems are highly dependent on the supply of this limited resource. To make matters worse, aging infrastructure and piping results in significant losses of potable water and leads to discharge of wastewater directly into groundwater resources and the environment. Municipal stormwater drains also draw in and combine enormous amounts of freshwater with wastewater, limiting the replenishment of groundwater aquifers and polluting a valuable resource.

The limited capacity of centralised municipal treatment systems to deal with excessive volumes of wastewater becomes clear during heavy rainfall and flooding events. This often results in the release of partially or untreated wastewater directly into the environment, counteracting efforts to collect and treat all wastewater entering the municipal sewage reticulation system. Furthermore, poorly maintained municipal infrastructure and technological failures combined with poorly trained or unskilled staff results is significant backlogs in treatment chains, ultimately leading to either direct release or limited treatment of wastewater which is discharged directly into oceans, rivers, lakes or other natural water bodies, with significant environmental and health impacts on downstream water users. These municipal systems also release huge amounts of greenhouse gasses such as methane and carbon dioxide, through non-functional flares and soured digesters, dumping of partially digested sludge in drying beds and through anaerobic digestion processes in storage lagoons and settling tanks. Decentralised systems such as constructed wetlands are far more resilient in buffering the effects climatic events and require less maintenance and capital investment and are able to self-regulate in response to climatic changes through feedback loops.

Given the exponential population expansion in developing countries such as South Africa, centralised wastewater treatment is often not possible or economically viable. Decentralised treatment systems can offer simple, easy to maintain, cheap and highly effective alternatives using simple technologies such as a closed septic tank (anaerobic reactor) combined with a FWS constructed wetland. Not only does this allow for effective and sustainable nutrient removal, but it can also provide reusable water resources and allows for nutrient recycling and carbon sequestration rather than solely emissions.

The current national water crisis has exposed the dire state of South Africa’s water resources and has illuminated our dependence on its availability. The vast number of dis-functional and non-functional municipal treatment works that release raw and partially treated sewerage directly into rivers systems has also been brought under the spotlight and has prompted extensive investment and research into developing and implementing new technologies that aim to reduce the water use and limit the environmental footprint of these institutions. Moving forward with sustainable development as the overarching goal, maximising the development of decentralised treatment systems will become key to achieving efficient and effective wastewater treatment, reducing the increasing pressure on municipal wastewater treatment systems and allowing for the reuse and recycling of water and nutrients in the face of climate change and population expansion. The question remains as to how much longer we can afford to use and pollute the large quantities of water needed by centralised wastewater treatment systems in areas and circumstances where sustainable alternatives may use much less water and protect what remains of this already limited resource.

References

Abbot LK and Murphy DV. (2007) *Soil biological fertility: A key to sustainable land use in agriculture*. Springer, Dordrecht, Netherlands.

Acreman MC and Fisher J. (2004) Wetland nutrient removal: a review of the evidence. Hydrology and Earth System Science 8(4): 673-685.

Allen RG, Pereira L S, Raes D and Smith M. (1998) Crop evapotranspiration: Guidelines for computing crop water requirements, *FAO Irrigation and Drainage Paper 56*, Food and Agriculture Organization of the United Nations, Rome, Italy.

Alvarez JA and Becares E. (2008) The effect of vegetation harvesting on the operation of a surface flow constructed wetland. *Water SA.* 34: 645-649.

Angelidaki, I. E. (2003). Applications of the anaerobic digestion process. *Advances in Biochemical Engineering/Biotechnology*, 117-129.

Baker TT, Lockaby BG, Conner WH, Meier CE, Stanturf JA and Burke MK. (2001) Leaf Litter Decomposition and Nutrient Dynamics in Four Southern Forested Floodplain Communities. *Soil Science.* **65**: 1334- 1347.

Bastviken S. (2006) Nitrogen removal in treatment wetlands: Factors influencing spatial and temporal variations. Doctoral Thesis, Linkoping University, Linkoping, Sweden.

