Wetland diversity and ecosystem services in the Tlokwe Municipal Area

C Pretorius

21726167

Dissertation submitted in fulfilment of the requirements for the degree *Magister Scientiae* in Environmental Sciences at the Potchefstroom Campus of the North-West University

Supervisor: Prof SS Cilliers

Co-supervisor: Dr MJ du Toit

May 2017

ABSTRACT

The last two decades saw rapid urbanization and development in South Africa, resulting in highly fragmented sprawling cities. In the Tlokwe Municipal Area (TMA), agriculture and urbanization are responsible for most of the irreversible transformation of natural areas including wetlands. The quantification of urbanisation change and the transformation of natural areas in the TMA over a 61 year time period revealed extensive land-cover changes. Changes in the coverage of wetlands designated a further study of wetland delineation within the TMA. Wetlands provide a range of ecosystem goods and services, but are also the least conserved ecosystems in South Africa. The alarming rate at which wetlands are being lost and degraded is increasing due to the lack of understanding of their ecological and socio-economic importance by planners, policy makers ad developers. The main aim of the study was to investigate the floristic composition and ecosystem service delivery of wetlands within the endangered Rand Highveld vegetation unit containing the Eastern Temperate Freshwater wetland type, in the TMA along an urban-rural gradient.

During the field investigation, wetlands within the TMA were identified, delineated and classified and the plant species composition and diversity thereof were determined. Plant species composition and diversity within the wetlands were determined along a number of 100m line transects across the wetland sites, where the vegetation was surveyed at each 10m interval along the transect, within a $1m^2$ quadrant. The plant species data collected is used to assess the Floristic Quality Assessment Index (FQAI) – used for evaluating habitat condition or status (with specific reference to the habitat) using the conservatism of the plant community; secondly the Wetland Index Value (WIV) – which measure changes in plant species that are indicative of changes in hydrology processes due to anthropogenic disturbance for each wetland. The potential ecosystem services each wetland may deliver, were scored according to the WET EcoServices rapid assessment method.

A total of 102 plant species (68 indigenous, 34 alien species) were identified within the 14 wetland study sites. Urban wetlands had a higher total vegetation cover and plant species richness than rural wetlands, as well as a different species composition. One of the rural wetlands can be seen as a ‘transition’ wetland, as it almost equally shares its species with both urban and rural wetlands. All wetlands surveyed was determined to be ‘true wetlands’ (all WIV <2.5). Two rural wetlands had lower WIV values than the others. The other rural wetlands had higher WIV values, mostly due to the higher abundance of facultative wetland species and lower abundances of obligate wetland species. The FQAI calculated for each wetland did not differ substantially between most of the rural sites, and all had relatively low FQAI values. Urban wetlands are considered to have relatively high FQAI values when compared to rural wetlands. The quality of the overall wetland habitat of the TMA could be considered as degraded.

Urban wetlands scored higher for direct ecosystem services (provisioning ecosystem services), which is ecosystem services such as tourism and opportunities for education and research (cultural ecosystem services), whereas rural wetlands scored higher for regulating ecosystem services (sediment and phosphate trapping, nitrate and toxicant removal and erosion control). From all the wetlands, it is evident that the presence and cover of vegetation can be considered a major driving force for delivering the ecosystem services, rather than the type of plant species (types of species) present in the wetlands.

The identification of ecosystem services being delivered by natural areas, has become a policy tool to protect biodiversity, however this concept is not readily implemented. The increase in urbanisation and anthropogenic activities surrounding wetlands, has decreased the available wetland area to deliver ecosystem services, thus reducing the ability of the wetlands to deliver ecosystem services. Several management and conservation measures are recommended for implementation by local residents and stakeholders of the municipality. Public participation of inhabitants surrounding the wetlands is encouraged; conservation of the entire wetland habitat, rather than focusing on a specific species, is encouraged; and regular monitoring of invasive plant species is recommended. The findings of this study should be tested in other urban areas in South Africa, to indicate potential general trends and strengthen the basis of ecosystem service delivery research of wetlands which is needed to adequately understand and ultimately limit further wetland loss in South Africa.

Keywords: Wetlands; urban-rural gradient; ecosystem services; plant species composition; plant species diversity; functional diversity

OPSOMMING

Die afgelope twee dekades was daar toenemende ontwikkeling en verstedeliking in Suid Afrika, wat gelei het tot hoogs gefragmenteerde stede. In ons studiegebied, die Tlokwe Munisipale Gebied (TMA), is landbou en verstedeliking verantwoordelik vir meeste van die onomkeerbare transformasie van die natuurlike gebiede, insluitend vleilande. Ekstensiewe grondgebruik veranderinge was gevind na die kwantifisering van die verstedeliking en die transformasie van natuurlike gebiede in die TMA oor ‘n periode van 61 jaar. Verandering in die bedekking van vleilande, het gelei tot verdere ondersoek van die vleilande. Vleilande verskaf ‘n verskeidenheid van ekosisteem goedere en dienste, maar ten spyte hiervan, is dit een van die minste bewaarde ekosisteme in Suid Afrika. Die ontstellende tempo waarteen vleilande vernietig en gedegradeer word is meestal as gevolg van die gebrek aan kennis van die ekologiese en sosio-ekonomiese belangrikheid daarvan deur beplanners, beleidmakers en ontwikkelaars. Die hoofdoel van hierdie studie was om die floristiese samestelling en die lewering van ekosisteemdienste van vleilande binne die bedreigde Randse Hoëveld Grasveld plantegroei-eenheid wat die Oostelike-Gematigde Varswater vleiland tipe bevat, langs ‘n verstedelikingsgradient in die TMA te ondersoek.

Gedurende die veld ondersoek is vleilande binne die TMA geïdentifiseer en geklasifiseer, die grense daarvan was bepaal, en die plantspesie samestelling en plant diversiteit bepaal. Plantopnames het geskied langs verskeie 100m lyn transekte wat uitgeplaas was deur die vleilande. By elke 10m interval, is plantegroei inligting binne ‘n 1m^2 kwadrant versamel. Die plantegroei inligting is eerstens gebruik om die Floristiese Kwaliteit Assesserings Indeks (FQAI) – wat gebruik maak van die konserwiteit van die plantgemeenskap, om die toestand of status van die vleiland plantegroei en habitat te bepaal. Tweedens, is die Vleiland Indeks Waarde (WIV) – wat veranderinge in die vleiland plantegroei en hidroliese prosesse aandui, wat veroorsaak word deur menslike impakte, van elke vleiland bepaal. Waardes vir ekosisteem dienste wat moontlik produseer kan word deur die vleilande was toegeken deur gebruik te maak van die WET-EcoService assesserings metode.

‘n Totaal van 102 plantspesies (68 inheemsespesies, 34 uitheemse spesies) is geïdentifiseer in die 14 vleiland studie gebiede. Vleilande in landelike gebiede het 'n laer plantspesie rykheid as vleilande in stedelike gebiede. Stedelike vleilande het ‘n hoër plantbedekking en verskillende spesie samestelling as landelike vleilande. Een van die landelike vleilande kan gesien word as 'n "oorgang" vleiland, aangesien meeste van die spesies ooreenstem met spesies van beide stedelike en landelike vleilande.

Alle bestudeerde vleilande kan as ‘ware’ vleilande beskou word (almal het WIV <2.5). Twee landelike vleilande het laer WIV waardes gehad as die res. Die ander landelike vleilande het hoër WIV gehad, as gevolg van die hoër volopheid van fakultiewe vleiland spesies en laer volopheid van verpligte vleiland spesies. Die FQAI waarde wat vir elke vleiland bereken was, het nie veel verskil tussen die landelike vleilande nie, en almal het relatief lae FQAI waardes getoon. Relatiewe hoë FQAI waardes was bereken vir die stedelike vleilande in vergelyking met die landelike vleilande. Die algemene kwaliteit van al die vleilande in die TMA word beskou as gedegradeer.

Hoër waardes is toegeken aan die direkte ekosisteemdienste (voorsienings ekosisteemdienste) van die stedelike vleilande, dit sluit in kulturele ekosisteemdienste (toerisme en opvoedkundige en navorsings geleenthede). Die reguleringsdienste (indirekte ekosisteemdienste) van die landelike vleilande het hoër waardes gekry as die van die stedelike vlielande. Hierdie dienste sluit in sediment en fosfaat opvanging, nitraat en toksiene verwydering en erosie beheer. Die lewering van ekosisteemdienste in die vleilande is meestal afhanklik van die bedecking van plantegroei, eerder as deur die tipe plantegroei (tipe spesies) teenwoordig in die vleilande.

Die identifisering van ekosisteemdienste wat gelewer word deur natuurlike gebiede, kan gebruik word as ‘n instrument om beleide op te stel vir die bewaring van biodiversiteit, maar hierdie konsep word nie geredelik ge-implementeer nie. Die toename van verstedeliking en antropogeniese aktiwiteite rondom vleilande, het die beskikbare vleiland oppervlakte wat moontlik ekosisteemdienste kan lewer, en so ook die vleiland se vermoë om ekosisteemdienste te lewer, laat afneem. Verskillende bestuur en bewarings maatreëls is voorgestel om te implementeer deur die plaaslike inwoners en medewerkers van die munisipaliteit. Publieke deelname van inwoners rondom vleilande word aangemoedig; bewaring van ‘n hele vleiland gebied en nie net die bewaring van ‘n enkele spesie nie, word voorgestel; en gereelde monitering van indringerplantspesies word ook aanbeveel. Die metodes en bevindinge van hierdie studie moet ook getoets word in ander stedelike gebiede in Suid-Afrika, om moontlike algemene tendense van stedelike gebiede aan te toon, om die basis van navorsing. navorsing oor ekosisteemdiens lewering te versterk wat benodig word om verdere verliese van vleilande in Suid-Afrika te beperk.

Sleutelwoorde: Vleilande; verstedelikingsgradiënt; ekosisteemdienste; plantspesie samestelling; plantspesie diversiteit; funksionele diversiteit

TABLE OF CONTENTS

LIST OF FIGURES

LIST OF TABLES

ABBREVIATIONS

CARA	Conservation of Agricultural Resources Act
CBD	Convention on Biological Diversity
CoGTA	Cooperative Governance and Traditional Affairs
CSIR	Council for Scientific and Industrial Research
DEAT	Department of Environmental Affairs and Tourism
DMA	Durban metropolitan Area
DRDLR	Department of Rural Development and Land Reform
DWAF	Department of Water Affairs and Forestry
DWS	Department of Water and Sanitation
FQAI	Floristic Quality Assessment Index
GIS	Geographical Information Systems
GPS	Global Positioning System
IPBES	Intergovernmental Platform for Biodiversity and Ecosystem Services
ISCW	Institute for Soil, Climate and Water
IUDF	Integrated Urban Development Framework
MEA	Millennium Ecosystem Assessment
MOSS	Metropolitan Open Space System
NMDS	Non-metric Multidimensional Scaling
NRC	National Research Council
NWDACE	North West Department of Agriculture, Conservation and Environment
PES	Payments for Ecosystem Services

SACS	South African Committee for Stratigraphy
SAfMA	Southern African Millennium Ecosystem Assessment
SAIRR	South African Institute of Race Relations
SANBI	South African National Biodiversity Institute
TMA	Tlokwe Municipal Area
UKNEA	United Kingdom National Ecosystem Assessment
UNDESA	United Nations, Department of Economic and Social Affairs, Population Division
USEPA	United States Environmental Protection Agency
WESSA	Wildlife and Environmental Society of South Africa
WfW	Working for Wetlands
WIV	Wetland Index Value
WWF MWP	World Wildlife Fund Mondi Wetlands Programme
WWF-SA	World Wildlife Fund South Africa

1. INTRODUCTION

1.1 General introduction and problem statement

Globally, more people reside within cityscapes than in traditional rural regions (United Nations, 2008). The 54% of the population that lives in these built-up areas is expected to grow substantially by 2050, with the proportion occupying these urban spaces increasing most dramatically in Latin America (to 90%), followed by Asia (65%) and Africa (62%) (UNDESA, 2015). In the 2014 revision of the World Urbanization Prospects by the UN's DESA's Population Division (UNDESA, 2015), they note that the most significant urban growth will take place in developing countries, particularly in Africa. Given these growth expectations, African countries will face numerous challenges in meeting the basic needs of their growing urban populations, including: housing, infrastructure, transportation, energy and employment, as well as in the provision of services such as education and health care. As is the general trend worldwide, the levels of urbanisation in South Africa have changed substantially, increasing from only 52% in 1990 to 62% in 2011 (World Bank, 2012). Levels of urbanisation are expected to increase rapidly in the coming years, exacerbating the strain on the countries already fragile infrastructure (SAIRR, 2013).

With rapid urbanisation on the horizon, South Africa has a number of challenges to address to aid in the effective and efficient development of its current urban areas as well as the rural regions. This transition is hampered by the fact that the country is of middle-income status and facing significant levels of unemployment (CoGTA, 2014). Urban areas within the country continue to be hampered by the legacy of Apartheid, resulting in a tradition of spatial sprawl, low density, functional segregation between home and work, as well as racial and class separations (CoGTA, 2014). According to the Integrated Urban Development Framework (IDF) (CoGTA, 2014), urban areas are dynamically linked to rural areas in that people move from rural to urban areas where jobs are being created and where household incomes are higher. This rural-urban migration has resulted in many South African cities experiencing an increasing number of informal settlements on the urban fringe. "These fringe-settlements, together with suburbanisation trends, have resulted in habitat fragmentation, as the landscape is transformed to accommodate a bigger urban population" (Cilliers and Siebert, 2011).

With the demand for water exceeding water supply, "water shortage has become more prominent in many cities in both the developed and developing world" (Brooks, 2007). Numerous cities, particularly in a semi-arid region such as South Africa, have outgrown or are on the verge of outgrowing their local water supplies. All available surface water sources which are close at hand and groundwater resources are currently being tapped and exploited in an unsustainable way (Brooks, 2007). In future, "cities will begin to rely on more distant water sources or alternatively, make use of small scale water sources such as wetlands" (Barthel and Isendahl, 2013). Thus, wetlands face potential exploitation, despite the increased difficulty in obtaining water from these areas.

Wetlands are important ecosystems as they provide many benefits, not only to the natural environment, but also to humans in the form of services (Ramsar Convention, 2000). These ecosystem services are provisioned by natural habitats and can be distinguished as provisioning, supporting or habitat services, regulating, or cultural services (MEA, 2005) (more detail in next chapter). In particular, urban areas benefit from wetlands by improved water quality; they serve as reservoirs, and also contain the runoff from roads, drains, roofs and storm water drains, reducing the risk of urban flooding (Braack *et al.,* 2000; Heydorn, 1996). Further, wetlands are also crucial filters of pollution for cityscapes, both in terms of toxins and organics (Kotze *et al.,* 1995). However, for many years, urban wetlands have been regarded as wastelands, with their importance and vital functions not having been fully understood (Cowan, 1999). Thus, we possess a limited knowledge of urban wetlands in South Africa. "Urban wetlands also sustain wetland-dependent organisms which live in multiple local populations and are sustained through migration to other local urban wetlands" (Gibbs, 2000). Maintaining minimum wetland densities in urban landscapes is thus fundamental to conserving these organisms as well as the important ecosystem services provided by wetlands.

Prior to widespread urbanisation, it is estimated that over 10% of South Africa was dominated by wetlands; however, this figure decreases significantly every year owing to unsustainable land-use practices (SANBI, 2014). South Africa has a large diversity of wetland types, the existence, characteristics and processes of which, are determined regionally by variation in climatic conditions, and locally, by variation in geomorphological setting (Ellerly *et al.,* 2011).

The Working for Wetlands Programme (2015) estimates that more than 50% of South Africa's wetlands have been destroyed through drainage for crops and pastures, poorly managed burning regimes, overgrazing, disturbances to wetland soils, vegetation clearing as well as industrial and urban development (including mining activities). Another major cause of the reduction of wetlands is due to the spread of invasive alien species. Alien species are problematic, given that these species consume significantly more water than do native species (Richardson and Van Wilgen, 2004). Once these invader species are introduced into the upper catchments of the wetland, they are quickly dispersed, causing a reduction in water flow. This in turn limits the availability of water to wetlands downstream, disrupting normal wetland ecosystem functioning (Richardson and Van Wilgen, 2004). South Africa is dependent upon water from wetlands as they supply freshwater to urban and rural areas. Plant invaders that restrict water to these areas threaten the already stressed water supply of South Africa

(Turpie *et al.,* 2008). The Working for Water Program in South Africa, an integrated multi-agency intervention that researches, addresses and initiates clean-up programs to affected areas, is an important body controlling the impact of alien invasive species on water courses in South Africa (Richardson and Van Wilgen, 2004)

"Although wetlands are high-value ecosystems that make up only a small fraction of the country, they rank among the most threatened ecosystems in South Africa" (Driver *et al.,* 2012). According to Nel and Driver (2012), South Africa's remaining wetlands were identified as the most threatened of all South Africa's ecosystems (48% of wetland ecosystem types considered critically endangered, 12% endangered and 5% vulnerable) (Mucina and Rutherford, 2006). A mere 11% of wetland ecosystem types are considered well protected, while 71% receive no protection at all. The remaining wetland systems suffer from severe erosion and sedimentation, undesirable plant species invasions, misuse of natural resources, artificial drainage and damming, and pollution (Collins, 2005). It is, therefore important to classify and identify the various characteristic of wetlands in South Africa (SANBI, 2012). Unfortunately, due to the focus on broad scale terrestrial vegetation studies, wetland mapping has for long been neglected in fine-scale vegetation maps (Mucina and Rutherford, 2006).

In 2012, a pilot study of the land-use transformation in the Tlokwe Municipal Area (TMA), surrounding the town of Potchefstroom, over a period of 61 years, showed significantly greater changes in the cover of natural areas versus urban areas (Pretorius *et al.,* 2013). Pretorius *et al.* (2013) found a 23% increase in urban land coverage and a 68% increase in cultivated land-uses, decreasing the coverage of natural habitats by 12% and thus also the coverage of wetland areas in the vicinity. In the TMA, the increase in urban areas and the consequent influx of urban population, lead to the demand for more built infrastructure, an increase in agricultural activities to provide food and an intensified use of the water resources (primarily from the Mooi River passing through Potchefstroom). This demand exacerbated changes to the landscape and hence to the biodiversity within it.

The ongoing agricultural and anthropogenic activities on the wetlands surrounding the Mooi River prompted the question: How does the degree of urbanisation influence the diversity of these wetlands and their ability to function properly and to provide valuable ecosystem services?

1.2 Research aim, objectives and hypothesis

The main research aim of this study was to determine the current floristic composition of wetlands along an urban-rural gradient within the Tlokwe Municipal Area (TMA). Additionally, the ecosystem services being delivered by these wetlands, which are subjected to long term anthropogenic influences and surrounding land-use changes over an urban-rural gradient, will be investigated using a rapid assessment method.

Furthermore, the specific objectives of this study were to:

1. Quantify the urban-rural gradient of the TMA, expressed through wetlands assessed;
2. Delineate and classify the different wetlands within the TMA;
3. Investigate the floristic composition of the wetlands along the urban-rural gradient, focusing on plant diversity and functional traits;
4. Calculate condition of the wetlands using two indicators, namely the Wetland Index Value (WIV) and the Floristic Quality Assessment Index (FQAI);
5. Apply a rapid ecosystem services assessment method (a scoring methodology) to estimate the extent of several ecosystem services provided by the wetlands along the urban-rural gradient, and to compare the ecosystem services delivered by wetlands in urban and rural areas of the TMA; and
6. Provide recommendations for the management and conservation of the wetlands of the TMA, to promote ecosystem services.

The overarching hypothesis of this project could be stated as the following:

When wetlands along an urban-rural gradient are compared, it is expected that the species and functional diversity and ecosystem service delivery of rural wetlands would be higher, than those of the urban wetlands which have more anthropogenic influences.

1.3 Dissertation structure and content

The extent and composition of wetlands and the ecosystem services they deliver are the two main themes explored in this dissertation. The dissertation can further be divided into five main parts:

Chapter 1 and 2 provide an overview, describing the Tlokwe Municipal area and also reviews the literature regarding urbanisation on wetlands and the ecosystem services they deliver, respectively. These chapters thus describe the broad context on which the rest of the dissertation is based.

Chapter 3 provides a full description of all materials and methods used to conduct this study. The location, topography and climatic conditions of the study area are also depicted. Wetland classification and assessment methodology, quantification of urbanisation gradient, the vegetation assessment methodology and methods for determining ecosystem services of wetlands are described. Lastly, the data analysis of each of these topics are explained.

In Chapter 4 the patterns of plant species composition and plant diversity of the selected wetlands within the study area are explored. The results obtained from species composition, diversity and functional diversity of the wetlands are discussed with regard to relevant literature. The condition of the wetland are also discussed in the latter part of this chapter.

Chapter 5 gives a description of the ecosystem services potentially delivered by the wetlands. The wetlands are compared to each other in terms of their wetland setting and by their urbanisation status.

Finally, chapter 6 provides insight to the conservation and management of urban and rural wetlands to prevent further degradation of these wetlands and to promote a better understanding of the preservation of wetland vegetation and ecosystem service delivery.

2. LITERATURE REVIEW

2.1 Introduction

The Ramsar Convention (2013) refers to a ‘wetland’ as “a wide variety of habitats such as marshes, peat lands, floodplains, rivers and lakes as well as coastal areas such as salt marshes, mangroves and sea grass beds, coral reefs (and other marine areas no deeper than six meters at low tide), as well as human-made wetlands such as waste-water treatment ponds and reservoirs”. In a South African context, and according to the National Water Act (No 36 of 1998) (RSA, 1998), a wetland is defined as “land which occurs in a transitional zone between terrestrial and aquatic areas. It occurs where the water table is usually at or near the surface, or where the land is periodically covered with shallow water and which, under normal circumstances, supports or would support vegetation, typically adapted to normal saturated soil conditions”. Thus, as per the above definitions, wetlands should consist of an abundance of water, waterlogged soils, and specialist fauna and flora (SANBI, 2012). They are classified according to their geographical location, the depth of the water covering them, the type of flow and their vegetation type.

Wetlands cover less than 10% of the terrestrial land surface area, but are “one of the most important ecosystem types as they provide various goods and services to both humans and the natural environment” (Ramsar, 2013). Since 1900 more than 50% of the world’s wetlands have disappeared (Stuip *et al.,* 2002). As most of Africa lies within semi-arid and arid climates, wetlands are “a key source of water and nutrients for biological productivity” and hence also a key source to the survival of people in these dry regions (Schuijt, 2005).

The continued degradation of wetlands will impact on biodiversity, ecological function, and the provision of ecosystems services, subsequently impacting on livelihoods and economic activity, as well as the health and wellbeing of local communities (Dini, 2012). Ehrenfeld (2000) listed many different impacts that cause wetland degradation via urbanisation, these include: an increasing number of alien species; disruptive soils suitable for weedy, alien species; solid waste that causes chemical and physical impediments to growth; displacement of certain species by humans through garden refuse; and causing the disappearance of upland habitats/ecotones adjacent to wetlands. The rehabilitation and conservation of wetlands should receive greater prioritisation, particularly in urban areas, since they are more likely to produce a higher quantity and quality of ecosystem services, due to their diverse nature, than would any other green urban area (Ehrenfeld, 2000).

In this literature study, I will expand on ecological studies and more specifically urban wetlands within larger wetland systems, their plant species composition and diversity as well as the ecosystem services they delivered. Although a global view will be given, a focus will be placed on South Africa and Africa in general.

2.2 Ecosystem services

The concept of ecosystem services went through considerable popularisation as a research theme and as a conceptual framework for numerous research projects over the past decade. Publications from de Groot (1992), Costanza *et al.* (1997) and Daily (1997) popularised the basic concept of ecosystem services. Since then, the integrated approach of ecosystem services research has been promoted through inter- and transdisciplinary research, and by defining and analysing the linkages and dependencies between natural and human systems (Burkhard *et al.,* 2009, 2010). Ecosystem services are "the benefits provided by ecosystems that contribute to making human life both possible and worth living" (Costanza *et al.*, 1997). Examples of ecosystem services include products such as food and water, regulation of floods, soil erosion and disease outbreaks, and non-material benefits such as recreational and spiritual benefits in natural areas (UKNEA, 2011). The term 'services' is usually used to "encompass both the physical and non-physical benefits that humans obtain from ecosystems, which are sometimes separated into 'goods' and 'services' "(UKNEA, 2011). Since ecosystem services are defined in terms of their benefits to people, it should be recognised that "ecosystem services are context dependant, that is, the same feature of an ecosystem can be considered an ecosystem service for one person but not valued by another depending on the situation specific to an individual" (UKNEA, 2011).

2.2.1 Classifying ecosystem services

The definition of ecosystem services as provided by by Costanza *et al.* (1997), includes both goods (i.e., resources) and services (i.e., ecosystem processes) to benefit humans. Ecosystem functions are the processes that deliver these services and are necessary for the self-maintenance of an ecosystem and its integrity, and include primary production, nutrient cycling, decomposition, etc. (de Groot. 1992; de Groot *et al.,* 2002). De Groot (1992) explained that a single ecosystem service "can be the product of two or more ecosystem functions, but a single ecosystem function can contribute to two or more ecosystem services".

The most widely used classification system of ecosystem services is the one used in the Millennium Ecosystem Assessment (MEA) (2005), which organizes services under sections of provisioning, regulating, supporting, habitat, or cultural services.

Provisioning services are those that include material outputs from ecosystems, including food, water, medicinal plants, and other resources (Haines-Young and Potschin, 2010). One of the most important provisioning ecosystem services is water. All other provisioning services are inherently dependent on water (e.g. food, fibre, raw materials), as are some regulating ecosystem services (e.g. flood protection, water treatment), supporting ecosystem services (e.g. primary production, photosynthesis), and cultural ecosystem services (e.g. recreation, aesthetic value) (Russi *et al.,* 2013). In South Africa, wetlands help secure water security for surrounding communities, as in the case of the communal wetlands in the Sand river catchment of the north-eastern region of South Africa, where wetlands provide water for mostly agricultural and harvesting purposes (Pollard and Cousins, 2008). Urban wetlands ensure water for urban and peri-urban agriculture, which then also contribute to food security and generate income for vulnerable urban households (Secretariat of the CBD, 2012). Provisioning services by wetlands to agriculture includes habitat for livestock, as well as grazing areas and drinking water for this livestock. Unfortunately, these wetlands are also over utilised by these agricultural practices, diminishing their ecosystem service capacity. Agricultural practices in wetlands can cause excess pollutant run-off, trampling of vegetation and destruction of the wetland setting, all negatively impacting the quantity and quality of available water (Biddle *et al.,* 2007).

Regulating services are the most diverse of the services provided by ecosystems (Brown *et al.,* 2006), "covering factors that affect the ambient biotic and abiotic environment, such as flood and disease control; the impacts of pollination; and pest and disease regulation on the provision of ecosystem goods such as food, fuel and fibre" (Haines-Young and Potschin, 2010). These services are strongly linked to each other and also to other kinds of ecosystem services. The example of water quality regulation provided by UKNEA (2011), links different types of ecosystem services as it is primarily determined by catchment processes. These processes then link to other regulating services, such as erosion and air quality control as well as climate regulation, and also supporting services such as nutrient cycling. Regulating services produced by urban parks and wetlands, for example, reduce the urban heat island effect. Urban temperatures can also be reduced when buildings are overgrown and covered with vegetation such as green roofs and green walls (Secretariat of the CBD, 2012). The vegetation of wetlands, located at, or near river banks or water bodies near an urban setting can help reduce erosion during major flooding as it stabilizes the soil, encourages deposition of sediments, and dampens the flow of water. Overall, urban green infrastructure can substantially contribute to climate regulation by reflecting and absorbing solar radiation, filtering dust, storing CO_2, serving as a windbreak, improving air quality, and enhancing cooling by evaporation, shading, and the generation of air convection (Secretariat of the CBD, 2012).

"Habitat or supporting services underpin almost all other services", in the sense that they maintain the basic conditions for life on earth (Brown *et al.,* 2006). Examples of supporting services on which all other ecosystem services depend includes primary production, soil formation and the cycling of water and nutrients in terrestrial and aquatic ecosystems (Brown *et al.,* 2006). Examples included networks of urban green infrastructure which provide critical resources for wild bees (Threlfall *et al.*, 2015) and avian populations (Marzluff *et al.,* 2001), which are important contributors to pollination in urban areas. Urban wetlands deliver supporting ecosystem services such as pest regulation (especially if there is nearby urban agriculture) (Bianchi *et al.,* 2006), disturbance prevention through water regulation (De Groot *et al.,* 2002), nutrient cycling through soil composition, and also contribute to gas-, climate- and water-regulatory functions (De Groot *et al.,* 2002).

Cultural services are the "non-material benefits people obtain from contact with ecosystems, including aesthetic, spiritual and psychological benefits" (Haines-Young and Potschin, 2010). It also includes places of human interaction and where nature provides these services. The environments delivering cultural ecosystem services result from "the outcomes of interactions between societies, cultures, technologies and ecosystems over millennia" (Haines-Young and Potschin, 2010). Interactions with natural areas affords the opportunity for outdoor learning and many other kinds of recreation, critical to social well-being. Brown *et al.* (2006) states that exposure to urban green infrastructure may have numerous benefits to visitors to these areas, including aesthetic satisfaction, improvement in overall health (Horwitz and Finlayson, 2011; Cools *et al.,* 2013) and fitness, and an enhanced sense of well-being. Jackson (2003), thus advocated the importance of the accessibility of urban dwellers to these urban green spaces, not only for their inherent recreational value but also for health reasons.

2.3 Urban ecosystems

Urbanisation is characterised by an increase in the diversity of human occupancy, together with extensive modification of the landscape, whereby a system that fails to take the depletion of natural resources into account, is created (McDonnell and Pickett, 1990). This urbanisation process is mostly driven by political, economic and cultural decisions, leading to extensive land-use conversions of natural environments to highly disturbed man-made regions (McDonnell and Pickett, 1990).

Niemelä (1999b) defined 'urban areas' as a fairly large, densely human populated area considered as having industrial, business and residential districts. This definition of Niemelä (1999b) is more fitting for the purpose of urban ecological research because it is often difficult to draw any definite ecological borders around an urban area. With urban areas being identified as having a human presence, they are often compared to a "natural area", which is considered to be an area in and around an urban space with a near absence of humans (McIntyre *et al.,* 2000).

