An analysis of the ability of small urban wetlands to treat stormwater: the case of Princess Vlei wetland, Cape Town

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ABSTRACT

The consequences of poor water quality on urban aquatic ecosystems have been well established by researchers worldwide. Stormwater management in the urban areas of South Africa predominately focuses on the collection and diversion of runoff into the nearest receiving water body, with little acknowledgement of the impacts on the environment. The City of Cape Town Metropolitan municipality is an authoritative entity on Cape Town pollution and has acknowledged that polluted stormwater is a significant contributor to the deterioration of its' urban aquatic ecosystems due to the persistence of conventional drainage systems. Small urban wetlands are often overwhelmed by the quality of stormwater and urban runoff. Thus water bodies receiving urban stormwater runoff often have elevated loadings of pollutants. In theory wetlands are capable of treating these pollutants and improving water quality through various ecosystem services, but understanding the performance of wetlands under varying conditions is difficult to determine. In South Africa, there is a paucity of studies focussing on the impacts of urban development on small, urban wetlands and thus their ability to provide ecosystem services. This study aimed to identify the surface water quality of Princess Vlei, a small urban wetland, over the past 8 years, and establish the ingress and outflow of the wetland. The pollutant concentrations within the wetland were best explained by the predictor variables of total rainfall and progression of time. Impacts of total rainfall differed with various parameters resulting in larger volumes of water entering the wetland either diluting pollutant concentrations or elevating pollutant concentrations. These inverse trends were proved through the significant correlations found between total rainfall and COD and total rainfall and EC, while the variable of time influenced the wetland's ability to provide ecosystem services, either through the accumulation, retention or flushing of pollutants. The accumulation of pollutants over time was identified through the increasing concentrations off COD and PO_4^{3-}, with the exception of NH_3-N that decreased over time. This implies that the wetland was able to assimilate the NH_3-N but not the COD and PO_4^{3-}. The results did not suggest that the wetland was able to treat the water, as the literature emphasises, rather, confirmed the pervasive impacts of the urban catchment on the ability of ecosystem services to treat water quality in the wetland.

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LIST OF ABBREVIATIONS

ARIES	Artificial Intelligence for Ecosystem Services
BASINS	Better Assessment in Science Integrating Point and Nonpoint Sources
BOD	Biological oxygen demand
CoCT	City of Cape Town
COD	Chemical oxygen demand
DO_2	Dissolved oxygen
EC	Electric conductivity
GIS	Geographical information systems
GLM	Generalised linear model
InVEST	Integrated Valuation of Ecosystem Services and Tradeoffs
SUDS	Sustainable urban drainage
SWAT	Soil and water assessment tool
TN	Total nitrogen
TP	Total phosphorus
TSS	Total suspended solids
WSUD	Water sensitive urban design
WQI	Water quality indicator

1. INTRODUCTION

The Greek philosopher, Plato understood the limitations in the capacity of the environment to provide sufficient resources for a growing population (Brauman et al. 2007). From the 1960s economists began to emphasise the benefits of services provided by natural systems and to quantify the functions of ecosystems services (Daily et al. 2009). A noteworthy report entitled the *Study of Critical Environmental Problems*, published in 1970 presented one of the earliest attempts to define and characterise the attributes of ecosystem services (Mooney & Ehrlich 1997). It was the first piece of work to describe ecosystems in terms of delivering services to humanity and to identify dominance of anthropocentric influence over nature (Mooney & Ehrlich 1997; Ernstson & Sörlin 2013). In the 1970s work on ecosystem services drew attention to wetlands as a rich natural resource with many researchers, mainly in the USA, highlighting the 'functions and values' of wetlands (Maltby & Acreman 2011). However, the concept of 'ecosystem services' was only coined in the 1980s recognising the importance of the link between nature's services and human well-being. (Daily et al. 2009; Sagoff 2011; Ernstson & Sörlin 2013). By the mid-1990s there was increasing use of the concepts such as ecology, resilience, landscape and planning literatures (Ernstson & Sörlin 2013). According to Ernstson and Sörlin (2013), who calculated the number of peer reviewed articles relating to ecosystem services and urban ecosystem services from 1989 to 2011 indicated an increasing trend of academic literature in the use of these concepts from 1998 and 2002 respectively. From 2002 there has been a noteworthy increase in the use of the 'ecosystem services' concept as well as an increase in the use of the 'urban ecosystem services' concept (Ernstson & Sörlin 2013). Ecosystem services have been divided into four main categories that include provisioning, regulating, cultural as well as supporting services which create the necessary conditions for the three remaining services to be delivered by the ecosystem (Brauman et al. 2007). Water treatment is a type of regulatory services that are performed by aquatic ecosystems and potentially provide water quality improvements (.ibid).

Urban development is one of the leading causes of deteriorating water quality in surface waters as well as wetland biological quality (Tong & Chen 2002; Angela et al. 2015; Patenaude et al. 2015; Yang et al. 2015). By early 2000, there was growing interest among researchers to understand the effects of urban development and land-use on the quality and quantity of water (Tong & Chen 2002). Streams receiving urban stormwater runoff were often characterised by elevated loadings of pollutants such as suspended solids, pesticides, nutrients and toxicants (Walsh 2000; Gurnell et al. 2007; Wenger et al. 2009).

It is well known that increased urban land cover alters the natural hydrology of a stream and elevates the volume of pollutants discharging into aquatic environments (Francis 2012; Sibanda et al. 2015). These pollutants are not readily biodegradable and tend to persist in the environment (.ibid). The modification of urban streams and rivers describes the 'urban river syndrome' and resulted in the expansion of scientific research on the health of aquatic environments in urban settings over the last 20 years (Walsh et al. 2005; Wenger et al. 2009).

Walsh et al. (2005) noted that increased urbanisation has prompted an increase in research in urban ecology over the last 20 years. However, more research needs to be conducted on the urban effects on stream ecosystem processes (.ibid). It was acknowledged by Tippler et al. (2012) and Beck and Birch (2012) in recent research that the degree of catchment impervious surface is recognised as an associated factor with the health of urban freshwater streams. It can be assumed that development contributes to the degradation of streams through increased runoff and pollutants but it is the magnitude of these increases on ecosystem service functioning that requires further investigation.

Urban ecosystems are typically classified by a high intensity of demand due to their proximity to the source of disturbances and immediate local beneficiaries to those in a rural setting (Elmqvist et al. 2015). These aquatic ecosystems are sensitive to catchment urbanisation and therefore provide a valuable site for assessment (Walsh 2000; Walsh et al. 2005). Catchment imperviousness and the design of drainage infrastructure are the primary determinants of the quantity and quality of stormwater discharged into streams (Walsh 2000).

Catchment hydrology is transformed by the construction of impervious surfaces and stormwater drainage systems (Gurnell et al. 2007). As seen by Figure 1 water and sediment quality is affected by the input of stormwater as well as point and diffuse pollutants through pipes and sealed drains. In conjunction with catchment water supply input and sewerage leaks these sources of influent have significant impacts on the flow and water quality within an ecosystem (.bid). Contaminant laden- stormwater runoff modifies stream hydrology and alters water chemistry and flow (Figure 1). Variations in stormwater runoff play a role in the ability of an ecosystem to process pollutants over time (Wenger et al. 2009).

As seen by Figure 1, a conceptual model identifies the significance of increased impervious areas on stream ecosystem flow and function (Grimm et al. 2000; Walsh et al. 2005). Human intervention and disruption of natural processes can alter ecosystem functions (Grimm et al. 2000; Walsh et al. 2005). These are defined by as "a suite of processes, such as primary production, ecosystem respiration, biogeochemical transformations, information transfer and material transport that occur within ecosystems" (Grimm et al. 2000: 574).

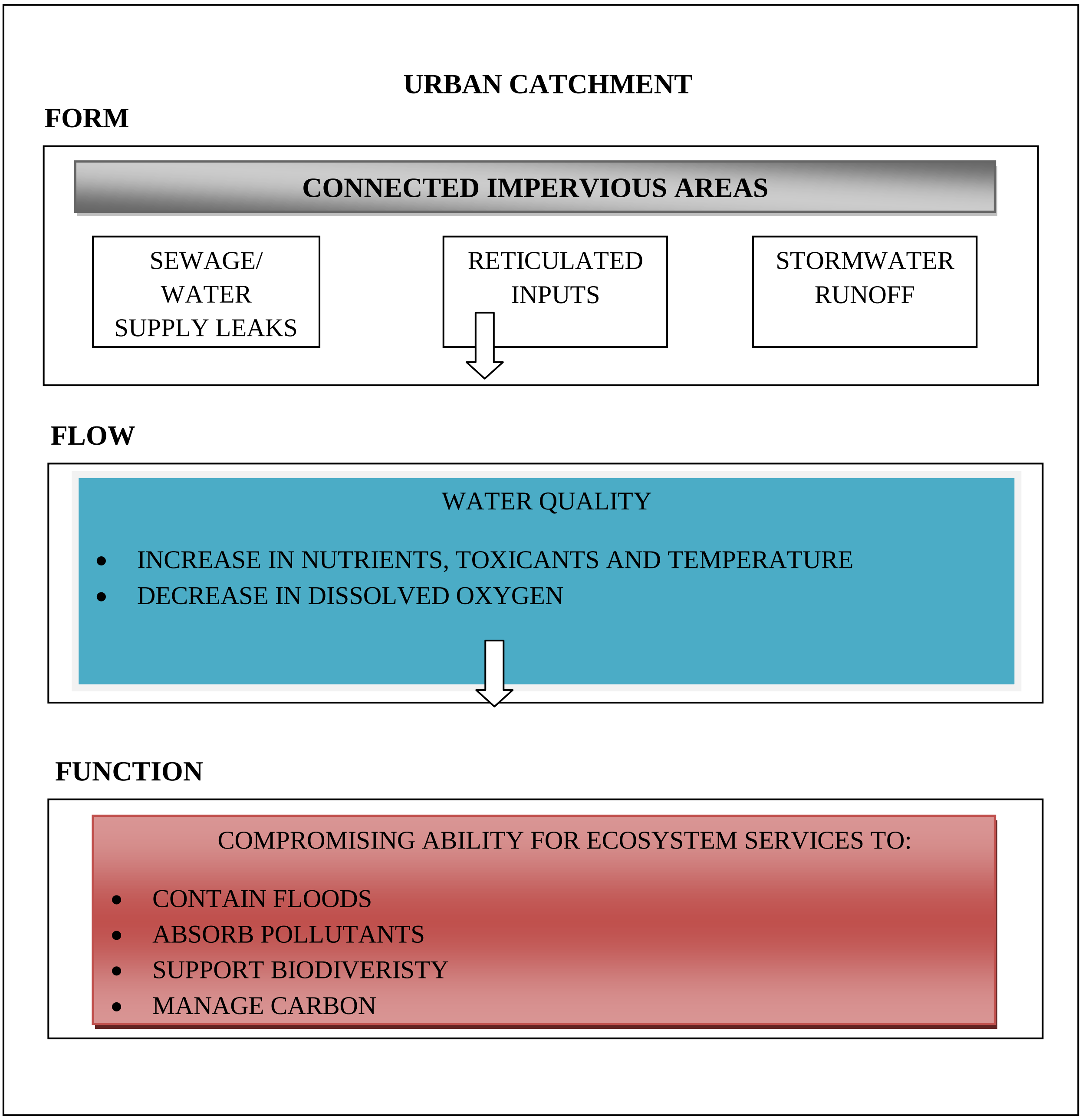

Figure 1: Conceptual model of mechanisms of major urban impacts on wetland ecosystems modified to highlight key areas appropriate to this research (Walsh et al. 2005).

Urban river syndrome describes the concept of degradation to streams that receive water that is draining from urban land (Walsh et al. 2005). Urban rivers are more prone to pollution due to their proximity to multiple sources of pollution. Symptoms include modified hydrographs, elevated concentrations of nutrients and contaminants, altered channel morphology and reduced biotic richness (Mbuligwe & Kaseva 2005; Walsh et al. 2005). Despite the focus on urban rivers, these symptoms can be translated or applied to various aquatic ecosystems within urban settings because the cause of the variation in water chemistry is primarily associated with the supply and characteristics of the pollution rather than the efficiency of the catchment or ecosystem to manage contaminates (Pinto & Maheshwari 2011). However, the impacts of urbanisation extend further than the immediate urban area because energy, water and food demands add a burden to the surrounding area resulting in further ecosystem degradation (Everard & Moggridge 2012).

The urban river syndrome provides a conceptual framework to identify the degree in which the form, flow and function of aquatic ecosystems are compromised as a result of increasing urbanisation (Everard & Moggridge 2012; Booth et al. 2016). The mechanisms that drive the syndrome are complex and interactive and thus the concept is able to illustrate an integrated approach of the factors contributing to changes in flow and function in aquatic ecosystems (Walsh et al. 2005). A combination of factors such as increased imperviousness surfaces and contaminants have degraded urban rivers to such an extent that they cease to provide regulating services (Everard & Moggridge 2012). These regulatory services functioning within a wetland are effective in improving water quality (.ibid). The concept offers a structured link between the catchment and the potential impacts on the final ecosystem service provisioning within an urban river (.ibid). The urban river syndrome concept is applied broadly to describe the impacts of an urban catchment on a receiving water body. Figure 1 does not specify the type of water body but rather the consequences of an urban catchment on the form, flow and function of the water body. An urban wetland is subjected to stormwater and urban runoff which alters its form, flow and function and therefore its ability to improve the water quality of the aquatic ecosystem.

Wetlands generally occupy lower landscape positions and are therefore linked to other ecosystems through hydrologic connections (Faulkner 2004). Habitat alteration in the form of urbanisation on watershed hydrology and nutrient cycling are particularly detrimental to wetland ecosystems (Faulkner 2004; Jacobson 2011). Increasing urbanisation modifies ecosystems and draws attention to changes in water quality through the increased turbidity, nutrients, metals and organic pollutants as a result of domestic or agricultural urban runoff (Tratalos et al. 2007; Tzoulas et al. 2007). However, wetlands are able to process, retain and decompose nutrients and organic wastes from urban effluent and runoff through dilution, assimilation and chemical re-composition (Faulkner 2004; Gómez-baggethun & Barton 2013).

According to Carleton et al. (2001) wetland performance in treating stormwater is generally a function of hydraulic loading rate and detention time. The inflow rate and the quantity of water held in the wetland influences the pollutant retention and determines the portion of runoff that is captured and made available for treatment (Carleton et al. 2001; Jacobson 2011). According to Mitsch and Gosselink (2000) the marginal value paradox describes how increases in human population from low to high result in an initial increase in the marginal per capita value of wetlands to a point and then begins to decrease as wetland functions deteriorate.

Urbanisation in the form of increased impervious coverage converts natural habitats to land uses with potentially poor planning and drainage. This habitat becomes fragmented and inhibits the infiltration of precipitation which contributes to the degradation of downstream ecosystems (Faulkner 2004). It therefore reduces surface storage of stormwater runoff resulting in an increase in surface runoff into wetland (Ehrenfeld 2000). An increase in stormwater discharge relative to base flow discharge enhances erosive forces in stream channels and increases sediment inputs (.ibid). Greater demand on water resources will decrease groundwater recharge resulting in decreased groundwater flow and therefore reduces base flow (.ibid). Impervious surfaces also serve as a transport system that channels pollutants directly into drainage networks and aquatic resources as storm runoff (Faulkner 2004). This results in higher peak flows, reduced time to peak flow, increased runoff volume and diminished baseflow all of which compromise stream habitat quality (.ibid).

Research seeks to understand how wetlands, as biological treatment systems are able to process urban runoff. The wetland is identified as an individually operating system disconnected from the complete urban water cycle and therefore the movement of water through an urban catchment.

1.1 Aim and Objectives

The aim of the research is to determine the potential of Princess Vlei wetland to process and treat contaminated stormwater and urban runoff entering the wetland. Rainfall events and dry antecedent months are considered when determining how the flow and function of natural ecosystems are compromised with elevated loadings of pollutants. Water quality will be used an indicator of the capacity of Princess Vlei wetland to process influent from the urban catchment. As the purification of water is an ecosystem service provided by wetlands this can be used as part of performance criteria to indicate the degree of processing of pollutants within the wetland (Armitage et al. 2013). The research aim is to determine the performance and operating conditions of ecosystem services, namely to evaluate the relative improvement in water quality of an urban wetland over time.

The objectives of the study were as follows:

1. To establish a baseline data record of trends and analyses of water quality of Princess Vlei obtained over a period of 8 years or more as reported by the City of Cape Town.
2. To determine water quality ingress and outflow at Princess Vlei over an 8 year period.
3. To determine whether the regulating services of a small urban wetland are capable of improving the condition of an urban water body or not.

1.2 Context of study

Princess Vlei is a small, shallow, freshwater coastal wetland with an estimated surface area of 35ha and located in the suburb of Retreat in Cape Town, South Africa (Harding 1992; Neumann 2011). Figure 2 indicates the greater catchment of Princess Vlei wetland and identifies that dense urbanisation within the catchment and directly surrounding the wetland. Princess Vlei wetland is located in Retreat, a suburb of Cape Town which has a population of approximately 25000 with an estimated area of 5.27km^2 resulting in a population density of 4900/km^2. Princess Vlei is boarded by other suburbs namely Elfindale, Southfield and Grassy Park. It forms part of the Cape Flats region which has been broadly described as a 400km^2 area of undulating sand with an average elevation of 30m. Princess Vlei is considered one of the four largest Vleis in the Cape Peninsula including Zandvlei, Rondevlei and Zeekoevlei (Harding 1992; Neumann 2011). However, the latter is the smallest and together they form the only substantial naturally-occurring inland water areas available to the public for recreational purposes (.ibid). Most of the Vleis and wetlands overlie the Cape Flats aquifer which has an estimated storage capacity of approximately 53Mm^3/a (Parsons and Harding 2002). At the beginning of the 20th century, prior to a significant increase in urbanisation the area supported numerous wetlands and shallow lakes (.ibid).

The surrounding vegetation has been severely degraded and invaded by alien plants. With the exception of Rondevlei which has been developed into the Rondevlei Nature Reserve and bird sanctuary, Zeekoevlei, Zandvlei and Princess Vlei are primarily used for recreational purposes by the public.

Princess Vlei is a freshwater lake with a catchment of 8000ha and a mean water depth of 2.4m (Neumann 2011). The water table of the Cape Flats aquifer is shallow and ranges widely from a few centimetres below the surface to a maximum of 4m in summer according to the winter-rainfall regime (.ibid). Freshwater sources into Princess Vlei include precipitation, urban runoff and the groundwater contributed from the aquifer. However, one main inflow in combination with runoff feeds the lake (Harding 1992; Neumann 2011). The outflow weir drains into a canal linking Princess Vlei to a parallel wetland (Harding 1992). A combination of canalisation and the degraded natural vegetation in the riparian zone has resulted in increased sediments loads reaching the vlei (Bickerton 1982). The Vlei is bordered by majority residential areas with minimal small-scale industrial areas (.ibid).

On the 27 April 1950, the apartheid government passed the Groups Areas Act which forced the segregation of the different races to specific areas (Princess Vlei Forum, n.d). The apartheid government designated the most attractive natural areas to whites only and the Vlei, due to its beauty and natural environment was to be used exclusively by whites (.ibid). However, confusion arose as it was one of the few recreational spaces that shared borders of both ‘white’ and ‘coloured’ areas. As it was seen as to be located too close to the ‘coloured’ Cape Flats it became one of the only natural open spaces local residents could visit and relax (.ibid). The vlei played a significant role in the surrounding neighbourhood of the vlei who were deprived of access to the majority of Cape Town’s recreational and scenic areas (.ibid). Legend states that the Vlei was named after a Khoisan Princess who was abducted and raped by Portuguese sailors (.ibid). The woman cried relentlessly and thus the Princess Vlei was named (.ibid).