Beebe DA, Castle JW, Molz FJ and Rodgers JH. (2014) Effects of evapotranspiration on treatment performance in constructed wetlands: Experimental studies and modelling. *Ecological Engineering* **71**: 394-400.

Bohdziewicz J and Sroka E. (2006) Application of Hybrid wetland systems to the treatment of meat industry water. *Desalinization Science* **1/3**: 33-40.

Borin M and Tocchetto D. (2007) Five-year water and nitrogen balance for a constructed surface flow wetland treating agricultural drainage waters. *Science and Total* Environment **1/3**: 38-47.

Breen PF. (1990) A mass balance method for assessing the potential of artificial wetlands for wastewater treatment. *Water Resources* **24**: 689-697.

Brix H and Schierup H. (1989) The use of macrophytes in water pollution control. *Ambio* **18:** 100-107.

Brix H. (1997) Do Macrophytes Play a Role in Constructed Wetlands? *Water Science Technology* 35 **(5):** 11-17.

Brown LM. (2007) Effect of evapotranspiration on phosphorus sequestration, macrophytes, microbial community and mineralogy in a municipal wastewater treatment wetland. Doctoral Thesis, University of Florida, Gainsville, Florida.

Campbell NA and Reece JB. (2002) Biology, sixth edition. Pearson Education Institute. San Francisco, United States of America.

Campbell Scientific. (1999) Application note: *On-line estimation of grass reference evapotranspiration with the campbell scientific automatic weather station*, Logan, Utah, USA.

Chung AKC, Wu Y, Tam NFY and Wong MH. (2007) Nitrogen and phosphate mass balance in a sub-surface flow constructed wetland for treating municipal wastewater. *Ecological Engineering* **32**: 81-89.

Chynoweth DP, Owens JM and Legrand R. (2001) Renewable methane from anaerobic digestion of biomass. *Renewable Energy* **22:** 1-8.

Clulow AD. (2007) The long-term measurement of *Acacia mearnsii* using Large Aperture Scintillometry. Master's Thesis, University of Natal Pietermaritzburg, South Africa.

Conley DJ, Paerl HW, Howarth RW, Boesch DF, Seitzinger SP, Havens KE, Lancelot C and Likes GE. (2009) Controlling Eutrophication: Nitrogen and Phosphorus. *American Association for the Advancement of Science* **323**: 1014-1115.

Correll DL. (1998) The role of phosphorus in the eutrophication of receiving waters: A review. *Journal of Environmental Quality* **27**: 261-266.

Cowling RH and Potts AJ. (2015) Variation in plant diversity in Mediterranean climate ecosystems: The role of climatic and topographical stability. *Journal of Biogeography* **42**: 552-564.

Cowling RM and Heijnis CE. (2000) Broad habitat units as biodiversity entities for conservation planning in the Cape Floristic Region. *South African Journal of Botany* **67**:15-38.

Cowling RM and Holmes PM. (1992) Endemism and speciation in lowland flora from the Cape Floristic Region. Biological Journal of the Linnean Society 47: 367-383.

Cowling RM. (1984) A syntaxonomic and syntological study in the Humansdorp region of the fynbos biome. *Bothalia* **15**: 175-228.

Cowling, R. 1982. Vegetation studies in the Humansdorp region of the Fynbos Biome. PhD Thesis. University of Cape Town.

Crites RW, Middlebrooks EJ and Reed SC. (2006) Natural wastewater treatment systems. *CRC Press* Boca Raton, Florida.

Cronk JK. (1996) Constructed wetlands to treat wastewater from dairy and swine operations: a review. *Agriculture, Ecosystems and Environment* **58:** 97-114.

Debusk W and Reddy KR. (2005) Litter decomposition and nutrient dynamics in a phosphorus enriched Everglades marsh. *Biochemistry* **75**: 217-240.

DEDEA. (2011) Department of Economic Development and Environmental Affairs. *Department of water affairs and Forestry Report 2882*. Eastern Cape, South Africa.

Dolan TJ, Bayley SE,ZoltekJ and Hermann AJ. (1981) Phosphorus Dynamics of a Florida Freshwater Marsh Receiving Treated Wastewater**.** *Journal of Applied Ecology* **18**: 205-219.