These natural areas are usually the vegetated parts of urban ecosystems and are known as 'urban green spaces' or 'urban green infrastructure'. The U.S. Environmental Protection Agency (EPA), defines urban green infrastructure as a concept used to describe an array of products, technologies, and practices that use natural systems or engineered systems (which mimics natural processes) to enhance overall environmental quality and to provide utility services (USEPA, 2014). At the scale of a city, green infrastructure is considered to be "natural elements" that provide habitat, flood protection, cleaner air, and purified water. For example, at the scale of a neighbourhood or site, green infrastructure may refer to storm water management systems that mimic nature by capturing and storing water (USEPA. 2014). Trees located in streets, open grassy areas in parks and lawns, urban forested areas, wetlands, lakes and streams are examples of green infrastructure and can also therefore be identified as natural urban ecosystems (Bolund and Hunhammar, 1999). Many authors also identify parks, urban forests, farmlands, natural areas, golf courses and sport fields as part of urban green infrastructure (Li *et al.,* 2005.; Sanesi and Chiarello, 2006; Konijnendijk *et al.,* 2006)

In terms of ecological studies, green infrastructure is one of the main contributors to conserving biodiversity in urban areas as these areas have a decreased number of available producers for ecosystem services in an urban setting (Bibby, 2009). However, it is possible to reverse these land-use changes, e.g., Bibby (2009) showed that most new housing is built within existing residential and housing development areas or in small rural developments, whilst crop and grazing land has been transformed into woodland rather than into housing. As discussed in Ehrenfeld (2000) and Baldwin (2004), urban habitats differ physically and biologically from rural systems in a number of ways. For example, when comparing urban wetlands to those situated in a more rural setting, urban wetlands function differently from rural wetlands as urbanization affects the hydrology, geomorphology, and ecology thereof. Furthermore, wetlands in urban regions may "take on anthropogenic values that they lack in rural environments, as they provide some contact with nature, and some opportunities for recreations that are otherwise rare in an urban landscape" (Ehrenfeld, 2000). Physical alteration of these habitats, such as ditching and diking, is also more common in urban settings. The species composition in urban habitats are often limited in its seed-dispersal capabilities or mutualistic interactions, such as pollination, and the possible range of habitat types is often limited. Green spaces help to improve connectivity within these areas as they function as corridors or enlarge the size of other urban habitats (Goddard *et al.,* 2009).

Urban habitats and ecosystems created by urban green spaces increase the overall vegetation cover, thus contributing to the conservation (Bratton, 1997) and integrity of habitat systems. Urban habitats may also provide the basis for urban ecological networks, thereby alleviating the ecological impacts of habitat fragmentation (Tzoulas *et al.,* 2007). Overall, urbanization impacts biodiversity and ecosystem services at various scales, resulting in the modification of existing ecosystems and result in creating ecologically different urban environments (Niemelä *et al.,* 2011; Williams *et al.,* 2009). Despite urbanisation leading to ecosystem simplification, biotic homogenisation (McKinney, 2006) and affecting local and regional energy balances, urban ecosystems can still provide valuable ecosystem goods and services (Boland & Hunhammar, 1999).

A widely and commonly used approach to studying urban areas is to quantify an urban to rural gradient. This spectrum of study, moving from the inner urban area to the less populated rural environment, are known as urbanisation gradient studies. Niemelä *et al.* (2000) defined an urbanisation gradient as "an urban landscape consisting of a densely built and developed core surrounded by an area of decreasing development and increasing 'naturalness'". The urban-rural gradient approach in urban ecology has allowed scientists to study the effects of urbanisation pressures on patterns, processes, fauna and flora of complex urban ecosystems (McDonnell and Pickett, 1990). This gradient approach is often subjectively determined based on geographical location in relation to urban cores (Cuevas-Reyes *et al.,* 2013). A gradient typically consists of a high human density urban area together with a high concentration of impervious surfaces at one end of the gradient (McDonnell *et al.,* 1997). Whilst rural environments, at the other end of the gradient, have less built infrastructure and lower concentrations of used energy, materials, water and waste products, due to the lower density of inhabitants (McDonnell *et al.,* 1997).

Many attempts have been made to study urban-rural gradients. Previously this gradient approach was successfully used to study the variation in plant species richness across urban-rural gradients (McKinney, 2008) in response to important urbanisation drivers, such as population density (Luck, 2007). Some examples of vegetation focussed studies include: Qureshi *et al.* (2010) who examined green space functionality along an urbanisation gradient in Pakistan; Hayasaka *et al.* (2012) who surveyed roadside vegetation along an urban-rural gradient in Japan to determine the variation thereof; and those of Hahs and McDonnell (2006) who studied measures to quantify Melbourne's urbanisation gradient. Hahs and McDonnell (2006) used a combination of landscape metrics and demographic- and physical variables for a more complete expression of ecological responses to urbanisation, to ultimately quantify the urban-rural gradient. Du Toit (2009) tested these measures of urbanisation whilst studying the grassland ecology for the South African city of Klerksdorp, and confirmed the usefulness of using landscape metrics in association with demographic- and physical variables. Thereafter, Du Toit and Cilliers (2011) identified several other aspects that must be considered when using urbanisation measures to quantify an urban-rural gradient. According to Du Toit and Cilliers (2011), these aspects include "analysis scale, spatial resolution, classification typology, input data accuracy, measure equations, statistical analysis type, and habitat context".

Ultimately, the use of urbanisation measures ensures a far more objective and statistically defined urbanisation gradient (Du Toit, 2009; Hahs and McDonnell, 2006; Lockaby *et al.*, 2005). Combining gradient analysis and landscape metrics, as in the study of Luck and Wu (2002) in Arizona, USA, it is clear that different land-use types can be characterised by distinct spatial signatures that can be correlated with certain spatial metrics. This highlights the importance of urbanisation gradient quantification for associating pattern and process in urban ecological research (Luck & Wu, 2002). McDonnell and Pickett (1990) stated that "by quantifying the urban-rural gradient of a certain environment, it contributes to the understanding of how organisms respond to the continuous process of urbanisation with humans as an integral part of urban ecosystems".

2.3.1 Urban ecological studies in South Africa

Since the adoption of the Southern African Millennium Ecosystem Assessment (SAfMA) in 2004, there has been an increase in the conservation and research of urban nature in South Africa. However, even before the SAfMA, the implementation of the "Metropolitan Open Space System (MOSS)" approach in 2009, which is based on biogeographical and ecological guidelines, was already being implemented in Durban (eThekwini Municipality Environmental Management Department, 2009). Since then, several other South African cities such as Johannesburg (Strategic Environmental Focus, 2002), Port Elizabeth (Stewart, 2006) and Cape Town (Hennessy, 2000) have adopted this system. This approach focusses on creating ecologically viable and self-sustaining systems within the urban context, ensuring the conservation of areas that would not have previously been prioritised for conservation (DMA, 1998).

Cilliers and Siebert (2011), provide a synthesis on South African urban ecological research in general. They highlight the concept of 'urban nature' which has been increasingly investigated in South Africa with the realisation of its ecological importance. Some examples of South African 'urban nature' studies include: the study of Du Toit (2009) focussing on grassland ecology in Klerksdorp; Van Der Walt *et al*. (2014) who compared plant species composition, plant species diversity and plant species functional diversity of grassland fragments in Potchefstroom; the investigation of avian communities in Pretoria (Van Rensburg *et al.*, 2009); the examination of a guild of nectar-feeding birds in a biodiversity hotspot in Cape Town and the effect of urbanisation on it (Pauw and Louw, 2012); the urban nature study of Dyssel (2013) focusing on urban nature conservation concerns in Bellville and, the study of Anderson *et al.* (2014) which explored the ecological advantages of using indigenous plants greening interventions in Cape Town.

An additional approach to examine the ecology of urban areas have also been identified. This approach "shows the importance of biodiversity to human well-being by evaluating the provision of numerous goods and services by natural ecosystems" (O'Farrell *et al.,* 2011). This ecosystem services approach is a fairly new concept to understand in ecology, differing from the usual urban ecological studies which are mainly driven by conservation concerns for urban nature (Cilliers and Siebert, 2011).

2.3.2 Urban ecosystem service studies in South Africa

Cilliers *et al.* (2013) reported that in general there is a lack of focus on ecosystem services in urban areas in developing countries, especially in Africa. Most studies on the status of biodiversity have focused on species composition (Biggs *et al*., 2004) and little research has been done on the link between biodiversity and ecosystem services, especially in developing countries (Mertz *et al*., 2007). Since the release of the Millennium Assessment in 2005, South Africa adopted the Southern African Millennium Ecosystem Assessment (SAfMA) which "provides southern African decision-makers with critical information about the state of important ecosystem services, helping them understand the types of policy interventions, trade-offs and management types required for achieving sustainable ecosystem service delivery in the region" (MEA, 2005). SAfMA developed several ways to measure the condition of biodiversity, one of them is the measurement of ecosystem services. Le Maitre *et al.* (2007) reported that there were only 18 studies in South Africa, which focussed on ecosystem services at that time, and were diverse in topics and none addressing urban ecosystems. According to Mertz *et al.* (2007) there "remains a lack of information about the link between biodiversity and ecosystem services, especially in developing countries". There are however, a few good examples of ecosystem service studies of cities in developing countries.

Roberts and Diederichs (2002) carried out one of the first studies to use the concept of ecosystem services in South Africa. As a result, the city of Durban, which has been regarded as a leader in urban open space planning in South Africa, has changed their conservation priorities to a more sustainable development mindset. This change increased the protection of the biodiversity within the city adding to the protection of the ecosystem goods and services being delivered to the surrounding communities (Roberts and Diederichs, 2002).

Several other ecosystem services studies have been conducted in the city of Cape Town in South Africa. O'Farrell *et al.* (2011) developed a rapid assessment method to identify areas where ecosystem services are deteriorating. Since Cape Town is situated in a biodiversity hotspot, priority conservation areas need to be identified to prevent further loss in biodiversity. Cilliers and Siebert (2011) also recognised the value in studying ecosystem services important to urban nature conservation. The study of Turpie *et al.* (2008) together with the Working for Water program addressed payments for ecosystem services (PES) systems which were the result of restoration of catchments in South Africa. With the success of the program, it then expanded into other types of ecosystem restoration projects that have the potential to merge into a general program of ecosystem service provision within a broader public works program. This started a strong case for focussing on the most valuable services provided by ecosystems, to use as 'umbrella services' to achieve more common environmental goals, such as conservation goals (Turpie *et al.,* 2008).

The link between ecosystem services occurrence and different aspects of biodiversity in South Africa was addressed in the study of Egoh *et al.* (2009). They aimed to determine whether biodiversity priorities, biomes, species richness and vegetation diversity hotspots co-occur in space with ecosystem services. Five ecosystem services were assessed in South African biomes, namely supplying surface water, regulation of the flow of water, carbon storage, accumulation of soil, and soil retention abilities. Some of the biomes deliver all five ecosystem services, but there is a low correlation between hotspots of ecosystem services and plant species richness and plant diversity hotspots (Egoh *et al.,* 2009). The result of this study indicated the need to prioritise biodiversity conservation actions in South Africa, which could lead to the protection of ecosystem services. Limiting the impacts on these ecosystem services, on the other hand, can also be used to reinforce biodiversity conservation in some areas.

The Council for Scientific and Industrial Research (CSIR) in South Africa together with the Intergovernmental Platform for Biodiversity and Ecosystem Services (IPBES) are examples of government entities who play key roles in the SAfMA in developing the science that is essential for ecosystem assessments and in providing information, tools and knowledge useful to integrate ecosystem services science into policy and planning. The condition of green spaces within urban areas underpins the functioning of urban ecosystems (Cilliers *et al.,* 2013). Thus, not only should government initiate the conservation of ecosystem services, local municipalities, residents and the surrounding communities should also form part of a joint venture to protect green remnants within their setting.

Studies such as those of Cilliers and Siebert (2011), Colding (2011) and Tzoulas *et al.* (2007), all stipulated the importance of including the total urban green infrastructure in the planning and management of green spaces, which will in turn increase the sustainability in urban areas through protecting the ecosystem services.

2.4 Wetlands in South Africa

"Wetlands are composed of a number of physical, biological and chemical components, such as soils, water, plant and animal species, as well as nutrients" (Dugan, 1990). Linkages among and within these components allow the wetlands to deliver certain functions and services (Dugan, 1990).

Wetlands are one of the most endangered ecosystems in South Africa. Unfortunately, the conservation of wetlands is not always prioritised, even though there is sound legislation in place for the conservation of natural areas within urban developments in South Africa (Working for Wetlands, 2015). The application and interpretation of legislation for the protection of wetlands are often left to be implemented by the local government, who is not always well informed or aware of unlawful wetland uses, or often neglects many of the requirements to implement such legislation (Lubbe *et al.,* 2010).

According to DWAF (2001), major threats to wetlands in South Africa include anthropogenic activities and human-induced land-use changes. Such changes are caused by artificial canalisation, drainage of the landscape, agricultural activities (specifically cropping), disposal of effluent into freshwater resources, disposal of sediment and/or rubbish/rubble, using wetlands as stormwater management areas, invasion by alien invasive plants, and extensive extraction of water. Further, Mucina and Rutherford (2006) identified the conversion of a wetland from one form to another as being the biggest threat to the conservation of wetlands. This conversion includes the establishment of built infrastructure in the wetland, causing changes to the general functioning of the wetland (Mucina and Rutherford, 2006). Both urban and rural wetlands have experienced some form of degradation. Impact drivers such as agricultural activities and overgrazing were evident in rural areas, whereas, urban wetlands were most influenced by infrastructure development and commercial and residential activities (McInnes, 2010; Lubbe *et al.,* 2010; DEAT. 2005; Driver *et al.,* 2005). In recent years, there has been a "widespread increase in fear of the future sustainability of these important ecological areas" (Masando, 2011). This concern was raised at the National Wetland Forum Indaba (October, 2009), focusing on of the diminishing conditions of some of South Africa's Ramsar-status wetlands (DEAT, 2012a). The indaba also highlighted that mining, urban development and alien plant invasions are significant threats to wetlands in South Africa (DEAT, 2012a).

In some areas, development surrounding urban wetlands has had a detrimental effect on these wetlands, and are then also considered as land for prime development (Govender-Ragubeer, 2014). An example of this is the Princess Vlei urban wetland area in Cape Town, Western Cape, which was threatened with destruction by the proposed construction of a shopping mall and taxi rank (Ernstson & Sorlin, 2013). Another example is the Libradene Wetland, in Boksburg, Gauteng, which was partially destroyed in an attempt to construct a petrol station on the land (Govender-Ragubeer, 2014). An example of a wetland still under threat from mining is the Wakkerstroom wetland in Mpumalanga (Sieben, 2012). According to the DEAT (2012a), existing wetland conservation policies fall short as economic development imperatives are undermining sound wetland conservation policies. An example of which is the 2009 application for approval for open-cast tungsten mining to commence near the Verlorenvlei wetland (a Ramsar site in the Western Cape) (Birdlife South Africa, 2009). Thus, it is clear, that many decisions made by both private landowners and public agencies can negatively affect wetlands (Govender-Ragubeer, 2014). This lack of awareness of the value of urban wetlands can drive the direct loss of terrestrial ecosystems (MEA, 2005).

Urban wetlands have, however, been shown to bring value to urban areas, as these ecosystems have been used to; increase efficient water usage (Rogers *et al.,* 2002), to decrease impacts on the hydrological cycle (Woods-Ballard *et al.,* 2007), manage urban water issues when linked to biodiversity (Brenneisen, 2006), mitigate and adapt to climate change (McEvoy & Handley, 2006) as well as to develop urban design solutions for issues pertaining to water areas (Hoyer *et al*., 2011). In addition, "to direct impacts on urban ecosystems, the densification and sprawl of built structures, if left unchecked, can generate impacts across a range of hydrological processes" according to Fitzhugh and Richter (2004). Cities can be an unsustainable strain on wetland water resources (Fitzhugh and Richter, 2004), by depleting groundwater (Bolund and Hunhammar, 1999) and polluting aquatic ecosystems (Longcore and Rich, 2004) should they not be properly managed.

Due to the vulnerability of wetlands in South Africa it is important to assess them for making comparisons on their health, to promote their protection (through conservation) and also to determine a better rehabilitation plan for already damaged wetlands. Examples of determining the vegetation compositions of wetlands, which can be used as an indicator to wetland status, are the studies of Cilliers *et al.* (1998) on the urban wetlands in Potchefstroom, the study of Van Wyk *et al.* (2000) on the urban wetlands in Klerksdorp and the report of Sieben (2011) regarding the vegetation of wetlands in in several South African provinces, including KwaZulu-Natal, Free State and Mpumalanga.

2.4.1 Wetland ecosystem services studies

Duan *et al.* (2011) showed that it is not only natural wetlands that can provide ecosystem services. Investigating the constructed Green Lake urban wetland park of Beijing, in terms of its ecosystem services and other environmental indices, Duan *et al*. (2011) showed that this park presents a combination of natural and urban ecosystems. It provides an urban refuge for species, eco-tourism, and environmental education and provides recreational opportunities, thereby creating a balance of conservation and sustainable utilisation of resources. Recognition of these cultural and habitat ecosystem services promotes better environmental support and ecological and economic benefits to the surrounding communities (Duan *et al.,* 2011).

Emerton *et al.* (1999) was the first to attempt to quantify the value of Ugandan wetland ecosystem services. The study of the Nakivubo urban wetlands in Uganda, aimed to quantify the value of wetland wastewater purification and nutrient retention functions to use for industrial and residential developments, but failed to evaluate the state of ecosystem biodiversity. Since then, further studies focussed on the Nakivubo urban vegetation structure and the nutrient retention capabilities which could be of great value to the urban areas surrounding the wetlands (Mugisha *et al.,* 2007).

Comparative studies have been undertaken between wetlands from Europe and Africa, to better understand barriers to the implementation of the Integrated Water Resources Management (IWRM) plan set out by the Ramsar Convention. An example of this is the case study presented by Rebelo *et al.* (2013), which indicated the lack of recognition of the ecosystem services provided by wetlands in developing countries when the management of ecosystem services from the Lobau wetland in Austria was compared to that of the Inner Niger Delta in Mali.

Lately, ecosystem service studies do not only focus on an entire region *per se*, but rather include elements or individual ecosystem services that could be beneficial. One of the most studied ecosystem services of wetlands is that of water quality regulation, especially in developing countries (i.e. Kenya) where there is a lack of wastewater treatment systems (Namaalwa *et al.,* 2013). Bateganya *et al*. (2015) found that this research alone was enough to include wetland ecosystems in the urban planning within the Masaka municipality of Uganda, to enhance municipal wastewater management and pollution control. Cools *et al.* (2013) also used wetland water quality regulation to assess the impact and adaptive capacity of water management options in the urban wetland context of the Inner Niger Delta in Mali.

O'Farrell *et al.* (2012) determined the degree of ecosystem services (including some wetland-related services) being contributed by the flora to the urban area of Cape Town. The rapid assessment method used suggested that the change in vegetation coverage within the municipal areas has impacted on the provisioning and regulating of ecosystem services (land capability, grazing, flood mitigation, soil retention, critical infiltration, coastal zone protection, groundwater recharge, groundwater yield and groundwater quality) being delivered. Since regulating ecosystem services only function *in situ*, the impact of the loss of properties that can deliver them (i.e. vegetation), has a more severe effect than the loss of provisioning services. Since Cape Town is situated in a biodiversity hotspot, this loss of regulating ecosystem services should encourage stricter management of biodiversity (O'Farrell *et al.,* 2012).

2.4.1.1 Methods of investigating wetland ecosystem services

Even though the rapid assessment method of O'Farrell *et al.* (2011) proved sufficient to investigate the ecosystem services being delivered by the urban flora of Cape Town, various other techniques have been developed to investigate ecosystem services.

Egoh *et al.* (2007) identified three main methods in which ecosystem services are currently being accounted for. These methods are based on the analysis of biodiversity patterns (weighting such as biodiversity units by some criteria or value to make some overall assessment), on ecological processes (focus on a particular service and seek to uncover the mechanisms by which it is generated), and mapping approaches (considers how biophysical units can be constructed and used to examine changes in service output over space and time). The method of ecosystem services assessment of wetlands in South Africa most often used is the WET-EcoServices method (Kotze *et al.,* 2008), a method based on ecological processes, which form part of a series of tools that has been developed to assist those wishing to undertake wetland rehabilitation in a well-informed and effective way in a South African context. These tools were developed as part of a comprehensive research programme on wetland management which was initiated by the Water Research Commission (WRC) to examine wetland rehabilitation, wetland health and integrity, the services they supply, and the sustainable use of wetlands (Kotze *et al.,* 2008).

In South Africa wetland ecosystem services are not essentially evaluated for the sake of research, but more often by environmental consultants for the compilation of environmental impact assessments (i.e. SRK Consulting, 2012; Baxter, 2013). By using the WET-EcoServices manual, wetland assessments and the reporting thereof, are conducted primarily to assess the potential impact of proposed developments in or near wetlands, while also supplying suitable recommendations in terms of rehabilitation practices where relevant. WET-EcoServices is a user friendly and accredited method evaluating the ecosystem services of the wetland at hand, based on its hydro-geomorphological (HGM) setting and the biota which inhabit it (Kotze *et al.,* 2008). As vegetation is one of the first determinants to identify a wetland, assessment of the wetland vegetation includes consideration of the extent of total removal of the (indigenous) vegetation and its replacement (notably by planted crops), as well as the extent to which areas of natural or semi-natural vegetation have altered species composition through increased abundance of ruderal (weedy) or invasive plants (MacFarlane *et al*., 2009).

2.5 Plant species diversity and plant species composition of wetlands

"Studies on wetland vegetation community structure and the role of environmental change in this structure is fundamental towards defining these ecosystems" (Ruto *et al.,* 2012). Every wetland has its own unique species diversity and community structure that is subject to changes (Ssegawa *et al.,* 2004). Wetland vegetation could be used as an indicator for responses to changes (whether it be chemical, physical, and biological) in an ecosystem (Cronk and Fennessy, 2001). To preserve the biological diversity for maximum ecosystem services delivery, the most direct approach is to measure the quality of a wetland by assessing its biota (Fennessy *et al*., 2004). Vegetation studies in urban environments are important to ensure and motivate the inclusion of green infrastructure in open space planning in urban areas (Roberts, 1993).

Due to these wetland habitats being exploited and degraded, focus has turned on understanding the response of the plant communities of these wetlands to these alternations (Cronk and Fennessy, 2001). Although high diversity in plant species composition has been recorded in various wetland habitats (Janousek, 2009), many African wetlands are characterized by only a few tropical species of the genera *Phragmites, Typha*, and species of *Cyperus* (Harper *et al*., 1999; Owino and Ryan, 2007).

Whittaker (2006) stated that "the diversity of plant species within a plant community reflects the complexity in their physical, chemical and human environments". This diversity and the complexity thereof can be to the result of the strong relationships between environmental quality and species composition and abundance (Gichuki *et al.*, 2001). This effect is even stronger, when coupling effects are in those areas prone to anthropogenic disturbances in the catchment of a water body.

Several studies, including that of Abila *et al.* (2008) and Stumm *et al.* (2009), have emphasized environmental change as “the major driving force in regulating plant species composition in wetlands”. The report of DWAF (2005) has shown that environmental disturbances could decrease species diversity to such a degree that wetland habitat is dominated by only a few species. Some studies, such as those of Allen et al. (2005) and Abila et al. (2008), have also shown that human factors have a fundamental influence over plant species composition of wetland environments. According to Reeves and Champion (2004) the species diversity of wetlands could be decreased by cultivation and farming activities, and by fire and grazing (by cattle and other livestock) within wetland environments. Habitat influences such as clearing land for cultivation, construction of settlements or draining of wetlands, degrade the habitat and can cause local extinction of some species, thus reducing habitat species diversity (Primack, 1993).

It is well documented that changes in land-use that involve a significant increase in impervious surfaces (i.e., concreting), result in an increased surface water runoff (Sajikumar and Remya, 2015), a situation which characterises urban areas. Change in land-use adjacent to wetlands influences the runoff characteristics of a drainage basin, which in turn, affects the surface and groundwater availability of the area (Sajikumar and Remya, 2015), and thus determines the water available to vegetation for survival and growth. Another common disturbance in wetlands is land-use change in the greater catchment area, which also includes the conversion to agricultural and urban land-use types (Gusewell and Klitzli, 1998; MEA, 2005; Zorrilla-Miras *et al.,* 2014) which then may significantly impact downstream wetland areas.

Mitsch and Gosselink (2000) noted that “a high proportion of global wetland loss is due to drainage for agricultural purposes”. Most remaining wetlands are consequently smaller and more isolated than in the past. For instance, in the USA, in fie agricultural counties in Ohio, 75% of the wetlands are less than 0.4ha in size (Fennessy *et al.,* 1994). Fragmentation of habitats on this large scale has had a profound impact on the ecosystem diversity, services and functions of these wetlands because of the reduction in available biota and hence reduced the quality and quantity of services that these wetlands can provide.

“The composition of a wetland plant community has also been shown to serve as a practical indicator for ecological stress” (Lopez and Fennesy, 2002). Changes in vegetation represent a community level response that integrates the effects of a wide range of ecological stressors. Predictable changes in vegetation community composition, abundance of different plant species, productivity, and other ecosystem properties occur as environmental conditions shift (Lopez and Fennesy, 2002).

Vegetation is the structural foundation of wetlands, and particular species and growth forms are indicative of different wetland types because of their sensitivity to certain processes such as the type of hydroperiod, hydrodynamics, nutrient availability, and other environmental conditions. Interactions between biota and these processes can cause subtle and dramatic shifts in wetland plant biodiversity (Mitsch and Gosselink, 2007). Urbanization creates a physically and biologically different environment from non-urban settings (Ehrenfeld, 2000), which in turn alters the fundamental controls on vegetation structure and function in urban wetlands. Wetland vegetation is adapted to soil that is deprived of oxygen. Their roots are also submerged for shorter to longer periods of time, requiring them to persist where most plants cannot (SWAF, 2005). This type of vegetation could be used as a reliable indicator of a wetland's presence, since only certain plant species are able to thrive in such conditions. Thus, wetland vegetation can be used as an indicator of habitat integrity and health as they show differential tolerance to environmental conditions (DWAF, 2005) and are thereby critically important to research, management, conservation efforts, and form a fundamental component in wetland delineation (for regulatory purposes). Vegetation studies in Europe are often used to implement and improve guidelines for the management of urban green spaces. Unfortunately, in South Africa it is mainly used to identify conservation targets for funded projects (Rutherford *et al.*, 2012) and not necessarily for use by municipalities to consider green spaces in urban planning.

Vegetation plays a critical role in nutrient cycles and productivity of many terrestrial and aquatic systems, and therefore accurate determination of plant cover is essential for broad-scale assessments of ecosystem structure and dynamics. Assessments over an urban-rural gradient are necessary to "measure the extent to which natural vegetation in the wetland has been replaced and invaded by alien species", or been substituted by non-wetland species (Wear *et al.,* 1998).

2.5.1 Assessment of wetland condition

Vegetation assessments, i.e. estimate of species richness and species diversity (Bavrlic and Bowers, 2010), provide several useful indicators of ecosystem health. Environmental degradation, which includes environmental pollution, and climate and land-use change could be detected early on, by utilizing the monitoring of herbaceous vegetation as an early warning system (Roberts-Pichette and Gillespie, 1999). Furthermore, assessing vegetation regeneration provides a measure of vertical structure and allows for an enchanced understanding of vegetation succession within each community. Knowledge on the different successional changes of wetlands, could influence the management strategies to be used. The vegetation of wetlands is known to be a useful indicator of biotic integrity, as plant communities respond to water quality, hydrologic modifications, chemical pollution and nutrient enrichment (Albert and Minc, 2004). According to Van Wieren and Zorn (2005) fluctuation in sediment levels, increases in the water nutrient content and turbidity affects the submergent species richness, whilst enriched inputs affects the emergent species. Furthermore, alien species often characterise wetland degradation (Zedler andKercher, 2004). The coverage of alien species is a good measure of wetland condition, as wetlands within heavily urbanized environments tend to be dominated by alien species and have a lower species diversity than what the previous natural state of the wetland would have (Albert and Minc, 2004). The goal of the vegetation assessment is to ultimately identify biological characteristics that provide reliable information to enable the determination of wetland condition.

The Floristic Quality Assessment Index (FQAI) is a recognized assessment technique for evaluating habitat condition or status (with specific reference to the vegetation habitat) using conservatism of the plant community, which has been impacted on by anthropogenic activities. This method was originally developed for prairies by Swink and Wilhelm (1979), but has since then been used for determining the floristic quality of woodlands (Francis *et al.,* 2000) and depressional marshes (Cohen *et al*., 2004), in the ecological assessment of open and natural land areas (Wilhelm, 1977; Wilhelm and Ladd, 1988), and determining the native species diversity in a managed grassland (Jog *et al.*, 2006). Despite its myriad of uses, this method is still considered most popular for wetland studies (Bried *et al.,* 2013).

A challenge in assessing the overall condition of wetlands is to ensure that measurements are related to function or condition of the wetland and not natural (be it temporal or spatial) variability, and thus, the basic principle of the FQAI method echoes human disturbance or on site condition with minimal natural ecological disturbance (Bried *et al.,* 2013). Other indices typically used to quantify habitat diversity (i.e. Shannon index of diversity) does not take the differences in the identity of species into account, but does make use of both plant species richness and plant abundance distribution (Botta-Dukat, 2005). These indices are then considered not of much use for conservation purposes, since they do not discriminate between the communities composed of widely and distributed species with those composed of specialized or rarer ones (Francis *et al.,* 2000). The FQAI method is thus used as an alternative to typical indices, since it is based "on the use of diversity indices that weight the 'value' of the species occurring within the plant communities and is thus more specialised" (Lopez and Fennessy, 2002).