According to Harding (1992) the high concentrations of phosphates and nitrates in the Vlei contribute to its’ eutrophic status. Vegetation bordering the Vlei includes grass species and stands of semi aquatic reeds *Typha Capensis* (hereafter referred to as Typha) that create a dense fringe around the inlet bay. Water Hyacinth (*Eichhornia crassipes*), Africa’s biggest aquatic invader, is known to infest the area (Harding 1992). Typha can be used as an indicator of the ecological value or integrity of wetlands and thus identify negative impacts on the Vlei (Govender 2004). The proliferation of Typha signifies a decline in habitat diversity and biodiversity and is influenced by stream flow and nutrient input (.ibid). In conjunction with poor water quality alien species ring the wetland and limit accessibility to the water. This threatens habitat establishment for vulnerable species such as the critically endangered Cape Flats Sand Fynbos (Harding 1992). Both Typha and Water Hyacinth are often considered as pests as it is known to spread quickly and invasively. However, the plants are able to provide water treatment in an aquatic system as part of the phytoremediation process through the removal or degradation of contaminates in soil, sediments, surface and groundwater.

In 2008 the CoCT produced a Biodiversity Network that included those areas that encompass a feasible minimum needed to conserve a representative sample of Cape Town's unique biodiversity (Ernstson 2011). The Biodiversity Network aimed to evaluate all remaining vegetation due to the majority of threatened plant species in the world, present in Cape Town (City of Cape Town 2016). It was shown that two vegetation types are present in the Vlei, Cape Flats Dune Strandveld and Cape Flats Sand Fynbos which are listed as endangered and critically endangered respectively (Ernstson 2011). This vegetation is only found in small patches where the soil is of a particular characteristic. Due to the presence of the original soil it has been said that the Cape Flats Dune Fynbos can be restored on the Eastern shore of the Princess Vlei (.ibid). The most recent study completed in 2008 upholds that if critically endangered vegetation is present the area must be conserved in order to meet national conservation targets (.ibid).

In the mid 1990s Princess Vlei acted as a flood attenuation pond (Brown and Magoba, 2009). Up until the late 1990s Princess Vlei drained into Rondevlei but the outflow was then diverted through a system of reed beds in order to reduce the pollution before entering the Vlei. Princess Vlei is influenced by the canalisation of both the input and outflow as well the drainage discharged through stormwater pipes. This intervention excludes the wetland as being defined as a natural wetland as described by Ramsar Convention Manual (2013).

Considering the high density urbanisation surrounding the wetland, Princess Vlei is subjected to increased impervious coverage and the associated contaminated pollutants (Figure 2).

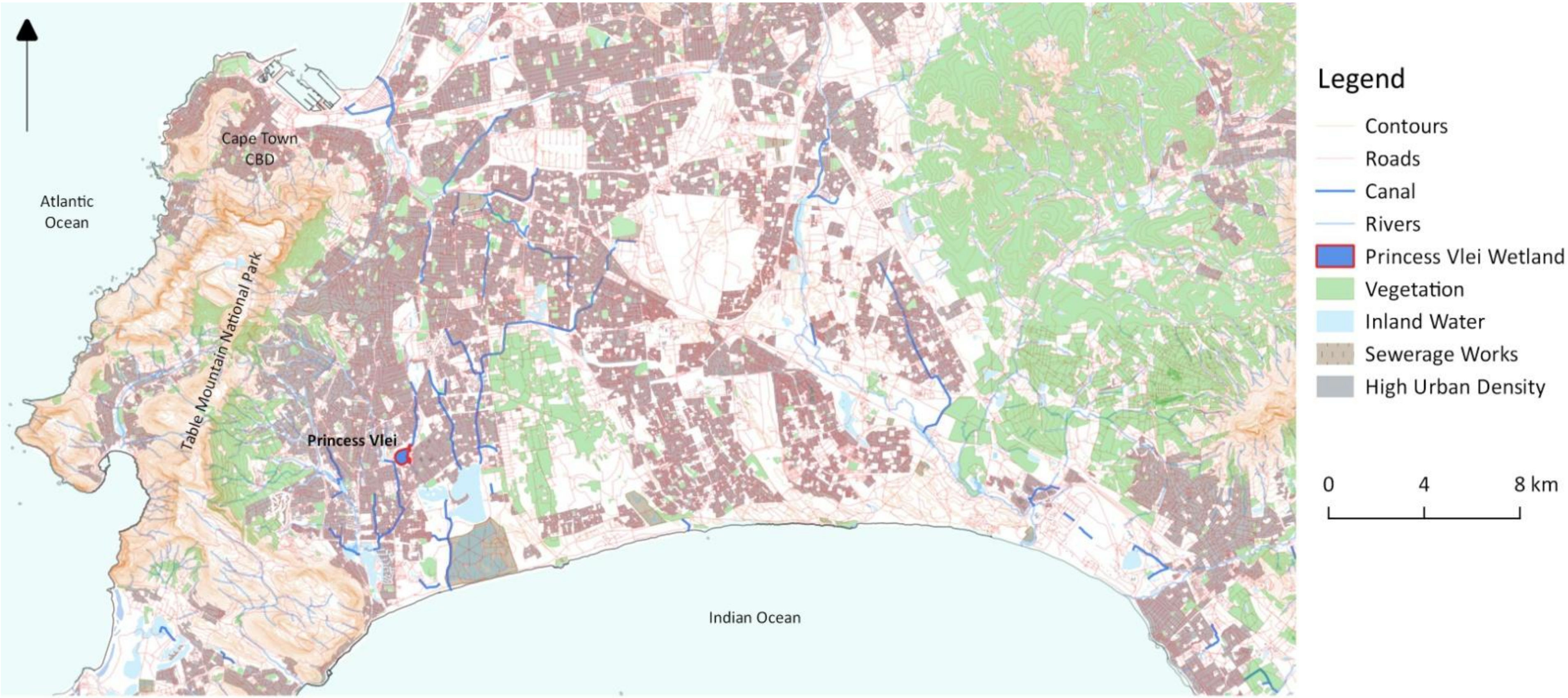

Figure 2: Study area showing land-use in the greater catchment of Princess Vlei wetland

1.3 Overview of research methods

Data was collected at the Princess Vlei wetland between September 2016 and January 2017. Five sampling sites were established namely the outflow weir, inflow weir and three stormwater pipes emptying into the wetland. Princess Vlei wetland is a suitable site to conduct research due to its location within an urban catchment (Figure 2) and potential consequences on water quality from stormwater runoff. In conjunction with primary data collection, secondary water quality parameter data was provided by the CoCT and rainfall data was sourced from a local resident who resides in Bergvliet, the same area as where the wetland is located. The resident used a standard conical rain gauge that was placed in an open area in their property’s garden, away from any potential overhanging trees or other obstacles that could block the rainfall entering the gauge. The rainfall data was consistently recorded throughout the described time period between 2009 and 2016 by a single individual to ensure consistency in data recordings.

1.3.1 Methods overview

The method for each objective is summarised below:

Objective 1: To establish a baseline data record of trends and analysis of water quality of Princess Vlei obtained over a period of 8 years or more as reported by the CoCT

Grab samples were recorded by the CoCT that were taken at irregular intervals and identified various water quality parameters in Princess Vlei wetland. This established water quality trends and patterns as well as provided an idea of potential land-use change in the surrounding area. The health condition of Princess Vlei between 2009 and 2016 of the wetland will be established.

Objective 2: To determine water quality ingress and outflow at Princess Vlei

Similar stormwater capture instruments were placed and secured strategically at the entrance of the three stormwater pipes. This instrument was designed to capture different phases or time periods of stormwater runoff during a rainfall event through the use of three bottles at various elevations. After each rainfall event the samples were removed from the sampling sites for laboratory analysis and the bottles were replaced to capture the subsequent rainfall event. A ‘current flow’ water sample was simultaneously collected using a water scoop within a 12 to 24 hour lag period of a rainfall event at all sampling sites.

‘Current flow’ samples were also collected during the dry antecedent months where the five sampling sites were subjected to minimal flow and the stormwater capture instruments could not be used. Hand held probes were used to determine basic parameters and standard methods were followed to measure nutrients using the HACH Water Analysis Handbook (filth edition) using the HACH DR 2700 portable Spectrophotometer pre-installed programmes.

Objective 3: To determine whether ecosystem services of a small urban wetland are capable of improving the condition of an urban water body or not.

Each water quality parameter gathered throughout primary data collection was compared between its inflow measurement and its outflow point. This has the potential to establish the wetlands removal efficiency therefore its ability to process contaminants. This allowed for an analysis of various water quality parameters between different sampling sites. The data provided by the CoCT established a trend as to the performance of the wetland over 8 years which was used to substantiate the primary data collected.

1.4 Limitations and assumptions

A number of limitations arise when determining the potential capacity of water treatment ecosystem services within an aquatic ecosystem such as Princess Vlei wetland. The wetland is saturated with multiple inflows from various water sources including groundwater, diffused sources and stormwater drainage pipes. Despite a wetland being identified as disconnected, they still have hydrologic connections to other waters through ground water systems. The relative importance of the groundwater contribution is dependent on the hydraulic conductivity of the underlying soils (Leibowitz & Nadeau 2003). Considering this, limitations in the stormwater instrument makes it unable to differentiate between groundwater and stormwater/ urban runoff contributing to the wetland. The positioning of each instruments and its limited capacity to indicate surface water quality will provide appropriate data to determine the overall water quality of the Vlei.

Only certain parameters have been identified as key indicators of water quality in the Vlei and therefore ecosystem service functioning. The performance of ecosystem services is confined by water quality indicators and does not include an analysis of biological indicators.

2. LITERATURE REVIEW

Cities need to be identified and managed as a functional ecosystem especially when addressing the complete urban water cycle. Stormwater is a valued component of this urban water cycle. It has been the primary focus in the implementation of SUDS in order to improve water quality and regenerate aquatic ecological systems. Urban development does however, create a disconnect between natural processes and society. This disconnect results in ecological degradation in urban waterways and poor water quality persistent in urban rivers and wetlands. It has been acknowledged that stormwater runoff is a significant contributor to the degradation of surface water quality and thus a suitable indicator of the health of aquatic ecosystems. This emphasizes the benefits of aquatic ecosystems in a city context in order to develop the diversification of water resources and an overall resilience.

2.1 From conventional drainage to SUDS

Drainage infrastructure is required to meet two converging impacts, that of climate change and urbanisation, the former caused by precipitation extremes and the latter, more especially in developing countries, that of an increase in population growth and development of the urban environment (Zhou 2014). Urban drainage has traditionally been managed to fulfill the primary objectives of flood protection and public hygiene, and more recently, environmental protection (Chocat et al. 2007). Developed countries have roughly achieved the first two objectives through flood relief methods and public awareness campaigns but have shifted their focus and resources to the third objective, environmental pollution control (.ibid). In developing countries, along with securing a safe and efficient water and food supply, flood protection and basic hygiene are still the key concerns (.ibid). Alternative pressing social requirements result in minimal acknowledgement of the health of receiving aquatic ecosystems (.ibid).

Traditionally, the ‘out of sight, out of mind’ attitude drove the perception that stormwater is environmentally benign with little social or economic value (Brown 2005; Wong 2009). The single solution to the drainage predicament established a combined sewer for all runoff and wastewater. These systems are comprised of a large number of structural measures such as concrete pipes and underground basins which are costly and high maintenance (Zhou 2014). This type of infrastructure is increasingly out of touch with environmental values and the greater goal in the pursuit of sustainable urban environments. Notwithstanding a few developments, Brown (2005) notes that the conceptual models employed in most cities have not changed much since Roman times.

Burns et al. (2012) describes two conventional approaches to stormwater management namely ‘drainage-efficiency focused’ and ‘pollutant-load-reduction’ methods. In the past urban stormwater management systems were designed to minimise the risk of flooding and to manage the volume of discharge (Brown 2005; Zhou 2014). These methods supported the traditional views regarding stormwater management and did not acknowledge the significant modifications and severe impacts they would have on aquatic ecosystems (Burns et al. 2012). The majority of impervious surfaces within urbanised catchments are connected to these conventional systems, particularly in developing countries, that effectively route stormwater runoff into receiving waters with little attenuation or treatment (.ibid). Conventional drainage systems focused mainly on water quantity control and had limited capacity and flexibility to adapt to future climatic variability and urbanisation (Zhou 2014).

This common approach has come under widespread criticism from a variety of commentators since the 80s (Brown 2005; Wong 2009; Barbosa et al. 2012). Conventional stormwater management and design has more specifically been criticised due to its costly nature and large-scale infrastructure as well as its facilitation of a design that enhances the wastage of a potentially valuable and typically overlooked resource (Brown 2005).

In response to these limitations, there has been a move towards sustainable strategies for urban water drainage. The movement away from the traditional approach has manifest itself the development of a sewer system where sanitary sewage and stormwater collection are directed into separate systems (Chocat 2007). These separate systems reduce the amount of water needing treatment as potentially clean stormwater can be diverted directly into a watercourse (.ibid). The 1980s was associated with a significant change in thinking where the quality of urban stormwater gained both international and local attention and the assumption that stormwater runoff was environmental benign was questioned (.ibid). The answers to these questions began to offer alternatives to the historic legacy of providing a single solution approach to the urban drainage problem. A sustainable drainage approach also advocates for decentralised or hybrid approaches at the local level that are environmentally sound and that focus on the integration of the urban water cycle into urban design (Cech 2005). Stormwater discussions throughout the 1980s and 1990s focussed on topics including water quality, pollution control and waterway quality. These discussions allowed for the establishment of the link between the quality of urban stormwater and its adverse impacts on waterway environments encouraged a shift towards sustainable stormwater usage (.ibid). It was stated by Delleur (2003) that this paradigm needed to be extended to incorporate the prevention of water quality deterioration.

The ability to design effective drainage systems received increased interest due to its positive effects within the urban landscape (.ibid). The urban water system is a combination of three elements namely the water supply, wastewater and stormwater. These three sources need to be recognised in the shift towards SUDS but more important is the recognition of stormwater as a resource and its management in water scare regions (Walsh et al. 2012; Zhou 2014). Acknowledging stormwater as a resource promotes the 'fit for purpose' approach to water where stormwater could potentially be supplied as a non-potable source of water for uses such as toilet flushing, laundry and garden watering (Wong 2009). The significant impact stormwater has on the flow regimes of urban streams and rivers and the necessity for stormwater management has been emphasised by Walsh et al. (2012).

According to Wong (2009), increased emphasis on improving stormwater quality for the protection of aquatic ecosystems has led to a re-examination of stormwater management practices. Brown (2005) states that urban stormwater can provide a valuable water resource. Despite this, assessments have not often considered stormwater as a suitable alternative source because of its perceived lower reliability (Wong 2009). It has however, been established that stormwater capture enables cities to have greater access to a diverse range of water sources in addition to the well established convention of capturing rainfall-runoff (Wong 2009; Walsh et al. 2012). Sustainable drainage is therefore a departure from the traditional approach towards the management of urban drainage systems.

The evolution towards a more sustainable approach in urban drainage is reflected through the rapid growth in literature and the various newly developed terms associated with this holistic approach (Fletcher et al. 2016). Examples of terminology that evolved in the 90s included 'Best Management Practices" (BMPs), "Low Impact Design"(LID), "Water Sensitive Urban Design"(WSUD), "Sustainable Urban Drainage "(SUDS), notably terms which are more commonly used in the UK. These terms provided support for an increasing societal interest in stormwater management as well as the integration into different approaches (.ibid).

The development and usage of SUDS correlates significantly with the expansion and advancement of water sensitive urban design which took place approximately 20 years ago

(Charlesworth et al. 2003). SUDS emerged in the UK in the late 90s and the term was formalised with working principles in 2000 (.ibid). SUDS promotes the management of stormwater in a sustainable manner by activating its natural behaviours and processes in an urban environment (Zhou 2014). Sustainable management was a result of the evolution over the past decade in the way stormwater is being understood and valued (.ibid). Charlesworth et al. (2003) stated that in order for the unsustainable effects of climate change and urbanisation to be addressed, the adaption of environmental systems needs to imitate or act in combination with natural systems. SUDS are intended to imitate natural conditions in order to create a self-sustaining solution (.ibid). The concept of SUDS aims to address the quality of stormwater discharge into receiving water as well as the people in direct contact with the system. This can be highlighted through the focus on drainage design, the layout of public spaces as well as different transport networks.

The components of the urban treatment train are identified within the development of sustainable urban drainage management systems (Figure 3). SUDS usually consists of a sequence of stormwater practices and technologies that work together to form a treatment train, as it is recognized that one process unit cannot solely provide treatment for a wide range of pollutants from the urban landscape. Sustainable urban drainage includes a number of different approaches in order to manage flow, volume and water quality and to enhance amenity and biodiversity benefits. This suite of components is connected by input sources from stormwater drainage that link the various treatment processing units until it is ultimately discharged into receiving waters (Figure 3).

Figure 3 represents a combination of processes occurring in a sequential manner to optimise treatment as stormwater moves through the urban treatment train. These processes modify the flow and quality characteristics of stormwater runoff and have the potential to improve water quality and ecosystem health. Three key objectives of the SUDS approach are highlighted to effectively provide stormwater management: reduce the quantity of runoff from the site, slow the velocity of runoff to allow for settlement and provide a passive treatment to collected surface water before discharge onto land or into a watercourse (Charlesworth et al. 2003).

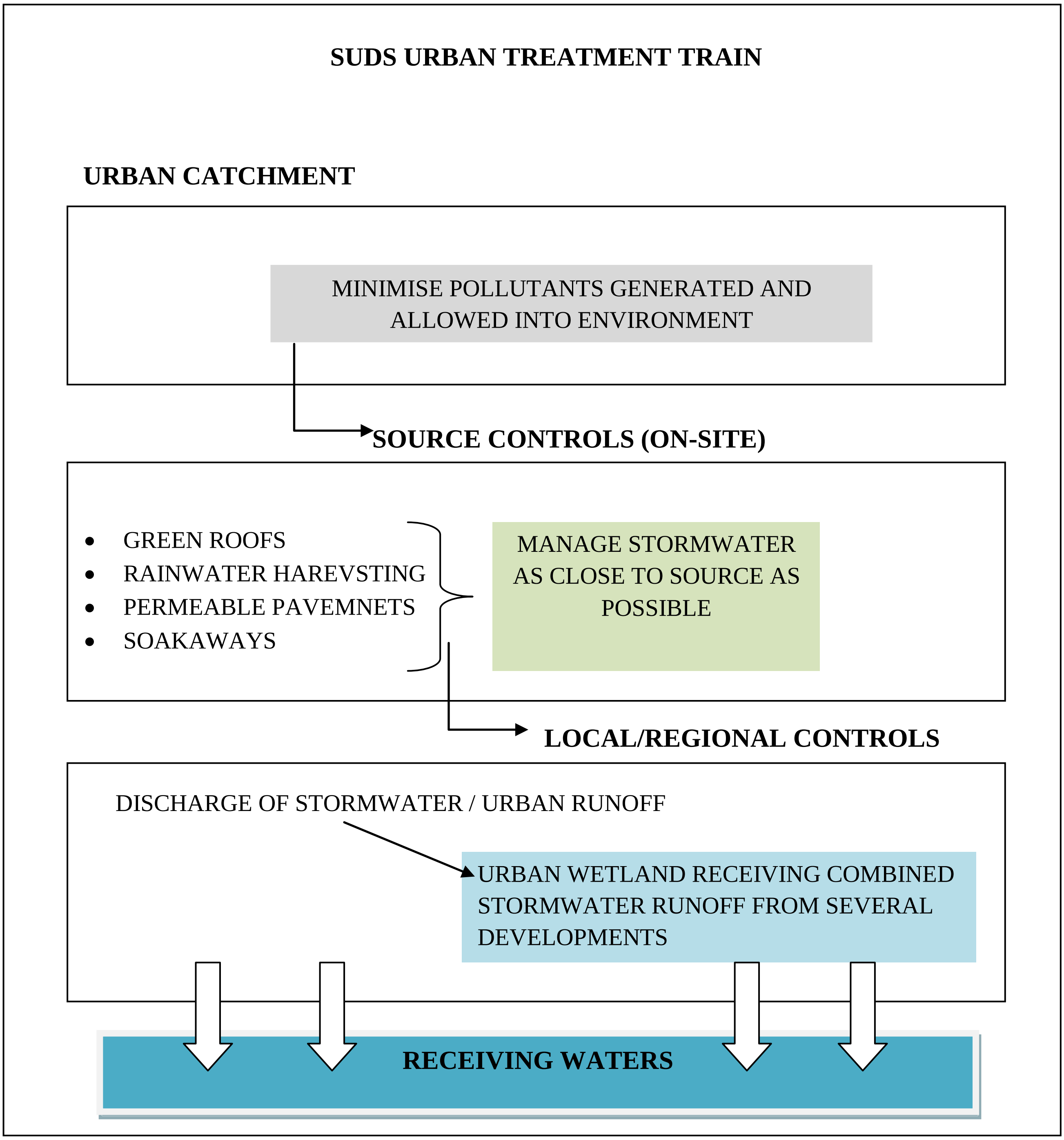

Figure 3: SUDS treatment train to optimally treat stormwater runoff adapted to highlight urban wetlands as SUDS component modified from Armitage et al. (2013).