Ellery W, Grenfell M, Grenfell S, Kotze D, McCarthy T, Tooth S, Grunding P-L, Beckedahl H, le Maitre D and Ramsay L. (2009) *WET-Origins: Controls on the distribution and dynamics of wetlands in South Africa.* WRC Report No. TT 334/08. Pretoria: Water Research Commission.

Faulwetter JL, Gagnon V, Sundberg C, Chazarenc F, Burr MD, Brisson J and Stein OR. (2009) Microbial processes influencing performance of treatment wetlands: A review. *Ecological engineering* ***35*****:** 987-1004.

Fennessy MS, Rokosch A, and Mack JJ. (2008) Patterns of Plant Decomposition and Nutrient Cycling in Natural and Created Wetlands. *Wetlands* 28: 300-310

Fontainea B, Mariottib A and Abbadie L. (2003) The priming effect of organic matter: a question of microbial competition? *Soil Biology & Biochemistry* **35:** 837-843.

Galloway JN, Aber JD, Erisman JW, Seitzinger SP, Howarth RW, Cowling EB and Cosby BJ. (2003) The Nitrogen Cascade. *American Institute of Biological Science* **4**: 341-356.

Gilvear D and Bradley C. (2009) Hydrological Dynamics: Groundwater and Hydrological Connectivity. *Wiley-Blackwell Publishers* **7**: 169-193.

Gilvear DJ, Andrews R, Tellam JH, Lloyd JW and Lerner DN. (1992) Quantification of the water balance and hydrogeological processes in the vicinity of a small groundwater fed wetland, East Anglia UK. *Journal of Hydrology* **144**: 311-334.

Greenway M and Wooley A. (2001) Changes in plant biomass and nutrient removal in a constructed wetland, Cairns, Australia. *Water Science and Technology* **44**: 303-310.

Growlland-Gualtieri A. (2007) South Africa's Water Law and Policy Framework: Implication for the right to water. *International Environmental Law Research Centre* **7**: 2-36.

Haigh EA, Grundling PL, Illgner P, Wilmotand J, and Buckle J. (2002) The Wetland Rehabilitation Project in the Kromme River System, Eastern Cape. *Department of Water Affairs and Forestry Project* X832633.

Headley TR and Kadlec RH. (2007) Conducting hydraulic tracer studies in constructed wetlands: A practical guide. *Ecohydrology and Hydrobiology* **8**: 3-34.

Henze M, Harremoes P, Jansen J and Arein E. (2001) Wastewater Treatment: Biological and chemical Processes. *Springer Publishers*, Berlin.

Ho G and Anda, M., (2006) Centralised versus decentralised wastewater systems in an urban context: the sustainability dimension. In: 2nd IWA, *Leading Edge on Sustainability in Water-Limited Environments*, IWA Publishing, 81 to 89.

Hosking J, du Preez M and Sharp G. (2015) Low-income residents' preferences for the location of wind turbine farms in the Eastern Cape Province, South Africa. Journal of Energy in Southern Africa. 26(3): 10-18

Howard-Williams C. (1985) Cycling and Retention of Nitrogen and Phosphorus in Wetlands: A Theoretical and Applied Perspective. *Freshwater Biology* **15**: 391

Jaddhav RS and Buchberger SG. (1995) Effects of vegetation on flow through free-water surface wetlands. *Ecological Engineering* **5**: 481-496.

Kadlec RH and Knight RL. (1996) Treatment Wetlands. *Lewis Publishers*. Boca Raton. FL.

Kadlec RH and Reddy KR. (2001) Temperature effects in treatment wetlands. *Water Environmental Research* **33**: 126-141.

Kadlec RH and Wallace SD. (2009) Treatment Wetlands, second edition. *CRC Press*, Boca Raton. FL.

Kadlec RH, Dunne EJ, Reddy KR and Carton OT. (2005) Constructed wetlands to remove Nitrate. *Wageningen Academic Publishers* **40**: 132-143.