The conceptual basis of this assessment type is that it allows the condition of an area to be evaluated by examining the degree of ecological conservatism or the tolerance (species conservatism) of a species present at that site (Andreas and Lichvar, 1995). Since this assessment technique is based on expert knowledge of each plant species' tendency to occur, adapt and survive in a certain habitat, the subjectivity associated with the assignment of conservancy values has led to scepticism from many environmental practitioners (Bourdaghs *et al.,* 2006; Johnston *et al.,* 2009). However, the use of FQAI has proven to be successful in many wetland ecological studies, such as the study of Miller and Wardrop (2006), which used the FQAI to indicate the level of anthropogenic disturbance to wetlands located in Pennsylvania, USA. The specialist study of Van Deventer *et al*. (2013) used the FQAI assessed for wetlands located in KwaZulu-Natal (RSA) to determine their condition for the allowance for possible future construction activities; and Cowden *et al*. (2014) used the FQAI calculated for two rural wetlands (KwaZulu-Natal, RSA), on two different occasions, to determine their condition after rehabilitation.

The study of Cowden *et al.* (2014) assessed the vegetation composition of two South African wetlands, pre- and post-rehabilitation using the FQAI method. The results of the Cowden *et al.* (2014) study indicated that one wetlands' vegetation has improved following rehabilitation, whilst the other wetland was still dominated by pioneer species and in a transformed state. This strongly suggested that the FQAI is a valuable tool, which could be repeated after a time period to provide indication of changes to a wetland.

Cowden *et al.* (2014) also determined the Wetland Index Value (WIV) for each of the wetlands they studied. The WIV was developed as a tool to measure changes in vegetation that are indicative of changes in hydrologic processes because of anthropogenic disturbance (Coles-Richie *et al.,* 2007). In the study of Carter *et al.* (1988) vegetation data along transects within a wetland-terrestrial transitional zone were collected to determine the wetland boundary of the Great Dismal Swamp in Virginia, USA. This index uses 'a single number, or index number, to summarize quantitative data on many species in a community or site and weight the contribution of each species to the final number through the use of an indicator value" (NRC, 1995). This reflects the affinity of a species to occur in a wetland (NRC, 1995). This indicator value is species specific, based on the classification of being likely to occur in more hydric (wetland) environment- also termed the species 'wetland indicator status' (See Section 3.4.3).

The application of the WIV is not only limited to determining wetland boundaries but could also be used to indicate the extent of certain impacting factors. Atkinson *et al.* (1993) used the WIV to evaluate early site conditions after artificially creating wetlands, to calculate the percentage wetland and non-wetland vegetation within these created sites, as an early monitoring tool. Even though the results indicated that wetland and non-wetland species estimates were similar within the surveyed sites, the vegetation may provide an early indication of conditions within the created wetlands, meaning that the plant species established in certain areas, are adapted to other factors such as soil moisture and overall hydrology of the sites. Summerford (2009) combined WIV and soil redox measurements to evaluate flood-irrigated land for potential changes in wetland or non-wetland vegetation communities after management has taken place. Since the wetland index for a site can be calculated as a weighted average of the wetland index for each species recorded at a site, riparian community types were used instead of the individual WIV of the species in the study of Coles-Richie *et al.* (2007).

WIV can also be used in combination with other impacting factors (i.e. soil redox, hydrology), as indicators of different wetland and non-wetland species occurrences due to the degree of wetness of a site, causing certain plant species to establish within certain sites. Ultimately the application of WIV in wetland ecology is wide spread and can be “used to monitor wetlands sites over time, compare wetland sites with similar environments, or compare sites with different environmental impacts” (Coles-Richie *et al.* 2007).

2.5.2 Plant species diversity

Few studies have investigated plant species diversity and community composition in the wetlands of the grassland biome in South Africa, perhaps because they are considered to be relatively small. The studies of Cilliers *et al.* (1998) and Van Wyk *et al.* (2000) are examples of urban wetland species composition studies done within the Tlokwe Municipal Area and in Klerksdorp respectively. Plant species diversity can be measured in different ways, most of the time focusing on measuring species richness and/or species evenness (Crist *et al.,* 2003). Species richness is simply the number of species per wetland, and species evenness is the relative abundance of species within these wetland sites (McGinley, 2014). Measuring these components of diversity will give an indication of the state of biodiversity and resources available within the wetlands and is also useful in tracking environmental changes (Rojo *et al.,* 2012)

Anthropogenic influences are one of the most dominant and persistent driving forces of species richness within urban areas (Lubbe, 2011). Species richness is found to be higher in urban areas due to the increase of alien species invading in the urban areas, mostly due to home gardening practices (Paker *et al.,* 2014; Wania *et al.,* 2006). Little has been reported on the plant species richness of wetlands, and especially of urban wetlands. Most studies of species richness in wetlands have focussed on amphibian species richness (Shulse *et al.,* 2012; Ishiyama *et al.,* 2014); the impact of soil elements on species richness (Wang *et al.,* 2013; Zhu *et al.,* 2012); or on the species richness of constructed wetlands (Wang *et al.,* 2013; Chang *et al.,* 2014).

Species richness is also influenced by a variety of different factors. Audet *et al.* (2015) investigated plant community characteristics, focussing on species richness, in Danish riparian wetlands to identify factors required to sustain species diversity. Results indicated that environmental variables (groundwater table being closer to the soil surface and low nutrient availability) are important factors for improving species richness in restored wetlands.

2.5.3 Plant functional diversity

Functional diversity is another means that can be used to provide insight into the diversity within a plant community (Li *et al.,* 2015). To determine the overall state of an ecosystem, "the relationship between species diversity and functional diversity is central to identifying the diversity effects on ecosystem functioning" (Flynn *et al.,* 2011). Functional diversity has been used to describe several different aspects of plant community or a certain areas' ecosystem structure, focussing on the variation in the functional characters of plant species (Petchey & Gaston, 2002; Mason *et al*., 2003) and the number of plant functional groups present (Diaz and Cabido, 2001). Functional diversity is also used to understand how species richness or diversity relates to ecosystem function (Petchey *et al.*, 2003; Cadotte et al., 2009; Flynn et al., 2011) or how diveristy responds to environmental stress or disturbance (Norberg *et al.,* 2001; Suding *et al.,* 2008).

Several models have been developed to explain the relationships between species and functional trait diversity (i.e. Díaz and Cabido, 2001; Mayfield *et al.*, 2005; Naeem and Wright, 2003). This set the tone for our current understanding of the diversity of plant species and their functional traits relationships, which stated that "if species diversity is lost due to human land-use change, functional trait diversity will also be lost" (Mayfield *et al.,* 2010), which will most likely influence the diversity of ecosystem services being delivered.

Several examples of functional diversity studies includes the study of Van Der Walt (2014), who used plant functional diversity as a means to investigate the impact of surrounding urbanisation on grassland fragments within the TMA. The study of Franklin (1988) used plant functional diversity to provide evidence of intensive human disturbance on temperate forests, and as motivation for the preservation and enhancement of the biotic diversity. Bouchard *et al.* (2007) used functional diversity measures to determine the effects of macrophyte functional group richness on emergent plant species functions within freshwater, finding that an increase in the richness of wetland plant functional groups enhanced the belowground processes (e.g., root dynamics, decomposition). Zhang *et al.* (2010) used functional diversity indices to ultimately understand the relations of plant diversity with soil microbial community patterns in constructed wetlands.

Overall, there is a lack of studies focussing on plant functional diversity to ultimately determine the state of an ecosystem as a whole. Most research into the relationship between species diversity and their functional traits has been based on the diversity of trait values for individual traits of potential importance to specific ecosystem functions (such as Specific Leaf Area (SLA)) (Diaz & Cabido, 2001; Naeem and Wright, 2003). Concentrating on only one aspect on which functional diversity may have an influence, does not provide insight into the entire ecosystem. Ultimately, the "key to understanding the relationship between species diversity and functional diversity, and how they affect ecosystem function, is to determine how abiotic factors influence these diversity measures" (Cadotte *et al.,* 2011) and further, to understand how the structure and function of the vegetation present in wetland regions varies (Li *et al.,* 2015). That is why the study of Flynn *et al.* (2011), which examined multi-trait diversity indices (that is, indices that incorporate multiple traits and their functional diversity) is more ideal for studying the influence of plant functional traits on ecosystem functioning, rather than just focussing on the diversity of trait values.

2.5.3.1 Plant functional traits

The range, values and distribution of key physico-chemical attributes in a given community or those components of plant diversity that influence ecosystem functioning (Naeem *et al.,* 2009) is measured when examining functional diversity. These morphological and physiological attributes are also known as plant functional traits, which are any measurable features of a plant species that potentially affects its performance or fitness within its local environment (Cadotte *et al.,* 2011).

Plant functional traits give better insight into the constraints and opportunities faced by plants in different habitats than does taxonomic identity alone (Grime, 1979). Functional traits are defined as "the attributes of species which influence ecosystem properties, or the response of a species to its environmental conditions, given the range of the specific trait in question" (Violle *et al.*, 2007). These plant traits define the processes individual plant species may contribute to a certain community levels (Roscher *et al.,* 2012). Ecosystem processes are determined by the product of the distribution in the relevant trait values and also the relative abundance of species exhibiting these traits in the community (Grime, 1998; Diaz *et al.,* 2007). Thus, traits provide an understanding of how functional diversity underpins ecosystem processes (e.g., decomposition/fertility, productivity etc.) and the benefits that people derive from them (Díaz *et al.,* 2007). Depending on the exact nature of the measured traits, traits can influence environmental tolerances and habitat requiremnts (Cadotte et al., 2011). For example, traits determine where a species can live, how a species interact with one another (interspecific interaction), or how it contributes to ecosystem function (for example, through differences in nutrient use and storage efficiency) (Lavorel *et al.,* 2011).

Cornelissen *et al.* (2003) developed the "handbook of protocols for standardised and easy measurement of plant functional traits" worldwide. Since then, studying traits, in a standardised manner, has provided greater understanding of the ecological and evolutionary patterns and processes within different ecosystems. A decade later, Pérez-Harguindeguy *et al.* (2013) developed a new handbook for the standardised measurement of plant functional traits to expand on previous work by using traits to help quantify a wide range of natural and human-driven processes, including changes in biodiversity, the impacts of species invasions, alterations in biogeochemical processes and vegetation–atmosphere interactions.

Most studies of plant functional traits have focused on grasslands (De Bello *et al.,* 2010). The study of Van Der Walt (2014) is an example of using plant functional traits in grassland fragments in the TMA, which differ from wetland habitats in terms of important processes and services. Lavorel *et al.* (2011) used plant functional traits for "the analysis of ecosystem services across landscapes, as an approach to understand the fundamental ecological mechanisms underlying ecosystem service provision, and the trade-offs or synergies among services, specifically in grassland habitats". Plant functional traits are known to be useful for evaluating engineered ecosystems, habitats commonly found in urban areas. Lundholm *et al.* (2015) examined plant traits as predictors of ecosystem properties and services in a planted green roof system. This study demonstrated that easily measureble traits could be used to select species that optimise green roof performance across multiple ecosystem services (Lundholm *et al.,* 2015).

Recently, more studies on wetlands investigated the applications of plant functional groups in wetland studies. Sutton-Grier *et al.* (2014) studied the response of two ecosystem functions effecting riparian nitrogen removal pathways in restored freshwater to the variation in plant trait composition. Results indicated that plant species with different trait values are essential to maintaining multiple ecosystem functions, and provided a trait-based link between the recent findings that higher biodiversity is necessary for multi-functionality within ecosystems (Sutton-Grier *et al.*, 2014). Moor *et al.* (2015) combined species distribution modelling and plant functional traits to estimate the direction of change of ecosystem processes in Swedish wetlands under climate change. The study of Moor *et al.* (2015) suggested that the possible changes to the increase in flood attenuation services and nutrient retention, are all useful information for wetland management and safety for surrounding communities.

2.6 Relation between biodiversity and ecosystem services

The concept of linking biodiversity to the delivery of ecosystem services is a relatively new concept, and the use of vegetation as the proponent of biodiversity, for identifying and evaluating underlying ecosystem services, is an acceptable method (Thackway *et al.,* 2006), applicable in both urban and rural settings. Valuation of the ecosystem services of an area, by means of a monetary valuation (De Groot *et al.,* 2002; Baveye *et al.,* 2013) or vegetation characteristics and condition (Yapp *et al.,* 2010), can be used as a tool for communicating the importance of nature and biodiversity as well as their delivery of ecosystem services to the appropriate management authority.

The valuation of ecosystem services is very complicated, and therefore, alternative means of apraising ecosystem services within urban and rural environments was established (Ring *et al.,* 2010). Ziter (2015) reviewed papers on urban ecosystem services in the ecological literature and found that species diversity was the most common biodiversity type measured within the ecosystem services categories. Functional diversity was the second most common measured type of biodiversity, but was found to be only measured for regulating services (Ziter, 2015).

Biodiversity, and in the case of this current study, vegetation, is crucial in ecosystem structure and processes, with complex contributions to ecosystem functions (Diaz and Cabido, 2001). An ecosystem can deliver many different ecosystem services, but no one-to-one correspondence can be defined between attributes of an ecosystem (i.e. biodiversity) and the ecosystem services they deliver (Aslaksen *et al.*, 2015). A single attribute of biodiversity can contribute to several ecosystem services. For example, beetle larvae in dead wood contributes as food for woodpeckers (provisioning ecosystem service) and also to decomposition and nutrient circulation (regulating ecosystem services) (Aslaksen *et al.*, 2015).

It is assumed that higher-level biodiversity is associated with an increased ecosystem function and the delivery of a wider variety of ecosystem services (Hooper *et al.,* 2005), but not all ecosystem functions require a large diversity of organisms (Aslaksen *et al.,* 2015). Thus, in the process of determining which aspects of biodiversity are delivering ecosystem services, variability based measures of biodiversity (Magurran, 2004) (i.e. species richness) and their state and abundance should be taken into account.

Yapp *et al.* (2010) developed the VAST (Vegetation, Assets, States and Transitions) Framework, which classifies landscape units according to the vegetation condition in a specific area. Assessing the vegetation as representative of the biodiversity of an area, was identified as a practical approach that can be applied to evaluate the identified ecosystem services. The seven classes used, indicate the likely balance of an ecosystem service being delivered, including what services have been benefitted or harmed in relation to changes in vegetation condition (Yapp *et al.,* 2010). The vegetation condition indicates the capacity of a unit of vegetation to produce ecosystem services. Condition changes as the vegetation type responds to pressures (i.e. land-use change in the surrounding environment) applied to it (Yapp *et al.,* 2010).

Biodiversity is important at all levels of ecosystem service delivery, as regulator of ecosystem processes that underpin ecosystem services, as an ecosystem service in and of itself (e.g. species can contribute directly to goods and their values), and as a good which is subject to valuation, economic or otherwise (Mace *et al.,* 2011).

Ecosystems produce multiple services that interact in complex ways, and different services are interlinked, both negatively and positively (i.e., feedback loops) (Elmqvist *et al.*, 2010). If an ecosystem is managed primarily for the delivery of one ecosystem service (only for food or carbon sequestration, for example), biodiversity and other ecosystem services may often be negatively affected (Elmqvist *et al.*, 2010). Thus, it is important to know the types of ecosystem services being delivered, what delivers these ecosystem services and how to preserve them, so as to ensure optimal delivery of ecosystem services to the surrounding environments.

Aslaksen *et al.* (2015) addressed the relationship between ecosystem services and the biodiversity that supports it, in the context of the Nature Index for Norway. The Nature Index for Norway is a species abundance index and a framework for integrated biodiversity measurement. It was designed with the primary focus being on biodiversity measurement, rather than an assessment of ecosystem services, but the value thereof can be interpreted as a measure of ecosystem capacity. In particular, for those ecosystem services where high biodiversity is of great importance, this tool is especially useful (Aslaksen *et al.,* 2015). Harrison *et al.* (2014) suggested that this approach allows for relating the provision of ecosystem services to attributes of biodiversity at different levels, such as species, functional group and community scales. Evaluation of such attributes will aid in creating a basis for an index of ecosystem capacity, which will add value to the index for biodiversity, and give opportunities to select particular attributes and develop indices for assessing individual ecosystem services (Aslaksen *et al*., 2015).

2.7 Management and conservation of South African wetlands

Due to ongoing rapid urbanisation in South Africa and the need to support this growing urban population, agricultural and forestry practices boomed, resulting in the loss of natural habitat (and therefore also wetlands) to provide for the increased dependants. From this, a challenge of linking the protection and conservation of natural environments (including wetlands), and sustainable development has emerged (Lélé. 1991). The DEAT have pledged to “take on the challenge to maintain and restore South Africa’s wetlands in order to ensure that the ecosystem service levels provided by wetlands (or all other natural environments) keep up with the pace of the developing population and its growing demands on the resources and services being delivered by the wetlands” (DEAT, 2012b).

A range of policy and legislative frameworks have therefore been developed by the various governmental entities (i.e. DWAF, DEAT) in response to this challenge, particularly in the water sector, as well as civil society and private sector initiatives.

2.7.1 South African policy and legislative frameworks

The Constitution of South Africa sets the basis for any environmental decision making, specifically through the environmental rights and various other rights relevant to natural resources, and thus provides an adequate platform for a legal regime governing wetlands. This platform is reinforced through the objectives of the National Environmental Management Act (NEMA) (Act 107 of 1998), which addresses several principals relevant to the protection of wetlands and also defines the principles which all organs of state (in the case of this study, the local municipality) are compelled to adhere to and implement.

In 1998, South Africa implemented the National Water Act (NWA), which “recognises aquatic ecosystems (including wetlands) as legitimate users of water, and makes provision for sufficient water to be set aside to ensure the sustainable functioning of these ecosystems” (NWA, 1998). In order for the effective implementation of legislation as advocated by NWA (1998), the strategies should include proactive preventative measures and remedial interventions as a result of past degradation, in order to sustain and improve the wetlands (DEAT, 2012b). A key mechanism to achieve the objectives of the NWA is the establishment of specific management actors within the local municipality to achieve localised and focused water resource management. Their responsibilities include the carrying out of scientific studies and engaging with various stakeholders which support effective decision making within the local environment to provide for the more focused protection of wetlands.

However, the implementation of the legislation regime for the protection of water resources is vulnerable due to the lack of effective implementation of the legal requirements. Therefore, other environmental entities are being established to include local residents and stakeholders in the protection of wetlands. The World Wildlife Fund Mondi Wetlands Programme (WWF MWP), has been at the forefront of pioneering the rehabilitation and restoration, and inadvertently, the conservation, of wetlands across South Africa (WWF, 2016). This initiative started in 1991 as a collaboration between World Wildlife Fund South Africa (WWF-SA) and the Wildlife and Environmental Society of South Africa (WESSA). This collaboration is a privately-funded ecological conservation programme, being the first to focus on the protection of wetlands outside of protected areas (WWF, 2016).

From the WWF MWP, a government sister project, the Working for Wetlands (WfW) initiative, was established in 2001, which received direct guidance and support from the MWP up until 2006. This initiative employs previously disadvantaged members of the surrounding community to rehabilitate degraded wetlands, thereby alleviating poverty whilst also maintaining biodiversity and ensuring long-term water security (WWF, 2016).

Currently, the WfW programme uses its resilient landscape-approach to water stewardship to focus on alleviating impacts on major water-stressed catchments with industries that have traditionally impacted wetlands (such as sugar production, dairy farming and forestry) (WWF, 2016). This is aimed at strengthening natural freshwater infrastructure, conserving the surrounding natural habitat to ensure better quality natural processes, by creating a deeper understanding of shared responsibilities and shared risks with everyone involved in a product's value chain (that is, the end product of farming, forestry, and also the ecosystem services being delivered), not just landowners and farmers.

Within the local setting of the municipality, the Integrated Development Plan (IDP) is a strategic plan which guides all planning, investment, development and implementation decisions across the different sectors, in consultations with communities and all relevant stakeholders. This plan links physical, social, institutional and economic components of planning with the management and development structure of the municipality to implement sustainable community planning and development (Tlokwe City Council, 2013). A Spatial Development Plan (SDP) is a critical and integral component of the IDP, which illustrates the form and extent of development that the municipality wishes to promote. An SDP is developed to address the spatial inequality and inefficiencies in the local municipalities of South Africa (DRDLR, 2011) and is aimed at facilitating integration, access to opportunities for all and the sustainable use of the natural resources (DRDLR, 2011).

Within the SDP of the TMA (Tlokwe Local Municipality, 2008), proposed spatial areas (within the urban area of Potchefstroom and surrounding rural areas) are dedicated to preserve the open spaces still undeveloped by infrastructure (such as urban wetlands 1 and 2), and promote and reinstate their ecological integrity (Tlokwe Local Municipality, 2008). Thus, the framework is available for the protection and promotion of especially wetlands in the TMA, but the implementation of this framework and the information thereof is not always relayed to the surrounding community members.

2.8 Summary

Despite wetlands being severely affected by anthropogenic activities, they are still considered to have conservation importance (Cilliers *et al.,* 2008) in the delivery of urban ecosystem services (Wittmer *et al.,* 2010). The following aspects will be examined in this study and were explored in this literature review chapter:

- Effect of urbanisation on green urban areas
- Urban ecology and the quantification of the urban-rural gradient
- The concept of ecosystem service and its application in South Africa
- Wetland ecosystem services
- The plant diversity and condition of urban wetlands
- Wetland biodiversity (vegetation) and ecosystem services they may deliver

In the current study on wetland diversity and ecosystem services in the Tlokwe Municipal Area (TMA), all the above mentioned aspects will be addressed. The challenge is to determine whether wetland vegetation condition has an effect on the delivery of ecosystem services.

3 MATERIALS AND METHODS

3.1 Study Area

3.1.1 Location

The study focused on the Tlokwe Municipal Area (TMA) in the North West Province of South Africa (Figure 3.1). This municipality covers 2672 km^2 and comprises the city of Potchefstroom and its surrounds. Major driving forces for the current development in and around Potchefstroom include the rapid expansion of previously called township areas (e.g. Ikageng, Mohadin and Promosa) at the city fringe. The TMA is characterized by a transformation of over 13% of the natural grasslands over a 25 year period. Land-uses such as residential, industrial and commercial areas, dominate (Cilliers and Siebert, 2011). The most current, accurate population estimates for Potchefstroom were collected in the South African census of 2011. The population estimate of the TMA (including Potchefstroom, and Ikageng, Mohadin and Primosa informal housing settlements) is approximately 250 000 (www.potch.co.za).

The Mooi River flows through Potchefstroom and includes rural upstream and downstream segments with a city segment shaped by decades of urbanisation followed by significant population growth. There are various dams situated in the Mooi River system and TMA where water for domestic use are withdrawn. The Mooi River is further used for angling and other general recreational purposes (e.g., boating, swimming etc.). Industrial use of water from the Mooi River is concentrated in and around Potchefstroom. Extensive water extraction from the down stream Mooi River for agricultural purposes has been identified by the DWS PES (Present Ecological State) and EIS (Ecological Importance and Sensitivity) Database for Secondary Catchments (DWS, 2014). The Mooi River and its tributaries are however, contaminated by a wide variety of point and diffuse sources, including agricultural and industrial effluents, which changed the size and occurrence of wetlands along the river (Barnard *et al.,* 2013).

3.1.2 Climate

The study area is located on the plateau of South Africa in the Highveld area characterised by the grassland biome (Mucina and Rutherford, 2006). The area experiences a cool dry steppe climate, a summer rainfall regime (October until March) (Van der Walt and Bezuidenhout, 1996), with an average annual rainfall between 570mm and 730mm (Mucina and Rutherford, 2006). Substantial seasonal and daily variation in temperature is common, with hot summers (daily average temperatures exceeding 32°C midsummer) and mild to cold winters and an average minimum monthly temperatures of ~-12°C (ISCW, 2003). There are no extreme weather conditions in this area, but hail occurs sporadically in summer and frost is prevalent in winters (Mucina and Rutherford, 2006). August to November are the windy months, with the wind being in a predominantly northern direction (Mangold *et al.,* 2002).

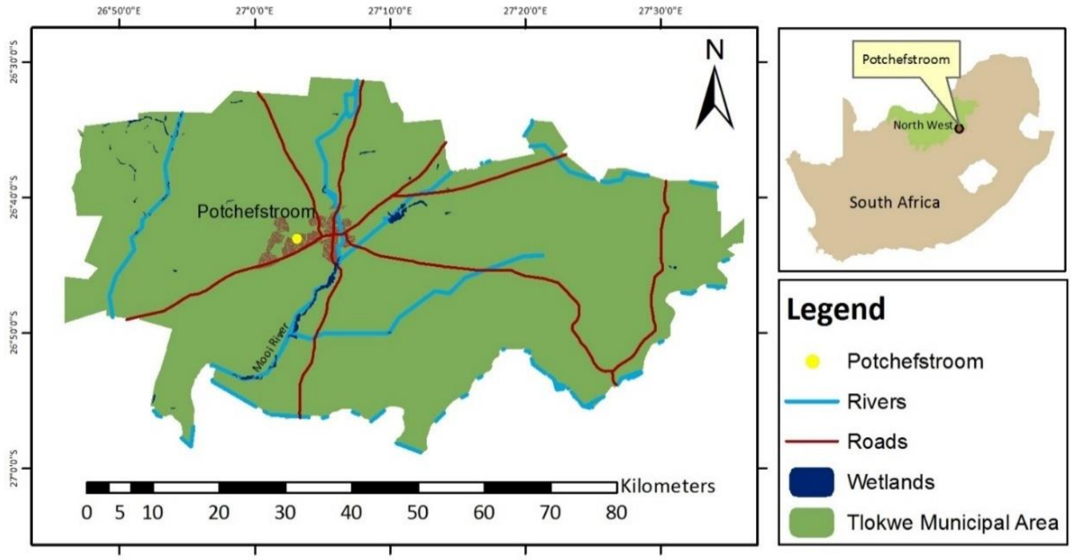


Figure 3.1: Map showing the Tlokwe Municipal Area (TMA) situated in the North West Province of South Africa.

3.1.3 Geology

The geology of the area is diverse however the Transvaal Sequence dominates. Three groups, based on lithological differences have been established under the Transvaal Sequence (Bezuidenhout, 1993), namely the Wolkberg, Chuniespoort and Pretoria Groups (SACS, 1980). The dominant rock type found in the area (Figure 3.1) is from the Pretoria Group of the Transvaal Sequence. This Group is composed predominantly of quartzite, shale and volcanic elements in the Hekpoort Andesite Formation, as well as diabase sills that intrude into the Strubenkop shale (Nel *et al.*, 1939; SACS, 1980).

3.1.4 Land types and Soil

The Potchefstroom area is classified under the Bc and Fb land types (Land Type Survey Staff. 1984). The Bc land type is characterized by flat or slightly undulating plains and widespread eutrophic, red soils, dominated by both Hutton and Mispah soil forms (21 % of the land type) (Land Type Survey Staff, 1984).

The soils of the Fb land type are without lime, but contain accumulations of soluble salts in one or more valley bottom soils (Land Type Survey Staff, 1984). Conversely, the dominant soil forms in the

Fb land type are Glenrosa (25 % of the land type) and Mispah (24 % of the land type) with exposed rocks being least dominant (20 % of the land type) (Bezuidenhout, 1993).

3.1.5 Vegetation

Potchefstroom lies within the Grassland Biome on the high central plateau of South Africa and contains vegetation types of the Dry Highveld and Mesic Highveld Grassland bioregions (Mucina and Rutherford, 2006).

It is mainly characterised by herbaceous vegetation, predominantly from the grass family, Poaceae (Mucina and Rutherford, 2006). The Grassland Biome is considered exceptionally biodiverse. Three vegetation types can be recognized in the Potchefstroom Municipal Area, namely the Carletonville Dolomite Grassland, Rand Highveld Grassland and Andesite Mountain Bushveld (Mucina and Rutherford, 2006). The study area is situated in the Rand Highveld Grassland of the Mesic Highveld Grassland bioregion and is considered endangered as almost 50% has been transformed for the purposes of cultivation, plantations, urbanisation or dam-building. With the conservation target being 24 % as calculated by Mucina and Rutherford (2006), it is still poorly conserved (only 1%), as only small patches are protected in statutory reserves. Van der Walt *et al.* (2014) has, however, shown that urban fragmented natural areas, which are often regarded as being degraded and transformed beyond conservation status, are still conservable given the appropriate action. Conserving these areas can add to the sustainability of the already endangered Rand Highveld Grassland vegetation unit (Van der Walt *et al.,* 2014).

The wetlands situated in this area are described as Inland Azonal vegetation occurring in the Eastern Temperate Freshwater wetland type (Mucina and Rutherford, 2006). Typical of this area is the flat landscape with shallow depressions filled with (ephemeral) water bodies supporting zoned systems of aquatic and hygrophilous vegetation of temporarily flooded grasslands. A conservation target of 24% is suggested for this vegetation type to ensure sustainability, but Mucina and Rutherford (2006) calculated that currently, only 5% thereof is being conserved.

The following section of this chapter will discuss the methodology used to delineate and classify the wetlands of the TMA (3.2), the quantification of the urban-rural gradient (3.3), the vegetation assessment conducted of the wetlands (3.4), assessment of the wetland condition (3.5), and ecosystem services being delivered (3.6).

3.2 Wetland delineation and classification

A field investigation within the aforementioned TMA wetlands was undertaken in 2013, in order to identify and delineate wetland habitats within the study area, to determine the plant species composition and diversity and also to assess the ecosystem services of the chosen wetlands. Prior to the field investigation ortho-photographs, topographical maps, contour data, the provincial wetland inventory, and Google Earth digital satellite imagery were used as reference material to identify the location of potential wetlands.

3.2.1 Wetland Delineation and Mapping

A thorough investigation of the Mooi River system throughout the Potchefstroom area as well as the periphery of Potchefstroom was conducted to find wetland study sites that were appropriate for this study. Wetlands considered for the purposes of this study were those ecosystems defined by the National Water Act (No 36 of 1998) (RSA, 1998), as "land that is transitional between terrestrial and aquatic systems where the water table is usually at or near the surface, or land which is periodically covered with shallow water, and land which, under normal circumstances, supports or would support vegetation typically adapted to live in saturated soils". Wetlands had to meet the following attributes as set out by the DWAF (2005):

- Wetland soils showing signs of prolonged saturation
- The presence, at least occasionally, of hydrophytes
- A high water table that results in saturation at or near the surface, leading to anaerobic conditions developing in the top 50cm of the soil.

'A Practical Field Procedure for Identification and Delineation of Wetland and Riparian Areas' (DWAF, 2005) was used to identify wetland boundaries located at the study sites. This requires the investigator to assess on-site wetland indicators, with a primary focus on wetland vegetation. Sampling sites were recorded with a Global Positioning System (GPS) and captured using ArcMap 10 (ESRI, 2015) for further processing.