Recognising the role of wetlands has been highlighted in its inclusion as a SUDS component (Charlesworth et al. 2003). The importance of wetlands at a local scale is reiterated by Niemczynowicz (1999) who states that it is generally accepted that stormwater should be attenuated locally where it can be kept on the surface, treated and potentially re-used (.ibid).

SUDS components rely on local treatment, retention, reuse, infiltration and conveyance of water runoff in urban areas better achieving sustainability principles (Zhou 2014). Examples of various local and regional treatment processes are highlighted in Armitage et al.(2013). The delivery of water is a key element in maintaining and enhancing the services provided by wetlands (Maltby & Acreman 2011). The link between the principles of the SUDS approach and the delivery and promotion of ecosystem services of aquatic ecosystems provides the fundamental focus of the sustainable stormwater management approach. The essential linkage to water supplies also places wetlands centrally in some of the most contentious and urgent issues governing the appropriate management of a resource. This type of resource attracts intense competition and increased uncertainty (.ibid).

These systems have gained growing public interest in recent years as a result of the positive effects on water quality and quantity (.ibid). SUDS have also been used in combination with natural systems to mitigate against flooding and pollution (Walsh et al. 2012). Environmental flow requirements for freshwater systems have broadened the focus of environmental flow management to include all aspects of land and water use in order to combine water quality with flow management (.ibid). Unlike conventional drainage, SUDS are largely able to alleviate the impacts of non-point sources of pollution in urban water bodies (Zhou 2014).

2.2 Ecosystem services and urban drainage

According to Bastian et al. (2012) the theoretical underpinning of urban ecosystem systems is less detailed than for rural or forest landscape in comparison to their relative importance. Similarly, the relative effects of wetland cover and urban development on the quality of the remaining wetland remains unknown (Patenaude et al. 2015). In contrast, Maltby and Acreman (2011) argue that the ecosystem service concept has been developed further in wetlands than any other ecosystem. It does however, remain a challenge to recognise the potential complexity of wetlands in the provision of services when it does not necessarily have the support of sufficiently robust evidence (.ibid). Earlier research in the value of wetlands was focussed narrowly on their description, origins of formation and ecological connections (.ibid). With wetlands assuming greater prominence, there has been improved scientific research and greater recognition of the wider consequences of wetland degradation and loss in wetland functioning and service delivery (.ibid).

In South Africa, there is a paucity of studies focussing on the impacts of urban development and more specifically urban drainage on small, urban wetlands and their ability to provide ecosystem services. In order to advocate effective water quality management of aquatic ecosystems in urban areas, it is important to identify the elements that significantly contribute to pollutant concentrations and elevated loadings (Hatt et al. 2004). According to Hatt et al. (2004) the drainage connections are proposed as a significant variable in explaining the relationship between pollutant concentrations and impervious surfaces. The stormwater drainage system consists of various elements including natural waterways, constructed channels and underground piped systems mainly transporting stormwater to aquatic receiving waters (Wong 2017). The urban drainage system therefore acts as a transporting network for stormwater and urban runoff bringing contaminates from the surrounding catchment into ecosystems such as wetlands. Pollutants running off an urban area are difficult to regulate and measure as they arise from a multitude of activities and can vary with time due to different weather effects (Hatt et al. 2004).

The CoCT has acknowledged that polluted stormwater is a significant contributor to the deterioration of its' urban aquatic ecosystems and in part, also due to the persistence of conventional drainage systems (Fisher-Jeffes & Armitage 2013). In cities with separate sanitary and storm sewage systems it has been determined that stormwater runoff is the primary variable in the degradation of streams and aquatic ecosystems (Walsh 2000). In combination with this, drainage infrastructure has been identified as a significant influence on changes to hydrology arising from stormwater runoff. This further highlights the interconnectedness of drainage systems, stormwater discharge and the provision of ecosystem services.

Water bodies receiving urban stormwater runoff are often characterised by elevated loadings of pollutants. In conjunction with stormwater, effluent from wastewater treatment plants can contribute up to 70% of the flow of urban rivers resulting in increased nutrient loadings and subsequent eutrophication (Matthews 2016). Similarly, Tabayashi and Yamamuro (2009) noted that many studies have reported rivers as major paths that bring contaminants into downstream wetlands causing eutrophication. This further reiterates the 'urban stream syndrome' which describes the degradation of aquatic ecosystem due to elevated contaminates in stormwater and urban runoff effluent. Ecosystem services are potentially able to filter out and decompose organic waste from effluent and thus improve the quality of both a rural and an urban setting (Gómez-Baggethun & Barton 2013). These ecosystem services are translated to wetlands and other aquatic systems and where they act as 'buffer zones' that facilitate the prevention of contaminates from agricultural and urban or industrial sources from reaching main water sources (Jeng & Hong 2005; Maltby & Acreman 2011; Gómez-Baggethun & Barton 2013; Yang et al. 2015). The quality of the stormwater can however adversely affect the health of aquatic ecosystems and therefore inhibit or create an impediment to ecosystem service functioning. This emphasizes the current problem of aquatic ecosystem protection and the demand to develop strategies to cope with pollutants entering the ecosystems (Zhou 2014). Without these sustainable strategies, elevated pollutants in stormwater and urban runoff are limiting the ability for ecosystem to provide efficient ecosystems services.

In theory, wetlands are capable of treating pollutants and improving water quality through various ecosystem services, but understanding the performance of wetlands under varying conditions is difficult to ascertain (Wenger et al. 2009). Increasing degradation of aquatic ecosystems can therefore be attributed to the non-integration of ecosystem services in spatial and land-use planning (Bastian et al. 2012). According to Armitage et al. (2013) the promotion of ecosystem services is directly related to the implementation of the SUDS principles. Ecosystem services can be used as a monitoring system to act as performance criteria in indicating whether a SUDS treatment train is functioning in a sustainable manner or not (.ibid). The guidelines set out by Armitage et al. (2013) promote the primary objective of protection, restoration and the improvement of ecosystems services.

In order to improve water quality in South Africa's urban aquatic systems, catchment wide strategies are required to address the complete urban water cycle (Fisher-Jeffes & Armitage 2013). The poor integration of stormwater management within the urban water cycle is highlighted in the inefficiency of urban drainage and the consequences for ecosystem delivery (.ibid). South African municipalities address the urban water cycle in a fragmented manner resulting in an inadequate and underfunded approach (.ibid). Municipalities need to invest in stormwater management approaches that follow the SUDS approach but this is however, hindered by financial constraints and minimal funding for basic maintenance (Charlesworth et al. 2003; Fisher-Jeffes & Armitage 2013).

This research highlights the ability or inability of a system to accept increased contaminate loadings due to catchment urbanisation and process these contaminates to a certain degree. The performance of the ecosystem is indicated and monitored through the quality of the water which will signify the efficiency of the ecosystem services within the system. As noted by Keeler et al. (2012) water quality is often misrepresented as a final ecosystem service rather than a contributor to many different services. Water quality is therefore the fundamental player, underlying the performance of other provisioning, regulating, supporting and to an extent cultural services. It is clear however that valuing water quality changes is particularly challenging in comparison to other ecosystem goods and services (Keeler et al. 2012).

The challenges in future urban ecosystem service research are comprehensively discussed in Luederitz et al. (2015). They result in a systematic review of the links between ecosystem services and urban development. Luederitz et al. (2015) clearly highlights the underlying complications of high resolution, scientific based research in order to properly understand urban ecosystem services and thus where gaps in research lie (.ibid). Despite the concept of ecosystem services and its application to urban environments gaining attention, the integration of urban wetland ecosystem services in research is inadequately addressed (.ibid). This is potentially attributed to the limited transferability of data and that ecosystem service research between various ecosystems is highly context specific (.ibid). This is further reiterated by Maltby and Acreman (2011) who acknowledge the complex relationships between wetland water regimes and catchment hydrology and the difficulties in extrapolating findings between wetlands.

2.3 Methods of analysing wetland performance

According to Angela et al. (2015) the development of readily measurable hydrological and biological indicators are required to describe the current health status of aquatic ecosystems. This is as a result of a combination of wetland loss due to urban and agricultural expansion as well as poorly managed industrial and household pollution contributing to contaminated urban surface runoff and affecting water resources and biodiversity (Matthews 2016). A number of modelling systems have been employed to investigate the relationship between land-use change and water-related ecosystem services often in conjunction with associated limitations (Angela et al. 2015; Gao et al. 2015; Francesconi et al. 2016).

Vigerstol and Aukema (2011) conducted a comparison between different tools for modelling freshwater ecosystem services including SWAT, InVEST and ARIES. The most common and investigated method is the SWAT model which has been proposed to assist in quantifying ecosystem services (Vigerstol & Aukema 2011; Francesconi et al. 2016). Modelling the volume and quality of water on a daily basis allows for the assessment of ecosystem service functions. Research conducted by Francesconi et al. (2016) compiled all available knowledge on the application of SWAT for addressing ecosystem service issues. It was found that provisioning and regulating processes were the most common ecosystem services evaluated using the SWAT method (.ibid). It should be highlighted that within regulating services, publications primarily analysed water quality to evaluate water pollution and purification processes emphasising its ability to indicate changes in contaminates in water (.ibid). Despite its broad application and its ability to model several hydrologic attributes that underlie water related elements, the model requires detailed data inputs for specific analysis which the user might not have (Vigerstol & Aukema 2011).

Both Vigerstol and Aukema (2011) and Berg et al. (2015) compare and make use of the InVEST tool. This tool models and maps a suite of ecosystem services across the landscape to elucidate general patterns and changes in ecosystem services (Vigerstol & Aukema 2011). This is used to generate and compare future land use scenarios (.ibid). According to Berg et al. (2015) land cover is used as a proxy for ecosystem varieties and InVEST can compare the future extent of certain ecosystem services under varying land cover.

A comparison was conducted by Dennedy-Frank et al. (2016) between InVEST, a simple hydrologic ecosystem service model and SWAT, a widely used ecohydrologic model. The comparison was made using the results the models implemented in two study sites. This analysis identified that the SWAT results were more accurate than the InVEST results because of the daily-scale function (Dennedy-Frank et al. 2016). Despite this, is it noted that SWAT, although estimating hydrologic responses at a higher temporal scale, does not present the results in an ecosystem service context (.ibid). Although Vigerstol and Aukema (2011) recommend the use of the SWAT model if the research requires interest in sediment and pollutants in addition to water yield, comprehensive data input is required which is a challenge in the developing world where data may be scarce (Dennedy-Frank et al. 2016; Vigerstol & Aukema 2011). The SWAT model was also selected by Karabulut et al. (2016) who conducted a study aimed at mapping and assessing water provisioning services. SWAT has been used to evaluate changes in water quality through the estimation of sediment yield as well as nutrient loss outputs such as nitrate loading and total nitrogen (Francesconi et al. 2016). This method was selected on its integration ability particularly related to water quantity, water quality, soil erosion and for testing best practice management (Karabulut et al. 2016).

Despite the use of these models for various water resource decisions, the implementation of these within wetland scenarios to investigate water quality and thus waste water treatment functions of the wetland are scarce (Tong & Chen 2002). Methods are used to identify aquatic ecosystem services and the potential future changes using predicted future scenarios but the complexity of urban ecosystem services needs further research (.ibid). More specifically, the relative impacts of land use on surface water are yet to be ascertained and quantified.

The BASINS tool was implemented to model the potential effects of land use on water quality in a local watershed. This is an integrative tool that combines statistical and GIS spatial analyses with land-use data and outputs from a number of hydrological models within the BASINS tool. Statistical results revealed a significant relationship between land-use and in stream water quality. It is clear that urbanisation has a significant impact on water quality but then the question of how this effects the ability of urban wetlands' to act as natural wastewater treatment systems arises.

Another model highlighted within the comparison conducted by Vigerstol & Aukema (2011) contrasts SWAT , InVEST and the BASINS tools. ARIES is a web-based tool that uses local data input in combination with probalistsic relationships based on data stored from similiar sites(Vigerstol & Aukema 2011). This tool does not solely rely on biophysical relationships and thus the user is able to apply varying levels of data input in order to compare multiple ecosystem services simultaenously (.ibid).

An example of primary data collection through field observations and measurements was implemented in a study conducted in the Rivers of the Magdalena-Eslava sub-basin within Mexico City (Angela et al. 2015). Amongst other indicators physiochemical parameters were sampled four times between September 2012 and September 2013, twice in the rainy season, once in the dry cool season and once in the dry warm season. The indicators were recorded *in situ* using a multi-parameter probe and included water temperature, electrical conductivity, DO_2, pH and discharge flow. This determined the water quality within two urban streams. A study conducted by Jeng and Hong (2005) aimed to assess the fate of selected nutrients and priority pollutants in both sediment and water with the purpose of determining the ability of a wetland system to assimilate nitrogen, phosphorus, carbon and other elements. Water and sediment samples were collected at three sampling points established within the bounds of the wetland. Water samples were collected in 250ml glass bottles but time frames between collections were not stipulated (Jeng & Hong 2005).

Table 1: A summary table to compare the different methods used to model ecosystem services

Modelling tool	Purpose of tool	Variables needed	Limitations	Comments
SWAT (Vigerstol and Aukema, 2011)	• Quantify ecosystem services	• Volume • Quality	• Requires detailed data inputs and daily monitoring	• Provisioning and regulating services most commonly evaluated
InVest (Vigerstol and Aukema, 2011; Berg et al. 2011)	• Ecosystem service patterns and changes • Compare future land use scenarios	• Land-use	• Less accurate results then SWAT	• Land cover used as a proxy for ecosystem services
BASINS (Tong and Chen, 2002)	• Effects of land use on water quality in a local watershed	• Land –use	• Requires significant data input	• Integrative tool that combines statistical and GIS spatial analysis with land-use data and outputs of hydrological models
ARIES (Vigerstol and Aukema, 2011)	• Web-based tool that allows users to evaluate trade-offs between ecosystem services • Identify beneficial stakeholders in area of interest	• All local data available • Stored data based on other sites	• Limited local data input results in an assessment with a high range of uncertainties	• Tool uses probability to uncover relationships between input data and ecosystem service values • Can be applied at varying levels of detail depending on local data input

It is clear from an analysis of freshwater ecosystem services that a range of methodologies in which to determine water quality have been implemented. Although several studies have examined the effects of both urban development and wetland cover on the quality of remaining wetlands, their relative effects remain unknown (Jiang et al. 2012; Patenaude et al. 2015). Similarly, it has been stated by Zedler and Kercher (2005) that there is an abundance of research considering wastewater treatment in constructed wetlands but comparatively few studies concerning water quality improvement in natural wetlands (Shutes 2001; Sani et al.2013; Tu et al. 2014).

It is clear from table 1 that the modelling tool selection is based on what data inputs are needed and available depending on the scale of the research. SWAT for example has been identified as being able to quantify ecosystem services but to achieve this, daily monitoring of volume and quality is required which is not likely to be available in developing countries.

2.4 Water quality as an indicator of performance

Ecological assessments explore how the supply and efficiency of ecosystem services change over time depending on various mechanisms. The study of pollution on biota has subsequently become a matter of interest due to the correlation between pollutants and the alteration of biota (Brander et al. 2006). Physical and chemical methods such as temperature and pH are ideal for instantaneous measurements allowing for short-term analysis. Biological methods are used to monitor long-term environmental variation in the water quality of various natural systems (Giorgio et al. 2016). However, the implementation and spatial and temporal characteristics of either method will influence its efficiency and ability to determine health and therefore water quality (Giorgio et al. 2016). A suite of tools allow for the analysis and determination of the health of different ecosystems and therefore indicate the performance of ecosystems services operating within (Hattam et al. 2015). These techniques measure the general condition of river health influenced by a number of factors but primarily water quality (Dickens & Graham 2002).

There are a number of complex methodologies used in different ecosystem valuation approaches (Pandeya et al. 2016). This makes it useful to explain ecosystem services at a macro level despite a locally relevant evaluation being hindered by data-scarcity (.ibid). This is however more challenging in developing countries where approaches and assessment methods are often diverse in order to accommodate different context and site-specific challenges. Stated by Pandeya et al. (2016) and reiterated by Shoyama and Yamagata (2014) an integrated approach with a well structured foundation is necessary to ensure consistency through assessment methods.

Water, biota and river geography are the endpoints of human induced pollution. According to Pinto and Maheshwari (2011) one can assume that the water chemistry is fundamental to river health assessments due to the consequences of multiple stressors on its quality. Subtle changes in water quality can be identified and measured before they are visible in the biological community and thus better identify the source of pollution (Pinto & Maheshwari 2011).

Water quality indicators can be used to characterise the status and quantify the change of aquatic ecosystems under different disturbance regimes (Wang et al. 2016). The accurate predictions of ecological indicators are essential in improving the understanding of water quality changes in aquatic ecosystems (.ibid). Wang et al. (2016) emphasises the importance of the incorporation of available historical datasets in a data-driven method based on observations although warns against the sensitivity and dependency on the amount and quality of available datasets. Water quality can be assessed with the measurement of the chemicals, pathogens, nutrients, salinity and sediment present in the surface and groundwater (Brauman et al. 2007). These reflect the biological, chemical and physical attributes of ecological conditions (Wang et al. 2016).

A study conducted by Sun et al. (2016) developed a modified water quality index that uses minimal parameters but is able to adequately reflect water quality as well as seasonal changes. An initial WQI was determined by taking into account approximately 15 water quality parameters (.ibid). This was further modified based on a Principal Component Analysis. The modified WQI showed similar results to the original WQI despite it being composed of merely five indices namely; temperature, pH, total suspended solids, ammonium and nitrogen (.ibid). It has been reiterated throughout research that WQI is a practical method to consider critical environmental variables which correspond to the pollution conditions in various water bodies (.ibid).

Water quality is an indication of ecosystem service performance as it influences the behaviour in which organisms respond to adverse conditions. Initial water quality improvements would indicate the evidence of an overall improvement in the ecosystem and synergistic relationships and improvements in biological services. The behaviour of inputs through to outputs will determine the degree and capacity of ecosystems to process contaminates. Water quality partially explains ecosystem performance as one can then assume improved or degraded water quality will move in unison with the state of biological services. Despite the availability of biological tools the use of water quality as an indicator determines variation at the source of the problem. The potential pollutant contamination from urban effluent can be seen initially in the quality of the water and therefore can be assessed immediately.