Kadlec RH. (1994) Detention and mixing in free water wetlands. *Ecological engineering* **4**: 1-36.

Kalbus E, Reinstorf F and Schrimer M. (2006) Measuring methods for groundwater – surface water interactions: A review. *Hydrology and Earth System Sciences* **10**: 873-887.

Keller C and Knight D. (2004) Sediment Accretionand Ageing in Treatment Wetlands. *Wetland Solutions Incorporated* **1**: 1-26.

Klopatek M. (1978) Nutrient dynamics of freshwater riverine marshes and the role of emergent macrophytes. *Ecological Processes and Management* **1**: 195-216.

Knight RL, Payne VW, Borer RE, Clarke RA and Pries JH. (2000) Constructed wetlands for livestock wastewater management. *Ecological engineering* **15:** 41-55.

Kusin FM, Jarvis AP and Gandy CJ. (2010) Hydraulic residency time and iron removal in a wetland receiving farraginous mine water over a 4-year period. *Water Science and Technology* **8**: 1937- 1946.

Lee S, Maniquiz-Redillas MC, Choi J and Kim LH. (2014) Nitrogen mass balance in a constructed wetland treating piggery wastewater effluent. *Journal of Environmental Sciences* **26**: 1260-1266.

Lewis CA. (2008) Geomorphology of the Eastern Cape. *South African Journal of Science* **81:** 675-687.

Lloyd CR, Rebelo LM and Finlayson CM. (2013) Providing low budget estimations of carbon sequestration and greenhouse gas emissions in agricultural wetlands. *Environ. Res. Lett*. **8**: 1-12.

Lorenzen B, Brix H, Mendelssohn IA, McKee KL and Miao SL. (2001) Growth, biomass allocation and nutrient use efficiency in *Cladium jamaocense* and *Typha* domingensis as affected by phosphorus and oxygen availability. *Aquatic Botany* **70**: 117-133.

Loupasaki E and Diamadopoulos E. (2013) Comparative evaluation of three attached growth systems and a constructed wetland for in situ treatment of raw municipal wastewater. Journal of *Environmental Technology* **34**: 1503-1512.

Machado AI, Beretta M, Fragoso R and Duarte E. (2016) Overview of the state of the art of constructed wetlands for decentralized wastewater management in Brazil. *Journal of Environmental Management* **X**: 1-11.

Maltby E. (2009) The changing wetland paradigm in: The wetlands Handbook, edited by Maltby E, Barker T, Oxford. *Wiley-Blackwell Publishing* **1**: 3-42.

Mara D. (2004) Domestic wastewater treatment in developing countries. *Earthscan*, London UK.

Marasco DE, Hunter BN, Culligam PJ, Gaffin SR and McGillis WR. (2014) Quantifying Evapotranspiration from Urban Green Roofs: *A comparison. Environmental Science and Technology* **1**: 1-33.

Mencio A, Galan M, Boix D and Mas-Pla J. (2014) Analysis of stream-aquifer relationships: A comparison between mass balance and Darcy's law approaches. *Journal of Hydrology* **517**: 157-172.

Miao SL. (2004) Rhizome growth and nutrient resorption: Mechanisms underlying the replacement of two clonal species in the Florida Everglades. *Aquatic Botany* **78**: 55-66.

Mitsch WJ and Gosselink JG. (2007) Wetlands, 4th Edition. *Ecological Engineering* **34:** 251-262.

Mitsch WJ, Horne AJ and Narin RW. (2000) Nitrogen and phosphorus retention in wetlands: Ecological approaches to solving excess nutrient problems. *Ecological Engineering* **14**: 1-7.

Mthembu MS, Odinga CA, Swalaha FM and Bux F. (2013) A future alternative wastewater treatment technology. *African Journal of Biotechnology* **29**: 179-199.

Muchina L and Rutherford MC. (2006) Vegetation of South Africa, Lesotho and Swaziland. Strelitzia, *South African National Biodiversity Institute*. Pretoria South Africa.

Mustafa A and Scholz KB. (2010) Carbon storage and fluxes within freshwater wetlands: A critical Review. *Wetlands Journal* **30:** 111-124.