For the purposes of delineating physical wetland boundaries of this study, wetland vegetation was used as indicators. Vegetation is a useful guide in determining the boundary of a wetland as plant communities generally undergo distinct changes in species composition along a moisture gradient from the centre of a wetland, outwards towards adjacent terrestrial areas. The soil within the chosen study site displayed signs of wetness (mottling and gleying) within 50cm of the soil surface for an area, as to be classified as a wetland according to requirements of the DWAF (2005). This method of delineation is considered the best estimate of the extent of a wetland at a specific site given the conditions at the time of the survey. Given this, it is unlikely that any significant wetland systems were missed during the course of the study.

Fourteen wetland study sites were subsequently selected from the city and downstream segments of the Mooi River and named according to the farm, region or owner of origin, of the wetland (Figure 3.2, Table 3.1). ArcGIS 10 was used to map the study sites (the coordinates of sampling sites were recorded as geographic projections, and projected using WGS 84 Datum (Smith, 1984). Mapping was based on the field data collected, and the interpretation of aerial photography and Google imagery of the area. No wetlands upstream of the Mooi River was identified which fulfilled the criteria for selecting wetlands. Also, no 'experimental control' wetland site could be identified, since all the wetlands identified to be sampled were all disturbed.

Vegetation surveys were done for each of the identified wetland sites in the TMA area. Table 3.1 shows the selected wetland sites with their respective sizes and the number of transects surveyed per wetland. The surveyed method is discussed in Section 3.4.

Table 3.1: The 14 wetlands in the greater TMA area, indicating the wetland size and the number of sample transects per wetland.

Nr.	Wetland Name	Location	Size (ha)	Number transects	Management Type
1	Van Der Hoff Park	26°41'51.2"S; 27°06'34.9"E	31.58	38	Recreational activities
2	Prozesky Bird Sanctuary	26°44'03.1" S; 27°06'07.7" E	17.01	30	Bird Reserve
3	Wilgeboom Farm	26°45'32.6" S; 27°05'56.8" E	2.85	7	Grazing
4	Castello Farm	26°45'25.4" S; 27°05'59.8" E	4.24	4	Grazing
5	Noric Farm	26°48'07.9" S; 27°04'04.0" E	1.76	4	Grazing
6	Van Zyl Farm	26°46'41.8" S; 27°04'57.5" E	2.95	6	Grazing
7	Swart Farm	26°50'57.4" S; 27°02'25.5" E	3.21	4	Grazing
8	Makouspan Farm	26°51'34.6" S; 27°02'02.5" E	3.41	5	Grazing
9	Cillier Farm	26°41'19.6" S; 27°05'23.7" E	1.14	3	Grazing
10	Kleyn Farm	26°53'12.1" S; 26°58'11.4" E	3.79	9	Grazing
11	Delport Farm	26°53'08.3" S; 26°59'40.6" E	2.58	5	Grazing
12	Transvalia Farm	26°53'12.2" S; 26°59'37.3" E	4.79	10	Grazing
13	Trompie Farm	26°45'44.6" S; 26°05'30.2" E	1.47	3	Grazing
14	Stam Farm	26°45'43.5" S; 27°05'33.8" E	1.44	3	Grazing

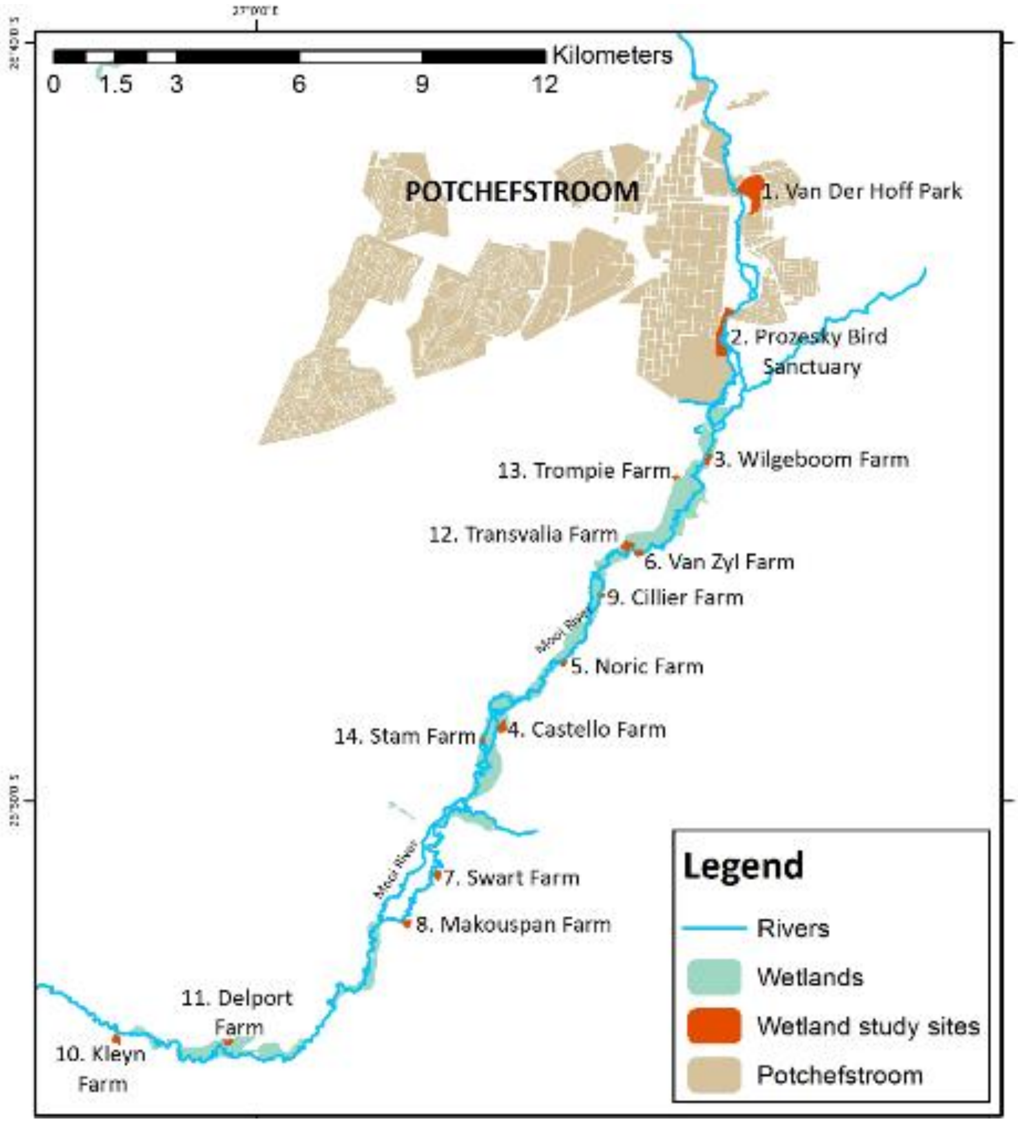


Figure 3.2: Map of identified wetland study sites along the Mooi River in the Tlokwe Municipal Area (TMA)

3.2.2 Wetlands Classification in South Africa

The need for a wetland classification system prompted SANBI to initiate a collaborative process with the WRC to develop a classification system whereby "wetland habitat types with shared natural attributes could be grouped together" (Ollis *et al.,* 2009). Ollis *et al.* (2015) reviewed this classification system to "reach a common language and terminology that can be used on inland aquatic ecosystems throughout South Africa, and which focus specifically on the different types of wetlands that occur in South Africa". This classification system provides wetland specialists, academics, the government and other role players with a common language to distinguish between the different types of wetlands for management and conservation purposes (DEAT, 2012a).

The classification system is six-tiered in its structure (Figure 3.3). The first four levels distinguish between different types of aquatic ecosystems on the basis of 'primary discriminators', which are "criteria that consistently differentiate between the specified categories at a particular level" (Ollis *et al.,* 2015). The tiered structure progresses from the broadest scale at Level 1 (The type of system the wetland is located in – marine, estuarine or inland), through to hydrogeomorphic (HGM) units (Level 4) as the core units of classification. At Level 5, 'Secondary discriminators' are applied to "classify the tidal/hydrological regime of an HGM unit", which is not applicable to inland wetlands, and 'Descriptors' at Level 6 to categorise a range of biophysical attributes. In this study, the wetlands will be classified up to level 4 (hydrogeomorphic units).

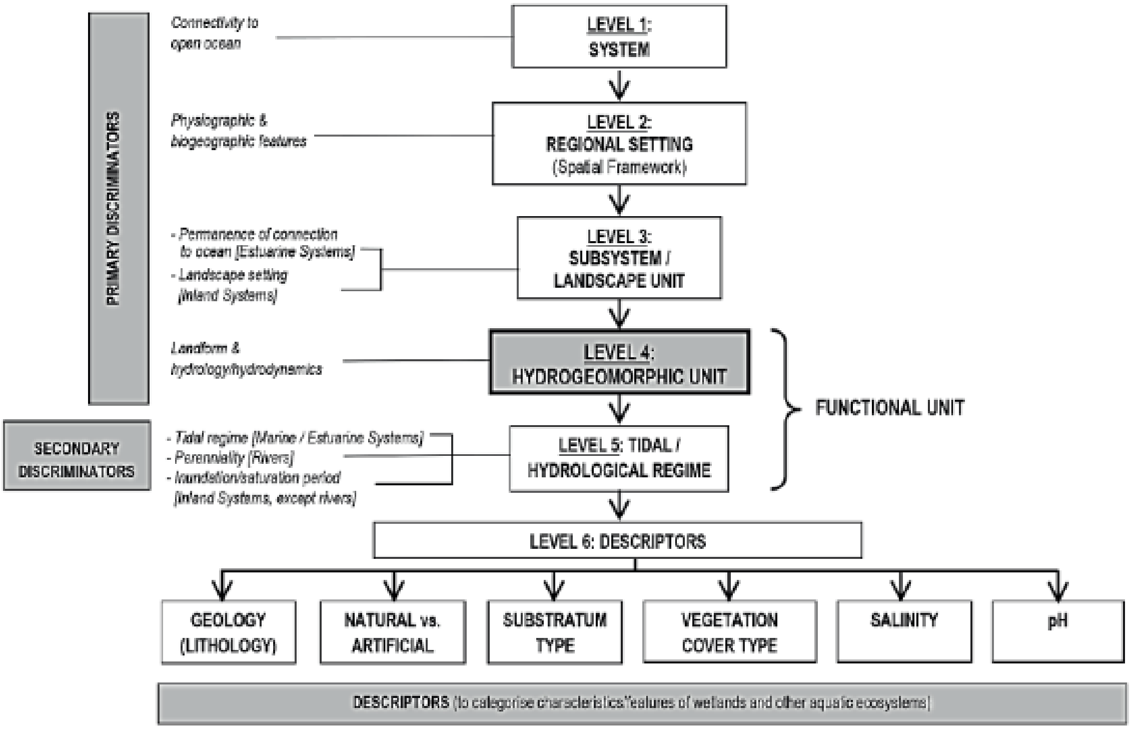


Figure 3.3: Conceptual overview of the classification system for wetlands and other aquatic ecosystems, indicating the six levels upon which a wetland can be classified, as in Ollis *et al.* (2015).

3.3 Quantification of the urbanisation gradient

The identification of possible conservation areas within urban and rural areas and to enable more sensible comparative studies, necessitates the quantification of the urbanisation gradient (Du Toit, 2009). Certain patterns and processes drive areas to become more urbanised. Assessing these patterns and processes will allow for comparisons to be made between urban and rural areas, in terms of their vegetation and ecosystem service delivery.

Physical measures were used to quantify the urbanisation gradient by identifying the percentage coverage of certain land-use types within a 500m buffer area around the wetland study sites. These measures are considered the most fundamental components of the urban ecosystem in terms of the contrast to the surrounding environment as well as within the city. The popular Vegetation-Impervious-surface-Soil (V-I-S) model proposed by Ridd (1995) is another example where physical measures can be used to determine an urban-rural gradient. Ridd (1995) explained that the objective of such measurements is to identify and characterize variable land cover patterns and not to identify land-use.

Following the success of the method used by Ridd (1995), for the method of quantification in the current study, classifying the different land-use types within a 500m buffer surrounding the different wetland sites was used to determine the degree of 'urbanness' of each wetland setting. Buffers of 500m were also used in the study of Van der Walt *et al.* (2014) on grasslands in the city of Potchefstroom in the North West province. Using the aerial photographs of the entire study area and ArcView 10, a polygon of each identified wetland site was digitized and a 500m surrounding buffer area created, in which different landscape metrics were calculated (Figure 3.4).

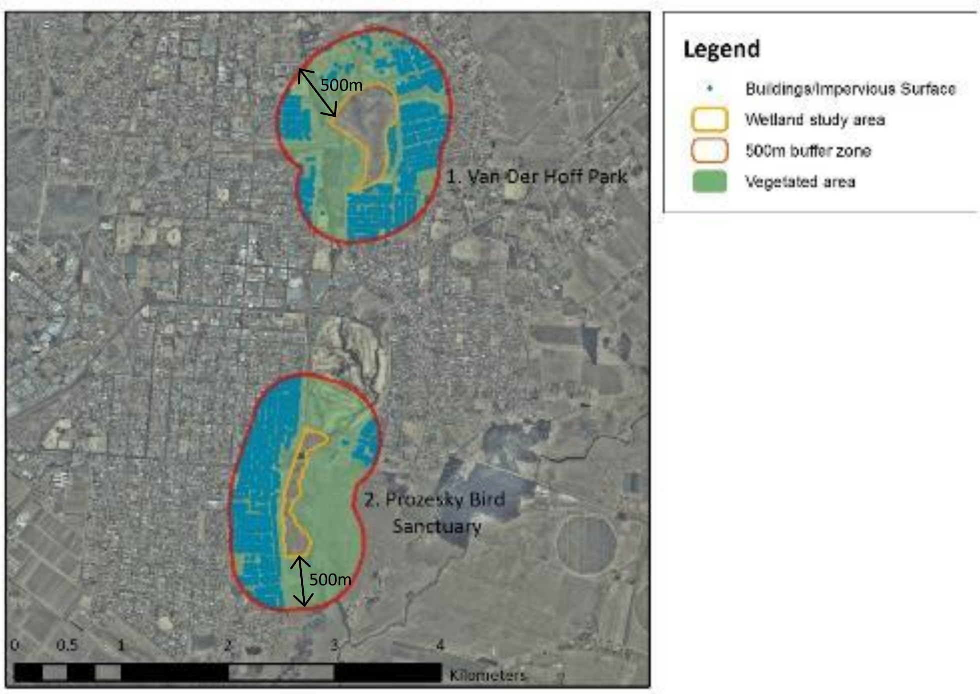


Figure 3.4: Two of the selected wetland sites within the study, indicating their variability in size and shape and the 500m buffer surrounding the wetlands, in which different metrics were used to calculate the urbanisation gradient.

3.3.1 Urbanisation measures

The selected wetland sites in the study area vary in shape and size resulting in different buffer area sizes and shapes thus different extents. This implies that area- and shape-sensitive landscape metrics could not be used for this study. The aim of the chosen metrics was then to select statistically sound measures of urbanisation that are of relevance to the entire study area that were rapidly measurable and available for use. Four urbanisation measures (edge density, percentage vegetation cover, percentage urban land cover, and density of dwellings) (Van Der Walt *et al.,* 2014), were used as indicators for patterns and processes associated with urban areas. The urbanisation measures was calculated for matrix areas with a 500m buffer surrounding each selected wetland (within the buffer areas indicated in Figure 3.5) in order to quantify the position of each wetland site along an urban-to-rural gradient.

- Edge Density (ED)

Edge Density was calculated to indicate landscape fragmentation. ED is the sum of the lengths (m) of all edge segments within the 500m buffer zones (E) surrounding each wetland, per hectare (A) (McGarigal and Marks, 1995) and was calculated as:

ED = E/A(10 000)

- Vegetation cover (PVEG)

The landscape metric of percentage vegetation cover was used to determine the extent of grassland and wetland habitat loss in the study area. This was determined by digitizing the current vegetation within the 500m buffer area surrounding the wetland sites and determining their size in hectares (ha). It is assumed that before urbanisation occurred, the buffer contained 'natural' vegetation. Once the total cover of the remaining vegetation is measured and a large portion thereof still contains vegetation, then the assumption is made then those areas have not undergone a significant degree of habitat loss. McGarigal and Marks (1995) calculated vegetation coverage as:

$$PVEG = \frac{\sum_{j=1}^{n} a_{ij}}{A}(100)$$

Therefore, PVEG equals the total vegetation patch area (a_{ij})(ha) within the 500m buffer zone, divided by total buffer area (A) (ha), multiplied by 100 (10 convert to a percentage); in other words, PVEG equals the percentage of the buffer area comprised of the corresponding vegetation patches.

- Percentage urban (PURB)

PURB indicates the percentage of each 500m buffer surrounding the wetlands that consists of impervious surfaces of urban built-up areas (e.g. roads, pavements, buildings, farmsteads). Percentage impervious surfaces is an often used measure as it is usually associated with urbanised areas (e.g. buildings, roads, pavements) (Hahs & McDonnell, 2006; Alberti, 2010), or alternatively, with the presence of humans in more rural areas (e.g. farmsteads). PURB was calculated the same as PVEG:

$$PURB = \frac{\sum_{j=1}^{n} a_{ij}}{A}(100)$$

- Dwellings (DWEL)

The count of all buildings per buffer area (per ha) (A) indicates the number of residences surrounding the wetland. The buildings were identified using SPOT 5 HRV satellite imagery, and was calculated as:

DWEL = Number of Buildings/A

3.3.2 Data Analysis

In order to determine if varying degrees of anthropogenic disturbances have a statistically significant impact on the different wetland sites along an urbanisation gradient, the urbanisation measures selected for this study (ED, PVEG, PURB and DWEL) were used as input for a hierarchical, agglomerative cluster analysis in PRIMER software (PRIMER-E, 2012). This method of analysis is used to objectively classify the selected wetland sites according to an increasing degree of urbanisation based on the different identified percentage cover classes.

3.4 Vegetation assessment

3.4.1 Vegetation sampling and design

Vegetation surveys were conducted from January to March 2014, during the flowering season of most plants, to aid in the identification of species. Plant species composition and abundance within the wetlands were determined by laying 100m line transects across each wetland (Ruto *et al.,* 2012; Flinn *et al.,* 2008). Transects were aligned along the longest axis of each wetland. Where sites were wide enough, adjacent transects were sampled parallel to one another, with 20 m between each (Flinn *et al.,* 2008). The number of transects per selected wetland was determined by the size of the wetland under observation. A minimum of 3 transects were undertaken per wetland (since the smallest size wetlands sampled (wetlands 9, 13 and 14, all less than 2 ha) could only fit 3 transects with 20m between each), the number of additional transects increased with increasing wetland size. The largest wetland (wetland 1) had 38 transects, since this wetland (Table 3.1).

The vegetation survey consisted of recording the presence, and estimating the percentage cover of plant species within a 1 x 1 m quadrant (thus, 1 m^2 in size) placed at 10 m intervals along the 100m transects (Ruto *et al.,* 2012; Flinn *et al.,* 2008) situated in different homogenous areas of each wetland sites (Figure 3.5). A homogenous area was identified by selecting assemblages of species which represent different plant communities in the wetlands over an extensive area within the wetland boundaries (Kent, 2011). A total of 131 transects (1310 quadrants) was sampled in the 14 identified wetland sites.

In each transect, the cover abundance of each species was subjectively determined. Although the visual estimation of plant cover within quadrants is subjective due to the chance of observer bias (Dethier *et al.,* 1994), it is far quicker and hence a more effective method, which allows for rare or low covering species to also be recorded (Hanley, 1978). A species which covered less than 0.1m² (1%) of the sample plots were given a percentage cover value of 0.5%. Unknown plant species were collected and identified with the use of herbarium specimens and various field guides of the regional vegetation (Van Oudtshoorn, 2006; Van Ginkel *et al.,* 2011). Plant species names are according to Germishuizen *et al.* (2006) and are all listed in Appendix A.

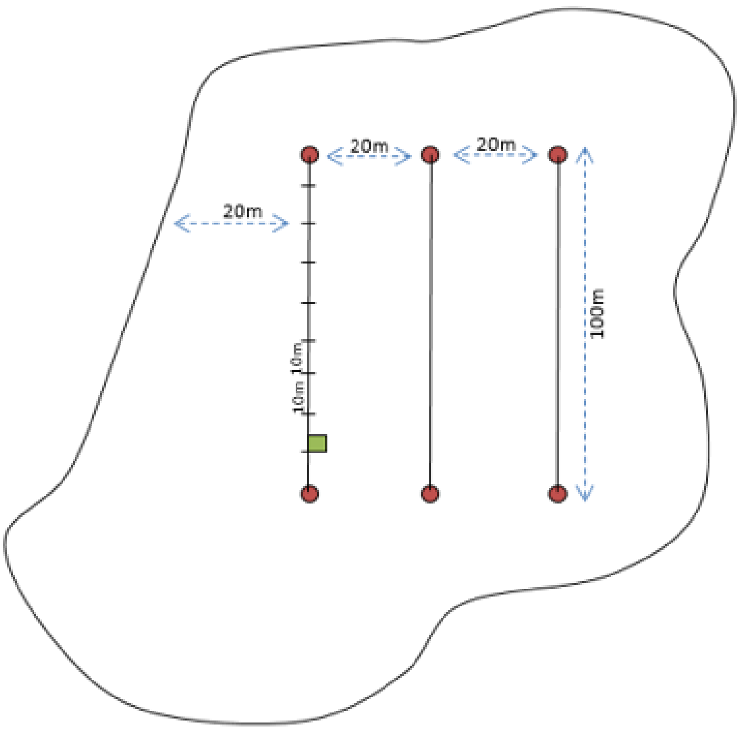


Figure 3.5: The design of the transect placement used to survey wetland vegetation. The 100m transects were placed 20m from the edge of the wetland border across the width of the wetland. Transects were placed 20m apart, parallel to each other. At every 10m interval on the transect, a 1x1m quadrant (thus, 1m^2 in size – indicated by the green block in the figure) was used to sample the vegetation

3.4.2 Plant species diversity and functional traits

Better understanding of the working, development and configurational changes of a plant community can be improved through the research of the diversity of these communities (Ganlin *et al.,* 2006). Wetlands can contribute disproportionately to landscape-level diversity because they often have high levels of both alpha diversity, also known as local species richness, and spatial variation in community composition (beta diversity) (Tiner, 2003).

The diversity of these wetland communities may be determined by local factors (e.g. surrounding land-uses, grazing) which interact with the landscape structure (Niemelä, 1999a). The most important and unambiguous measure of plant species diversity is species richness. Species richness may be used in association with other indices such as evenness, distribution, variation and abundance (Tiner, 2003). Whittaker (1972) described three terms for measuring biodiversity over spatial scales: alpha, beta, and gamma diversity. Gamma diversity is a measure of the overall diversity of the different ecosystems within a region (Whittaker, 1972). Alpha diversity refers to the "diversity within a particular area or ecosystem, and is usually expressed as the number of species (i.e., species richness) in that ecosystem" (Primack, 2002). Alpha diversity takes into account the number of species and the proportion at which each species is represented in the community. In short, a community will have a high alpha diversity, when there is a high number of species and their abundances are very similar.

Beta diversity was defined by Whittaker (1972) as "the extent of species replacement or biotic change along environmental gradients". If there is no change in the species composition along the gradient, then the beta diversity will be low. Measuring the change in species diversity between ecosystems, allows for the comparison of diversity between ecosystems, or in this case, among wetland sites. Beta diversity allows one to identify both shared species amongst compared sites, as well as also unique species.

3.4.2.1 Plant functional traits

"Functional diversity reflects the variety of functions performed by species within a community" (Diaz and Cabido, 2001; Petchey and Gaston, 2006). To understand how biodiversity affects ecosystem function, we must examine how the functional characteristics (traits) of organisms and functional diversity within the wetlands relate to ecosystem function (Hooper *et al.,* 2002; Petchey and Gaston, 2006).

A group of functionally similar species that occur in/utilise the same resource pool would not be much impacted by the loss of a single species, as this loss would not have a significant impact on the functional diversity of the group, whereby another similar species would compensate for its role in the resource reserve utilisation.

However, "if a species that is functionally unique (i.e. functionally dissimilar from other species) is lost, both resource pool utilisation and species functional diversity will decrease" (Petchey *et al.,* 2009). Invasion of alien plant species is a key concern for biodiversity conservation. These types of invasions (biotic invasions) occurs in susceptible ecosystems with lowered functional diversity. This susceptibility is due in part to the underutilisation, or not utilised at all, trait niche, which provides an "entity that can be exploited by an alien species possessing the functional characteristics to do so" (Diaz and Cabido, 2001). It is thus also possible for alien species to fulfil the functional roles of indigenous species in an ecosystem. Schwartz *et al*. (2000) stated, however, that ecosystem function should depend on indigenous species, because if ecosystem processes are maintained by conserving alien species, indigenous species would not hold conservation priority over alien species, leading to uncontrolled alien species invasions.

Functional traits are defined as "the attributes of species which have an effect on ecosystem properties, or the species 'response to their environmental conditions" (Violle *et al.*, 2007). "Soft" traits were used to determine how certain species contribute to the wetland community. "Soft" traits are those that are easily and effectively determined for a great number of sites and species. "Soft" traits are said to "be effective correlates of harder traits, and are therefore usually recommended, due to quicker uncomplicated measurement" (Cornelissen *et al.*, 2003), and are thus suitable for use in studies on floristic change and ecological processes. The traits used in this study included identifying the origin of each species (indigenous/alien), its life span (annual/perennial), growth form (tree, shrub, herb, graminoid and geophyte) and wetland indicator status. Wetland indicator status divides plants into categories based on their expected frequency of occurrence in wetlands (Tiner, 1999). The naming of these categories may differ from that proposed by Tiner (1999) (i.e. those described in Van Ginkel *et al.* (2011) differs from Tiner (1999)) but the category descriptions remain the same. The following wetland indicator status descriptions are based on those originally described by Tiner (1999):

- Obligate Wetland Plants - occur almost exclusively in wetlands under natural conditions (>99% of occurrences);
- Facultative Wetland Plants - usually occur in wetlands, but can occasionally be found on dry land (67-99% of occurrences);
- Facultative Plants - equally likely to grow in wetlands and non-wetlands (34-66% of occurrences);
- Facultative Dryland Plants - usually occur outside of wetlands, but occasionally found in wetlands (1-34% of occurrences); and
- Obligate Dryland Plants - occur almost exclusively outside of wetlands under natural conditions (<1% of occurrences).

Plant functional traits were identified at the species level (taxon-explicit (Lavorel *et al.*, 2008)) to group species together based on shared functional attributes. Functional properties to be considered are listed in Table 3.2. The list has been compiled from a combination of soft traits listed in Cornelissen *et al*. (2003) that are easy to determine, and the wetland indicator status was determined from Van Ginkel *et al.* (2011) and Cowden *et al*. (2014). These plant traits were chosen as it is best to reflect the status of vegetation to the environmental conditions in wetland study sites, situated in areas of differing matrix quality (e.g. from highly urbanised to no urbanisation).

Species trait information was obtained from a variety of sources including SANBI (2012), Van Oudtshoorn (2006) and Van Ginkel *et al.* (2011), as well as via *in-situ* observations in the field and via herbarium specimens.

Table 3.2: List of traits and their categorical units determined for the species encountered in selected wetland study sites (adapted from Cornelissen *et al.*, 2003, Van Ginkel *et al.* (2011) and Cowden *et al.* (2014).

Plant trait		Categorical Unit
1. Status	1	Indigenous
	2	Alien
2. Growth Form	1	Geophyte
	2	Graminoid
	3	Herb
	4	Shrub
	5	Tree
3. Life Span	1	Annual
	2	Perennial
4. Wetland Indicator Status	1	Facultative
	2	Facultative Dryland
	3	Facultative wetland
	4	Obligate wetland
	5	Obligate Dryland

3.4.3 Data analysis

3.4.3.1 Diversity indices

To determine how equally abundant the species are distributed throughout all the wetland sites, species evenness was determined using the Pielou's evenness index (J'). This technique uses frequency data with Primer 5 software (Clarke and Gorley, 2001) and was calculated as:

$$\mathbf{J' = H' / H'max\ ,\ H'max = H' / \ln S}$$

The maximum diversity (H'max) that could possibly occur would be found in a situation where all species exhibit equal abundances (Magurran, 2004).

3.4.3.2 Non-metric multidimensional scaling (NMDS) ordinations for wetland plant species composition and functional diversities

The floristic composition of each of the wetlands was compared using an ordination method known as non-metric multidimensional scaling (NMDS). The ordinations were performed using the Primer 5 software (Clarke and Gorley, 2001). Data collected during the vegetation surveys was used to compile ordinations for percentage vegetation cover and presence/abundance data. The species abundance data for each of the transects were converted to pair-wise distance matrices per wetland (Clarke & Ainsworth, 1993). The ordinations derived illustrate that the distance between each transect is in rank order with their difference in species composition as determined by the Bray-Curtis dissimilarity coefficient (Williams, 2005; Kent, 2011).

To address the stress related to ordinations, NMDS "minimises stress by adjusting transect positions so that the fit between ordination distances dissimilarities is enhanced" (Williams, 2005). This stress function of an NMDS ordination was defined by Kent (2011) as a measure of how well two points fit or match. According to Clarke (1993), "the simplest indicator of the accuracy with which an NMDS ordination perceives the relationship among the transects, is this stress value".

Clarke (1993) also suggested the following rule of thumb for interpreting stress values:

- Stress value smaller than 0.05 (< 0.05) provides a good representation with no opportunity of misinterpretation;
- Stress value smaller than 0.1 (< 0.1) corresponds to a good ordination;
- Stress value smaller than 0.2 (< 0.2) can still provide a good ordination;
- Stress values larger than 0.2 (> 0.2) have the potential to be misleading; and
- Values that reach 0.35-0.4 are randomly placed and tolerate very little relation to the original similarity ranks.

However, according to Clarke (1993), these guidelines "are simplistic and stress tends to increase with increasing numbers of samples". The NMDS ordinations generated were used to analyse the coverage of the wetland plant species, their abundances, and to indicate functional diversity within the wetland sites.