The use of water quality to determine the performance of a wetlands ability to provide ecosystem services can be confirmed by Keeler et al. (2012). A framework developed by Keeler et al. (2012) links actions to a measured change in water quality and thus its implications on the provision of ecosystem goods and services. Biophysical models such as SWAT and InVEST inform the link and determine the consequences on ecosystem service provision of water quality modifications.

2.5 Developing a resilient city

Urban areas are central to environmental change across multiple scales (Grimm et al. 2008). These can either promote solutions to sustainability challenges or further the degradation and loss of ecosystems (.ibid). The potential of urban ecosystem services for improving city resilience can be realised through the properly managed reconnection between humans and the biosphere (Jansson 2013). Cities within this context can move towards sustainability through the promotion of green infrastructure which combines SUDS components and landscape planning (Ahern 2007). Evolving conceptual frameworks view cities as heterogeneous and dynamic landscapes where the provisioning of ecosystem services links society and ecosystems (Grimm et al. 2008). This illustrates the interaction between landscape structure and function and the degree of connectivity. In highly modified landscapes and in particular urban environments, connectivity is reduced resulting in fragmentation and further impacts on ecological processes (Ahern 2007). According to Ahern (2007), in a human dominated urban environment water connectivity is essential when pursuing sustainability. Societies’ significant reliance on water encourages the maintenance of a connected and healthy hydrological system that is able to provide aquatic ecosystem functions (Ahern 2007). However, Wong (2009) states that a critical challenge in the move towards resilient cities is the sustainable management of water resources and the protection of aquatic environments.

In order to create resilience within a city, there is a necessity to promote the role and value of aquatic ecosystems in an urban environment. This is emphasised by Gomez-Baggethun et al. (2013) who states that cities depend on ecosystems and their components to sustain long-term conditions for health and quality of life. Ecosystems are also able to play a substantial role in the reconnection between humans and the biosphere which is a vital relationship identified by Jansson (2013). Service provisioning is especially relevant in the urban contexts including urban temperature regulation, noise reduction, recreation, reduced air pollution and improved cognitive development. These in conjunction with the potential to improve water quality in an urban environment, are major sources of resilience for cities (Gomez-Baggethun et al. 2013).

According to the 2017 Market Intelligence reports developed by Green Cape, South Africa is ranked as the 30th driest country in the world ear marking it as highly water stressed with extreme climatic conditions and fluctuating rainfall. More specifically, climate models estimate that the Western Cape will get drier and hotter with reduced water availability. Water resources are becoming increasingly vulnerable in cities where water resources are already constrained and water quality is compromised in urban settings. Aquatic ecosystems respond to this by enhancing the capacity of urban environments to deal with environmental shocks (Gomez-Baggethun et al. 2013).

According to Gomez-Baggethun et al. (2013) in order to achieve enhanced sustainability of cities the value of ecosystems needs to be identified through the conservation and regeneration of ecological infrastructure and ecosystem. Strategies that focus on an interdisciplinary approach can combine knowledge from all fields concerned with the urban environment in order to establish an effective framework. Urban communities are therefore seeking resilience to future uncertainties associated with climate change and population growth (Wong 2009). Cities need to look at alternate water sources which will provide access to a diverse range of water sources in addition to the traditional capturing of rainfall. These alternatives allow cities to be flexible and have access to a diversified number of water sources each with their own characteristics of reliability, environmental risk and associated costs (.ibid). A move away from conventional and traditional approaches that draws on the depleting ecosystems and natural environments is essential to establish resilient, water sensitive cities.

3. Research Methods

3.1 Study Design

According to Chandra and Azeez (2010) and reiterated in recent literature, wetlands situated in the vicinity of cities and surrounded by dense urbanisation undergo rapid degradation (Seilheimer et al. 2007; Stander & Ehrenfeld 2009; Abiye 2015). It is well known that wetlands in urban landscapes are subjected to a variety of hydrologic alterations that include quality and quantity (Stander & Ehrenfeld 2009). Linked to this statement, Leibowitz and Nadeau (2003) noted that water quality is one of the least studied characteristics of a disconnected wetland. The purpose of this study is to analyse the performance of a small urban wetland system in order to determine whether it is able to mitigate the impacts of runoff in the case Princess Vlei wetland. Jiang et al. (2012) claims that changes in urban wetlands and there driving forces have become a focus of attention in wetland science and ecological studies. Paul and Meyer (2001) highlighted the link between landscape transformation and the prevalent increase in impervious surface cover in urban catchments. This increase consequentially increases runoff and therefore the discharge of contaminants into streams (.ibid). As seen in Figure 4 and 5 the immediate surroundings (an estimated 2km radius from Princess Vlei wetland) of Princess Vlei is significantly covered by 'high urban density' identifying substantial impervious surface coverage. It is noted that a 75-100% increase in impervious surface cover will result in an increase of surface runoff by greater than 55% prior to urbanisation (.ibid).

Water quality data was used throughout this study to determine the three objectives as described earlier. Multiple water quality parameters were either collected or sourced to establish the degree of ecosystem services or lack of, occurring within the wetland. This resulted in two datasets namely the primary data collected over five months and secondary data that included water quality parameter measurements for Princess Vlei wetland provided by the CoCT.

Primary data was gathered by collecting water samples at five different sample sites (Figure 4). In order to determine objectives listed as number two and three, multiple water quality parameters were analysed including pH, dissolved oxygen (DO_2) and electrical conductivity (EC), total suspended solids (TSS), ammonia nitrogen (NH_3-N), orthophosphates (PO_4^{3}-), nitrates (NO_3-) and nitrites (NO_2-) in order to establish the quality of stormwater runoff entering the wetland as well as the quality of the water leaving the wetland. Primary data collection aimed to differentiate between current inflow and outflow water quality conditions with the influence of rainfall variations. The assumption was that winter and summer conditions potentially influence the manner in which ecosystem services function in an aquatic system. Water samples collected during the winter months were used to determine water quality which was influenced by rainfall and stormwater runoff.

The use of secondary water quality data allowed for the establishment of a baseline data record of trends and analyses of water quality obtained over a period of 8 years at Princess Vlei wetland. Data were collected by the CoCT in two different sampling sites namely PVWEIR and PV03 at Princess Vlei Wetland (Figure 5). PVWEIR indicates the outflow weir and PV03, a representation of conditions in the Vlei as a water body. This contributed towards the first objective by providing patterns and trends over the 8 year period. Water quality parameters represent monthly, usually mid-month, grab samples taken between the years 2009-2016 by the CoCT Scientific Services Division. The data were limited as samples were taken from two sample sites at Princess Vlei at regular intervals which does not take into account changes in environmental conditions. Measurements were often represented as a 'greater than' value which indicates range beyond capabilities of the instruments and therefore excluded these data points from analysis. A limited number of parameters were available in the water quality data provided by CoCT which limited the pool of parameters that could be selected from. Eight parameters were selected from the secondary dataset as representatives of the physical properties of stormwater and urban runoff.

This prompted the selection of similar parameters to be analysed for the primary data as representatives of stormwater properties and ease of comparability with the secondary data.

Rainfall data was sourced from a resident in Bergvliet using a standard conical rain gauge. Bergvliet is a suburb located in the same catchment as the study site but closer to the Cape Peninsula, the Eastern face of Table Mountain with a steep elevation of approximately 1085m. This area is subjected to higher rainfall due to orographic rainfall conditions related to Table Mountain. The daily rainfall data was consistently recorded throughout the described time period between 2009 and 2016 by a single individual to ensure consistency in data recordings.

3.2 Project development

3.2.1 Study site selection: Princess Vlei wetland

The Cape Flats region has been identified by Kruger (2004) as part of the South Western Fynbos-type climatic region. The resistant sandstones of the Table Mountain has given rise to steep cliffs on the Peninsula as opposed to the undulating characteristics of the Cape Flats (Harris et al. 1999). The geographical and topographical features influence the climate of the region, a typical Mediterranean climate (.ibid). Precipitation generally occurs between May and August with highly localised microclimates (Harris et al. 1999; Kruger 2004). Rainfall is unevenly distributed due to the mountainous nature of the area with the Cape Flats receiving between 590-980mm/yr and the Cape Peninsula receiving approximately 1200mm/yr (Kruger 2004; Neumann 2011).

It has been previously mentioned that the Cape Flats region has been significantly impacted by human activities attributed, but not limited, to urban sprawl and the introduction of exotic species which are out-competing natural vegetation such as Cape Flats Sand Fynbos and Cape Flats Dune Strandvlei (Bickerton 1982). The majority of the area surrounding the Vleis has been converted from its natural state for agricultural purposes, residential housing as well as industrial activities (.ibid). According to Bickerton (1982) the upper catchment is characterised by urbanisation resulting in increased impervious surfaces and canalisation and thus a proportionate increase in runoff via stormwater pipes. Despite most of the Vleis being recognised for recreational purposes Princess Vlei is of particular importance both historically and culturally.

According to Chandra and Azeez (2010) inflows of wastewater carrying elevated loadings of nutrient can overwhelm its natural assimilation and capacity of the water body over time. Natural wetlands located within cities near urban, industrial and agricultural activities may experience higher levels of degradation due to contaminated effluent discharges. The study area was chosen because of its location within a formal urban area dominated by residential areas and some small scale infrastructure. It has varying degrees of drainage infrastructure, impervious surface coverage and the adjacent highway which together have the potential to increase runoff into the wetland It is also linked within its catchment to a variety of land-uses however the majority indicates high urban density, followed by industrial areas and then inland lakes and sewage works.

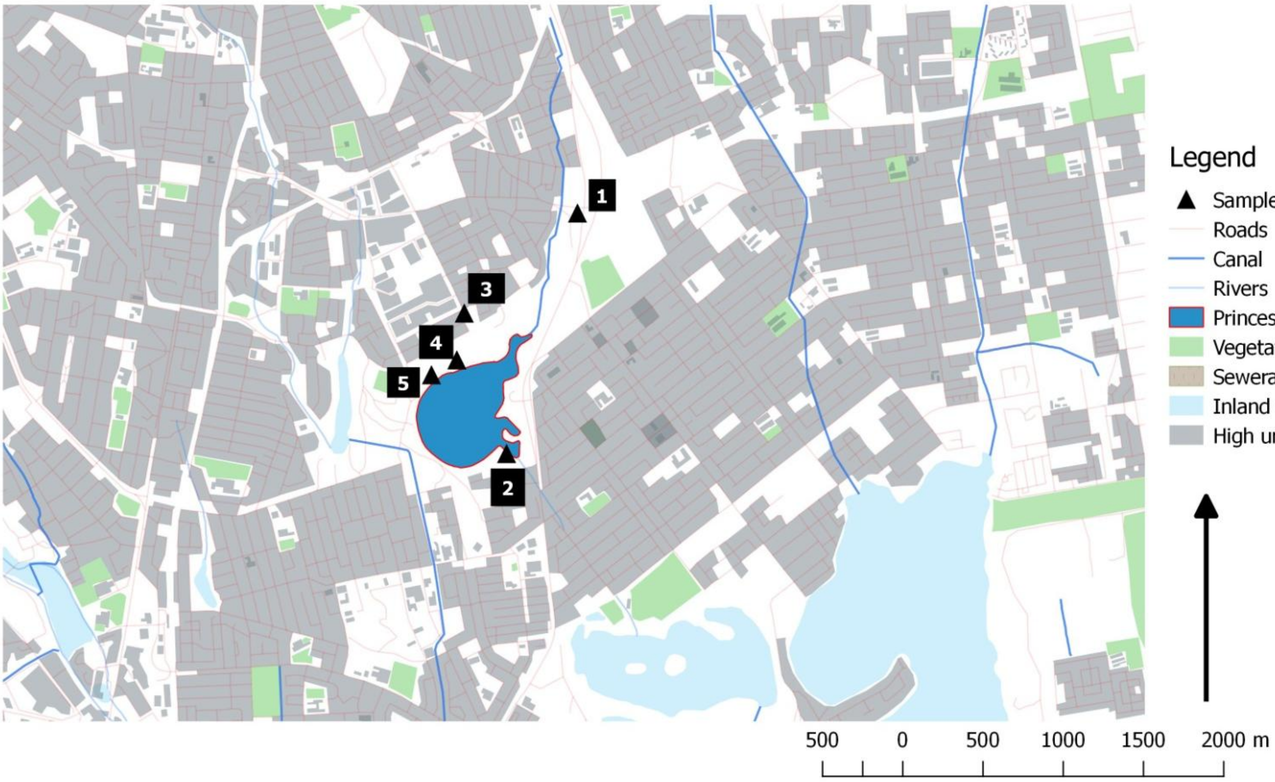

Figure 4: Study area highlighting sample points used for primary data collection

Figure 5: Study area highlighting sample points used for secondary data collection

3.2.2 Sample site selection

As previously stated freshwater sources flowing into Princess Vlei include precipitation, river and stormwater runoff as well as groundwater from the Cape Flats aquifer. According to Parsons and Harding (2002) of the 550mm mean annual precipitation, approximately 30% contributes to groundwater recharge. This flows in a southerly direction in the vicinity of Princess Vlei wetland (.ibid). During the summer months when rainfall and river inflow is limited, groundwater is often the only source of freshwater recharging the wetland (Parsons & Harding 2002). However, water can drop to several metres below the surface in very dry conditions resulting in the exposure of sandy beaches along the shoreline (Harris et al. 1999; Neumann 2011). Rivers arising from the surrounding mountain ranges feed into some of the Vleis however, most of the streams and rivers are non-perennial and dependent on winter rainfall and surface runoff (Harrison 1962).

Of the five chosen sample sites, four represent influent pipes feeding the wetland. Input from the catchment is directed into stormwater pipes indicated as sample sites three, four and five before emptying into the wetland (Figure 6). These stormwater channels are located on the northern perimeter bank of the wetland and are influenced by both residential and industrial land use activities. Sample sites four and five are not photographed due to bad visibility of the stormwater pipes. Sample site one namely the Southfield canal is a weir approximately 1km upstream from the main water body (Figure 7). This canal was built to drain surrounding residential areas which further drains into Princess Vlei wetland. In addition to the urban stormwater runoff the canal receives sewage effluent from the Cape Flats sewage works.

Sample site two is a canal which is located South East of Princess Vlei and serves as a drainage outlet (Figure 8). During high levels of rainfall Princess Vlei connects with Rondevlei (a small lake to the West of Zeekoevlei) via a flood control weir. The outflow canal provided a suitable sampling site that indicates ‘the end of the line’ whereby the water had passed through the system and been subjected to ecosystem services. It is assumed that the outflow weir water has been processed by the wetland system as a whole and that water quality sampling at this point is some measure of how ecosystem services have processed, polished and treated the quality of water.

In order to determine an overall understanding of the performance of Princess Vlei wetland, water samples were used to determine the degree of functioning ecosystem services related to water treatment within the wetland. Due to difficulties in separating the groundwater contribution to the runoff input attributed to the 30% groundwater recharge, surface water samples were used to indicate the overall water quality of the wetland.

a) b)

Figure 6: Sample site three a) front view b) towards Princess Vlei

a) b)

Figure 7: Sample site one a) front view b) towards Princess Vlei

a)

b)

c)

d)

Figure 8: Sample site two a) front view b) back view c) drainage outlet d) towards Princess Vlei

3.3 Parameter selection

Water quality parameters where chosen due to their relative importance as stormwater contaminants and their ability to indicate biological and ecological processes in aquatic environments. However, further evidence to aid in parameter selection was guided by the Department of Water Affairs and Forestry who established the South African Water Quality Guidelines for Aquatic Ecosystems (Volume 7). These are a derived set of water quality criteria to protect freshwater ecosystem. These guidelines govern the surface water quality required to protect fresh water aquatic ecosystems by providing quantitative and qualitative criteria for a list of water quality constituents. From these guidelines a comparison was made with the secondary data set and parameters were selected based on the water quality parameters that were similar. The parameters were selected and analysed and used for the primary data collection.

The parameters chosen for sampling and analysis included water quality constituents within the system variables, non-toxic inorganic and nutrients categories thus ensuring selection covers an adequate basis of all water quality variables. System variables were chosen including pH, dissolved oxygen (DO_2) and electrical conductivity (EC) which combined are able to regulate essential ecosystem processes. The natural concentration of total suspended solids (TSS) is generally dependent on localised physical and hydrological processes and is selected as a representative of the aquatic ecosystems characteristics. These variables were used to determine the physical properties and conditions of the stormwater and urban runoff flowing into the wetland. Selected nutrients were also examined, including ammonia nitrogen (NH_3-N), orthophosphates (PO_4^{3}-), nitrates (NO_3-) and nitrites (NO_2-). This is significantly linked to the stimulation of eutrophication and will represent potential elevated loadings of nutrients in the system which can be attributed to changes in land- use activities.

3.4 Instrument design

Different sampling techniques were undertaken to consider flow rates, runoff levels and the infrastructure of the sampling site. In the construction of the instrument these aspects were considered in order to ensure the accurate collection of primary data. A combination of two forms of instruments were used namely the ‘stormwater capture’ and ‘water scoop’ in order to consider various flow rates entering the wetland. This also assisted in ensuring a water sample could be collected by combating a potential issue of minimal flow rates during the dry summer months. These instruments were designed using basic, inexpensive materials in order to take into account the context of Princess Vlei wetland and surrounding social structure where equipment could potentially be stolen or vandalised. It was determined materials used to assemble them needed to be relatively cheap and easily concealed from sight.

3.4.1 Stormwater capture

Initially a trial and error approach was adopted in terms of instrument design in order to ensure it could cope with the elevated flow of water as a result of a storm event as well as the ability to collect stormwater runoff as opposed to natural flow. These were two essential requirements. The instruments consisted of three standard size bottles attached along a PVC pipe (Figure 8). Bottle one and two were connected to the pipe using ‘T’’ joint fittings whereas the third bottle making up the ‘stopper’ at the end of the pipe was attached via an ‘L’ joint fitting. All bottles were at approximately 20cm intervals. It was assumed that the three bottles could capture various stages of the stormwater runoff and that the third bottle being the ‘stopper’ would illustrate higher levels contamination according to the first flush theory. The instruments in sampling sites three, four and five were attached to the stormwater pipe (or as close as possible to the initial flow) using dowel rods and cable ties. The bottles were secured at various elevations sloping down from the stormwater pipe in order to take advantage of gravity. Instruments could not be attached at sampling sites one and two namely the inflow and outflow weir due to the surrounding infrastructure.

3.4.2 Water sample scoop

A simple ‘scoop’ was designed in order to take samples easily and efficiently (Figure 9). The top of a plastic 300ml bottle was removed and attached with cable ties to PVC pipe (approximately 2m in length). The ‘scoop’ allowed for a sample to be taken regardless of flow rates entering into the wetland.

Figure 9: Example of stormwater capture instrument secured in sample site three

Figure 10: Various angles of water sample scoop

3.5 Sampling

The stormwater capture instrument is used to capture and store stormwater runoff post storm event. Three instruments were secured to sample sites three, four and five which were represented by stormwater pipes. Sample site one and two did not provide adequate infrastructure to enable the attachment of an instrument to the weir, visibility and challenges in positioning the instrument to collect stormwater runoff as oppose to natural flow. The 'current flow' was taken at these two sites throughout the field study.

Primary data was collected by undertaking field studies during winter months including September and October 2016 post rainfall events as well as during summer months between November, December (2016) and January (2017) to determine the wetlands reaction to the dry antecedent season. Each sample site called for a combination of sampling procedures due to varying seasonal conditions.

This is detailed below:

3.5.1 Sample sites one and two

- September and October 2016: current flow samples were taken using the water scoop post rainfall events.
- November, December 2016 and January of 2017: current flow samples were taken on a weekly basis at both the inflow and outflow weir. These samples were identified as 'current flow'.