Mustafa A and Scholz KB. (2011) Nutrient Accumulation in *Typha latifolia* L. and Sediment of a Representative Integrated Constructed Wetland. *Water Air soil Pollut.* **219**: 329- 341.

National Water Act, Act 36 of 1998. Republic of South Africa. The Department of Water Affairs and Forestry. Government Gazette 19182, Government Printer, Pretoria.

Nedwell DB and Watson A. (1995) CH_4 production, oxidation and emission in a UK ombrotrophic peat bog. *Biochemistry*. **27**: 893-903.

Newman JM, Clausen JC and Neafsey JA. (2000) Seasonal performance of a wetland constructed to process dairy milkhouse wastewater in Connecticut. *Ecological Engineering* **14:** 181-198.

Owen CR. (1994) Water budget and flow patterns in an urban wetland. *Journal of Hydrology* **169**: 171-187.

Page KL, and Dalal RC. (2011) Contribution of natural and drained wetland systems to carbon stocks, CO_2, N_2O, and CH_4 fluxes: an Australian perspective. *Soil Research* **49**:377-388.

Palintest Ltd. Test Instruction Manual. (2012) Water Analysis Technologies. Gateshead, United Kingdom.

Pester M, Knorr KH, Friedrich MW, Wagner M and Loy A. (2012) Sulfate-reducing mocroorganisms in wetlands - fameless actors in carbon cycling and climate change. *Frontiers in Microbiology.* **4:** 1591-1620.

Pienaar GJ and Van der Schyff E. (2007) The reform of water rights in South Africa. *Law Environment and Development Journal* **3/2**: 179-199.

Pritchard D. (2009) Reducing Emissions from Deforestation and Forest Degradation in developing countries- the link with wetlands. *Foundation for International Environmental Law and Development. F.I.E.L.D* Background paper.

Pulley S, Foster I and Antunes P. (2015) The dynamics of sediment associated contaminants over a transition from drought to multiple flood events in a lowland UK catchment. *Hydrological Processes*. University of Northampton, Northampton, United Kingdom.

Rebello AJ. (2013) *An Ecological and Hydrological Evaluation of the Effects of Restoration on Ecosystem Services in the Kromme River System, South Africa*. Stellenbosch; Stellenbosch University (MSc Thesis) [PDF].

Reddy KR and Debusk WF. (1985) Nutrient removal potential of selected aquatic macrophytes. *Journal of Environmental Quality* **14**: 459-462.

Reddy KR, Kadlec RH, Flaig E and Gale PM. (1999) Phosphorus Assimilation in Streams and Wetlands. *Environmental Science and Technology* **29**: 1- 16.

Samie A, Obi CL, Bessong PO, Potgieter N and Igambor EO. (2004) Physio-chemical quality of water supply in rural Venda communities in South Africa. *Water SA* **4:** 515-519.

SANS. (2005) South African National Standard for drinking water quality (SANS 241). South African Bureau of Standards Pretoria South Africa.

SAWS. (2017) South African Weather Services. Available Online at: //www. Weathersa.co.za/.

Schierup H, Brix H and Lorenzen B. (1990) Wastewater treatment in constructed wetlands in Denmark: State of the art. *Pergamon Press* **1**: 495-501.

Shamir E. (1998) Nitrogen accumulation in a constructed wetland for dairy wastewater treatment. Master's Thesis, University of Arizona, Arizona, United States of America.

Smith EL, Gordon R, Madani A and Stratton G. (2006) Year-round treatment of dairy wastewater by constructed wetlands. *Wetlands* **2**: 349-357.

Smith VH and Schindler DW. (2009) Eutrophication Science: Where do we go from here? *Trends in Ecology and Evolution* **24:** 201-207.

Solano ML, Soriano P and Ciria MP. (2004) Constructed wetlands as a sustainable solution for wastewater treatment in small villages. *Biosystems Engineering* **1**: 109-118.