3.4.3.3 Spatial representation of plant species diversity and functional diversity

The plant species attributes and functional diversity occurring in the wetlands of the TMA were also spatially represented. For this purpose, surface interpolation rasters were created in ArcMap of ArcView 10.3.1 (ESRI, 2015). The use of interpolation rasters allows spatially accurate comparisons of the measured indices (i.e. plant species richness or plant functional traits) distributions in the wetlands. The visualization of the specific spatial locations in accordance with the measured indices of each wetland could allow quick identification of possible influences affecting the diversity/ecosystem services delivered by the wetlands in the TMA, allowing identification of anomalies or influences of possible disturbances to be identified.

The IDW (Inverse Distance Weighted) interpolate tool was used to create rasters of the different measured indices of the 14 wetland sites in the TMA in a 2km buffer area surrounding each wetland site. IDW is a method of interpolation that "estimates cell values by averaging the values of sample data points in the neighbourhood of each processing cell" (ESRI, 2015). The nearer a point (input data) is to the centre of the cell being estimated, the higher the influence or weight the point has in the averaging process (ESRI, 2015). An image is created using gradients of the input data and therefore, only the values immediately surrounding the wetland are the corresponding measured indices (i.e. growth form) of that wetland. However, the surfaces between the patches should not be taken as an accurate representation of the real situation since the values are interpolated from the wetlands.

3.5 Wetland condition determination based on plant species composition and diversity

Further analysis of the wetland vegetation sampling data was based on methods used by Carter *et al.* (1988) and Miller and Wardrop (2006), which are detailed below. The analysis made use of indices to gauge the extent of change in the proportion of wetland plant species and percentage of native species occurring within the 14 wetland sites in the TMA. This was used in order to determine the degree of "wetland-ness" of the sampled wetland sites, in other words, which sites are considered to be more of a wetland type site versus a more terrestrial site, based upon their vegetation composition. The two specific indices that were used for this purpose, includes:

3.5.1 Wetland Index Value (WIV)

Data on the abundance of plant species and their wetland indicator status were used to determine the Wetland Index Value or WIV for each wetland site surveyed (i.e., community weighted mean). The WIV provides a useful way of interpreting the status of wetlands based on their vegetation status and is based primarily on the relevant species wetland indicator status, which provides a useful means of addressing the question relating to whether the surveyed site could be considered a true wetland, a transitional wetland, or a non-wetland status (Cowden *et al*., 2014). An ecological index value was awarded to each wetland indicator category (Table 3.3).

Table 3.3: Ecological index values assigned to each wetland indicator status category for the determination of the WIV for the wetland in the TMA (Carter *et al.,* 1988).

Wetland Indicator Status Category	Wetland frequency	Ecological Index
Obligate wetland	> 99%	1
Facultative wetland	67 - 98%	2
Facultative	34 - 66%	3
Facultative dryland	1 - 33%	4
Obligate dryland	< 1%	5

Using the wetland indicator status categories and the associated ecological index values assigned to each species, the WIV was calculated for each of the 14 surveyed wetland site as follows (Atkinson *et al.,* 1993):

$$WIV = (Y_1U_1 + Y_2U_2 + Y_3U_3 \ldots\ldots + Y_MU_M)/100$$

Where:

$Y_1, Y_2, Y_3...Y_m$: Represents the relative cover estimated for each species in the wetland sites surveyed
$U_1, U_2, U_3...U_m$: Represents the ecological index value allocated to the species wetland indicator status

Specific thresholds recommended by Wentworth and Johnson (1986) were used in defining the level of wetness of the surveyed sites (Table 3.4):

Table 3.4: Thresholds used to interpret the WIV of each of the 14 wetland sites surveyed in the TMA (Carter *et al.,* 1988)

WIV	Status
<2.5	True wetland
2.6-3.5	Transitional type
>3.6	Non- wetland

3.5.2 Floristic Quality Assessment Index (FQAI)

Data on the abundance of plant species and their classification status was used to determine the Floristic Quality Assessment Index (FQAI), as defined by Miller and Wardrop (2006). The FQAI is an evaluation procedure that provides an indication of the quality of wetland habitat based on the relative abundance of indigenous, weedy, pioneer or alien invasive species within each surveyed site. Species were assigned a "coefficient of conservatism" which is "a subjective rating indicating a species' preference for non-degraded natural communities" (Tiner, 1999). This was based on available literature (Tiner, 1999; Van Ginkel *et al.,* 2010) indicating the status for each particular species and specialist opinion where such data was not available. Specialist contacted in this regard was Donovan Kotze and Craig Cowden, wetland ecology researchers who have conducted several wetland studies in KwaZulu Natal and have expert knowledge of wetland vegetation. Within the selected wetland sampling sites, each native plant species was allocated a specific coefficient of 0 to 10 based on its conservatism relative to other native species in the surrounding area. Species with very low tolerances to disturbance and high fidelity to habitat integrity (also known as very conservative species) were assigned a coefficient of 10, "while a species that tolerates almost any disturbance and can be found in almost any habitat type", were assigned a coefficient of 0 (Mushet *et al.,* 2002). "Species with conservatisms falling between the two extremes are assigned appropriate coefficients ranging from 0 and 10 based on the professional judgement of the ecologists familiar with the regions flora" (Mushet *et al.,* 2002). Ultimately the FQAI of a sampled site is calculated by weighing the species richness to the mean conservatism value of that specific site, with the use of a formula (Chamberlain *et al.,* 2016).

Thus, the coefficient of conservatism scores is assigned a priori, based on an individual plant species' fidelity to specific habitat types and its tolerance to both natural and anthropogenic disturbance (Taft *et al.,* 1997). Miller and Wardrop (2006) recommended specific thresholds based on the species classification status (Table 3.5).

Table 3.5: The coefficient of conservatism awarded for the different plant species based on their specific classification status (Miller and Wardrop*,* 2006).

Coefficient of conservatism	Species status
0	Alien invasive species
1	Ruderal plants or weeds
4	Occasionally ruderal or weedy species
6	Non-ruderal but pioneer species
10	Sensitive native plants intolerant of disturbance

The FQAI for each of the 14 wetland sites were then calculated using the following equation (Miller and Wardrop, 2006):

$$\text{FQAI} = \bar{C} \times \sqrt{N}$$

Where:

$\bar{C}$ = Mean coefficient of conservatism (determined by dividing the sum of coefficient of conservatism values for each species)

N = Indigenous species richness (count of native/indigenous species recorded within the site)

3.6 Ecosystem services of wetlands

Based on the HGM approach, WET-EcoServices was developed for South African conditions by Kotze *et al.* (2008). The overall goal of WET-EcoServices is to “assist decision makers, government officials, planners, consultants, and educators in undertaking quick assessments of wetlands, specifically in order to reveal the ecosystem services that they supply” (Kotze *et al.,* 2008). This allows for more informed planning and decision making. This assessment could be used for:

- Education and raising of awareness (influence perceptions about the values of wetlands).
- To flag important wetland benefits that need to be considered when managing an individual wetland.
- Prioritization for the allocation of management and rehabilitation resources
- During catchment planning, to determine the importance of the wetland in a catchment context (Kotze *et al.,* 2008).

This assessment has some limitations as indicated by Kotze *et al.* (2008). This is a subjective assessment and “not designed to provide a single overall measure of value or importance of a wetland, or to quantify the benefits supplied by a wetland” (Kotze *et al.,* 2008). WET-EcoServices only assists in assigning indices to ecosystem services for comparative purposes. Rountree *et al.* (2009) had some “frustration” with the use of this assessment, as the descriptors of a number of categories in WET-EcoServices are broad and are accompanied by a high level of uncertainty which not all evaluators are familiar with. WET-EcoServices is, however, a useful tool to categorise and broadly describe the value of some of the more important services provided by the wetlands, and is regarded as sufficient for this study (see discussion in 2.4.1.1).

3.6.1 Levels of assessment of the WET-EcoServices manual

After the individual HGM units are distinguished based on their HGM type, they are then assessed based on the two levels of subjective assessment that the WET-EcoServices provides.

3.6.1.1 Level 1 Assessment

The Level 1 assessment is conducted at desktop level, and is based on existing knowledge of the chosen wetlands. Level 1 is used to establish whether these wetlands are likely to be providing any hydrological benefits and is thus mostly used on a broad scale for strategic assessment purposes (Kotze *et al.,* 2008). The different ecosystem services being assessed (Table 3.6) includes indirect benefits such as: flood attenuation, streamflow regulation, erosion control, sediment trapping, phosphate, nitrate and toxicant assimilation and carbon storage; and direct benefits such as: biodiversity maintenance, provision of water for human use, provision of harvestable resources, provision of cultivated foods, cultural heritage, tourism and recreation and education and research are verified by limited fieldwork.

3.6.1.2 Level 2 Assessment

Level 2 is a field-based assessment that is reliable in such a way that a high level of confidence can be attached to the results and ensures that both direct and indirect benefits (Table 3.6) are determined within the wetland. These aspects are scored based on the existence and extent to which the wetland in question provides specific goods and services. The assessment is undertaken by “determining the likely ability of a wetland to deliver an ecosystem service, and the extent to which the wetland is delivering said ecosystem service” (Kotze *et al*., 2008). There are 5 classes of scores which can be allocated to the ecosystem service being delivered. Score classes vary between 0 (absence of ecosystem service provision) to 4 (maximum possible level of ecosystem service delivery) (Table 3.5) (See Appendix B for WET-EcoServices score sheet of the current study).

Specific information required to be entered into the predesigned WET-EcoServices spreadsheet is gathered during the field visit and during a desktop analysis using ArcView GIS 10 (ESRI, 2015). Once all the required information is entered into the spreadsheet, the effectiveness, opportunity and overall functional scores for each the ecosystem services provided by the wetland units is generated.

Table 3.6: Ecosystems services included and assessed using WET-EcoServices (Table adapted from Kotze *et al.*, 2008).

Ecosystem services supplied by wetlands	Indirect benefits	Regulating and supporting benefits	Flood attenuation		The spreading out and slowing down of floodwaters in the wetland, thereby reducing the severity of floods downstream
			Streamflow regulation		Sustaining streamflow during low flow periods
			Water quality enhancement benefits	Sediment trapping	The trapping and retention in the wetland of sediment carried by runoff waters
				Phosphate assimilation	Removal by the wetland of phosphates carried by runoff waters
				Nitrate assimilation	Removal by the wetland of nitrates carried by runoff waters
				Toxicant assimilation	Removal by the wetland of toxicants (e.g. metals, biocides and salts) carried by runoff waters
				Erosion control	Controlling of erosion at the wetland site, principally through the protection provided by vegetation.
			Carbon storage		The trapping of carbon by the wetland, principally as soil organic matter
	Direct benefits	**Biodiversity maintenance**[2]			Through the provision of habitat and maintenance of natural process by the wetland, a contribution is made to maintaining biodiversity
		Provisioning benefits	Provision of water for human use		The provision of water extracted directly from the wetland for domestic, agriculture or other purposes
			Provision of harvestable resources		The provision of natural resources from the wetland, including livestock grazing, craft plants, fish, etc.
			Provision of cultivated foods		The provision of areas in the wetland favourable for the cultivation of foods
		Cultural benefits	Cultural heritage		Places of special cultural significance in the wetland, e.g., for baptisms or gathering of culturally significant plants
			Tourism and recreation		Sites of value for tourism and recreation in the wetland, often associated with scenic beauty and abundant birdlife
			Education and research		Sites of value in the wetland for education or research

Table 3.7: Classes for determining the likely extent to which an ecosystem service is being delivered (Kotze *et al.,* 2008)

Score:	<0.5	0.5 – 1.2	1.3 – 2.0	2.1 – 2.8	>2.8
Rating of the likely extent to which the ecosystem service is being delivered	Low	Moderately Low	Intermediate	Moderately High	High

3.6.2 Data analysis

All scores gathered from the assessments of the different wetlands sites, are logged into the accompanying WET-EcoServices datasheet (Kotze *et al*., 2008) (Appendix B). The score for each of the fifteen identified ecosystem services being evaluated is calculated. It is then presented in radar graphs (see Section 4.5) for easy comparisons and further evaluation thereof.

3.6.2.1 Spatial representation of plant species diversity, functional diversity and ecosystem services

The ecosystem services being delivered by the wetlands of the TMA were also spatially represented on surface interpolation rasters. The analysis of this it the same as presented in Section 3.4.4.2

4 RESULTS AND DISCUSSION

4.1 Introduction

In this chapter, the results of all different aspects of the wetlands studied within the TMA are provided and thereafter discussed. It is the objective of this chapter to provide the results obtained from the quantification of the urban-rural gradient of the TMA, and thereafter, the classification of the different wetlands along this gradient. The results and discussion regarding the quantification of the floristic composition of the identified wetlands (focussing on plant diversity and functional traits), and the condition thereof, using two indicators (WIV and FQAI) is provided. Lastly, the scores for the ecosystem services potentially being delivered by the wetlands are provided and discussed.

4.2 Urbanisation gradient of the TMA

To effectively compare the wetlands in terms of their degree of urbanness, the wetland sites were ultimately identified as 'urban' and 'rural' wetlands, based on selected urbanisation measures (ED, PVEG, PURB and DWEL). The subsequent calculations of the cluster analysis are presented in Figure 4.1, where two urbanisation classes, namely, 'urban' and 'rural' have been identified at a cut-off point of 70% Bray-Curtis similarity.

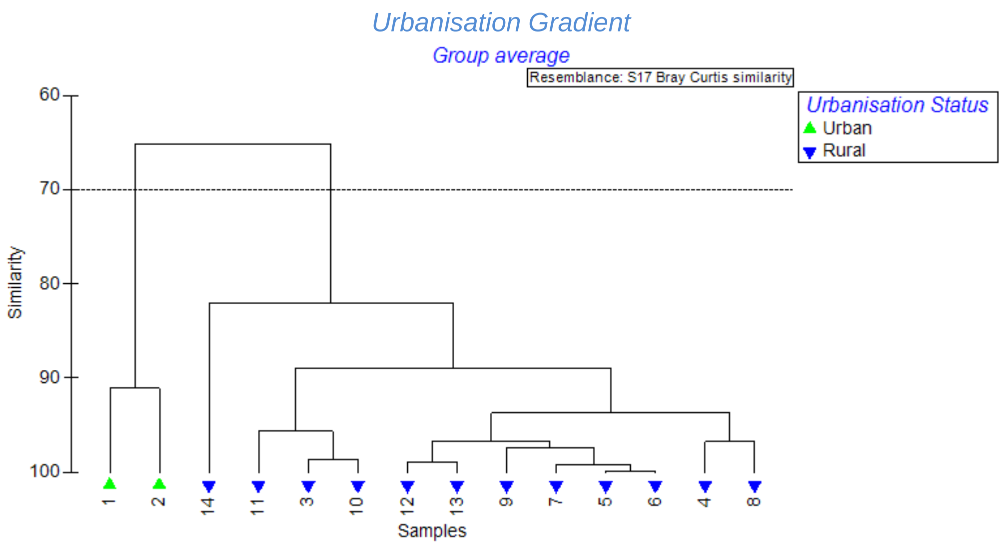


Figure 4.1: Cluster analysis results (dendrogram – group average) for the urbanisation measure values of the selected wetland site (based on Bray-Curtis similarity index). Wetlands were classified into two classes, namely urban and rural, at a 70% Bray-Curtis similarity (grey dashed line).

Table 4.1 provides the specific urbanisation measure values for each selected wetland study site, categorised as urban or rural based on the aforementioned urbanisation measures. Rural wetlands are all situated in matrix areas that have less than 3% impervious surfaces (% urban coverage) (Table 4.1), thus being the main factor for the grouping of certain wetlands as 'rural'. Urban wetlands are all situated in matrix areas that have less than 65% of natural vegetation remaining, and a density of dwellings exceeding 3 per hectare. For the purpose of confirming the urban and rural classification of the wetland sites, an NMDS ordination was created (Figure 4.2). In Figure 4.2 all the urbanisation measure values were used as presented in Table 4.1, indicating a clear differentiation of urban and rural wetlands. In terms of the location of the two urban wetland sites (Figure 3.2) they can be regarded as urban central (wetland 1) and urban edge (wetland 2). No clear grouping in terms of location of the rural wetland sites from the city of Potchefstroom is apparent from the urbanisation measure values (Table 4.1). Photographs of some of the different urban and rural wetlands are provided in Figure 4.3.

Table 4.1: The urbanisation measure values for the 500m buffer areas surrounding the 14 selected wetland sites along the urbanisation gradient as determined by the selected urbanisation measures. The wetland study sites were classified as urban or rural based on the PRIMER cluster analysis, and is arranged by decreasing urbanisation (from top to bottom) based on all urbanisation measures (as in Figure 4.2).

	Wetland site	**Wetland size (ha)**	**Buffer size (ha)**	**% Urban coverage**	**Fragmentation (ED)**	**% Vegetation cover**	**Dwellings**	
URBAN	**1**	31,58	216,09	43,77	477,458	55,34	3,661	Decreasing urbanisation
	2	17,01	225,429	35,9	397,045	63,79	3,66	
RURAL	**11**	2,58	111,1733	1,25	293,14	87,59	0,107	
	10	3,79	117,668	2,43	263,765	87,93	0,256	
	3	2,85	116,856	1,98	266,72	93,53	0,094	
	9	1,14	100,518	1,51	230,765	93,83	0,05	
	7	3,21	112,972	0,04	218,076	99,62	0,018	
	6	2,95	117,25	1,63	218,115	96,22	0,051	
	5	1,76	104,247	2,46	217,995	96,29	0,201	
	12	4,79	126,138	0,72	201,93	97,5	0	
	13	1,47	104,897	1,97	205,553	95,5	0,086	
	8	3,41	116,025	2,75	188,98	90,57	0,121	
	4	4,24	119,667	0,51	177,39	94,92	0	
	14	1,44	101,608	0,03	130,043	96,72	0	

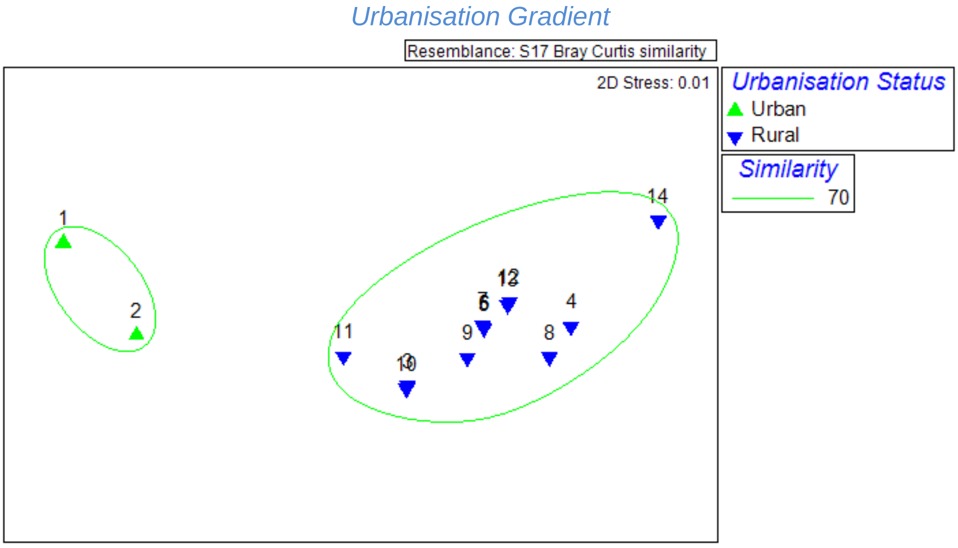


Figure 4.2: NMDS ordinations to express the wetland sites classified as urban or rural based on the values of the four main urbanisation measures. Two distinct clusters (indicated by circles) are visible, indicating that wetlands 1 and 2 are clearly dissimilar from the rest of the wetlands, mainly in terms of % urban coverage.

(a)

(b)

(c)

Figure 4.3: Examples of the urban wetlands ((a) urban wetland 1 and (b) urban wetland 2) in which residential areas and other infrastructure is visible. A photograph (c) of rural wetland 14, in which no building infrastructure is visible.

4.3 Wetland classification

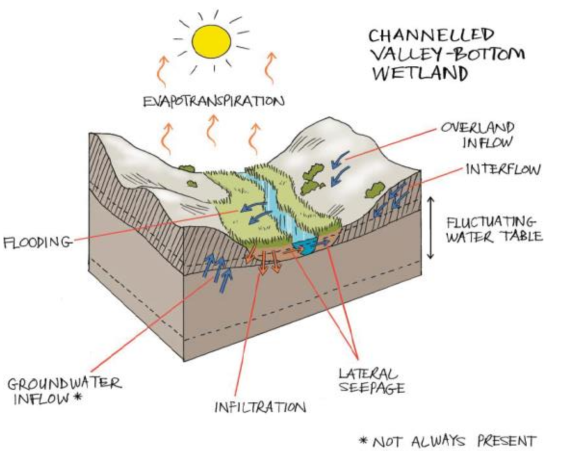
Hydrological and geomorphological characteristics of the two urban and twelve rural wetland sites were used to distinguish between the wetlands based on how they function. These distinctions are known as hydrogeomorphic (HGM) units and are determined by (1) the shape and localised setting of a wetland; (2) hydrological characteristics, which describe the nature of water movement into, through and out of the wetland; and (3) hydrodynamics, which describe the direction and strength of flow through the wetland (Ollis *et al.,* 2009). All wetlands surveyed in the TMA were classified as valley bottom wetlands using the classification methodology proposed by Ollis *et al.* (2015) (Table 4.2). Valley bottom wetlands receive the majority of their water from the movement of water down the drainage network, rather than through local rainfall (Garden, 2008). Valley bottom wetlands may also be classified as channelled or unchannelled, suggesting variations of water inputs from surface channels, surface runoff and groundwater (see Figure 4.4). Where there is a channel identified in the wetland, there is no lateral migration of water through the wetland that could cause channels (Garden, 2008).

Channelled valley bottom wetlands identified within this current study included rural wetlands 3, 4, 5, 10, 11, 12,13 and 14 (Figure 4.4 a and b). These wetlands are situated at the bottom of a river valley, and are marked by small channels and are fed by groundwater from the surrounding slopes and occasionally through overbank flooding. Unchannelled valley bottom wetlands, identified to be the two urban wetlands (wetland 1 and 2) and four rural wetlands (wetlands 6, 7, 8 and 9) (Figure 3.4c), which have no channels, and are fed by groundwater from the surrounding slopes.

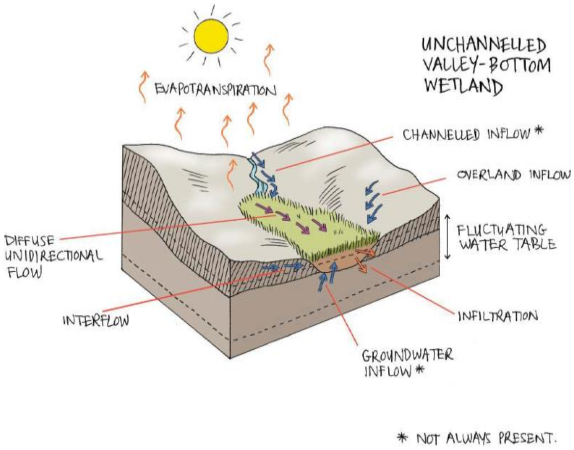
Based upon site characteristics, no significant differences could be noted between channelled and unchannelled valley bottom wetlands in terms of plant species composition and diversity (see section 4.3) and ecosystem services being delivered (see section 4.5). The focus of this chapter will therefore be on the differences between urban and rural wetlands.

Table 4.2: The classification of the wetland sites identified within the TMA, according to the classification system proposed by Ollis *et al.* (2015)

Level 1: System	Level 2: Regional Setting	Level 3: Landscape unit	Level 4: Hydrogeomorphic (HGM) unit
Inland: An ecosystem that has no existing connection to the ocean but which is inundated or saturated with water, either permanently or periodically.	**Ecoregion:** The study area falls within the Highveld Ecoregion	**Valley floor:** The typically gently sloping, lowest surface of a valley	**Channelled valley bottom wetland:** A valley bottom wetland with a river channel running through it.
			Unchannelled valley-bottom wetland: A valley bottom wetland without a river channel running through it.



(a) (b)



(c) (d)

Figure 4.4: Conceptual illustration of a (a) channelled valley-bottom wetland and a photograph of (b) rural wetland 13 indicating flooding from the adjacent river channel; and a conceptual illustration of (c) an unchannelled valley bottom wetland and a photograph of (d) urban wetland 2, indicating no obvious channelling in the wetland. The illustrations show the typical landscape setting and the dominant inputs, throughputs and outputs of water in each type of wetland. Figure 4.4a and c from Ollis *et al.* (2015).

4.4 Plant species composition and plant diversity of wetlands

Many types of wetlands are characterised by distinct plant communities and vegetation can be used to easily identify these wetlands. Plant communities of urban wetlands are often represented by plants that can grow equally well in surrounding habitats and in wetlands (Tiner, 2003). This can make it difficult to simply use vegetation to identify these wetlands, but Cilliers *et al.* (1998) showed that it is possible to identify plant communities within urban wetlands, which gives an indication of the presence of a wetland. Studies by Cilliers *et al.* (1998) and Van Wyk *et al.* (2000) in Potchefstroom and Klerksdorp urban areas respectively, showed that it is possible to also determine the effect of neighbouring land-use changes on wetlands. Results of the current study can be compared to these earlier studies.

It is thus important to study wetland vegetation to correctly identify wetlands, with specific focus on its floristic composition (with specific mention of the plant diversity and functional traits) and the condition thereof, for management and rehabilitation practices and to ultimately determine the effect of urbanisation on wetland functioning.

4.4.1 Results

4.4.1.1 Vegetation cover

The vegetation cover for each of the wetlands are indicated in the figure below (Figure 4.5). This indicates the vegetation cover of each wetland site independent of their size.

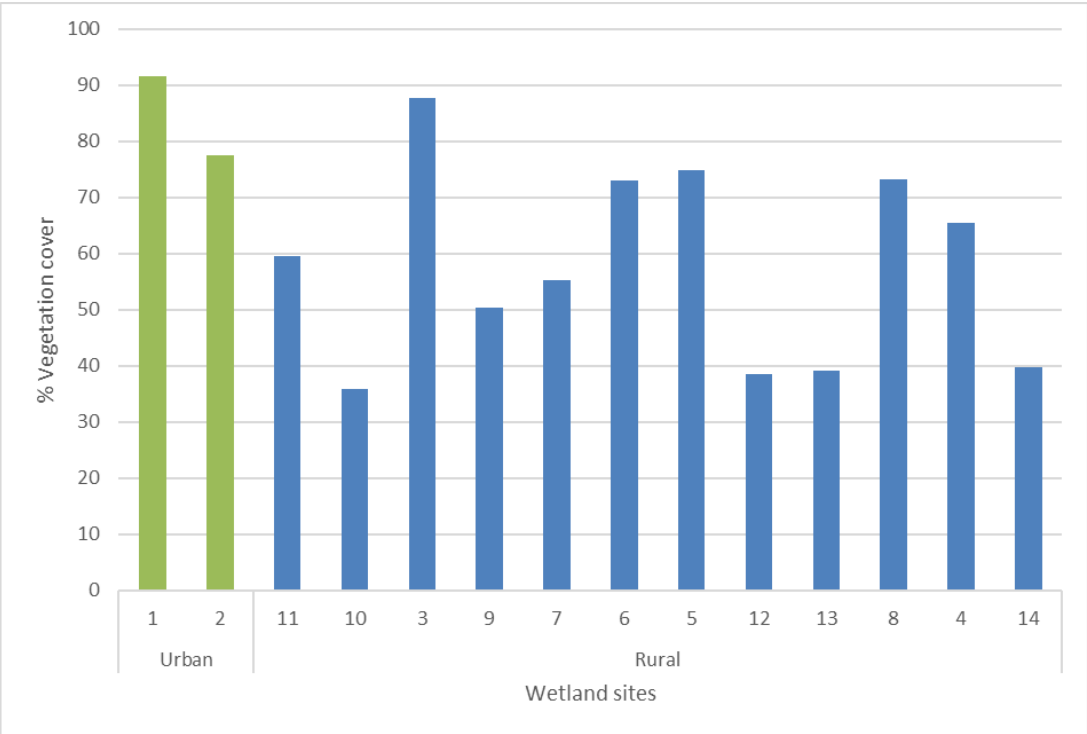


Figure 4.5: The percentage vegetation cover of each of the wetland sites investigated in the TMA.

4.4.1.2 Plant species composition

Figure 4.6 describes the distribution of the transects within the ordination space based on their species composition and percentage cover. No distinct boundaries between the sites can be observed, however, transects of rural wetlands 10 and 11 differ in terms of species composition from the rest of the wetland transects. Transects of the urban wetlands (wetlands 1 and 2) also grouped to one side of the ordination, a clear separation from the remaining sites. There was no clear distinction in species composition between channelled valley bottom wetlands and unchannelled valley bottom wetlands, as indicated under section 4.3.