3.5.2 Sample sites three, four and five

- September and October 2016: samples were collected using the stormwater capture instrument post rainfall events. The three bottles attached to the instruments were emptied into sample jars and the bottles were re-secured onto the instrument. Data was ad hoc using the instruments due minimal flow. The 'current flow' was simultaneously collected unless there was no visible or accessible sample to collect.
- November, December 2016 and January of 2017: current flow samples were taken at sample site three as sample site four and five were dry. These samples were identified as 'current flow'.

3.6 Laboratory methods

Various methods were used to quantify water quality variables in order to produce primary data. Duplicate methods including hand-held probes and standard methods using pre-installed programmes using the HACH Spectrophotometer were used to analyse each sample. Variables were measured at the water analysis laboratory at the University of Cape Town.

3.6.1 Chemical analysis

Basic parameters were measured using a combination of hand held probes

1. pH – Martini Instruments (pH 55)
2. Dissolved Oxygen (DO_2) – Milwaukee (MW 600)
3. Electric Conductivity (EC) – Hanna (HI 8733)

Nutrients were measured following standard methodology set out in the HACH Water Analysis Handbook (filth edition) using the HACH DR 2700 portable Spectrophotometer pre-installed programmes.

4. Total Suspended Solids (TSS): 630 Suspended Solids
5. Ortho-phosphates: 490 P React. PV
6. Ammonia Nitrogen: 385 N, Ammonia, Salic
7. Nitrates: 352 N, Nitrate MR PP

3.7 Data analysis

Both regression and linear statistical analysis were conducted using IBM SPSS Statistics 24. Initially, primary, secondary and rainfall datasets were analysed individually. A mean table was produced for the secondary water quality data parameters comparing the mean, standard deviation, standard error of mean, minimum and maximum at both sites (PV03 and PVWEIR). The data was then categorised into four seasons. December, January and February months were classified as Summer, March, April and May were classified as Autumn, June, July as August as Winter and September, October and November as Spring. This allowed the changes in water quality parameters to be depicted over time by season. Summer and winter seasons were chosen to show extreme seasonal changes. A similar mean table was produced for the water quality parameters from the secondary dataset comparing the mean, standard deviation, standard error of mean, minimum and maximum at both sites (PV03 and PVWEIR) and by all four seasons.

3.8 Data analysis discussion

The limitations of both datasets inhibited a more comprehensive analysis of temporal water flow variability at a finer weekly and/or daily scale. This would have improved the understanding of variable inflows and fluctuating water quality as well as the response to this by the regulatory services performing within Princess Vlei wetland.

3.8.1 Primary dataset

The context of Princess Vlei wetland and use of various instruments their effectiveness and ability to produce feasible data may have been restricting. The stormwater capture instrument is limited to sampling that is aimed at surface runoff and therefore cannot distinguish between the impacts of the groundwater contribution and diffused sources of water. Similarly, the inability to monitor the instruments once placed in the field during and post rainfall events resulted in some doubt as to whether the instrument was purely capturing the stormwater runoff as expected. It was also assumed that the positioning of each instrument was correct and would allow for the capture of stormwater runoff rather than natural flow. Due to the high visibility of the sample sites which were used as a walkway one was unable to place more sophisticated type sample stormwater instruments. The need to collect samples in summer proved difficult due to the minimal flow rates entering into the Vlei. Sampling was therefore a trial and error process which has to take into account summer and winter rainfall conditions which directly affected flow rates. As indicated by Table 2, all water quality parameters indicated a large number of missing values. This can be attributed to a combination of low flow rates and time restraints in laboratory analysis of collected samples.

Statistics

		pH	DO	EC	TSS	OP	AN	N
N	Valid	61	58	58	57	61	30	11
	Missing	30	33	33	34	30	61	80
Mean		6.989	4.090	399.72	12.96	.3666	.3063	.945
Std. Deviation		.5520	1.3184	143.861	14.422	.49794	.34055	.6729
Minimum		6.1	.6	91	-4	.02	.00	.1
Maximum		8.3	6.3	731	70	2.94	1.60	2.1

Table 2: Frequency table produced for primary dataset

3.8.2 Secondary datasets

The secondary water quality data received from the CoCT was incoherent whereby measurements were recorded as ‘less than’ values and not as an absolute value. As indicated in Table 3, Ammonia as Nitrogen, Nitrate and Nitrite as Nitrogen and Ortho-phosphates had the three highest recorded non-absolute values at approximately 55, 47 and 63 respectively. These values had to be excluded from the analysis reducing the number of data points.

Statistics

		AN	COD	EC	DO	N	OP	TP	TSS
N	Valid	170	219	223	222	178	162	221	225
	Missing	55	6	2	3	47	63	4	0
Mean		.075229	36.26	58.622	8.0876	.21131	.02739	.17487	32.222
Median		.035500	36.00	58.000	8.5000	.07700	.01550	.16400	29.000
Mode		.0100	26[a]	54.0	8.60	.002[a]	.010	.127[a]	22.0
Std. Deviation		.1328791	9.849	11.8240	2.37651	.334648	.037798	.089762	15.7992
Minimum		.0000	14	26.0	.30	.000	.000	.039	1.0
Maximum		.9290	65	99.0	17.00	1.690	.224	.988	102.0

a. Multiple modes exist. The smallest value is shown

Table 3: Frequency table produced for secondary dataset

Statistics

RAINFALL

N	Valid	426
	Missing	0
Mean		15.44
Median		9.00
Mode		1
Std. Deviation		16.988
Minimum		1
Maximum		100

Table 4: Frequency table produced for rainfall dataset

Rainfall data provided a comprehensive pattern of daily rainfall from 2009. As indicated by Table 4 there were 426 days of rainfall between April 2009 and November 2016 with an average rainfall of 15.44mm.

4. Results and discussion

In order to assess the ability of Princess Vlei wetland to function as a water treatment system, concentrations of water quality parameters were measured. According to Cech (2005) surface water runoff from precipitation flush a range of contaminants from the urban environment including streets, construction sites, agricultural fields, golf courses, factories and sediment. These pollutants join the water system at stages and are ultimately washed into the receiving bodies such as streams, rivers and lakes. The amount and type of pollutants carried in stormwater runoff is dependent on land use, intensity of rainfall and time between rainfall events (Greenway 2010). Rainfall plays a role in stormwater quality and quantity, as time between rainfall events fluctuates this will influence whether concentrations of pollutants will accumulate or disperse.

4.1 Overview

The ability of a wetland to provide ecosystem services can be partially determined by the capacity of the wetland to process pollutants. Two different statistical techniques were applied to the secondary dataset and used to determine the capacity of the wetland to clean water. The statistical models used were determined according to their distribution in order to take into account the variation between the water quality parameters within the dataset. Multiple regression models use several explanatory variables to predict the outcome of a response variable. This aims to model the potential relationship between the explanatory variables (water quality parameters) and predictor variables (Table 5). The explanatory variables were selected as indicators of fluctuating water quality and the predictor variables were selected based on the potential influence of these on the water quality parameter concentrations. Water quality parameters were plotted to determine their distribution. If the parameter demonstrated a normal distribution this then supports a linear regression analysis that could be fitted to the water quality parameter. A GLM is a flexible generalisation of linear regression that enables the response variables that have distribution other than normal distribution. This was fitted to water quality parameters that displayed a skew distribution. Once the model was selected, the significance value was ascertained between the water quality parameters and predictor variables.

COD, EC, NH_3-N and PO_4^{3-} indicated an overall model significance as well as significant relationships between the water quality parameter and combination of time, month or total rainfall as predictor variables. As none of the models indicated significance between the response variable and the predictor variable ‘sample point’ it can be assumed that there is no significant difference in pollutant concentrations between PV03 and PVWEIR as initially expected (Figure 5). This was determined by using sample site PV03 as a reference point whereby PVWEIR was compared too. A cycle plot was produced for each water quality parameter. This plotted the absolute values collected in January between 2009 and 2016 as a single month with a similar method applied to each consecutive month. This produced a graph representing each sample measurement taken between 2009-2016 and allowed for seasonal trends to be established over a single year.

The positive or negative significant relationship indicated the effects time had on the concentrations of pollutants. Time indicated 12 months from January to December and this indirectly used months of the year as a proxy for seasonal variation over a year. The 12 months illustrated the change in season and the change in temperature and how that influenced the concentrations of pollutants in the wetland. A significant positive relationship implies that pollutant concentrations would increase overall in concentrations. A significant negative relationship would then imply that pollutant concentrations would decrease overall.

Representation	Predictor variable	Explanation
spoint	Sample point	Used as an indicator variable to determine differences in concentration between PVWEIR and PV03. Where PV03 is used as a reference point (Figure 5)
month	Month (January-December)	Indirectly indicates season
site_month	Sample site versus month interaction	To determine whether a single month effected concentrations at a specific sample site
sin_12	Period T=12 (over 12 months)	Used to fit the periodic structure of the data
cos_12	Period = 12 (over 12 months)	Used to fit the periodic structure of the data
time	Progression of time	Whether concentrations increased or decreased over time
Tot-rainfall	Total rainfall	The rainfall that fell before a sample was collected (and post the previous sample) was totalled. Rainfall was then attributed to both sample sites on the date the sample was collected

Table 5: Predictor variables used in regression analysis equations

4.1 Model diagnostics

Histograms were generated to determine the frequency of data for each water quality parameter. This then identified the distribution of the data and the selection of the model that fit the distribution (Appendix 1a: Mixed regression Model Diagnostics- Histograms). Residual analysis was undertaken to determine how well the models represented the data and whether the underlying assumptions were met or not (Appendix 1b: Mixed regression Model Diagnostics- Residuals).

4.1.1 Linear Regression

A linear regression model was selected for COD, EC, DO_2, TP and TSS with the assumptions that:

- The relationship between the water quality parameter and the regressors is (approximately) linear
- The errors are normally distributed

The assumptions of the model are evaluated by plotting the residual values which were determined by calculating the deviation between the observed values and the predicted values. Residual values reflect the deviation between the data and the fit and are a measure of the variability in the response variable that is explained by the model. The residual plots were checked for:

- Normality using the kernel density plot: a symmetric bell-shaped curve
- Constant variance using the scatterplot of the residuals against the fitted values: the residuals are contained in a horizontal band.

Residual analysis showed no serious violation against these assumptions (Appendix 1b: Mixed regression Model Diagnostics- Residuals).

4.1.2 Gamma Regression

A gamma regression model was selected for NH_3-N, PO_4^{3-} and N (NO_2^{-}N and NO_3^{-}N). A GLM was selected because the standard assumptions of normality and constant variance were not satisfied with these water quality parameters. For non-normal response distributions in GLMs, the Pearson residuals are often skewed and thus the Anscombe residuals were preferred because they are expected to closely follow a normal distribution. Similar to the linear regression the residual plot is used to evaluate the assumptions (normality and constant variance). Anscombe residuals from the model did not deviate seriously from a normal distribution. A scatterplot of the Anscombe residuals verses the natural log of the variance showed the residuals contained in a horizontal band. (Appendix 1b: Mixed regression Model Diagnostics- Residuals). Anscombe residual analysis showed no serious violation against these assumptions.

4.2 Water quality

4.2.1 Linear Regression analysis

A linear regression was performed between five water quality parameters and the predictor variables including sample site, month, total rainfall and time. The model also included an interaction effect between site and month which describes how two predictor variables interact if one of the variables differs depending on the level of the other variable. A sine and cosine pair of terms were used to fit the periodic structure of the data within a 12 month period, hence time (T) is T=12. Identical linear regression models were applied to COD, EC, DO_2, TP and TSS to determine the water quality parameters relationship with the predictor variables (Table 5). The model used the secondary data set provided by CoCT (between 2009-2016) for each water quality parameter excluding the non-absolute values but including any outliers that can be visually identified in the histogram distributions.

4.2.1.1 Chemical Oxygen Demand

This model best describes the relationship between COD and predictor variables. Data included a total of 164 observations. Table 5.1 presents the regression summary.

Table 5.1: Multiple regression summary for COD

Variable	**Co-efficient**	**Std. Err.**	**t**	**P>l t l**	**[95% Conf. Interval]**	
spoint	-0.875	1.848	-0.47	0.637	-4.524	2.775
month	0.145	0.239	0.61	0.545	-0.327	0.617
site_month	0.108	0.249	0.43	0.667	-0.385	0.51
sin_12	8.035	1.098	7.32	0.000	5.867	10.203
cos_12	1.787	0.804	2.22	0.028	0.199	3.376
time	0.001	0.000	2.25	0.026	0.000	0.003
tot-rainfall	-0.033	0.009	-3.53	0.001	-0.05	38.536

Table 5.1 shows that:

- The regression model was significant ($F(7,156) = 39.63$, $P<0.01$) with $R^2 = .64$ (adjusted $R^2 = .62$).
- Significance on the sine and cosine terms suggests an improved fit.
- There is a significant negative relationship between COD concentrations and total rainfall at a threshold $P<0.05$, where increased rainfall is associated with decreased COD concentrations
- There is a significant positive relationship between COD concentrations and time at a threshold $P<0.01$, where progression of time is associated with increased COD concentrations.

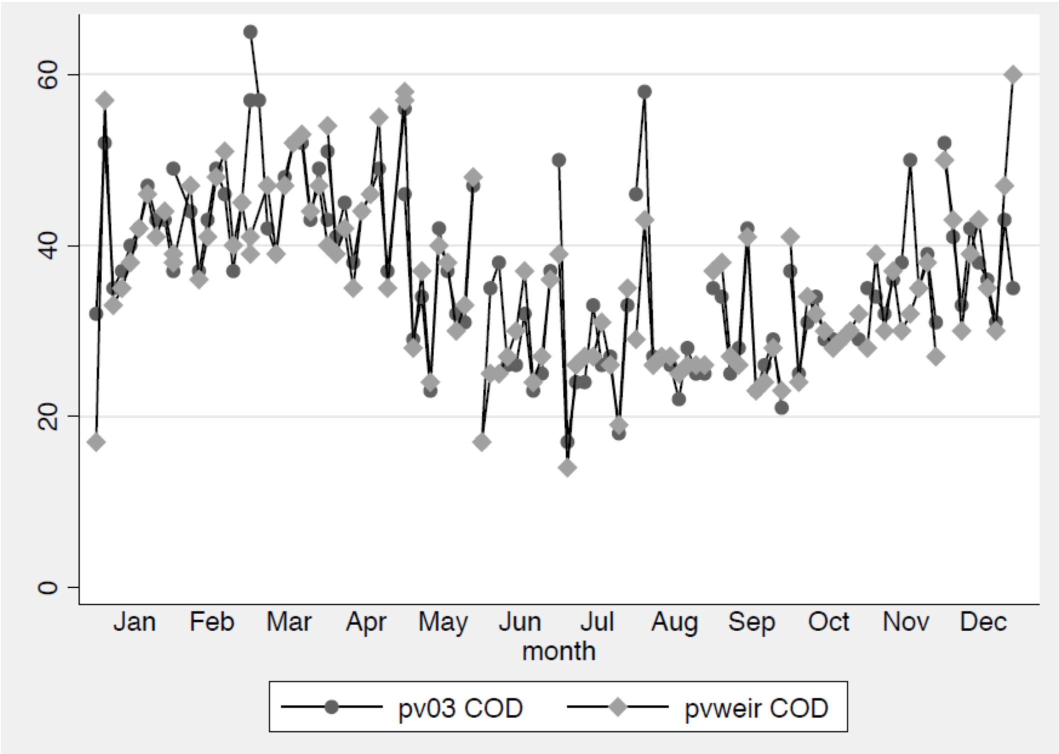

Figure 11: Cycle plot for COD representing samples collected for the years 2009-2016 shown collectively for each month

The trends for COD displayed that an increased rainfall resulted in reduced COD concentrations. This is reiterated by the cycle plot produced for COD in which COD levels are clearly over June, July and August which are considered winter season, rainfall months. It can also be determined that the COD concentrations for PV03 and PVWEIR follow a similar trend which reiterates the lack of significance between the water quality parameter and the sample point predictor variable.

Both COD and BOD parameters were positively correlated to gross organic content of waters and are therefore related to one another (Hemond 1988), that is - COD provides a metric to determine the effect an effluent will have on the receiving body in a similar way BOD does. The COD or BOD measurement provides an indicator of the lack of 0_2 caused by organic pollutants on living aquatic organisms therefore it is the amount of oxygen that can be consumed by reactions in a measured solution. Excess decaying plant material causes increased organic requirements for DO_2 followed by high COD and BOD requirements (Cech 2005). Decaying plant matter provides a food source for microorganism that depletes the DO_2 levels through aerobic respiration (.ibid). A high proportion of decaying organic material results in an increased COD/BOD and as a consequence, a larger amount of oxidizable organic material. COD is an organic based water quality parameter where significant amounts of BOD and COD can be introduced into wetlands through sources of nutrients such as municipal effluent, fertilisers, and septic tanks.

The significant negative relationship between reductions in COD, due to increased rainfall, reduced the amount of organics in the system and therefore the COD rate. Organics in surface water occur as small particulates and therefore have the same movement patterns as flowing surface water (Cech 2005). Organic compounds can float on the surface water of a groundwater aquifer and migrate in the same direction as the groundwater (.ibid). Additional flows from stormwater and urban runoff into the wetland can create dispersion by driving organics downstream and removing them from the system. According to Hemond (1988) wetlands are also able to decrease COD/BOD through the decomposition of organic materials during aerobic bacterial respiration however its efficiency is dependent on resident time. However, inputs of pollutants can accumulate by chemical sorption in wetland sediments. The progression of time results in the accumulative impact of these inputs and ultimately higher loadings in the system (.ibid). This is confirmed by the significant positive relationship where the progression of time is associated with increased COD concentrations. Within an organically bound environment if nitrate-laden wastewater is loaded into a water body like a wetland and stimulates BOD/COD rates this could lead to an anaerobic environment (Cech 2005). Ultimately the system is moving towards a eutrophic state.

4.2.1.2 Electric Conductivity

This model best describes the relationship between EC and predictor variables. Data included a total of 166 observations. Table 5.2 presents the regression summary.

Table 5.2: Multiple regression summary for EC

Variable	Co-efficient	Std. Err.	t	P>| t |	[95% Conf. Interval]	
spoint	0.245	2.253	0.11	0.914	-4.204	4.694
month	-0.031	0.292	-0.11	0.916	-0.608	0.547
site_month	-0.143	0.306	-0.47	0.640	-0.748	0.461
sin_12	6.342	1.350	4.70	0.000	3.677	9.009
cos_12	-1.688	0.964	-1.75	0.082	-3.592	0.216
time	-0.000	0.001	-0.29	0.773	-0.002	0.001
tot-rainfall	-0.084	0.011	-7.39	0.000	-0.106	-0.061

Table 5.2 shows that:

- The regression model was significant ($F(7,158) = 40.08$, $P<0.01$) with $R^2 = .64$ (adjusted $R^2 = .62$).
- There is a significant negative relationship between EC concentrations and total rainfall at a threshold $P<0.01$, where increased rainfall is associated with decreased EC concentrations
- Significance of the sine term at a threshold $P<0.01$ suggests an improved fit and an overall significance for the pair.