Speece RE (1983) Anaerobic biotechnology for industrial wastewater treatment. *Environmental Science and Technology* **17:** 416-427.

Spieles DJ and Mitsch WJ. (1999) The effects of season and hydrologic and chemical loading on nitrate retention in constructed wetlands: A comparison of low and high nutrient riverine systems. *Ecological Engineering* **14**: 77-91.

Steinbachova-Vojtiskova L, Tylova E, Soukup A, Novicka H, Votrubova O, Lipavska H and Cizkova H. (2006) Influence of nutrient supply on growth, carbohydrate and nitrogen metabolic relations in Typha *angustifolia*. *Environmental and Experimental Botany* **57**: 246-257.

Stottmeister U, Wiebner A, Kuschk P, Kappelmeyer U, Kastner M, Bederski O, Muller RA and Moorman H. (2003) Effects of plants and microorganisms in constructed wetlands for wastewater treatment. *Biotechnology Advances* **22:** 93-117.

Tanner C, Sukias JS and Upsdell MP. (1998) Organic matter accumulation during maturation of gravel bed constructed wetlands treating farm dairy wastewaters. *Water Research.* **32:** 3046-3054.

Tanner CC, Clayton JS and Upsdell MP. (1995) Effect of loading rate and planting on treatment of dairy farm wastewaters in constructed wetlands - II. Removal of nitrogen and phosphorus. *Water Research Commission*. **29**: 27-34.

Tewari DD. (2009) An evolutionary history of Water Rights in South Africa. *Water SA* **35**: 693-707.

Townsend AR, Howarth RW, Bazzaz FA, Booth MS, Cleveland CC, Collinge SK, Dobson AP, Epstein PR, Holland EA, Keeney, DR, Mallin MA, Rogers CA, Wayne P and Wolfe AH. (2003) Human health effects of a changing global nitrogen cycle. *Ecol Environ.* **5**: 240-246

Truu M, Truu J and Heinsoo K. (2009) Changes in soil microbial community: The effect of irrigation with secondary treated municipal wastewater. *Ecological Engineering* **35**: 1011-1120.

Van Deun R. (2015) Treatment Wetlands: Wetlands in Environmental Management. *Agro and Biotechnology*. Thomas More University, Geel, Belgium.

Villa JA, and Bernal B. (2017) Carbon sequestration in wetlands, from science to practice: An overview of the biogeochemical processes, measurement methods, and policy framework. *Ecological Engineering*. ECOENG **4843:** 1-14

Vitousek PM, Aber J and Howarth RW. 1997. Human alteration of the global nitrogen cycle: causes and consequences. Ecol App **7**: 737–50.

Vymazal J. (2007) Removal of nutrients in various types of constructed wetlands. *Science of the Total Environment* **380:** 48-65.

Vymazal J. (2009) The use of constructed wetlands with horizontal sub-surface flow for various types of wastewater. *Ecological Engineering* **35:** 1-17.

Vymazal J. (2011) Plants used in constructed wetlands with horizontal subsurface flow: A review. *Hydrobiologia. 674* **(1):** 133-156.

Vymazal J. (2012) The use of hybrid constructed wetlands for wastewater treatment with special attention to nitrogen removal: A review of recent developments. *Water Research* **47**: 4795-4811.

Vymazal J. (2013) Emergent plants used in free water surface constructed wetlands: A review. *Ecological Engineering* **61**: 582-592.

Wu S, Kuschk P, Brix H, Vymazal J and Dong R. (2014) Development of constructed wetlands in performance intensifications for wastewater treatment: A nitrogen and organic matter targeted review. *Water Research* **57:** 40-55.

Yao F, Sun J, Tang C and Ni W. (2011) Kinetics of Ammonium, Nitrate and Phosphate Uptake by Candidate Plants in Constructed Wetlands. *Procedia Environmental Sciences* **10:** 1854-1861.

Zhang DQ, Jinadasa KBSN, Gersberg RM, Liu Y and Tan SK. (2015) Application of constructed wetlands for wastewater treatment in tropical and subtropical regions. *Journal of Environmental Science*. **30**: 30-46.