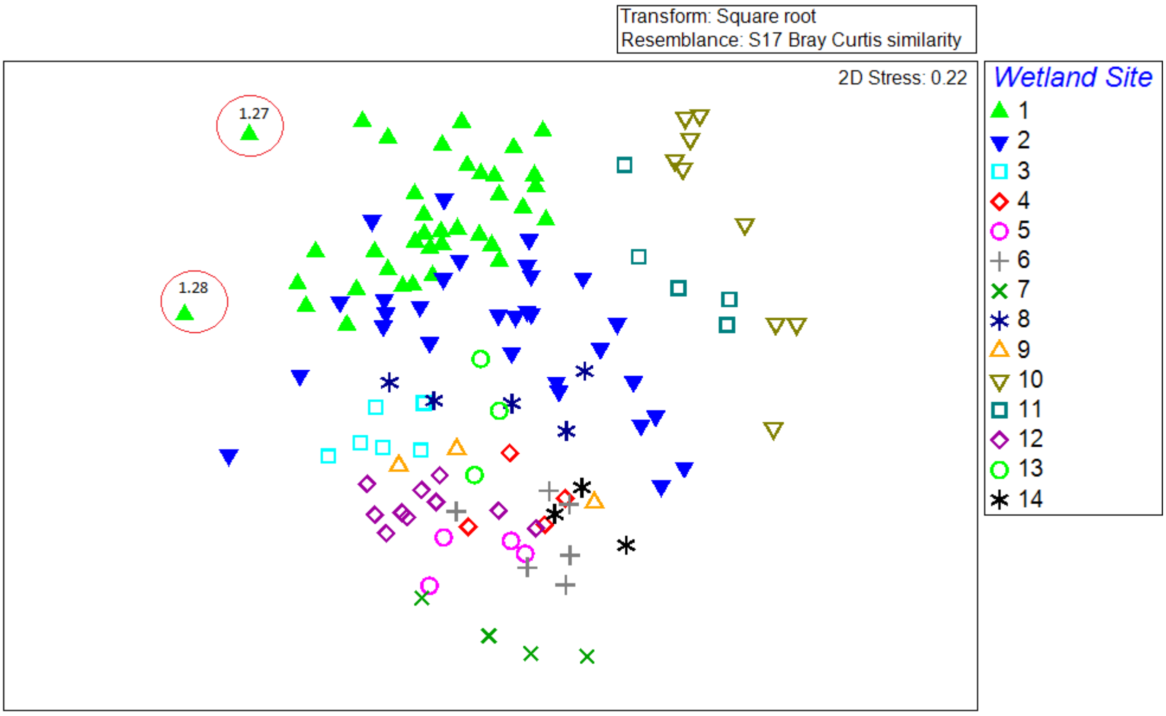


Figure 4.6: NMDS ordination of the transects of all the wetland sites of the TMA based on percentage cover (abundance) of all the species (indigenous and alien). Transects 1.27 and 1.28 is marked as outliers

A cluster analysis (based on the Bray-Curtis similarity indices) of the transects based on percentage cover of each plant species within each wetland site (Figure 4.7) indicates that the rural wetlands (A) and most of the urban wetlands (B) form two separate clusters. Transects from wetlands 10 and 11 clustered together (A1) within the rural wetlands cluster, confirming the ordination results. Some of urban wetland 2 transects group with the rural wetlands (A2 and A3). Transect 28 from wetland 1 and transect 3 from wetland 2 (B1) are different in terms of species composition from all other transects. There is also a division of transects within the urban wetland cluster (B). Transects 23 and 27 from urban wetland 1 (B2) differ in species composition from the rest of the urban cluster. The location of each of these transects (transect 1.27 and 1.28) is also indicated by the NMDS ordination in Figure 4.5, indicating that it differs from all other transects in wetland 1.

A NMDS ordination of the transects based on presence/absence data was created to compare the different wetland in terms of species occurrence within each wetland site (Figure 4.8). It is clear that the rural wetlands 10 and 11 differ from both the urban wetlands, just as much as a second cluster of rural wetlands (wetlands 3, 4, 5, 6, 7, 9 12, 13 and 14) differ in species composition from the urban wetlands (Figure 4.8). Transect 3 from wetland 2 is the urban transect that is most similar in species composition to that of wetlands 10 and 11. Rural wetland 8 most closely resembles plant species composition of an urban wetland in comparison to any other rural wetland.

Determining which species occur more often than others within the wetlands, will aid in explaining the plant species composition occurring in each wetland site. Table 4.3 provides a list of the 20 most frequent species within the 14 wetland sites. No single species occurred in all 14 wetland sites, but 3 species (*Cyperus longus, Falckia oblonga* and *Hemarthria altissima*) occurred in 13 of the 14 wetlands sites.

The 20 species which dominated the urban wetlands, based on the percentage cover of each species in all the urban wetlands, was also determined (Table 4.4). *Cyperus longus* had the highest mean percentage cover of plant species in urban wetlands. *Paspalum dilatatum,* an alien species, had a total cover of 43% within all the urban wetlands.

Table 4.5 provides a list of the 20 most dominant plant species occurring in rural wetlands, based on the percentage cover of each species in all the rural wetlands. *Brachiaria eruciformis* dominates the rural wetlands, occurring in all but one rural wetland site. Only one species (*Cyperus longus)* are one of the most dominant plant species in both urban and rural wetlands.

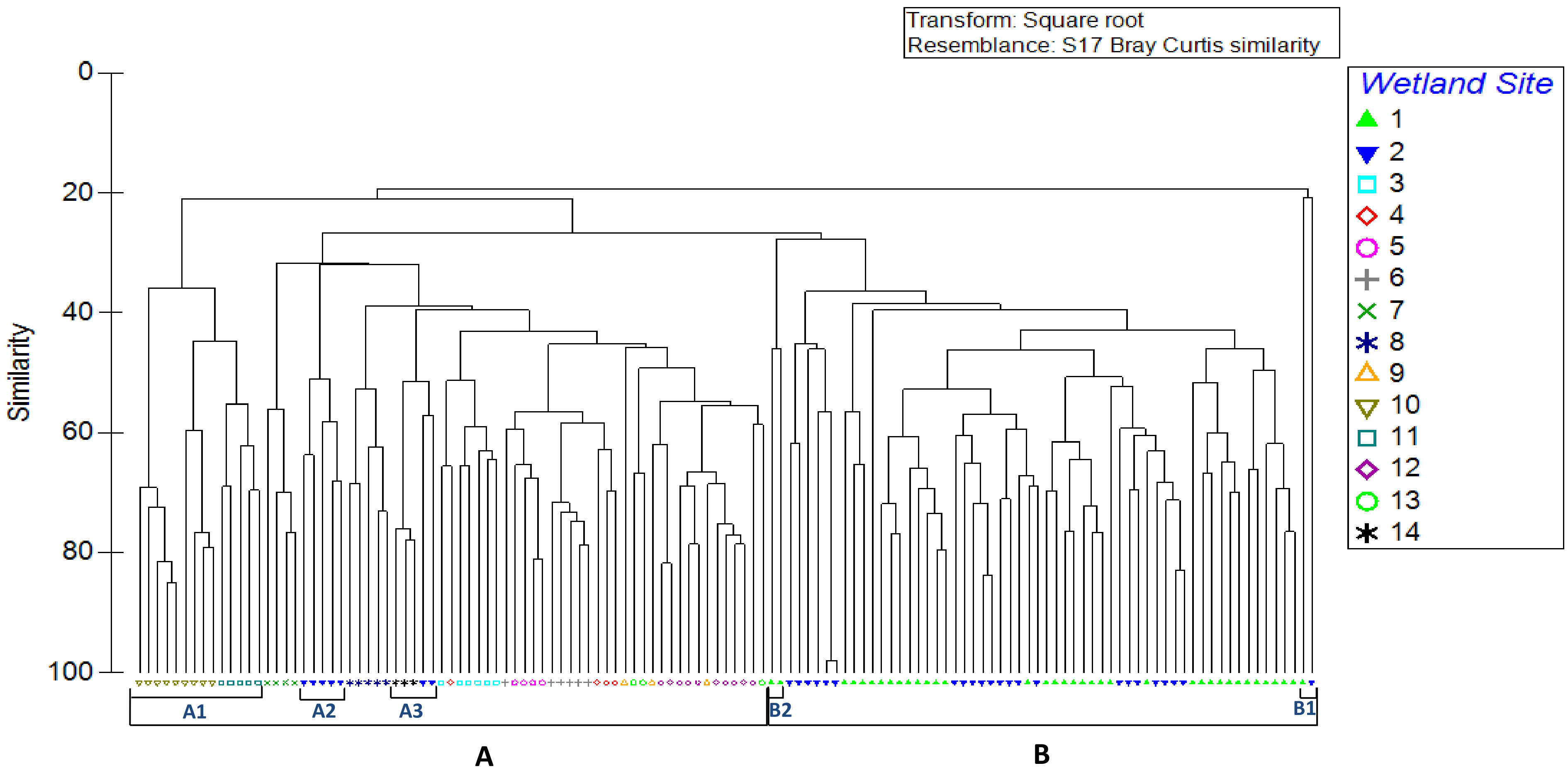


Figure 4.7: Cluster analysis of the transects of all the wetland sites surveyed in the TMA, based on species composition (percentage cover). (A = rural wetlands; B = urban wetlands; A1 = transects of wetlands 10 and 11; A2 and A3 = transects of wetland 2 grouping with rural wetlands; B1 = Transect 28 from wetland 1 and transect 3 from wetland 2; B2 = transects 23 and 27 from urban wetland 1

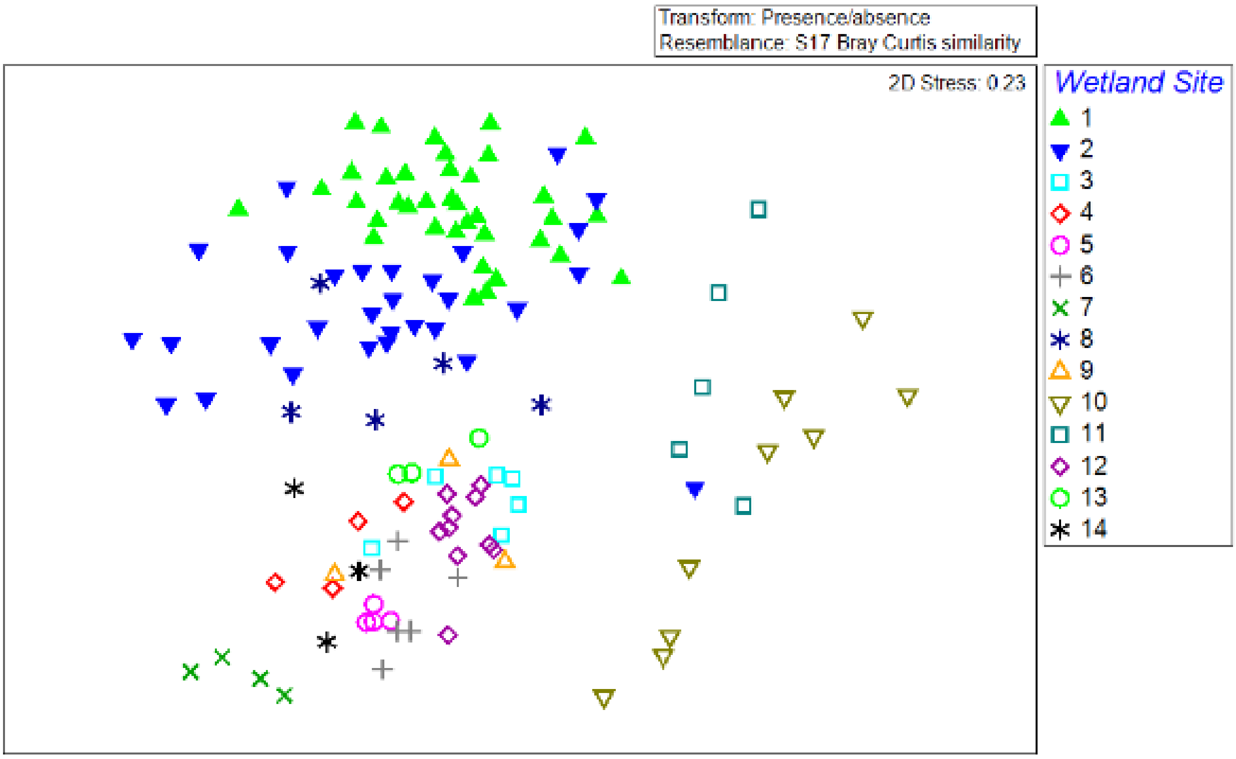


Figure 4.8: NMDS ordination of the transects of all the wetland sites of the TMA based on the presence/absences of all the species (indigenous and alien).

Table 4.3: The 20 most widely distributed species within the studied urban and rural wetland sites in the TMA – as frequency out of 14 wetlands (* indicates alien species).

Species	Occurrence in wetland sites
Cyperus longus	13
Falckia oblonga	13
Hemarthria altissima	13
Brachiaria eruciformis	12
Cyperus fastigiatus	12
*Leersia hexandra**	12
*Verbena officinalis**	12
Echinochloa holubii	11
Marsilea capensis	11
*Rumex crispus**	11

Species	Occurrence in wetland sites
Crinum bulbispermum	10
Cynodon dactylon	10
*Paspalum dilatatum**	10
Paspalum distichum	10
Persicaria decipiens	10
*Persicaria lapathifolia**	10
*Aster squamatus**	9
Cynodon transvaalensis	9
Cyperus laevigatus	9
Setaria pallide-fusca	9

Table 4.4: The 20 most abundant species in the urban wetland sites, ranked as mean % cover per wetland in the TMA (* indicates alien species)

Species	% cover
Cyperus longus	56
*Paspalum dilatatum**	43
Carex glomerabilis	40
*Leersia hexandra**	34
Cyperus laevigatus	30
Hemarthria altissima	27
Falckia oblonga	20
*Rumex crispus**	18
Marsilea capensis	17
Typha capensis	14

Species	% cover
Cynodon dactylon	12
Paspalum distichum	12
Berula erecta	12
Cyperus fastigiatus	9
Phragmites australis	9
Lactuca capensis	6
*Euphorbia helioscopia**	6
Eleocharis dregeana	5
Veronica anagalis-aquatica	5
Schoenoplectus brachyceras	5

Table 4.5: The 20 most abundant species in the rural wetland sites of the TMA, ranked as mean % cover per wetland (* indicates alien species)

Species	% cover
Brachiaria eruciformis	39
Cyperus longus	38
Cyperus fastigiatus	36
Paspalum distichum	34
*Leersia hexandra**	30
Hemarthria altissima	28
Falckia oblonga	28
Echinochloa holubii	27
Marsilea capensis	19
Cyperus laevigatus	19

Species	% cover
*Alternanthera sessilis**	18
*Rumex crispus**	18
*Verbena officinalis**	16
Setaria pallide-fusca	12
*Paspalum dilatatum**	12
Persicaria decipiens	11
*Persicaria lapathifolia**	10
Pycreus macranthus	9
*Aster squamatus**	7
Eragrostis heteromera	6

When comparing the indigenous species occurrence in each transect of the wetlands (Figure 4.10), urban wetlands form more distinct groupings than rural wetlands. A clearer division between urban wetlands 1 and 2 can be seen, as in the ordinations of all the species (Figures 4.7 and 4.8). This indicates that the differences between the two urban wetlands is mainly caused by the differences in the composition of indigenous species.

Although the majority of wetland sites in the TMA was composed of > 50% indigenous species, four alien plant species were identified as either declared invaders or problematic species, as classified according to the Conservation of Agricultural Resources Act (CARA) (1983). These species however, contributed < 1% vegetation cover to each wetland. *Cirsium vulgare* (class 1: declared invader) was identified only in rural wetlands 7 and 10. *Gleditsia triacanthos* (class 2: problematic plant) was identified only in urban wetland 1 and rural wetland 7. *Salix babylonica* (class 2: problematic plant) was identified to only occur in urban wetland 1. *Xanthium strumarium* (class 1: declared invader) was the most abundant declared weed in the study area and occurred in urban wetlands 1 and 2, and in rural wetlands 5, 6, 9, 11, and 13. None of these species were included in the lists of 20 most abundant species of urban and rural wetlands (Tables 4.3 and 4.4).

4.4.1.3 Plant species diversity

A total of 102 species were recorded in the different wetland sites studied in the TMA (Appendix A). In the figure below (Figure 4.9) it is clear that the two urban wetlands are considered to have a large number of species than the rural wetlands.

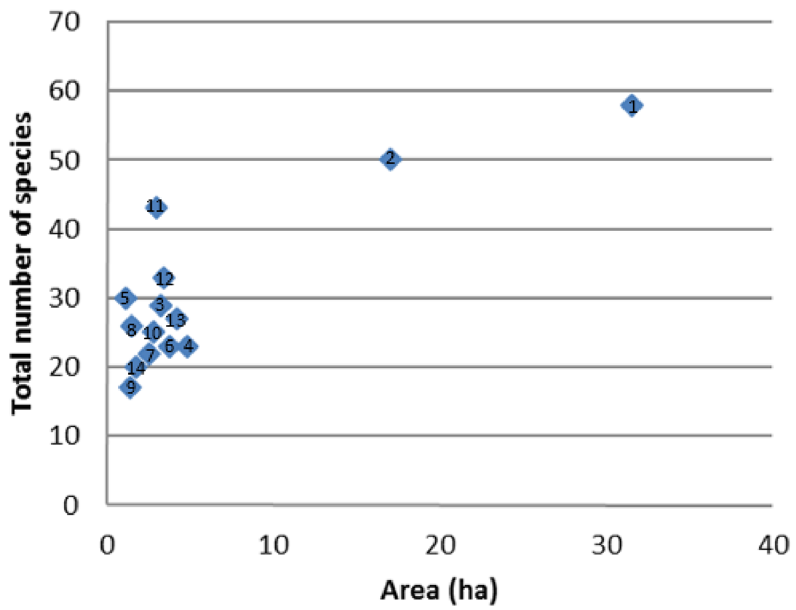


Figure 4.9: Species-area curve for the 14 wetland sites studied in the TMA

The total number of species and the numbers of indigenous and alien species of the different wetland study sites were compared (Figure 4.11 a and b). A clear difference in gamma diversity between urban wetlands and rural wetlands is evident. Urban wetlands had a substantially higher total number of species (or gamma diversity) when compared to the rural wetlands. Wetland 8 has a much higher number of species (48 species) than all the other rural wetlands (Figure 4.11 a). In contrast wetland 14 had the lowest number of indigenous species (10 species) of all the rural wetlands (Figure 4.11 a). Wetland 7 was the only wetland in which more than 50% of the species occurring in the wetland were identified to be alien species (Figure 4.11 b).

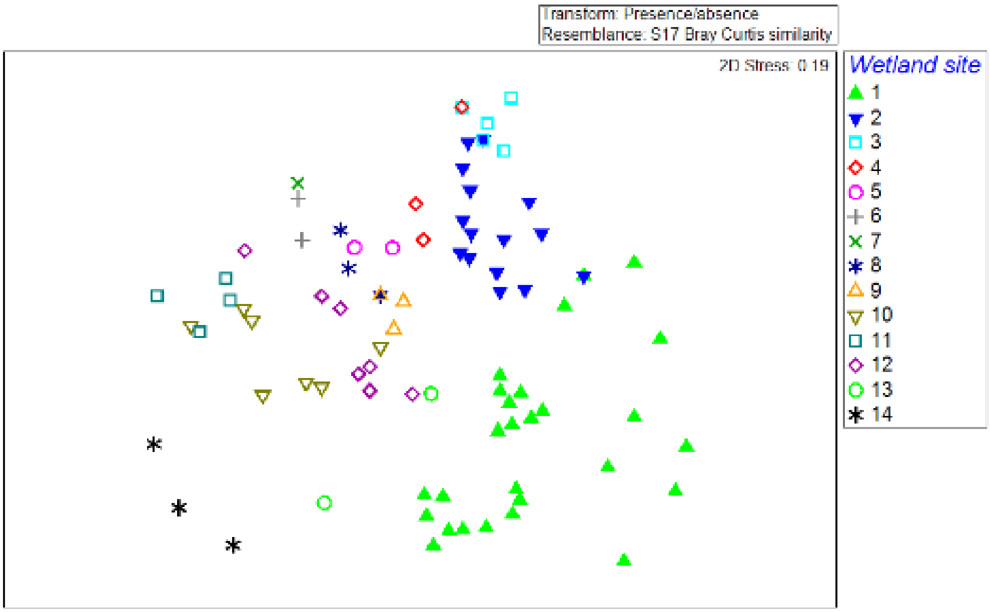
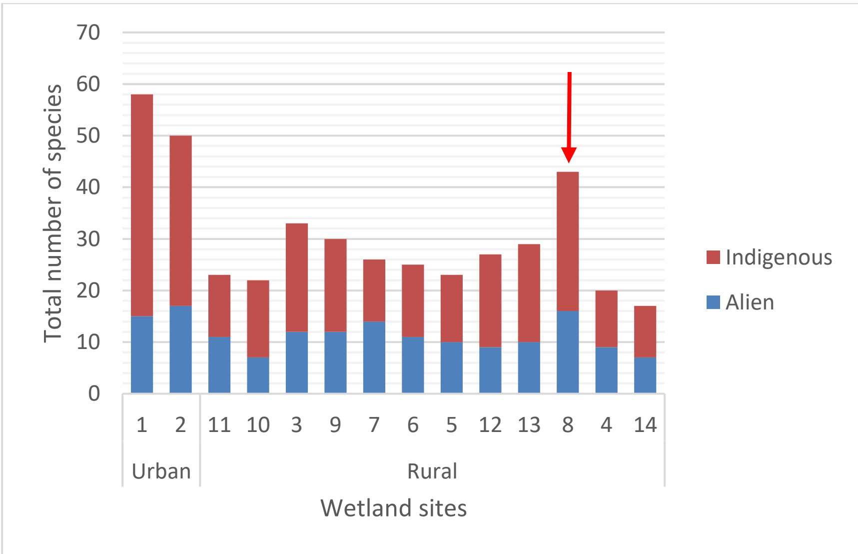
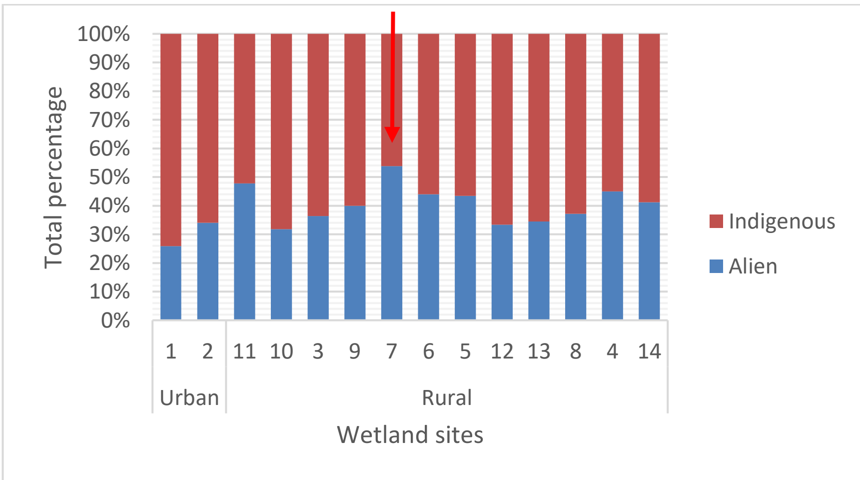


Figure 4.10: NMDS ordination of the transects of all the wetland sites in the TMA based on percentage cover (abundance) of indigenous species.



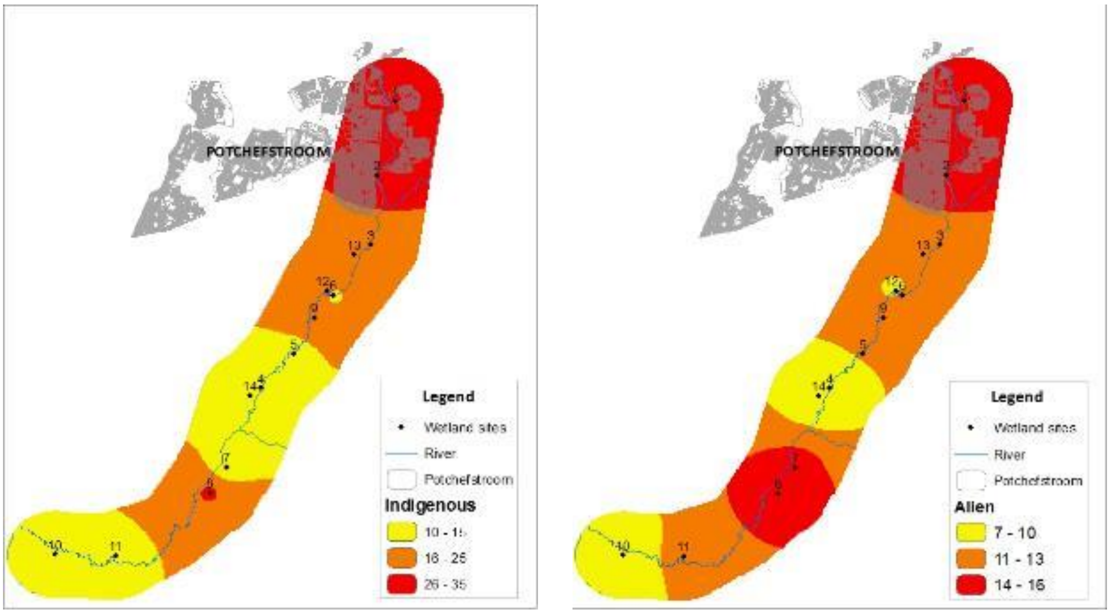
(a)



(b)

Figure 4.11: (a) Gamma (γ) diversity of total species, indigenous species and alien species is presented for each of the wetland sites. (b) The percentage of indigenous or alien species present within each wetland site of the TMA. Red arrows indicate variation in indigenous and alien species in that wetland.

Figure 4.12 shows the IDW surface rasters created by the interpolation tool. The wetlands are compared with regard to their total number of indigenous (Figure 4.12 a) and alien (Figure 4.12 b) species present. The urban wetlands (wetlands 1 and 2) show a higher overall occurrence of both indigenous and alien species as they exhibit a higher gamma diversity (Figure 4.11 a) than all the rural wetlands. Wetland 8 has the highest occurrence of indigenous species (Figure 4.12a) than all other rural wetlands. In Figure 4.12 b it is evident that rural wetland 7 had the most alien species present, alien species were also present in all other wetlands, irrespective of being urban or rural.



(a) (b)

Figure 4.12: IDW surface rasters of the total number of indigenous (a) and alien (b) species recorded per wetland site in the TMA.

Figure 4.13 presents the mean species richness calculated per transect (alpha diversity) for each of the urban and rural wetlands. Wetland 8 (rural) had the highest mean number of species (43 species, of which 73% were indigenous), whilst wetland 10 (rural) had the lowest mean number of species (22 species of which 68% were indigenous). In contrast with their gamma diversity (Figure 4.11a) the alpha diversity (species richness) of the two urban wetlands is lower than that of six of the rural wetlands. Even though it is indicated in Figure 4.12 that the urban wetlands have more indigenous and alien species, when the means of the species richness per transect is compared (see Figure 4.13), it is evident that six rural wetlands does have a higher species richness than the two urban wetlands.

There is a clear difference between urban and rural sites in terms of beta diversity (Table 4.6). Table 4.6 shows that the two urban wetlands have 38 shared species between the two sites and few unique species. When the urban wetlands were compared to the rural wetlands, 45 unique species and 19 shared species were identified. This demonstrates a greater dissimilarity when species composition is compared between urban and rural sites. Rural sites were similar to one another (i.e. 21 unique species and 18 shared species were found when wetland 9 and 12 were compared). Wetland 8 had the greatest difference in beta diversity when it was compared to the rest of the rural wetlands, displaying an average of 35 unique species and sharing on average 16 other species with the other rural wetlands.

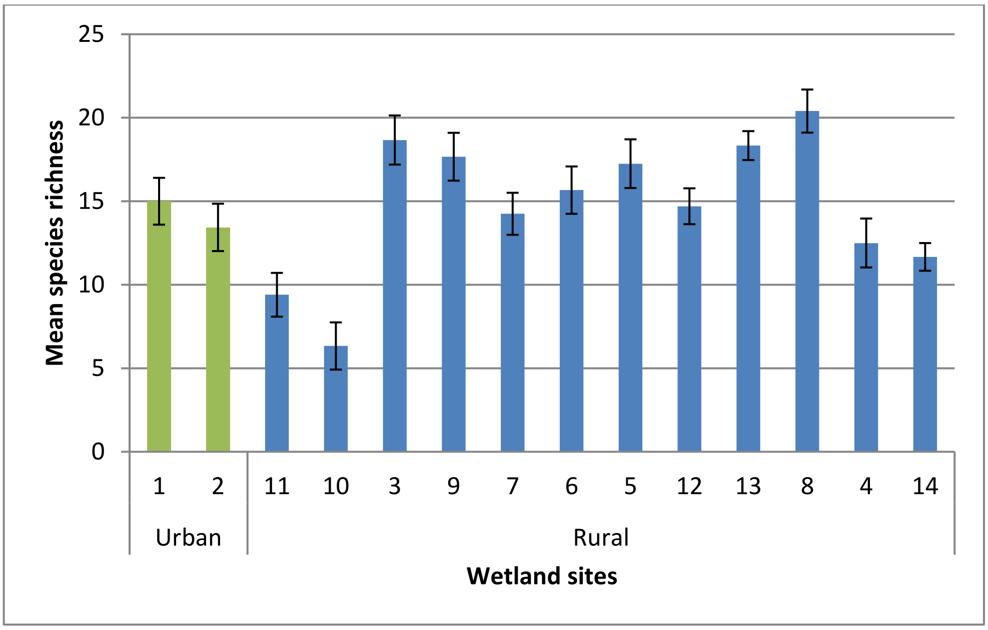


Figure 4.13: Mean species richness per 100m transect (alpha diversity) of all wetland sites in the TMA

Table 4.6: Beta diversity between each wetland site. The unique species of each the wetlands compared is indicated and the value in brackets indicates the shared species between the compared wetlands in the TMA.

1														
2	32(38)													
3	53(19)	35(24)												
4	52(13)	36(17)	19(17)											
5	53(14)	41(16)	24(16)	15(14)										
6	51(16)	39(18)	20(19)	13(16)	16(16)									
7	58(13)	48(18)	29(15)	22(12)	19(15)	25(13)								
8	50(25)	42(25)	29(23)	32(15)	37(14)	27(20)	38(15)							
9	44(22)	34(23)	25(19)	18(16)	21(16)	15(20)	26(15)	30(21)						
10	50(15)	40(16)	25(15)	16(13)	21(12)	23(12)	32(8)	36(14)	32(10)					
11	55(13)	45(14)	38(9)	27(8)	34(6)	28(10)	39(5)	41(12)	27(13)	29(8)				
12	49(18)	39(19)	24(18)	17(15)	20(15)	20(16)	25(14)	39(15)	21(18)	32(13)	30(10)			
13	51(18)	37(21)	24(19)	15(17)	20(16)	18(18)	27(14)	31(20)	21(19)	27(12)	26(13)	14(21)		
14	57(9)	41(13)	24(13)	13(12)	20(10)	14(14)	29(7)	31(14)	21(13)	23(8)	24(8)	22(11)	16(15)	
	1	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**	**11**	**12**	**13**	**14**

Pielou's evenness index (Figure 4.14) is constrained between zero and one, where a value of one indicates equal abundance of all species. Evenness differs between the wetland sites. Wetland 10 has a low evenness value indicating that there is a single species dominating within the wetland. The differences of the urban wetlands are negligible (p = 0.003) pointing out that the species are distributed evenly per transect in the different wetlands.