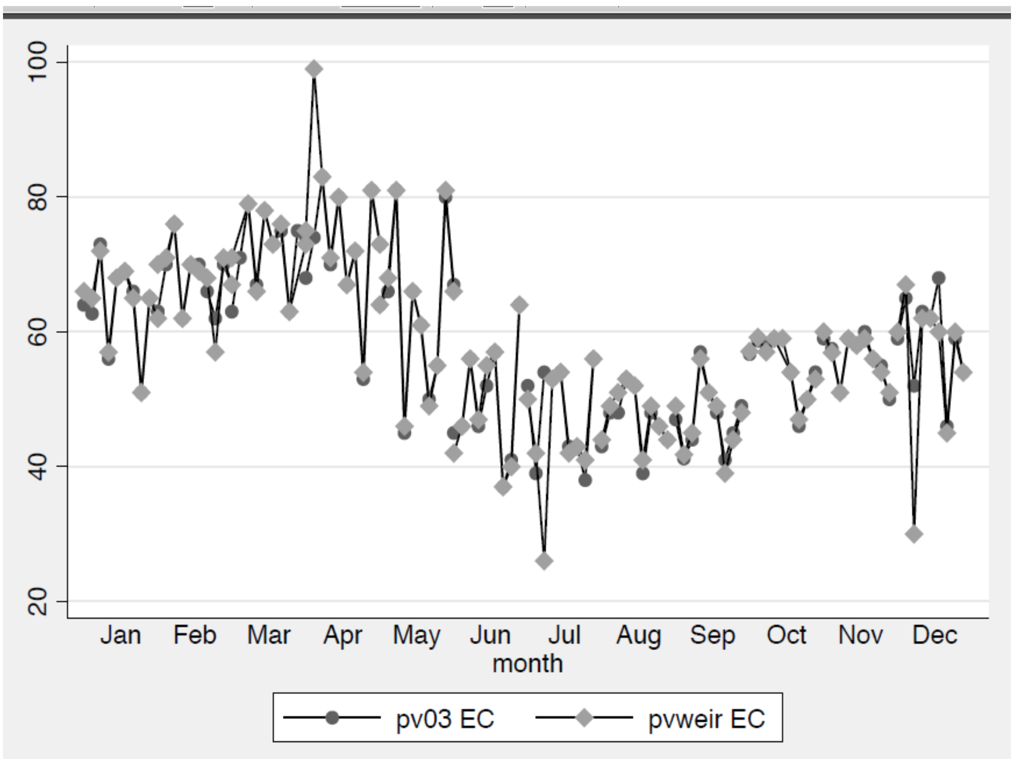

Figure 12: Cycle plot for EC representing samples collected for the years 2009-2016 shown collectively for each month

Figure 12 reiterates the significant negative relationship whereby increased rainfall has been associated with decreased concentrations. During the rainfall months, including June, July and August, EC concentrations are reduced in comparison to the dry, summer months (December, January and February). It suggests that precipitation did not result in the discharge of suspended solids, heavy metals and silt into the wetland contrary to what is expected in the literature. According to theory, stormwater runoff increases the discharge and loadings of suspended matter including silt, small organic particles or decaying particles which in turn increases the turbidity of the system and therefore increases the EC measurement. A possible explanation of the unexpected relationship is attributed to the limitations of the secondary data collected by the CoCT and the time lag between the EC sample collection and the rainfall events. A system receiving stormwater and surface runoff and a large turnover of water is expected to refresh the water body resulting in a reduction in EC. A large overall volume of precipitation over a year could potentially result in suspended solids and heavy metals being removed from the system.

4.2.1.3 Dissolved Oxygen

This model best describes the relationship between DO_2 and predictor variables. Data included a total of 166 observations. Table 5.3 presents the regression summary.

Table 5.3: Multiple regression summary for DO_2

Variable	Co-efficient	Std. Err.	t	P>│ t │	[95% Conf. Interval]	
spoint	-0.071	0.718	-0.10	0.921	-1.490	1.348
month	0.042	0.093	0.45	0.655	-0.142	0.226
site_month	-0.026	0.097	-0.27	0.786	-0.219	0.166
sin_12	-0.037	0.429	-0.09	0.931	-0.884	-0.810
cos_12	-0.256	0.312	-0.82	0.412	-0.872	0.360
time	0.000	0.000	3.02	0.003	0.000	0.001
tot-rainfall	0.002	0.004	0.66	0.513	-0.004	0.009

Table 5.3 shows that:

- There were no significant difference in the regression model (F (7,158) = 1.78, P>0.05) with R^2 = .73 (adjusted R^2 .032).

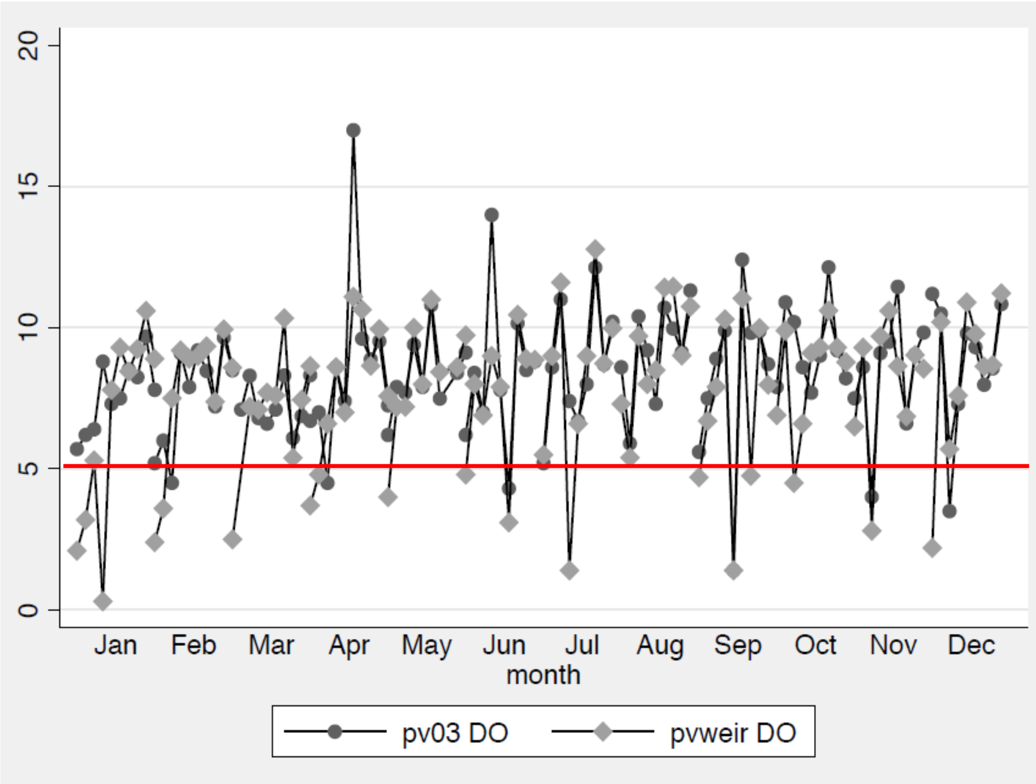

Figure 13: Cycle plot for DO_2 representing samples collected for the years 2009-2016 shown collectively for each month with red line indicating baseline DO_2 level.

When DO_2 is compared to the variables established for the statistical model, the differences are not large enough to determine a statistical significance. Figure 13 highlights a relatively linear trend throughout the year despite a number of outliers. This linear pattern implies that there is nothing or very little driving environmental change. Lack of significance can also be partly attributed to the lack of sensitivity of the secondary data over the interval periods.

4.2.1.4 Total Phosphorus

This model best describes the relationship between total phosphorus and predictor variables. Data included a total of 164 observations. Table 5.4 presents the regression summary.

Table 5.4: Multiple regression summary for TP

Variable	Co-efficient	Std. Err.	t	P>| t |	[95% Conf. Interval]	
spoint	-0.014	0.031	-0.45	0.651	-0.076	0.048
month	0.003	0.004	0.63	0.532	-0.006	0.011
site_month	-0.002	0.004	-0.46	0.643	-0.01	0.006
sin_12	-0.002	0.019	-0.13	0.896	-0.04	0.035
cos_12	0.014	0.014	1.00	0.318	-0.013	0.041
time	-0.000	0.000	-1.01	0.316	-0.000	9.93e-06
tot-rainfall	-0.000	0.000	-1.09	0.276	-0.000	0.000

Table 5.4 shows that:

- There were no significant differences in the regression model (F (7,156) = 1.58, P>0.05) with R^2 = .07 (adjusted R^2 .02)

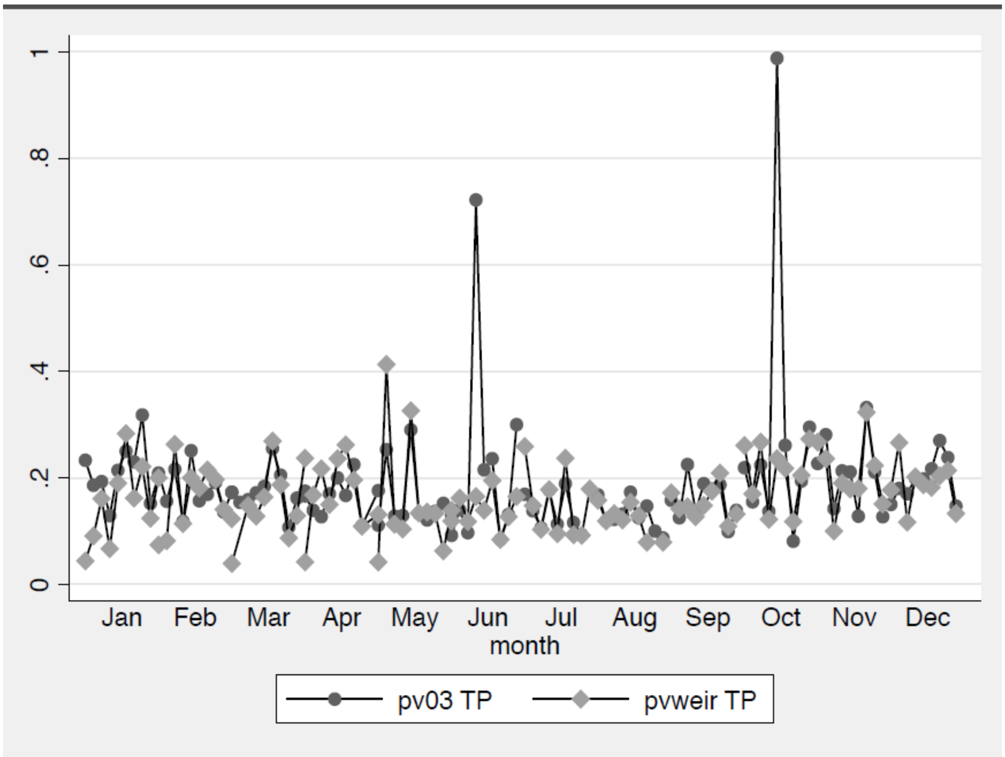

Figure 14: Cycle plot for TP representing samples collected for the years 2009-2016 shown collectively for each month

When TP is compared to the variables established for the statistical model, the differences are not large enough to determine a statistical significance. . Figure 14 highlights a relatively linear trend throughout the year with three notable outliers. Similar to DO^2, lack of significance can be partly attributed to the lack of sensitivity of the secondary data over the interval periods.

4.2.1.5 Total Suspended Solids

This model best describes the relationship between TSS and predictor variables. Data included a total of 168 observations. Table 5.5 presents the regression summary.

Table 5.5: Multiple regression summary for TSS

Variable	Co-efficient	Std. Err.	t	P>| t |	[95% Conf. Interval]	
spoint	-2.903	4.896	-0.59	0.554	-12.573	6.766
month	1.122	0.638	1.79	0.081	-0.139	2.383
site_month	-0.037	0.665	-0.06	0.956	-1.351	1.277
sin_12	8.629	2.931	2.94	0.004	2.841	14.417
cos_12	3.816	2.108	1.18	0.072	-0.345	7.929
time	-0.001	0.002	-0.57	0.57	-0.004	0.002
tot-rainfall	-0.018	0.025	-0.74	0.459	18.167	41.852

Table 5.5 shows that:

- The regression model was significant (F (7,160) = 4.37, P<0.01) with R^2 = .16 (adjusted R^2 = .12).
- Significance of the sine term suggests an improved fit and an overall significance for the pair.

The models for DO_2, TP and TSS show no significance between the water quality parameters and the predictor variables. The lack of significance can be partly attributed to the limitations in the sampling regime rather than accurately representing reality. The samples were collected once a month, mid-month and are fixed according to a regular interval. This however does not account for changes in environmental conditions before or during the collection period.

DO_2 is often described as the single most important measure of habitat quality and thus a significant parameter to investigate (Cech 2005).According to Cech (2005) there is a clear link between COD and DO_2 levels explained by the excess decaying plant material providing a food source for microorganisms causing depleted DO_2 and increased COD requirements. A system that is operating under a 5mg/L DO_2 (indicated by red vertical line on Figure 13) condition is known to adversely affect the survival of aquatic ecosystems (.ibid). Oxygen depletion is more common in standing waters such as wetlands than flowing rivers which can be oxygenated by wind and turbulence (.ibid). Figure 13 displayed sample measurements by plotting the absolute values for each consecutive month from 2009-2016 represented over a year. There are a number of measurements recorded under the 5mg/L limit throughout the years of sample collection which indicates adverse, anaerobic water quality conditions. The cycle plot also reiterates there is no significance between the sample points whereby PV03 and PVWEIR and thus no difference in water quality between the two sites. This is despite the water flowing through the wetland and being subjected to ecosystem services in an environment that is unable to cope with elevated loadings of stormwater and urban runoff.

Phosphorus and suspended solids enter surface water through nonpoint source pollution as a primary mechanism. Concentrations of suspended solids and phosphorus increase with increasing rainfall which transports an elevated discharge of domestic sewage, animal waste, artificial fertilisers, industrial effluent and urban runoff into rivers and wetlands. Suspended solids can prove detrimental and may serve to transport other materials that effect water quality (Hemond 1988). They are also known to increase turbidity which attenuates the transmission of light, decreasing the process of photosynthesis and oxygen production (.ibid). The phosphorus adsorption in wetlands is limited by the sorptive capacity of the suspended solids and sediment present (.ibid). The retentive capacity of nutrient removal tends to decrease over time therefore the introduction of high levels of nutrients over time can lead to deterioration of water quality (.ibid). However, according to Johnston and Niemi (1990) wetlands are more effective in removing suspended solids, total phosphorus and ammonia in high flow periods, which highlights the limitations of the sampling regime and the time lag between sample collection and analysis and rainfall events.

4.2.2 Gamma Regression analysis

A gamma regression was performed between three water quality parameter and the predictor variables including sample site, month, total rainfall and time. The model also included an interaction between site and month which describes how two predictor variables interact if one of the variables differs depending on the level of the other variable. A sine and cosine pair of terms to fit the periodic structure of the data with a 12 month period so time (T) is T=12. Identical gamma regression models were repeated for NH_3-N, PO_4^{3-} and N (NO_2^{-}N and NO_3^{-}N) to determine the water quality parameters relationship with the predictor variables (Table 5). The model used the secondary data set provided by CoCT (between 2009-2016) for each water quality parameter excluding the non-absolute values but including any outliers that can visually identified in the histogram distributions.

4.2.3.1 Ammonia nitrogen

This model best describes the relationship between NH_3-N and predictor variables. Data included a total of 113 observations. Table 5.6 presents the regression summary.

Table 5.6: Gamma regression summary for NH_3-N

Variable	Co-efficient	Std. Err.	z	P>∣ z ∣	[95% Conf. Interval]	
spoint	2.024	5.269	0.38	0.701	-8.302	12.351
month	0.156	0.898	0.17	0.862	-1.604	1.917
site_month	0.052	0.964	0.05	0.957	-1.837	1.941
sin_12	-5.809	4.966	-1.17	0.242	-15.542	3.925
cos_12	-0.579	3.541	-0.16	0.870	-7.519	6.361
time	-0.005	0.002	-2.56	0.010	-0.010	-0.001
tot-rainfall	-0.023	0.045	-0.52	0.603	-0.110	0.064

Table 5.6 shows that:

- There is a significant negative relationship between NH_3-N concentrations and time, at a threshold $P<0.01$, where progression of time is associated with decreased NH_3-N concentrations.

As previously mentioned a decline in water quality can be attributed to higher nutrient concentrations. According to Cech (2005) ammonia and ammonium are rich in nitrogen and serves as a well know fertiliser. As a benchmark, NH_3-N levels at 0.1mg/L (indicated by red vertical line on Figure 15) usually indicate polluted surface waters with recordings over 2mg/L described as toxic for aquatic species (.ibid). Figure 15 shows are a number of samples collected from Princess Vlei that display polluted and even toxic levels of water for aquatic species. High levels of ammonia are often found downstream of wastewater treatment plants and in wetlands receiving landfill and sewage leachate. It was determined that time was a significant predictor of NH_3-N concentrations with decreased NH_3-N concentrations associated with progression of time.

This result is contrary to theory that assumes an increase in NH_3-N concentrations over time within a stressed, urban environment. Despite the significant negative relationship between NH_3-N concentrations and time, Figure 16 indicates increasing trends in reality for both PV03 and PVWEIR. The increasing trendline over the years 2009 – 2016 indicates that there is an overall tendency for NH_3-N concentrations to increase.

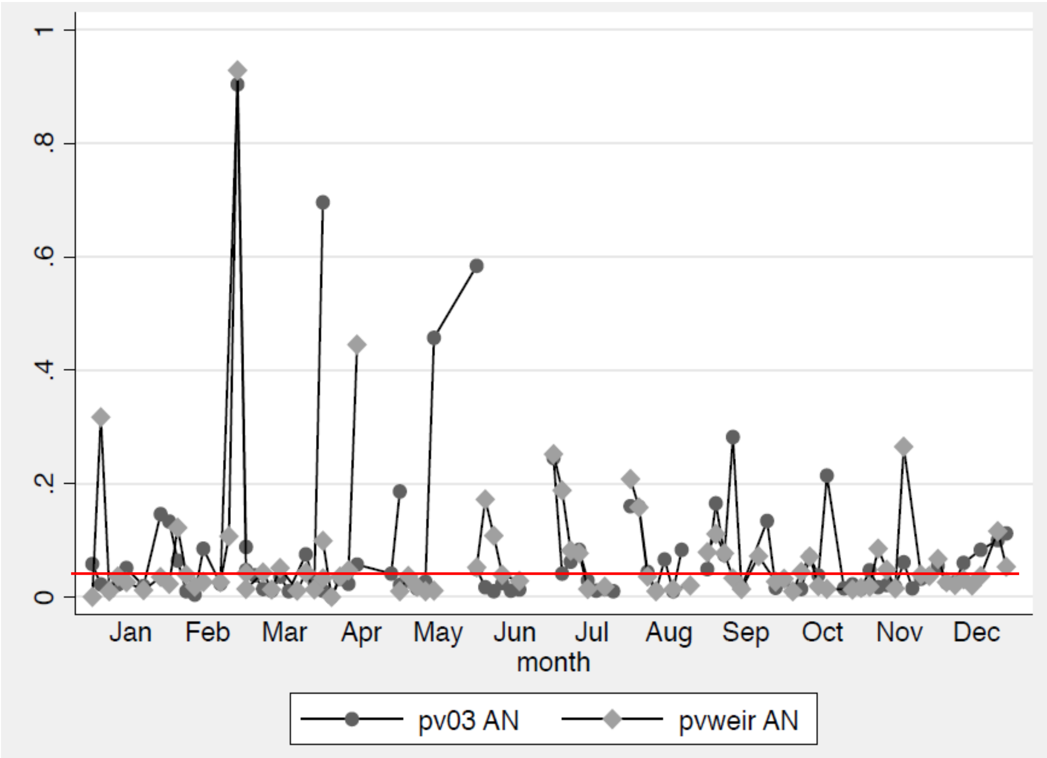

Figure 15: Cycle plot for NH_3-N representing samples collected for the years 2009-2016 shown collectively for each month with red line indicating baseline NH_3-N levels

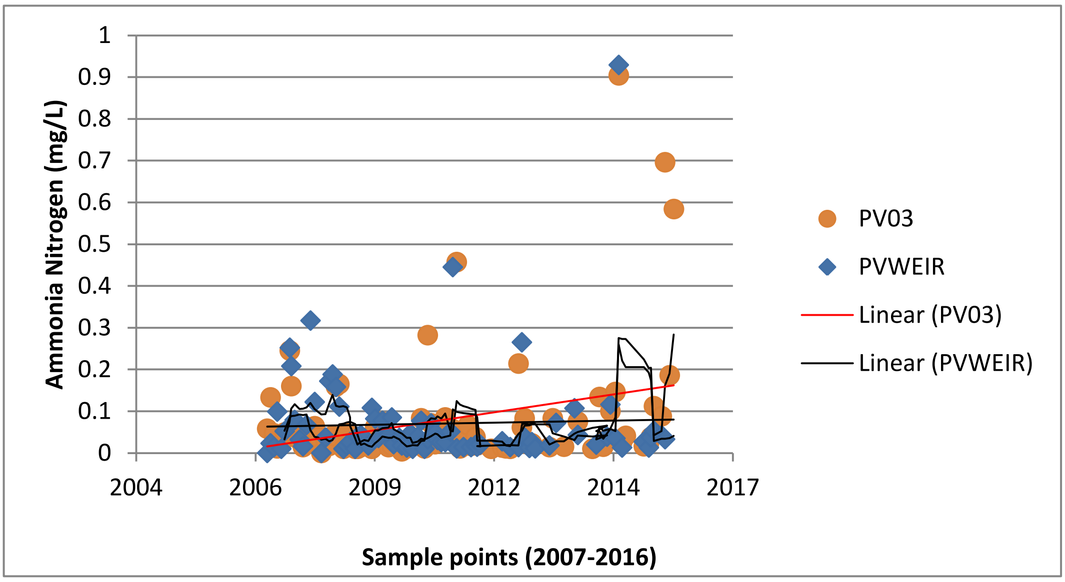

Figure 16: NH_3-N concentrations determined for two sample sites (PV03 and PVWEIR) taken on the same dates for both sites

4.2.2.1 Nitrates and Nitrites (as Nitrogen)

This model best describes the relationship between N ($NO_2^{-}N$ and $NO_3^{-}N$) and predictor variables. Data included a total of 121 observations. Table 5.7 presents the regression summary.

Table 5.7: Gamma regression summary for N ($NO_2^{-}N$ and $NO_3^{-}N$)

Variable	Co-efficient	Std. Err.	z	P>l z l	[95% Conf. Interval]	
spoint	-0.260	0.484	-0.54	0.591	-1.208	0.688
month	0.000	0.061	0.01	0.995	-0.121	0.122
site_month	0.009	0.065	0.13	0.893	-0.119	0.137
sin_12	-0.677	0.277	-2.44	0.015	-1.221	-0.134
cos_12	0.317	0.195	1.63	0.104	-0.065	0.7
time	0.000	0.000	1.72	0.086	-0.000	0.000
tot-rainfall	0.003	0.002	1.41	0.158	-0.001	0.008

Table 5.7 shows that:

- Significance of the sine term at a threshold $P<0.05$ suggests an improved fit and an overall significance for the pair.

4.2.2.2 Ortho-phosphates

This model best describes the relationship between PO_4^{3-} and predictor variables. Data included a total of 105 observations. Table 5.8 presents the regression summary.

Table 5.7: Gamma regression summary for PO_4^{3-}

Variable	Co-efficient	Std. Err.	z	P>l z l	[95% Conf. Interval]	
spoint	0.572	0.455	1.26	0.209	-0.32	1.463
month	0.159	0.059	2.71	0.007	0.044	0.274
site_month	-0.100	0.061	-1.65	0.098	-0.219	0.019
sin_12	0.213	0.251	0.85	0.397	-0.28	0.706
cos_12	-0.157	0.198	-0.79	0.428	-0546	0.232
time	0.000	0.000	3.18	0.001	0.000	0.000
tot-rainfall	-0.002	0.002	-1.03	0.303	-0.006	0.002

Table 5.8 shows that:

- There is a significant positive relationship between PO_4^{3-} concentrations and month at a threshold P<0.01, where a specific month is associated with increased PO_4^{3-} concentrations.
- There is a significant positive relationship between PO_4^{3-} concentrations and time at a threshold P<0.01, where progression of time is associated with increased PO_4^{3-} concentrations.

Phosphates exist in three forms including orthophosphates, metaphosphate and organically bound phosphate (Hemond 1988). Orthophosphates are produced by natural processes as well as significant anthropogenic sources such as partially or untreated sewage, runoff from agricultural sites and fertilisers, similar sources of ammonia (.ibid). In water, phosphates are transformed into a dissolved phase, orthophosphates, to become available for the uptake of plants (.ibid). Both monthly variables and the progression of time are significantly associated with an increased concentration of PO_4^{3-}. Over time, water quality has been compromised through stormwater drainage driven by stormwater runoff entering the wetland and thus the accumulation of phosphates. Retention of sediment and nutrient loads in the wetland is associated with the sedimentation processes. The accumulation of organic sediments by wetland vegetation can facilitate long-term nutrient retention (Hemond 1988).

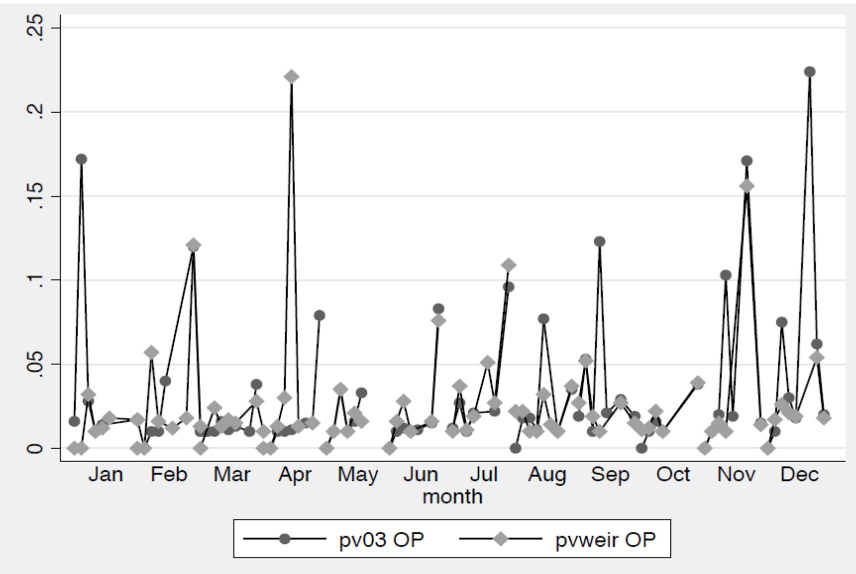

Figure 17: Cycle plot for PO_4^{3-} representing each sample collected for the years 2009-2016 for each month

A number of significant peaks identified in Figure 17 can be potentially attributed to the first flush phenomenon described by Lee et al. (2002) as the initial period of stormwater runoff during which the concentration of pollutants is substantially higher than during late periods.

Taking into account the inconsistent results for some of the water quality parameters when compared to theory, it is important to identify the elements of urbanisation that contribute most to pollutant concentrations and loads. This can add to the explanation of ad-hoc or extreme results in an urban environment. According to Hatt et al. (2004) drainage infrastructure is proposed as a variable whereby degree of imperviousness, drainage connection, unsealed road density, elevation, longitude, septic tank density and basin area are partial explanations of concentration patterns. A single variable cannot be attributed to the levels of contaminants running off urban areas in a wetland because they ultimately arise from a multitude of activities which are variable in time and effects due to the weather (.ibid).

Heavy rain falls results in significant runoff and solid material transportation result in increased turbidity and potentially suspended solids. In contrast, for low river flow rates, a temperature increase results in a concentration increase of dissolved substances in water but a concentration decrease of DO_2 (Prathumratana et al. 2008; van Vliet & Zwolsman 2008). However, a positive correlation implies a decrease in the concentration of some pollutants due to a low water velocity and longer resident/time. This is reiterated by van Vliet and Zwolsman (2008) who concludes that a decline in water quality is primarily due to favourable conditions for the development of algae blooms including higher temperatures, long residence time and high nutrient concentrations.

4.3 Primary data discussion

Primary data collection was undertaken between November 2016 and February 2017 in order to understand what water quality constituents and concentrations levels that were entering the wetland. Samples were taken within a 24hour period of a rainfall. Although sampling could not be carried out systematically due to site-specific limitations, the primary data identified a number of water quality parameters exhibiting elevated loadings of nutrients as well as depleted oxygen levels displaying a stressed aquatic environment. The primary data is displayed in table 5 and discussed below.

4.4 Comparisons between primary and secondary parameters to two sets of water quality guidelines

Table 5 illustrates a comparison between the minimum, average and maximum recorded values for the primary and secondary water quality data against a combination of South African water quality guidelines for aquatic ecosystems and the Ecological Reserve water quality benchmarks (Department of Water Affairs and Forestry 1996; Rossouw 2004). These guidelines essentially specify the surface water quality required to protect fresh water aquatic ecosystems (Department of Water Affairs and Forestry 1996).

The regression analysis conducted for the secondary dataset concluded that there was no statistical significance between the two sample sites, namely PV03 and PVWEIR (Figure 5) for all water quality parameters. This prompted the statistical summary in Table 5 for the secondary data to combine the water quality parameters for both sample sites and therefore calculate and analyse the minimum, average and maximum as an overall system. The five sample sites selected for the primary data (Figure 4) did not differentiate between sites, thus also combining the water quality parameters and calculating and analysing the minimum, average and maximum as an overall system (Table 5).

Despite the maximum DO_2 measurements for both secondary (17mg/L) and primary (6.3mg/L) data recorded within the natural category, the minimum DO_2 rates, 0.3mg/L and 1.3mg/L respectively, are well below the unacceptable category identified by DWAF. This indicates severe anaerobic conditions and an inability to support aquatic plants and wildlife. This is in conjunction with a maximum of 65 mg/L and an average of 36.26mg/L for COD for the secondary dataset, also considerably greater than the unacceptable range (>30mg/L) established by DWAF.

In combination with depleted dissolved oxygen levels, the wetland experienced extremely high nutrient levels as indicated by observations of PO_4^{3-} and NH_3-N in Table 5.
An average value of 0.23mg/L and 0.08mg/L for NH_3-N was recorded for primary and secondary data respectively. This indicates that while the secondary data average value is less than the natural state limitation (<0.015mg/L) the primary data recording exceeds the unacceptable range (>0.2mg/L). The maximum values for the primary and secondary data are significantly higher than the unacceptable value highlighting at some point or over a period of time there has been an excess of nutrients in the system. An average value of 0.34mg/L and

0.03mg/L for PO_4^{3-} was recorded for primary and secondary data respectively. While both the average and maximum (0.22mg/L) values recorded for the secondary dataset are within the natural limitation, the average value and maximum value (2.94mg/L) for the primary dataset is significantly greater than the unacceptable range. A combination of depleted oxygen levels, high COD and nutrient levels compromise the water quality and as a result the ecological integrity of surface waters (Smith et al. 1999; Nyenje et al. 2010).

Table 6: Comparisons between primary and secondary water quality parameter datasets with two sets of water quality guidelines
*values not recorded

Data	**SECONDARY DATASET**			**GUIDELINES** (Department of Water Affairs and Forestry 1996; Rossouw 2004)		**PRIMARY DATASET**		
Parameter	**Min**	**Average**	**Max**	**Natural**	**Unacceptable**	**Min**	**Average**	**Max**
NH_3-N (mg/L)	0	0.08	0.93	<0.015	>0.2	0	0.23	0.84
COD (mg/L)	14	36.26	65	<3	>30	*	*	*
EC (mS/cm)	26	58.79	99	150 (Depends on background EC, limited to 15% change)	500	201	406.71	731
DO_2 (mg/L)	0.3	8.09	17	6	4	1.3	4.11	6.3
Nitrate and nitrite as N (inorganic nitrogen) (mg/L)	0	0.21	1.69	>0.5	5-10	*	*	*
PO_4^{3-} (mg/L)	0	0.03	0.22	<0.005	>0.25	0.02	0.34	2.94
TP (mg/L)	0.39	0.17	0.98	*	*	*	*	*
TSS (mg/L)	1	32.33	102	Background concentrations: <100mg/L	Any increase must be limited to 10% of background TSS	0	11.41	42

Princess Vlei wetland is being subjected to elevated loadings of pollutants whereby the ecosystem services within the system are unable to cope resulting in the loss of the services that the system provides (Smith et al. 1999).

4.5 Trophic classes

A wetland ecosystem can be significantly altered by nonpoint and point source loadings such as nitrogen and phosphorus through the process of eutrophication (Hemond 1988). According to Nyenje et al. (2010) most nutrients causing eutrophication originate from agricultural and urban areas and can lead to highly detrimental changes to ecosystem structure and function as well as the quality of receiving fresh water. In developing countries in Sub-Saharan Africa, the majority of problems associated with nutrients is related to urbanisation with over 80% of wastewater produced in large cities untreated and either discharged via on-site sanitation systems or directly into rivers and lakes (Nyenje et al. 2010). However, the distribution of the nutrient load differs across the urban areas. In sewered parts, underlying aquifers receiving untreated wastewater from leaking sewers is relatively low whereas wastewater concentrations are estimated to be low to medium strength (.ibid). This is contrasted by the majority of the urban population using on-site sanitation systems producing low volumes of waste with high concentrations (.ibid). It has previously been described that wetlands are known to act as a buffer to eutrophication through the retention of nitrogen and phosphorus. However, encroachment on wetlands and increased wastewater production has increased nutrient loadings beyond this buffering capacity (Smith et al. 1999; Nyenje et al. 2010).

As previously mentioned, Zeekoevlei is a shallow lake located to the West of Princess Vlei wetland and within the same catchment. This has been identified as a freshwater system that has been suffering from hyper-eutrophic conditions in the last few decades. According to Das et al. (2009) the vlei has been draining a catchment covering residential, industrial, agricultural and horticultural areas. Zeekoevlei is also the receiving water body for the Great and Little Lotus Rivers which are heavily polluted with agricultural and urban runoff (.ibid). This research identifies similar characteristics to Princess Vlei wetland and can be used as support, highlighting the potential consequences for Princess Vlei wetland if circumstances remain the same.

Table 7: Relationships between trophic classes and nutrient ranges adapted from Smith et al. (1999)

Trophic state **(Lakes)**	**TN (mg/L)** (Smith et al, 1999)	**TP (mg/L)** (Smith et al. 1999)	**Phosphorus (mg/**L) (Carlson 1977)
Oligotrophic	<0.35	<0.01	0- 0.12
Mesotrophic	0.35 - 0.65	0.01 - 0.03	0.012 - 0.024
Eutrophic	0.65 - 1.2	0.03 - 0.1	0.024 - 0.096
Hypertrophic	>1.2	>0.1	0.096 - 0.384+

The classification of lakes by water quality and lake trophic status has been a common mechanism used to compare lakes, evaluate management interventions and establish future objectives (Nürnberg 1996). A number of trophic classes were established in order to categorise the relative magnitudes of their nutrient inputs (Smith et al. 1999). The four main trophic states are identified in Table 6. The terms oligotrophic, mesotrophic and eutrophic refer to system receiving low, medium or high levels of nutrients (.ibid). Hypertrophic is a term given to a system receiving greatly excessive nutrient inputs (.ibid).

According to Nürnberg (1996) TN is highly correlated to TP and can be used a trophic class predictor variable with almost the same efficiency as TN. TN and TP are therefore interchangeable as indicators of trophic classes. As indicated in Table 5, the minimum, average and maximum for the secondary data has been calculated for TP represented by 0.39mg/L, 0.17mg/L and 0.98mg/L respectively (Table 5). The average concentration of TP between the years 2009-2016 represents a concentration greater than the TP hypertrophic benchmark as 0.17>0.1mg/L. This therefore classifies Princess Vlei as suffering from hypertrophic conditions. In conjunction with this, the minimum TP measurement is even greater than the average TP measurement as 0.39>0.1mg/L, which indicates that the least attained measurement is also greater than the TP hypertrophic benchmark value. A combination of high nutrient levels, depleted oxygen levels and an extremely high TP concentration this describes a wetland that is experiencing significant loadings of stormwater and urban runoff is unable to cope with these elevated loadings.

5. Conclusion and recommendations

The consequences of poor water quality on the performance of urban aquatic ecosystems have been well established by researchers worldwide (Fisher-Jeffes & Armitage 2013) . Stormwater management in the urban areas of South Africa predominately focuses on the collection and diversion of runoff into the nearest receiving water body, with little concern for the impacts on the environment. Conventional drainage systems primarily focus on water quantity control, and have limited capacity and flexibility to adapt to future climatic variability and urbanisation (Zhou 2014). This has resulted in many cases similar to that of Princess Vlei wetland, a small urban wetland that is being overwhelmed by the quality of stormwater and urban runoff.

This study aimed to identify the surface water quality of Princess Vlei wetland over the past 8 years, and establish the ingress and outflow of the wetland. The results generated multiple insights into the capacity of an urban wetland to treat and retain components of stormwater and urban runoff and therefore its potential ability to provide ecosystem services in terms of water treatment within an urbanised catchment. The results did not suggest that the wetland was able to treat the water, as the literature emphasises; rather, they confirmed the pervasive impacts of the urban catchment on the health and functioning of the wetland. It can be speculated that the increased impervious coverage and conventional drainage systems associated with the urban catchment are most likely the significant producer of contaminated stormwater. The multiple pollutant inputs entering the wetland appeared to have resulted in deteriorating water quality and the hampering of ecosystem services of Princess Vlei wetland.

Despite the results inability to confirm the literature that discusses the benefits of wetlands as natural treatment systems, it highlighted the negative impacts of an urban catchment on a small aquatic system, its deteriorating water quality and its impact on the functioning of ecosystem services. This suggested a link between urbanisation in the form of increased imperviousness surfaces with urban wetland functioning with regards to water treatment.

The pollutant concentrations within the wetland were best explained by the predictor variables of total rainfall and progression of time, indicated by the significant estimated correlations. Rainfall events, named as total rainfall, resulted in larger volumes of water entering the wetland, which had a twofold effect on the wetland ecosystem, either diluting pollutant concentrations or elevating pollutant concentrations. These inverse trends were proved through the significant correlations found between total rainfall and COD and EC.

The decreased COD concentrations implied that the system was flushed of organics as rainfall creates dispersion and pushes the organics downstream. EC concentrations were similarly decreased with increased rainfall. The analysis of the secondary data revealed that varying rainfall over wet and dry antecedent periods influenced the ability of Princess Vlei wetland to provide ecosystem services.

The other predictor variable, progression of time, similarly influenced the wetland's ability to provide ecosystem services, either through the accumulation, retention or flushing of pollutants. The accumulation of pollutants over time was identified through the increasing concentrations off COD and PO_4^{3-}, with the exception of NH_3-N that decreased over time. This implies that the wetland was able to assimilate the NH_3-N but not the COD and PO_4^{3-}. This was either due to the constant inflow of orthophosphate enriched runoff into the wetland, or due to the wetlands inability to deal with the orthophosphates. The increased COD and PO_4^{3-} are indicative of sewage overflows into the wetland. Of the five predictor variables, total rainfall and progression of time were the dominant drivers of water quality parameters in the wetland. These primary predictors influenced the environment that the wetland operated within and its degree of potential water treatment.

Ultimately, this manipulated how the wetland was able to manage the ingress of stormwater runoff and the resultant quality of the outflow water. Despite the water being subjected to ecosystem services between the inflow points and outflow weir, the deteriorating quality was able to emphasise the limited degree of functioning ecosystem services. The results suggest that the performance of ecosystem services, namely water treatment is considerably impacted by rainfall levels. In combination with this, the progression of time will continually relate to the accumulation of nutrients if the catchment in which Princess Vlei operates within remains the same or becomes increasingly urbanised.