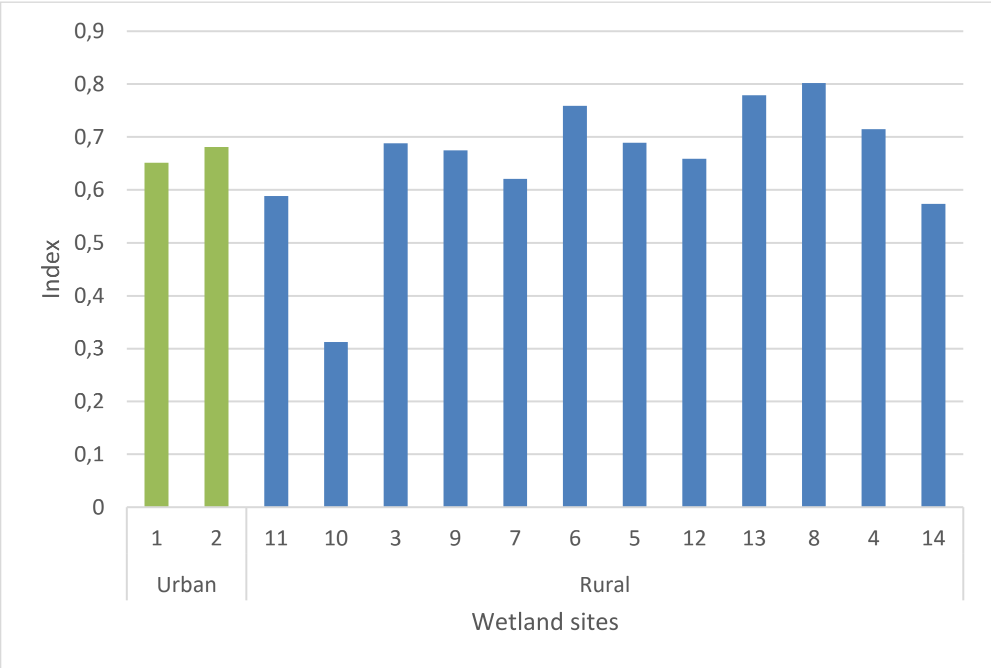


Figure 4.14: Comparative mean values of Pielou's evenness index for the different wetland sites in the TMA.

4.4.1.4 Functional diversity

The number and proportion of plant species and associated plant functional traits recorded during this study is provided in Table 4.7. The proportion of each indigenous and alien species possessing certain plant functional traits is also indicated. Many alien species were also annual species and shrub like in growth form.

The dominant occurrence of herbaceous species (herbs), provides a clear picture of the structure of the vegetation in all the wetlands. Sixty-eight of a total of 102 species are herbs, of which 60% are indigenous. Urban wetland 1 has the highest occurrence of herb species (Figure 4.15), followed by rural wetlands 11, 8 and 9. Shrubs only occurred in the urban wetlands and in one rural wetland (wetland 11).

Table 4.7: Properties of the plant functional composition for all different wetland sites in the TMA.

Category	Trait	Number of species	% species richness	Indigenous		Alien	
				Number	%	Number	%
Status	Indigenous	68	67%	-	-	-	-
	Alien	34	33%	-	-	-	-
Growth form	Geophyte	1	1%	1	100%	0	0%
	Graminoid	30	29%	25	83%	5	17%
	Herb	68	67%	41	60%	27	40%
	Shrub	2	2%	1	50%	1	50%
	Tree	1	1%	0	0%	1	100%
Life span	Annual	28	28%	12	43%	16	57%
	Perennial	72	72%	54	75%	18	25%
Wetland Indicator species	Facultative species	13	13%	9	69%	4	31%
	Facultative dryland species	45	44%	25	55%	20	45%
	Facultative wetland species	20	20%	15	75%	5	25%
	Obligate wetland species	19	18%	19	100%	0	0%
	Obligate dryland species	5	1%	0	0%	5	100%

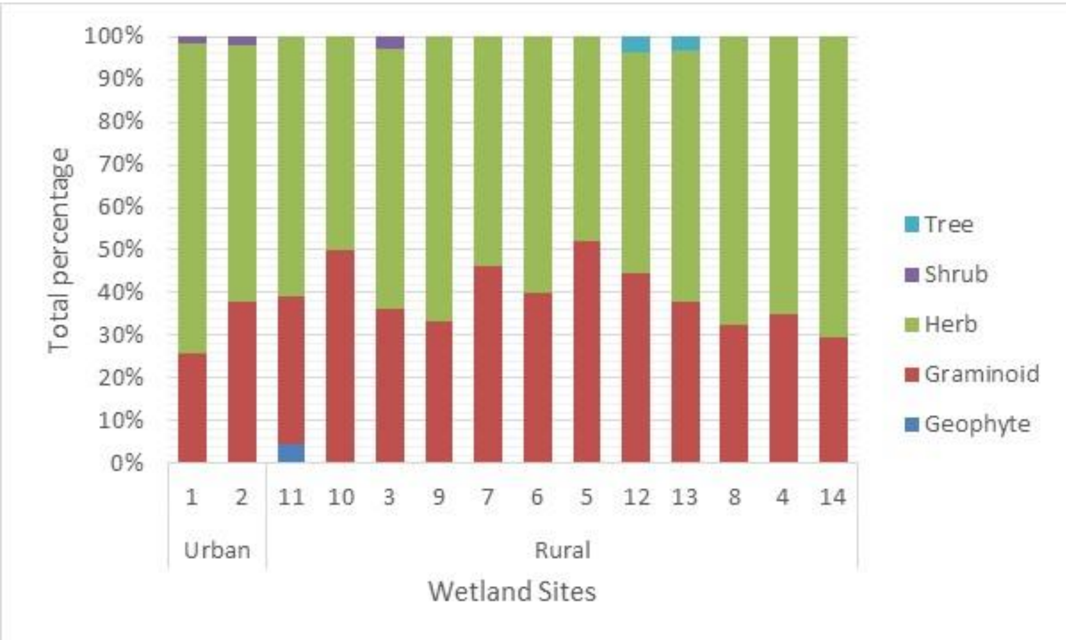
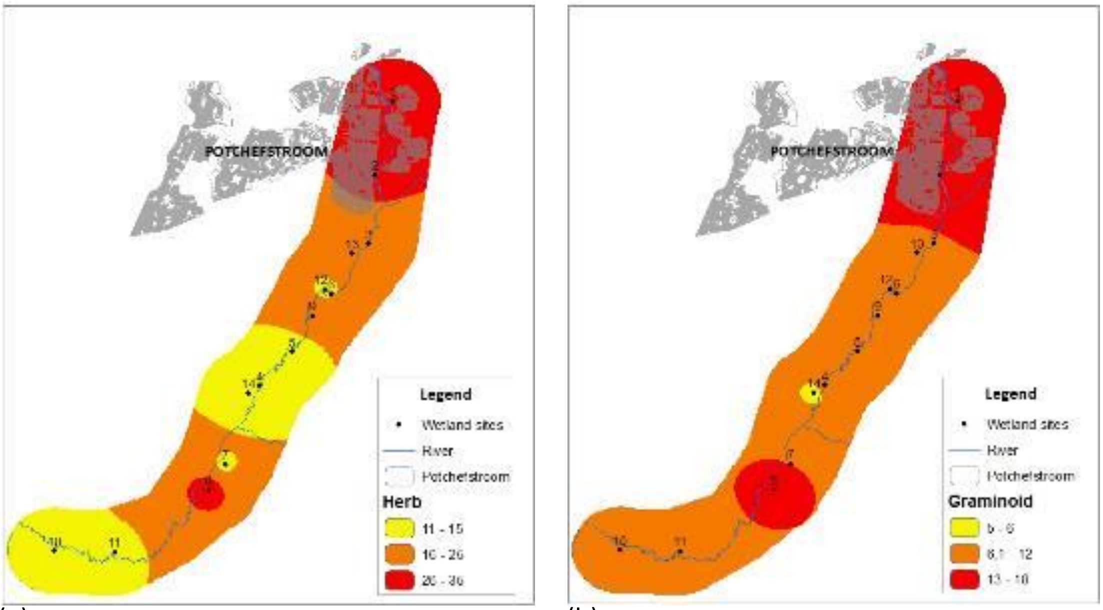


Figure 4.15: Proportions of the different wetland growth forms of plant species occuring in sampled wetlands of the TMA.

Figure 4.16 shows the IDW surface rasters created by the interpolation tool indicating the spatial distribution of the herb and graminoid growth forms in each wetland site (Other growth forms are not shown due to low occurrence thereof). These figures confirm the results of Figure 4.15, indicating that most wetlands are dominated by herb species (Figure 4.16 a) and graminoid species (Figure 4.16 b).



(a) (b)

Figure 4.16: IDW surface rasters of the distribution of the different growth forms of the species, namely (a) herb and (b) graminoid growth forms recorded per wetland site in the TMA.

In the ordination of the transects based on growth forms, no clear groups are visable (Figure 4.17), although the different transects from wetland 10 are positioned further away from the rest of the wetland sites.

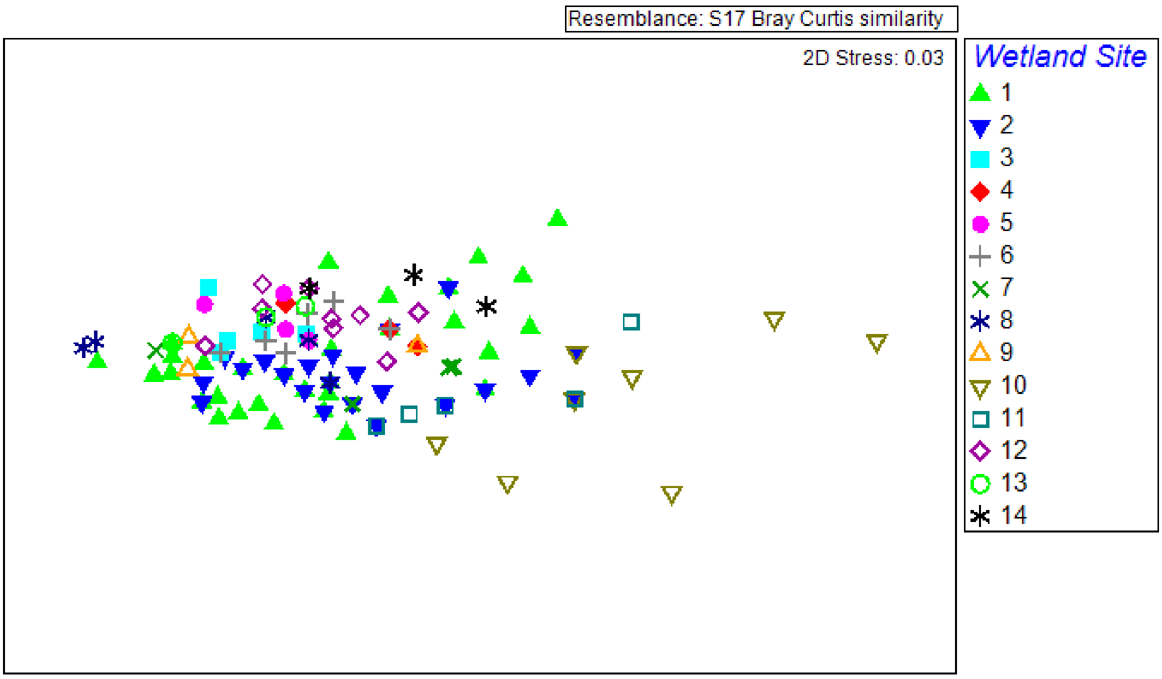


Figure 4.17: NMDS ordination of the transects based on the different species growth forms within the wetland sites of the TMA.

Plant species are grouped into categories based on the differences in the expected frecuency of occurance in wetlands. Facultative dryland species are the most dominant occuring wetland indicator type within all wetland sites (Figure 4.18), of which 55% are indigenous species. Second to that are facultative wetland species, of which 75% are indigenous species. Obligate wetland and obligate dryland species occured in both the urban wetlands, but were absent in some of the rural wetlands.

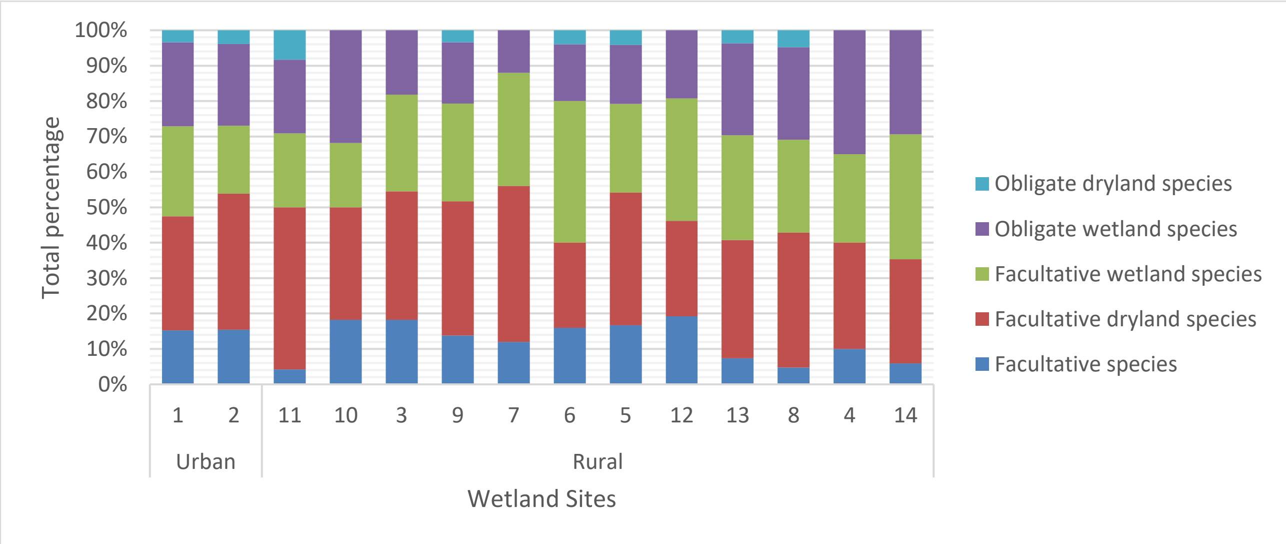
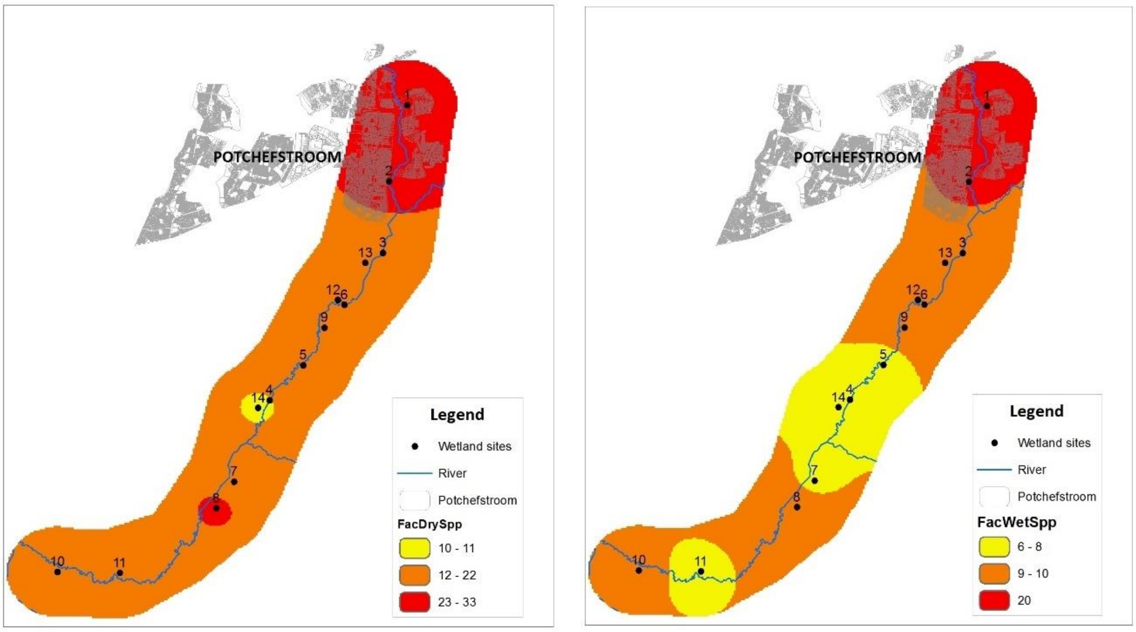


Figure 4.18: Proportions of different wetland indicator species types occuring in the wetland sites of the TMA.

The distribution of the wetland indicator species throughout the wetlands in the study area can be seen in IDW raster images of Figure 4.19. Figure 4.19 represents the results of Figure 4.18 and confirms visually that the most common type of wetland indicator species are those of facultative dryland (Figure 4.19 a) and facultative wetland (Figure 4.19 b). It is also evident that most of the facultative dryland and facultative wetland species occur within urban wetlands, and overall, there were more facultative dryland species identified throughout all wetlands, than facultative wetland species.

Figure 4.20 indicates that no clear groups formed based upon the wetland indicator species present within the different wetland sites. These species are distibuted evenly throughout the wetlands. Wetland 10 is an exception, as it diverts significantly from the remaining wetlands.



(a) (b)

Figure 4.19: IDW surface rasters of the distribution of the wetland indicator species recorded per wetland site in the TMA, namely (a) facultative dryland and (b) facultative wetland species.

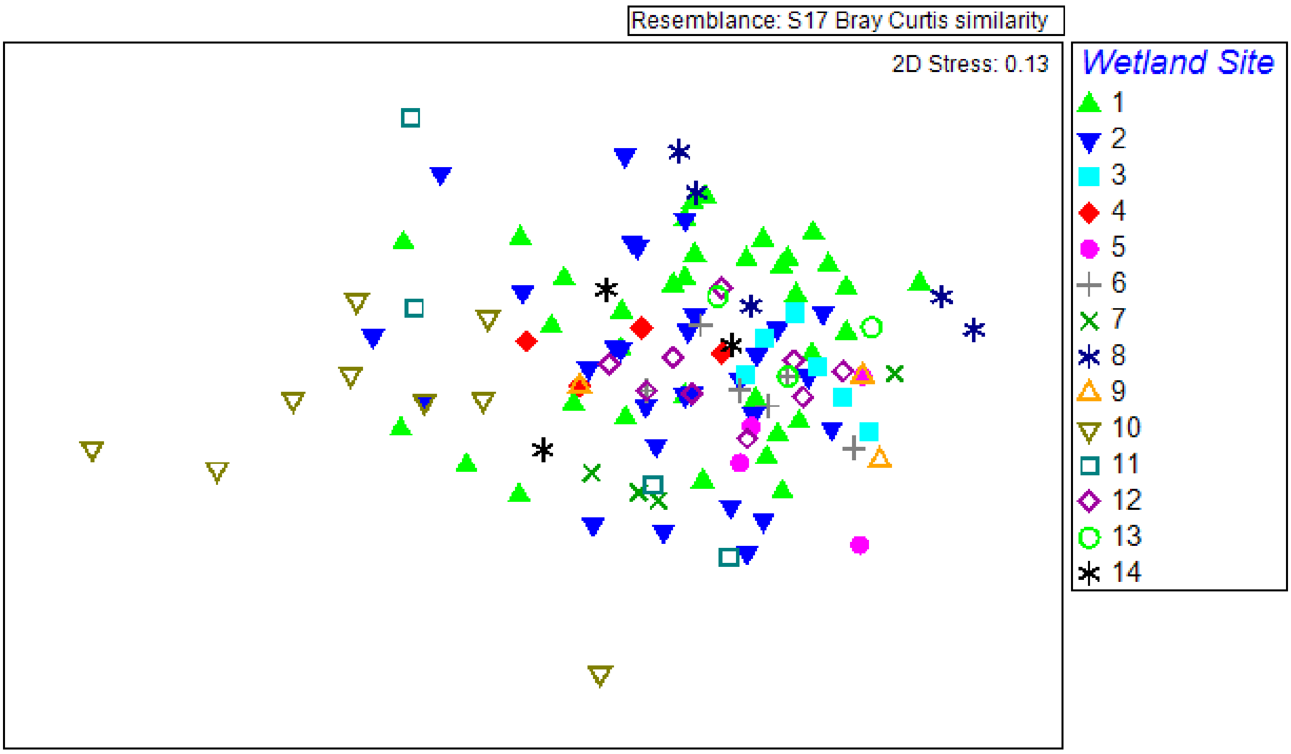


Figure 4.20: NMDS ordination of the transects based on the wetland indicator species type within the wetland sites.

4.4.1.5 Wetland condition determination based on plant species composition and diversity

The Wetland Index Value (WIV) of each wetland site was calculated to indicate the degree to which hydric vegetation was present (Figure 4.21). According to Figure 4.21, all 14 of the wetland sites surveyed in this study could be considered 'true wetlands', since all have WIVs of below 2.5. Rural wetlands 10 and 14 have the lowest WIV and rural wetlands 5 and 6 have the highest WIV.

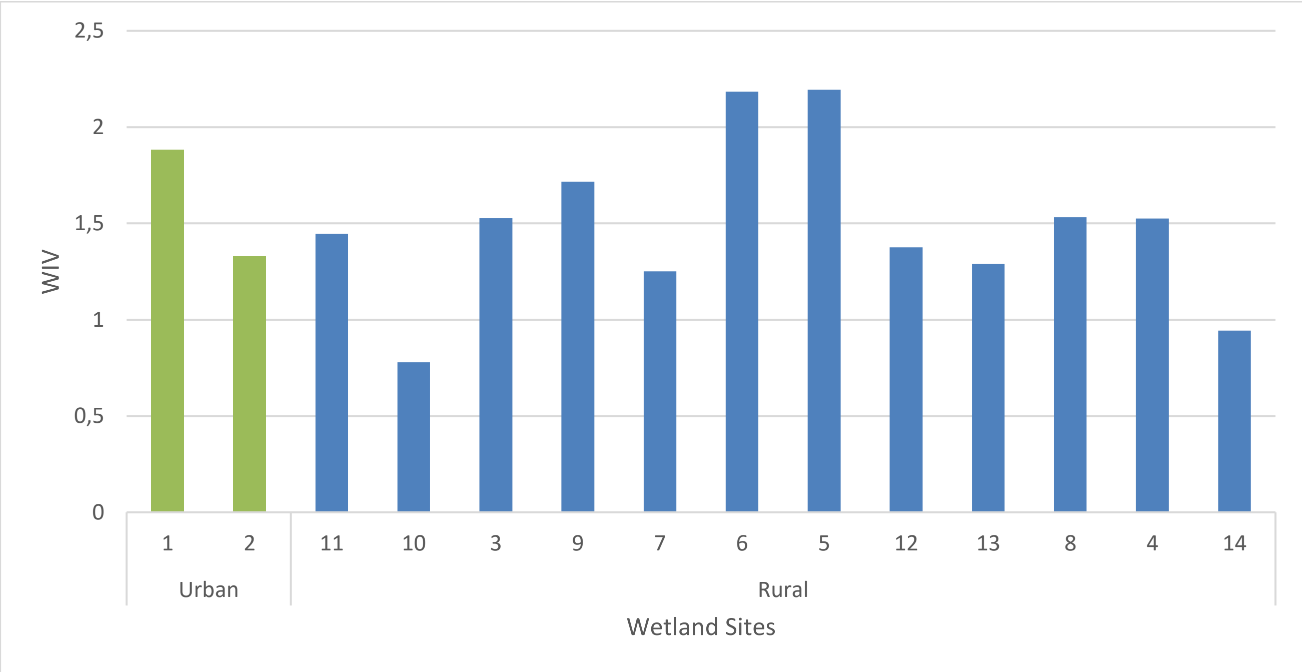


Figure 4.21: The Wetland Index Value (WIV) recorded in each of the wetland sites of the TMA.

The Floristic Quality Assessment Index (FQAI) calculated for each of the 14 wetlands in the TMA (Figure 4.22), indicate that the degree to which the vegetation present reflects the reference wetland condition, did not differ substantively between most of the rural sites (3, 4, 5, 6, 7 and 9). However, wetland 8 was the exception (FQAI of 40.7). Both the urban sites (wetlands 1 and 2) are considered to have relatively high FQAI values when compared to those of the rural wetlands (FQAI of 49.2 and 39.2, respectively) with rural wetlands 10 and 11 with almost the same FQAI values (FQAI of 36.2 and 35.2, respectively) (Figure 4.22).

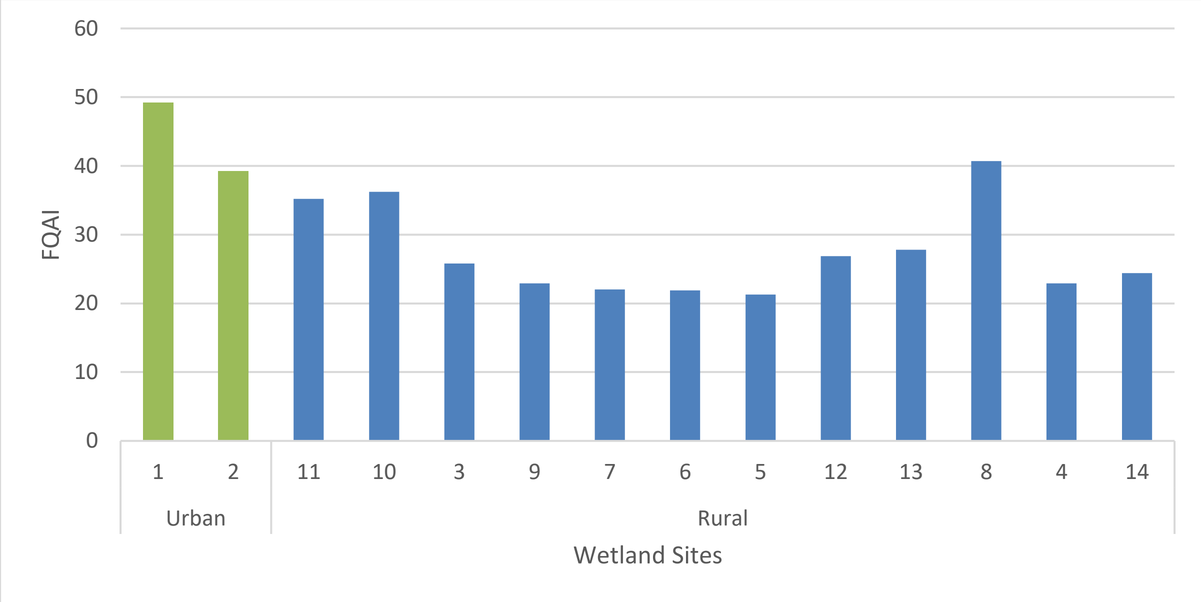


Figure 4.22: The Floristic Quality Assessment Index (FQAI) values recorded in each of the wetland sites of the TMA.

4.4.2 Discussion

4.4.2.1 Plant species composition

"Vegetation cover and structure are key determinants of urban biodiversity" (Oneal and Rotenberry, 2008). Hence, without these structural habitat components, it is unlikely that biodiversity would flourish or ecosystem services be successfully delivered in urban areas. Some of the important Eastern Temperate Freshwater Wetlands taxa identified in urban and rural areas in this study includes *Carex glomerabilis*, *Hemarthria altissima, Leersia hexandra, Paspalum dilatatum, Cyperus longus, Pycreus macranthus, P. nitidus, Falckia oblonga, Phragmites australis* and *Typha capensis* (Mucina and Rutherford, 2006).

Results show a difference in species composition between urban and rural wetlands, and also between some of the different rural wetlands (Figure 4.6). A greater difference in species composition was expected between urban and rural wetlands, since transformation and fragmentation caused by land-use change in the urban areas could alter species composition in urban wetlands more so than in rural wetlands. Figure 4.9 indicates the percentage of vegetation cover of the different wetland sites investigated. These results contradict the results found in the study of Albert and Minc (2004), who found that urban areas had lower and rural/natural wetlands higher vegetation coverage. "Vegetation cover is often negatively correlated with measures of urbanization, such as housing density and impervious surface cover" (Davies *et al.*, 2008). The fact that the urban wetlands in the TMA still have relatively high vegetation cover (Figure 4.9), could be explained by the fact that these wetlands have long been used for recreational purposes and that the municipality and local stakeholders did not allow for any infrastructure development to take place, transforming the wetlands. Tzoulas *et al*. (2007) recognise the importance of retaining an abundance of high vegetation cover in urban areas, and especially in wetlands, as it promotes aesthetic beauty, species persistence and provides a wide range of ecosystem services. The lower vegetation cover of some of the rural wetland sites could be due to vegetation removal as a result of extensive trampling by cattle or the case of old agricultural fields which are still re-establishing, although urban wetlands are also grazed (Figure 4.3) but probably less intensive than the rural wetlands. Louw (1951) also recognised this phenomenon six decades earlier, where wetland vegetation was replaced by agricultural crops and was constantly being grazed and trampled, especially in wetlands areas on the banks of the Mooi River, where these impacts are concentrated as the river provides access to drinking water.

Louw (1951) stated that the sedge family forms the bulk of the wetland vegetation, with some hygrophilous grasses also present. *Cyperus longus* Major Wetland Communities were identified in both the urban wetland sites (wetland 1 and 2) of the TMA in the study of Cilliers *et al.* (1998). The occurrence of seven *Cyperus* species identified in the entire study area, indicates the plasticity of these species in establishing under different environmental conditions (Louw, 1951). Louw (1951) noted that these species are commonly situated along edges of streams and in more marshy areas, however, in the current study, *Cyperus* species were found to be located within different areas of all wetlands. This is proof of the resilience of these species to not only survive ever changing conditions but also adapt to these situations (White and Stromberg, 2011).