The primary data was able develop the above description and provide support to the analysis of water quality parameters concentrations post rainfall events. This expanded on the objective determining the capacity of Princess Vlei to process pollutants. It was suggested from the significantly high nutrient levels of PO_4^{3-} and NH_3-N the wetland is heading towards a eutrophic state. Depleted DO_2 levels inhibit the survival of aquatic organisms. A combination of high nutrient levels and anaerobic conditions suggests a wetland that is unable to cope with elevated pollutants.

When comparing the water quality parameter measurements recorded for both primary and secondary datasets to the South African Water Quality guidelines for freshwater ecosystems all parameters were shown to exceed the guidelines and were often recorded well above the 'unacceptable' category. This was able to highlight the excessive levels of nutrients and low DO_2 levels within the system according to the South African guidelines which represent a primary source of reference information specifically aimed at water quality managers.

Stormwater is highly variable over time and space, with multiple environmental variables influencing the water quality of a receiving water body. SUDS aim to treat pollutants as close to the source as possible which highlights the need to research site-specific pollutant sources. The storrmwater system plays a role as a component of the treatment train indicated in Figure 3. More specifically, wetlands are one element of the stormwater system and ultimately an element of the ongoing treatment train at a local scale Although quantifying the depth and complexities of all influential variables on the functioning of the wetland in terms of catchment and wetland hydrology, sediment, vegetation and soil characteristics is beyond the scope of this thesis, results indicated an overall trend in the water quality of Princess Vlei wetland. Water quality parameters were used to indicate to some degree the health of the aquatic ecosystem and therefore the treatment capacity of the wetland to retain and treat urban water.

In order to further this study there are a number of recommendations to comprehensively research other variables influencing the provisioning of ecosystem services in an urban wetland:

- Designing a continuous monitoring regime to monitor inflows and discharge volumes in order to ascertain uninterrupted water quality measurements to establish seasonal characteristics within the wetland.
- Various stormwater interventions would be beneficial by sampling inputs and determining methods that could improve the treatment and cleaning of the stormwater before entering the wetland. This would contribute to the development of sustainable urban drainage guidelines.
- Research to improve the understanding the benthic ecological regime within vlei and the anaerobic/anoxic conditions in the lower regions of the wetland.
- Determine the contribution and influence of groundwater on the water quality or further investigate the overall water balances and various inputs.

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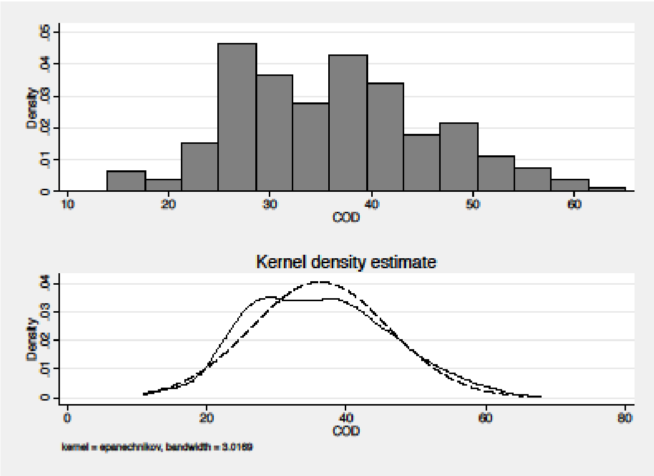
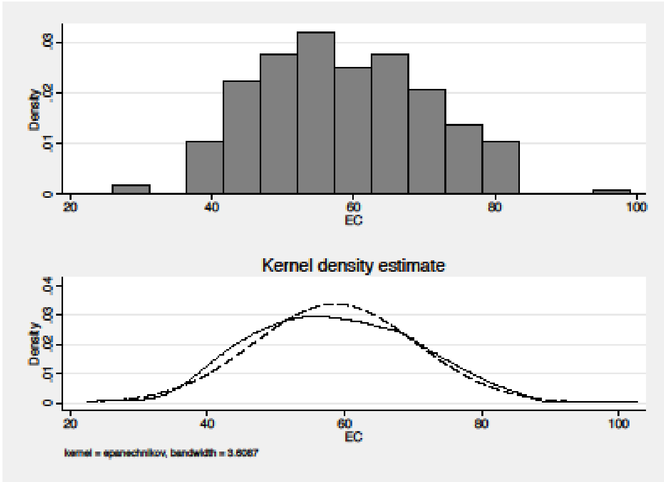
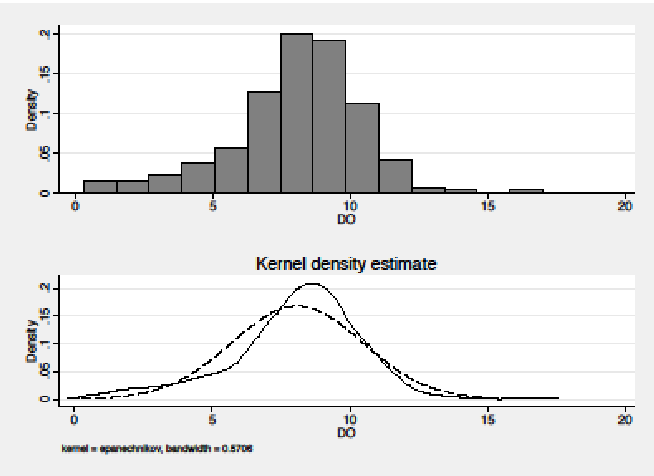
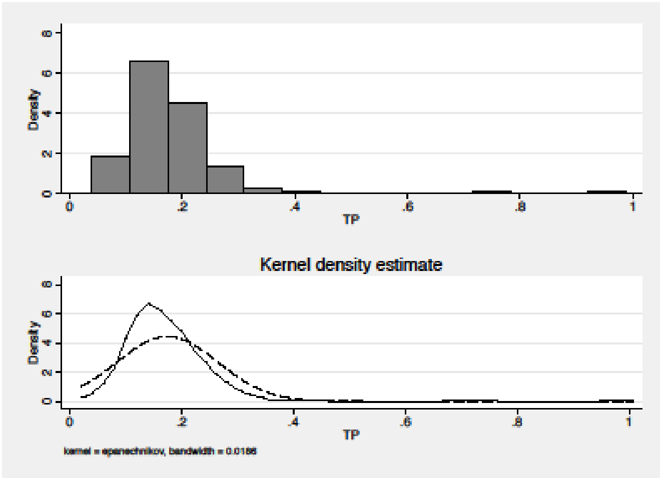
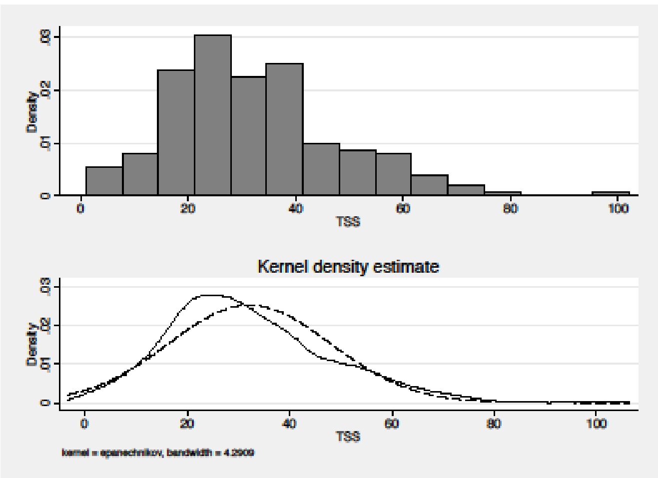
Appendix 1a: Mixed regression Model Diagnostics - Histograms

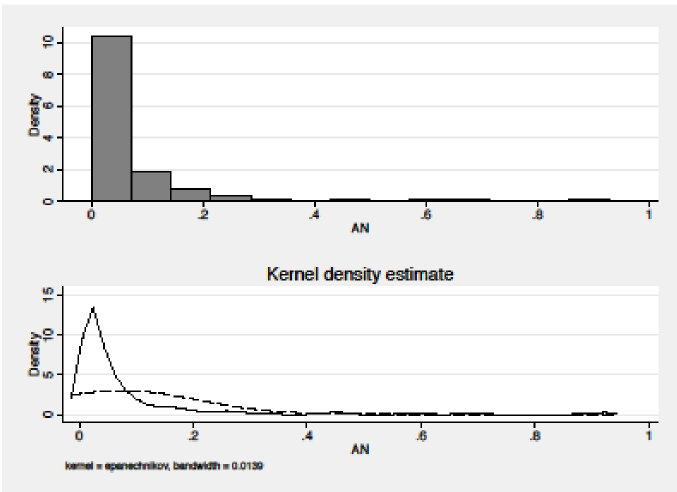


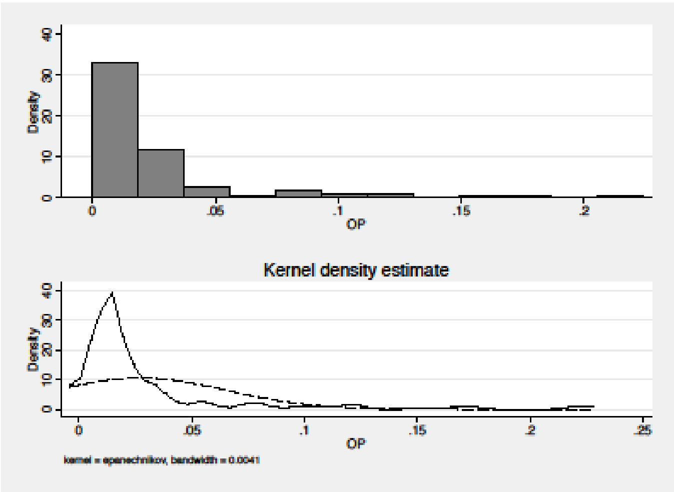


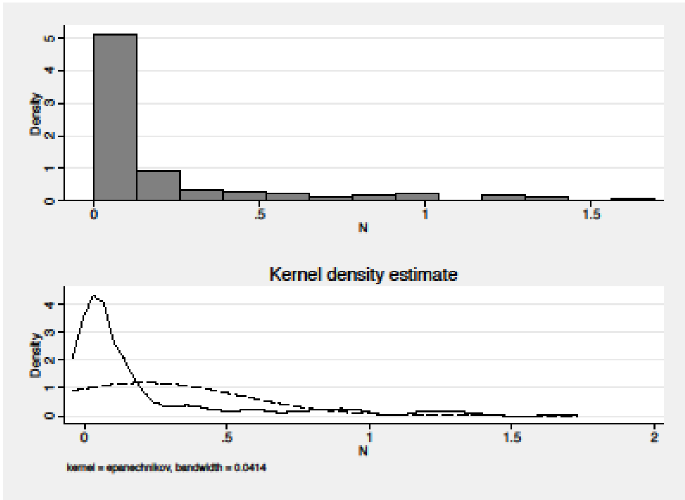




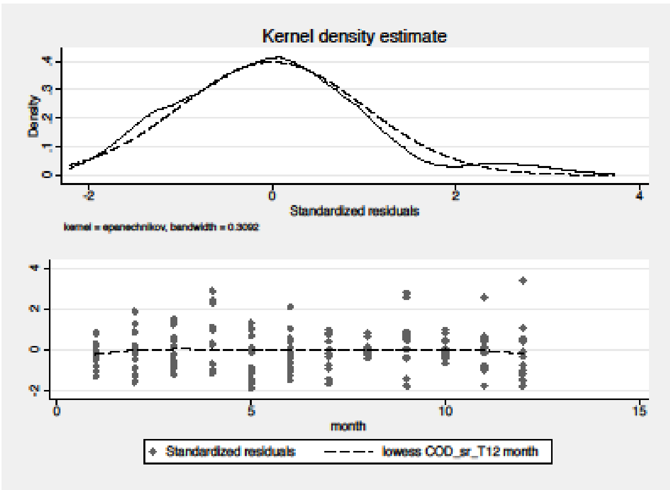


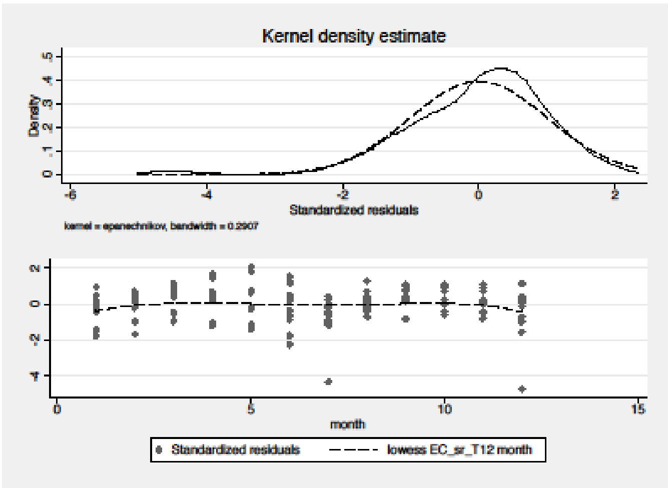


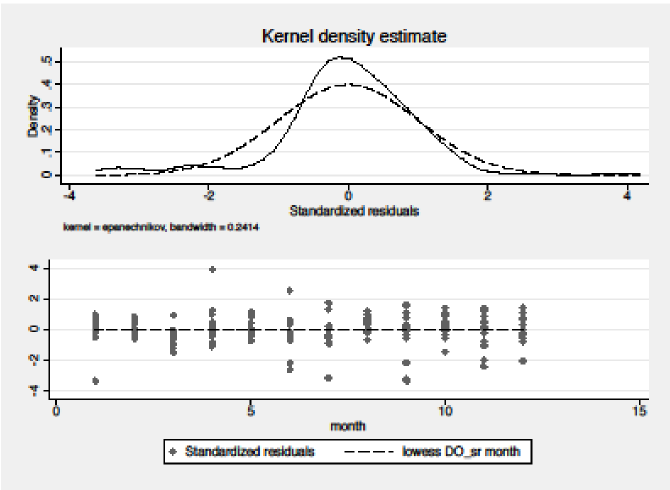


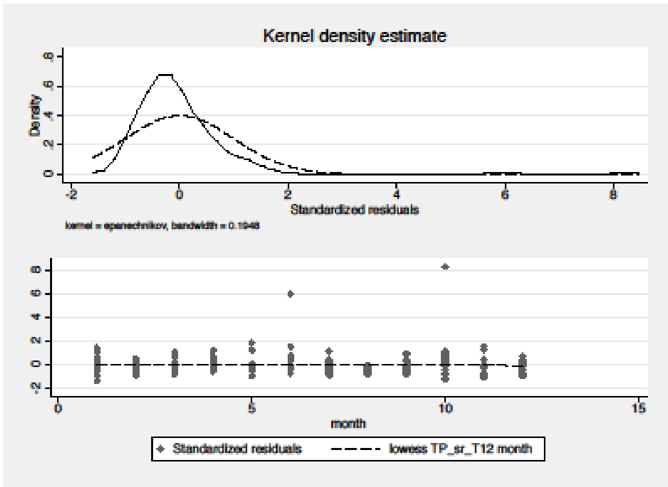


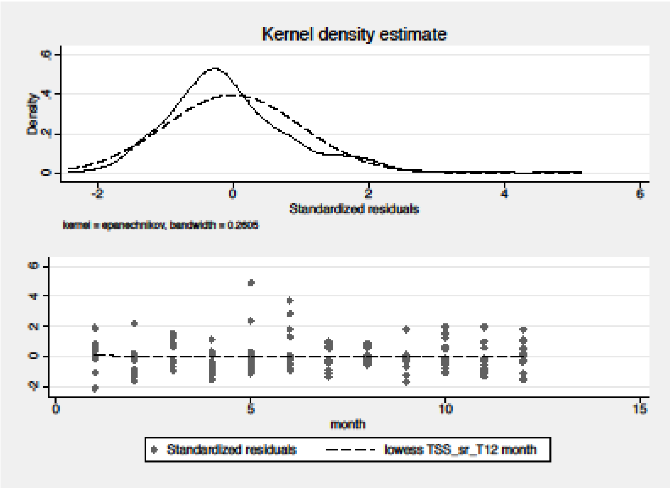
Appendix 1b: Mixed regression Model Diagnostics - Residuals







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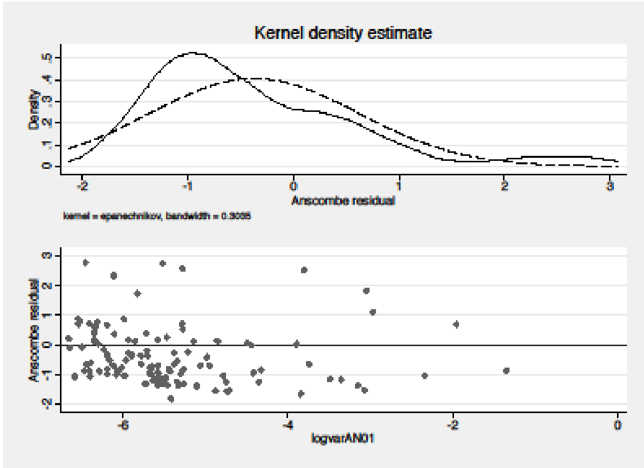
Density
Standardized residuals
kernel = epanechnikov, bandwidth = 0.2606
month
Standardized residuals
lowess TSS_sr_T12 month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Density
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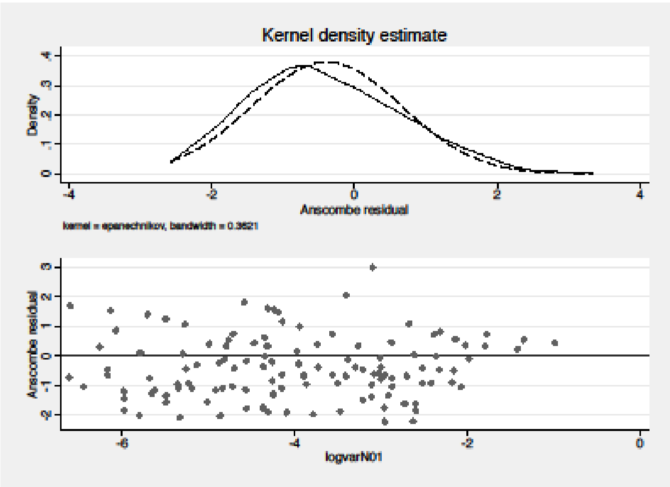
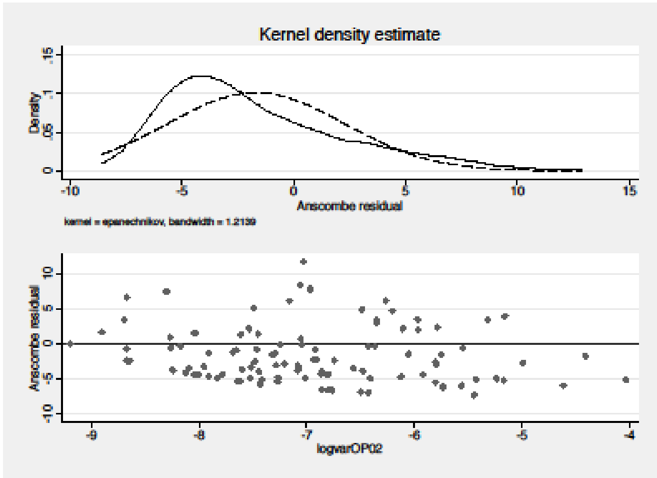
Anscombe residual
logvarAN01


Kernel density estimate
Density
Anscombe residual
kernel = epanechnikov, bandwidth = 0.3621
Anscombe residual
logvarN01


Kernel density estimate
Density
Anscombe residual
kernel = epanechnikov, bandwidth = 1.2139
Anscombe residual
logvarOP02