In the current study, *Cyperus longus* is still the dominant species within the urban wetlands which still contain a high abundance of the diagnostic species (i.e. *Paspalum dilatatum, Falckia oblonga*) from the original Cilliers *et al.* (1998) classification. Cilliers *et al.* (1998) also studied urban wetlands 1 and 2 with the aim of describe their wetland plant communities. Many of the buffer zones identified around the wetlands of the current study were also classified by Cilliers *et al.* (1998), mostly as grassland communities. The classification of the inner parts of the wetlands by Cilliers *et al.* (1998) is therefore comparable with vegetation identified in the current study. The two urban wetlands (wetland 1 and 2) clustered together due to the high abundance of *Cyperus longus*, *Cyperus laevigatus, Paspalum dilatatum* and *Leersia hexandra* (Figure 4.7). The co-occurrence of *Paspalum dilatatum* and *Leersia hexandra* was also noted by Louw (1951), especially in wetter soils. Two urban transects can be seen as outliers (B1) in Figure 4.7. Both these transects exhibit low overall species abundance with only two species (*Hemarthria altissima* and *Leersia hexandra*) with noteworthy abundances. This may be because transect 28 from wetland 1 was situated in a shallow depression that may be more saturated than other regions. Transect 3 from wetland 2 were sampled on the edge of a storm water canal with only 20% of the wetland consisting of vegetation, as the remainder was water. Cilliers *et al.* (1998) identified the presence of *Cyperus marginatus* in both wetlands 1 and 2, stating that it seems to be decreasing. The low abundance of *Cyperus marginatus* found in this current study confirms this, due to the effects of trampling and grazing, because it was originally situated in relatively small patches in the lower-lying wetter areas of the wetlands (Cilliers *et al.,* 1998). Cilliers *et al.* (1998) identified the *Cynodon dactylon* Invasive community in both urban wetlands 1 and 2, stating that it expands yearly. Louw (1951) already identified vast quantities of this species in disturbed areas within wetlands and surrounding terrestrial areas. Despite the wetlands still being used for grazing by cattle and horses (see Figure 4.3), in this current study, there were other species with much higher abundances making use of available wetland surfaces and not allowing further expansion of *Cynodon dactylon.*

Wetland 8 is considered a 'transition' wetland, as it shares almost equal numbers of species with both urban and rural wetlands. *Cyperus longus*, *Cyperus laevigatus* and *Leersia hexandra* have a high abundance in this wetland, making it similar in species composition to urban wetlands, however, these wetlands still share overlapping species that coincide with the remaining transects in grouping A (Figure 4.7). All other rural wetlands cluster together due to similar abundances of *Cyperus fastigiatus*, *Paspalum distichum*, *Leersia hexandra*, *Falckia oblonga* and *Echinochloa holubii*. Few species occur within these wetlands when compared to the urban wetlands, but those that do occur, have a high enough abundance to be grouped together. There is a small cluster of urban wetland 2 transects (A2 – Figure 4.7) within this greater rural cluster. This can be explained by the abundance of *Paspalum distichum* within these transects which is higher than in any of the other transects occurring in urban wetlands. Wetland 7 is noted to be further away (composition based on its % cover abundance) from the main rural wetland cluster. This wetland did not exhibit significant species abundance, with *Cyperus fastigiatus* dominating this wetland and being the only species that overlaps in abundance with that of all other rural wetlands.

Wetlands 10 and 11 cluster together based on cover (Figure 4.5 and 4.6) and presence/absence of certain plant species (Figure 4.8). Both these wetlands had few species in common with other rural wetlands (Figure 4.10). Only three out of the 20 most abundant species occurring in rural wetlands were found occurring, albeit in low frequencies, in these wetlands (Table 4.5). The species occupying these wetlands were unique to the area (e.g. *Eragrostis micrantha*, *Juncus punctorius* and *Juncus rigidus* only occurs in wetland 10 while *Bidens bipinnata*, *Bidens pilosa*, *Chenopodium album* and *Sonchus wilmsii* only occurs in wetland 11). Both *Cyperus longus* and *Pycreus macranthus* are on the list of the 20 most abundant plant species occurring in rural wetlands (Table 4.5). The abundance of *Cyperus longus* and *Pycreus macranthus* in wetlands 10 and 11 contributes the most to these values.

When comparing only the indigenous species present within the wetlands, it is clear that urban wetlands differ markedly from rural wetlands (Figure 4.10). The indigenous species of wetland 1, *Cyperus longus* and *Carex glomerabilis,* which occurs in high abundance, are setting this wetland apart from the indigenous species that occur with high abundances (*Cyperus longus* and *Cyperus laevigatus*) in wetland 2. All rural wetlands had similar indigenous species present (e.g. *Brachiaria eruciformis, Cyperus longus, Cyperus fastigiatus* and *Paspalum distichum*), thus grouping them together.

Interestingly, the tropical species that commonly occur in African wetlands (*Phragmites, Typha* and *Cyperus* species) (Harper *et al.,* 1999; Owino and Ryan, 2007), all occurred in urban wetlands (Table 4.4), whilst only *Cyperus* species occurred within the rural wetlands (Table 4.5). Louw (1951) noted these reed species (*Phragmites* and *Typha* species) occurred in high abundance in the wetlands surrounding Potchefstroom during his study, suggesting a reduction in the dominance of these species over time. Even though these species are able to thrive in different environments (i.e. different water depths, soil moisture content), since the original study done by Louw (1951), these factors have undergone significant alteration, possibly contributing to the change in abundance of the species in question. The local decrease in the cover of these species could be the general land-use changes which have characterised these wetlands over time, causing fragmentation of large stands of reeds, and ultimately destroying it to make way for crops, infrastructure etc.

The question arises – why were these reed species noted only to occur in conspicuous abundances in urban wetlands? The nutrient-rich water and variable water levels within the urban wetlands, mainly due to runoff from roads and inflow from storm water channels, often favour aggressive and tolerant plant species such as *Phragmites* and *Typha* species (Boers *et al.,* 2007). Thus, the enriched soil and water of the urban wetlands could be considered a more favourable growing environment. This also explains why these macrophyte species (*Phragmites* and *Typha* species) are more often used for pollution treatment in urban areas and constructed wetlands, as they have the ability to remove contaminants, whilst still maintaining normal growth (Ellis *et al.,* 1994).

4.4.2.2 Plant species diversity

The urban wetlands have a considerably higher gamma diversity than the rural wetlands (Figure 4.9a). One reason for this could be that the two urban wetland sites cover a much larger surface area than the twelve rural wetland sites (Table 3.1). The species area curve (Figure 4.9) shows that this is indeed the case. The species area curve for the 14 wetland study sites has reached an asymptote (Figure 4.9), demonstrating that species richness is positively related to habitat area. Urban areas have a much higher species diversity than the rural wetlands and correlate with the surface area of each of the wetlands.

The wetlands in this study showed a relatively high species richness, when compared to the study of Brand *et al.* (2013) in the high altitude montane wetlands of the eastern Free State, South Africa. The wetlands in the study of Brand *et al.* (2013) had an average of 13.56 species per 30 m^2, ranging from 7 to 29 species per sample plot in high altitude grassland wetlands. Although Figure 4.11 indicates that ten of the wetlands have mean values above 13 species, it is important to note that species richness in this study, was calculated per $1m^2$ quadrant (thus 10 x $1m^2$ for each 100m transect).

Overall results suggest that plant species richness of the wetlands increases with the influence of human disturbances (e.g. land-use changes, built-up areas). Thus, some of the rural wetlands (rural wetlands 3, 5, 6, 8, 9 and 13), which had a higher species richness than the urban wetlands (Figure 4.13), are more impacted on, for example by agricultural activities. However, "species richness is likely to vary with proximity to the centre of a city" (McKinney, 2008). In the study of Van Wyk *et al*. (2000), the influence of the surrounding urban environments enhanced the species richness of the (urban) wetlands surveyed. This was mainly due to the presence of the invasive *Eucalyptus camaldulensis* species, which created microhabitats suitable for other introduced wetland species (Van Wyk *et al.,* 2000). Although the current study did not show extensive invasion of woody alien plant species such as the study of Van Wyk *et al.* (2000), it might be possible that the vegetation present in wetland sites with higher species richness or the type of disturbance the wetlands experience, consequently creates unique niches for other species to survive in.

Urban wetlands had a higher gamma diversity than the rural wetlands (Figure 4.11). Similar results were found in the wetland study of Deutschewitz *et al.* (2003) in Germany, where urban wetland areas showed increases in urban plant species richness with increasing anthropogenic influences and structural heterogeneity. Reeves and Champion (2004) said that the lower species diversity of rural wetlands is mainly caused by agriculture, fire and grazing, which all are causes which may lead to the complete removal of vegetation, instead of just impacting on the growth thereof (by means of enriched water, higher temperatures, etc.). The diversity of human activities in cities creates and maintains a variety of habitats ranging from those that are seemingly natural to those that are highly modified, some of which do not occur elsewhere. Thanks to this diversity of habitat types (i.e. open lots, urban parks, wetlands), urban landscapes often have a high species diversity (Shepherd, 1994). Socio-economic factors and household behaviour can also affect species richness (Hope *et al.*, 2003). Disturbances such as garden refuse, storm water runoff and roads influence the vegetation composition of urban sites. Wetlands heterogeneity probably has the largest influence on these urban wetlands, chiefly due to the different anthropogenic factors mentioned before. Propagules may be dispersed through running water or wind to other wetlands downstream influencing their species richness (Dallimer *et al.,* 2012).

The high number of species with low percentage cover (abundance) values present in the wetlands suggests that they are likely to contribute considerably to gamma diversity. This was also the case for the uncommon and rare species present in temporary ponds in England and Wales, which made the biggest contribution to the gamma diversity in that study (Nicolet *et al.,* 2004).

The urban wetlands, which are situated most closely to Potchefstroom, had a higher proportion of indigenous species to alien species than all other wetlands (Figure 4.12), *Cyperus longus* being the most frequently recorded indigenous species in urban wetlands 1 and 2. This finding differs from previous studies (Albert and Minc, 2004). Dallimer *et al*. (2012) stated that alien plant species richness is usually higher with raised levels of built surface such as in urban areas. Indigenous species are usually present in very low quantities or even absent from urban areas as they are not adapted to the inner urban climates and nutrient rich soils and are “restricted to natural and semi-natural habitats, and are often adapted to specific environmental conditions” (Schmidt *et al*., 2014). “Indigenous species are often replaced by alien species or their habitat negatively affected by urbanization through fragmentation or habitat loss” (Dolan *et al.,* 2011). The occurrence of more indigenous species in urban wetlands could be because some of these species (i.e. *Kniphofia ensifolia*) find refuge in urban wetlands (“safe sites”), were protected from other anthropogenic influences. However, the large size of the urban wetlands could increase the habitat integrity for the presence of both indigenous and alien vegetation without there being invasion and dominance of one part of vegetation. Louw (1951) noted the occurrence of *Kniphofia* species in greater quantities along the Mooi River, whereas in the current study only single individuals were noted. It is likely that these species were eradicated and trampled along the river edge, and some remnants now exist only in the urban wetlands.

Overall, rural wetlands had the lowest proportion of indigenous species to alien species within the wetlands studied in the TMA (Figure 4.12). This can be caused by cattle trampling through the wetlands which may destroy the natural wetland vegetation, dispersing the species from one site to another, and may also introduce new (alien) species to the wetland (Reeves and Champion, 2004; Primack, 1993). It should also be noted that the urban wetlands had evidence of horse and cattle trampling, but not necessarily with the same intensity experienced within the rural wetlands. Also, these rural wetlands are surrounded by agricultural areas, thus decreasing the potential suite of species capable of replacing existing vegetation. Schmidt *et al.* (2014) found a lower species richness in areas with surrounding agricultural use as the nutrient runoff enriched soils, favouring the growth of alien versus native vegetation. Fire within the wetlands is normal, and some evidence of burning was evident in rural wetlands 10 and 11. Fire within wetlands could result in the long-term loss of habitat and species richness as shallower wetlands will fill in with sediments and grassland species will replace wetland species subsequent to burn events (Brand *et al.,* 2013).

The variation in species richness (alpha diversity) between the rural wetlands indicates that they are diverse in species abundances (Figure 4.13). A low alpha diversity indicates an underutilisation of potential resources within the wetland site (Mason *et al.,* 2003). This reduces the overall productivity of the wetlands, which is undesirable for a number of reasons (Petchey, 2003). Figure 4.13 also indicates that both the urban wetlands had a lower alpha diversity than six other rural wetlands. This is due to the stands of homogenous species (*Phragmites* and *Typha* species) encountered in different areas of these large wetlands, whereas no large stands (when compared to those found in the urban wetlands) were found within the rural wetlands.

The much higher beta diversity (Table 4.6) in the urban wetlands is due to the high number of species occurring in the urban wetlands, but with low cover (i.e., abundances). This could also explain why the urban wetlands had a higher gamma diversity than all other wetlands. Pielou's evenness index (Figure 4.12) indicated that these urban wetlands were not dominated by only one species, but there were high abundances of *Cyperus longus, Paspalum dilatatum*, *Leersia hexandra, Hemarthria altissima, Carex glomerabilis* and *Falckia oblonga* in both urban wetlands. Most of these were indigenous species, with the exception of *Paspalum dilatatum*. Louw (1951) considered *Paspalum dilatatum* to be a naturalised alien species which usually became weedy in areas of great disturbance.

When all the rural sites were compared to each other, high beta diversity was evident, but this was also accompanied by greater variation in Pielou's evenness index (Figure 4.14) as more alien species were present in them. This uneven distribution of species within the rural wetland indicates a possible underutilisation of resources available in the wetlands (Mason *et al.,* 2005), which can be due to some rural wetlands being more grazed than others (being over utilized – Wetland 10) or perhaps too few grazers to use all the possible resources (underutilized – Wetland 8 and 13). Underutilization tends to decrease the productivity of the wetland and increases the opportunity for invasion of alien species (such in the case of wetland 11, and 4 – Figure 4.11 b) (Mason *et al.,* 2005).

4.4.2.3 <u>Plant functional diversity</u>

Different plant species are more successful in specific parts of the landscape because they have different quantitative traits, such as life span, rooting depths, growth forms, leaf sizes and potential canopy heights (Westoby and Wright, 2006). These traits promote survival in different microclimate environments in the wetlands. To understand ecosystem properties, it is important to address the question of how and why plant traits vary among species and sites. All wetlands in this current study are dominated by certain wetland indicator species, namely, facultative wetland or dryland species (Figure 4.19), and by species which have either herbaceous or graminoid growth forms (Figure 4.16). The spatial distribution of these functional traits in all the wetlands gives the impression that the co-existence of many species collectively promotes stability (Sieben, 2012), and that some vegetation will be able to survive in the case of a sudden environmental change.

The vegetation structure was similar among all wetlands, with low growing herbaceous or graminoid species dominating, with few shrubs and tree species in some areas. The absence or near absence trees and shrubs is characteristic of grassland vegetation, specifically, Eastern Temperate Freshwater wetlands type (Mucina and Rutherford, 2006). This type of structure was also originally described in the study of Cilliers *et al.* (1998) which was characteristic of the *Cyperus longus* Major Wetland community. This consistency in growth forms throughout the different wetland study sites implies limited deviation in environmental conditions since the first study in 1998, and that all wetland sites experience similar disturbances (i.e. trampling, grazing) (Ramsay and Oxley, 1997).

Plants growing in wetlands (hydrophytes) (Tiner, 1999) are characterized by their ability to grow in anaerobic soil or soil saturated with water. However, many of the plants found in wetlands also grow in terrestrial habitats. These plants, although being more common in the latter, are also tolerant in varying degrees to anoxic soil conditions (Tiner, 1999). The plant species composition of wetland sites in this study likely result, at least in part, from the range of facultative wetland and facultative dryland species they support. Surrounded by grassland and agricultural land-uses, the wetland sites in this study contained almost as many facultative dryland species as facultative wetland species. Flinn *et al.* (2008) also found that wet habitats in southern Quebec, Canada (in the St. Lawrence River valley), had as many dryland species as wetland species. This was also the case for dryland and wetland species of riparian forest habitats in the Adirondacks, New York (Wright *et al.,* 2002). The mixture of facultative dryland and facultative wetland species occurring in the wetlands of the TMA, had a considerably higher presence of herbs being classified as facultative dryland species and was also reported by Flinn *et al.* (2008). As wetlands become drier towards the upslope terrestrial areas of the wetland, seasonally to temporarily saturated soils are found here and are typically dominated by more facultative wetland species .Although a high dominance of facultative wetland species may indicate that a wetland is possibly present, the occurrence of facultative species is *per se* not diagnostic of the presence of wetlands since these species can also occur in terrestrial environments and that other wetland characteristics (i.e. soils and geomorphology) should also be used to positively identify the presence of a wetland.

Wherever functionally similar species (*Cyperus longus, C. fastigiatus, C. laevigatus, Hemarthria altissima, Typha capensis* and *Carex glomerabilis*) occur together in a wetland, such as in both urban wetlands, these species compete with each other for resources, but where a single species (i.e. *Typha capensis* or *Cyperus longus*) outcompetes all other species, a monoculture is formed. This was also found in the study of Sieben (2012) who studied the Wakkerstroom wetland in Mpumalanga. Here four functionally similar species (*Typha capensis, Carex acutiformis, Pycreus nitidus*, and *Cyperus fastigiatus*) competed with each other, as in the case of the current study. Sieben (2012) explained that this "demonstrates niche separation between differing functional types with functionally similar species inhabiting the same niche". This means that competition between species that are "functionally similar is more likely to drive out one of the competitors, whereas co-existence is more likely among functionally dissimilar species and would allow for a more species rich community" (Pugnaire and Valladares, 2007).

Vegetation patterns like this likely reflect gradients of light, soil nutrients and moisture availability (Flinn *et al.,* 2008). At the same time, wetlands in the TMA provide habitat for obligate wetland plants (*Carex acutiformis* and *C. glomerabilis*) that would not otherwise occur in the surrounding grassland vegetation. These obligate wetland plants occurred in both urban wetlands and in some rural wetlands in this study (Figure 4.17), indicating that urban wetlands have larger saturated areas in which these obligate wetlands survive, whereas some rural wetlands have temporary dry periods. According to Van Ginkel *et al.* (2011) obligate wetland species spend "most of their life cycle inundated (either submerged or emergent), in hydromorphic soils, and can endure anaerobic soil conditions".

The vegetation of the wetlands in this study thus combined species from the surrounding grassland/dryland matrix with species typical of wetland types in the larger region such as *Cyperus longus, Hemarthria altissima* and *Falckia oblonga* (Mucina and Rutherford, 2006). The presence of a wide range of facultative species (wetland or dryland species) highlights the supporting role they could potentially play in the population dynamics and ecosystem functioning within the different wetland sites (Nicolet *et al.,* 2004).

4.4.2.4 Wetland condition determination based on plant species composition and diversity

The degree of wetland specific vegetation present, confirms that all of the wetland sites studied are considered true wetlands, since 51% of the species (52 species) found in this study were hydric species occurring at least 50% of the time or more (facultative, facultative wetland and obligate wetland species) in wetlands (Figure 4.21).

Rural wetlands 10 and 14 had the lowest WIVs. The low WIV of rural wetland 10 and 14 (WIV of 0.8 and 0.9, respectively) could be ascribed to the higher occurrence of obligate and facultative wetland species than most of the other wetlands (Figure 4.18). Wetland 10 and 14 were both considered to have a low gamma diversity (wetland 10 had 22 species, whilst wetland 14 had only 17 species) and alpha diversity, and ultimately the high dominance of a these few species and the type of hydric status of the species determined the low WIV of these wetlands.

The low WIV for wetland 10 is due to the high frequency of indigenous obligate wetland species such as *Pycreus macranthus* (12% frequency) and *Cyperus longus* (19% frequency). For wetland 14, the low WIV is due to the high frequency of indigenous obligate wetland species such as *Paspalum distichum* (21% frequency), *Leersia hexandra* (2% frequency) and facultative wetland species, *Hemarthria altissima* (5% frequency).

Similar results were also found in the study of rural wetlands in the specialist wetland assessment report of Van Deventer *et al*. (2013). In the study of Van Deventer *et al*. (2013), rural wetlands with neighbouring agricultural fields, were assessed for a proposed bulk water pipeline development between Lions River and Howick, and were found to be a true wetland (WIV = 1.99; 1.12), dominated by *Cyperus* spp. and *Leersia hexandra.* Although the wetlands studied by Van Deventer *et al.* (2013) experienced more intense disturbances (artificial draining, headcuts and being infilled) than the wetlands investigated in this study, the impacts of these activities created different habitat types for different species to thrive in. Therefore, it may also be that wetlands 10 and 14 promoted the development of microhabitats which favour the establishment of hydric species.

Even though rural wetlands 5 and 6 are still considered to be true wetlands, they have the largest WIV of all the wetlands studied (Figure 4.21). This is mostly due to the abundance of facultative wetland species (*Echinochloa holubii*) and the lower abundances of obligate wetland species (*Cyperus fastigiatus*). Wetland 6 is constantly being traversed and indiscriminately driven through by agricultural machinery, since a cultivated field is directly adjacent to it. Some infilling of the boundary of the wetland was also visible, influencing the soils capability to retain water and thus develop hydromorphic soils. Even though facultative species were present in this wetland, facultative dryland species were more abundant. The lower abundances of hydric species at wetland 6 could be the result of the drier and sandier soils found in patches of bare areas in the wetland, most likely due to grazing and trampling of cattle, and the presence of more obligate dryland species which could establish in these drier areas (Figure 4.18). There could also possibly be a wetness gradient present, where these facultative dryland species occur (closer to the adjacent agricultural fields), with the facultative wetland species occurring in wetter areas (closer to the Mooi River) which has consequently been affected by the aforementioned impacts. This response of vegetation to a wetness gradient within a wetland was also explained by Cowden and Kotze (2009). Although the case study presented by Cowden and Kotze (2009) examined the effects of vegetation in rehabilitated wetlands, the same principle of the wetness gradient applies to the establishment of vegetation within a pre-rehabilitated wetland, where more hydric vegetation will establish in wetter areas than in the drier areas. It should be noted that the drier areas found within wetland 6 was still within the boundaries of the delineated wetland. Another possible reason why (obligate/facultative) dryland species were more dominant in, especially the rural wetlands, is possibly due to the high water extraction rate from downstream of the Mooi River, and also due to the upstream dams (Potchefstroom dam – located upstream of wetland 1). Extraction of water from the river have created less frequent flooding events, diverting from the normal wetting regime of the wetland areas along the Mooi River. Kingsford (2000) also reiterated the effect of reduced flooding of rivers due to the diversions of rivers through the building of dams and the extraction of water. Kingsford (2000) found that wetland areas (especially floodplains) have turned into more terrestrial like ecosystems. This might also be the case for the dryland species dominated wetlands of the TMA, however, as mentioned previously, it is not only the plant species which determine the classification of a wetland, but the soil conditions (even though with a less frequent wetting regime) and topography of the wetland site should also be taken into account.

Cowden *et al*. (2014) also found rural wetlands, with neighbouring agricultural activities, to have more or less the same WIV as the rural wetlands of this current study. Cowden assessed the response of two wetlands (the Killarney and Kruisfontein wetlands) located in KwaZulu-Natal, to rehabilitation activities. The WIV of the Killarney wetland (prior to rehabilitation, since it could be considered the same as the current state as the rural wetlands in the TMA) averaged 2.1 (close to the values of wetlands 5 and 6).

Cowden *et al.* (2014) prescribed the high WIV value to the degraded seedbank in the wetland, due to prolonged influences form agricultural activities. The intensity and duration of such activities can deplete the seed bank of hydric species and replace it with crop seeds or allow alien species to establish. However, the intensity and duration of impacts alone cannot be used to explain why the WIV scores differ between urban and rural wetlands, as each has received different anthropogenic impacts at different intensities since the establishment of Potchefstroom over a long period of time.

Although the overall abundance of alien species within the wetlands of the TMA is considered low (only 33% of all species recorded were alien, Table 4.7), relative high abundances of the alien facultative dryland species, *Alternanthera sessilis,* and alien facultative wetland species, *Rumex crispus*, are present in wetlands 5 and 6, and could thus also have taken over the available habitat of indigenous facultative wetland species, as *Paspalum dilatatum* did in the Kruisfontein wetland surveyed by Cowden *et al.* (2014). Louw (1951) also noted that the mat-forming *Alternanthera sessilis* were very common in disturbed wetlands, and it is more likely that these alien species and facultative dryland species would establish faster than the facultative/facultative wetland species.

The quality of the overall wetland habitat of the TMA could be considered degraded, since the highest FQAI score was 49.2 (urban wetland 1) and the lowest 21.3 (rural wetland 5). Rural wetlands 8, 10 and 11 and the two urban wetlands are considered to have similar FQAI (FQAI all above 35), whilst the rest of the rural groupings all had FQAI scores of below 28.

FQAI values for rural wetlands 5, 6 and 7 are the lowest for all the wetlands. The low FQAI can be attributed to the high species richness of species with low coefficients of conservatism (species such as *Setaria pallide-fusca, Falckia oblonga* and *Brachiaria eruciformis*). Louw (1951) stated that *Setaria pallide-fusca* sometimes becomes weedy in areas of historical cultivation, which is another indication of lowered floristic quality. The relative high FQAI of wetland 10 (FQAI = 36.2), is due not only to the high abundance of the indigenous obligate wetland species, but also due to their high species richness and coefficient of conservatism. The same is not necessarily true for wetland 14. The FQAI for wetland 14 is 24.4, which is regarded as low when compared to the other wetlands FQAI. However, even though wetland 14 has a high abundance of indigenous obligate wetland species, the species richness of species with lower ranked coefficients of conservatism is higher (such as *Brachiaria eruciformis*).

Urban wetland 1 had the highest FQAI (FQAI = 49.2) which could be explained by the high species richness of *Cyperus laevigatus, C. longus* and *Carex glomerabilis*, all of which had the highest value coefficient of conservatism allocated to them. Although urban wetland 2 also has a high FQAI (FQAI = 39.3), it is considerably lower than urban wetland 1. The difference in the FQAI of these two wetlands could be ascribed to the overall lower indigenous species richness of wetland 2, which also had species such as *C. longus* and *Leersia hexandra* that enjoyed the highest species richness in wetland 2 and had the highest coefficient of conservatism values. Despite the difference in FQAI values, based upon their indigenous species, urban wetlands 1 and 2 still grouped together based on the abundance (percentage cover) of the indigenous species (Figure 4.10). Similarly, wetlands 8, 10 and 11 are also located within the same area on the NMDS ordination of Figure 4.4, implying that the species richness of the indigenous species such as *C. longus, C. fastigiatus* and *Hemarthria altissima,* groups them together.

When the FQAI values of the wetlands of the TMA were compared to those of the wetlands assessed by Van Deventer *et al.* (2013), the results are quite different. Some wetlands have much higher FQAI scores (FQAI = 77.9; 76) than those of the current study. This could again be explained by the microhabitats that were formed, but in the case of the rural wetlands assessed by Van Deventer *et al.* (2013), the wetlands had more indigenous species that established within the microhabitats created by the disturbances. Whereas in the wetlands of the current study, disturbances created habitats where species with lower coefficients of conservatism (such as *Brachiaria eruciformis, Falkia oblonga, Marsilea capensis*) were established, or the type of disturbances to the wetlands in the TMA had a more profound effect than those impacts in the study of Van Deventer *et al.* (2013).

The FQAI values from this current study are comparable to the FQAI for the rural Kruisfontein wetland assessed by Cowden *et al.* (2014). The Kruisfontein wetland was divided into five areas to which different rehabilitation techniques were applied. The pre-rehabilitation FQAI of the area named 'untransformed' had a FQAI of 46.38, comparable with the FQAI of urban wetlands 1 and 2. A further similarity, is the high abundance of the indigenous obligate wetland species *Carex glomerabilis*. Although no comparison could be made regarding the extent of disturbance between the rural Kruisfontein wetland and the urban wetlands of the TMA, disturbance ultimately does lower the FQAI, since the 'untransformed' area has been previously subjected to disturbances before assessment by Cowden *et al.* (2014).

The study of Miller and Wardrop (2006) examined a variety of wetlands from a range of different disturbances (such as agriculture, mining, and development) and then were ranked as low, medium or high, and in either a rural or urban wetland according to O'Connell *et al*. (2000). When the urban wetlands 1 and 2 are compared to the wetlands located within or close to urban areas of the study of Miller and Wardrop (2006), the wetlands in the TMA are considered to be of better quality, since the Miller and Wardrop (2006) urban wetlands FQAI were all below 19. The rural wetlands studied by Miller and Wardrop (2006) had relatively the same FQAI (FQAI between 19 and 30) as the rural wetlands found in the TMA (FQAI between 21 and 40).

Overall, many factors could have caused the low FQAI determined for the wetlands of the TMA. Most likely being the general transformation of the surrounding areas. FQAI has been shown to respond to fragmentation (Fennessy, 2002) and that environmental factors such as wetland soil moisture, soil nutrients, and flow patterns could change as a result of this fragmentation. It is important to consider the functional traits of vegetation when interpreting the FQAI for an area. In the wetlands of the TMA, all wetlands were dominated by graminoid and herb growth forms (Figure 4.15), which is characteristic of the grassland area these wetlands are situated in (Mucina and Rutherford, 2006). The TMA wetlands were all dominated by facultative dryland species, rather than facultative, facultative wetland, and obligate wetland species (Figure 4.18), indicating that more opportunistic species, most likely adapted to changed hydrology, drier soils or enriched conditions (caused by disturbances to the wetlands) (Ravit *et al.,* 2006) have established within the wetlands. The coefficient of conservatism of these species (such as *Leersia hexandra, Berula erecta*, and *Falkia oblonga*) were amongst the lowest (between 1 and 4), however due to their high abundance, they determined the low FQAI of all the wetland in the TMA.

4.5 Wetland ecosystem services

Determining the ecosystem services (in ecological, social, or economic terms) being delivered by the wetlands of the TMA could be "used as a modifying or motivating determinant in the selection of the management measures for these wetlands" (DWAF, 1999). The assessment of the ecosystem services scored to give the likelihood that the service is being provided. These scores could then be used to ecosystem services assessment method (scoring method) of Kotze *et al.* (2008). The characteristics were used to quantitatively determine the ecosystem services being delivered. Each characteristic was supplied by the identified wetlands was conducted according to the guidelines as described by the estimate the extent of several ecosystem services provided by the wetlands along the urban-rural gradient, and to compare the ecosystem services delivered by the (urban/rural) wetlands in the TMA

4.5.1 Results

4.5.1.1 Urban wetlands

Both of the urban wetland sites (wetlands 1 and 2) have similar scores for the ecosystem services being delivered (Figure 4.23). The two urban wetlands indicated in Figure 4.23, had high scores (score above 2.1) for flood attenuation and erosion control (regulating service) and for cultural services (tourism and recreation and education and research), but low scores for provisioning services (water supply for human use, natural resources and cultivated foods). Although having high scores, the greatest differences between the two urban wetlands is between the indirect ecosystem services, namely between streamflow regulation, sediment trapping and nitrate removal.

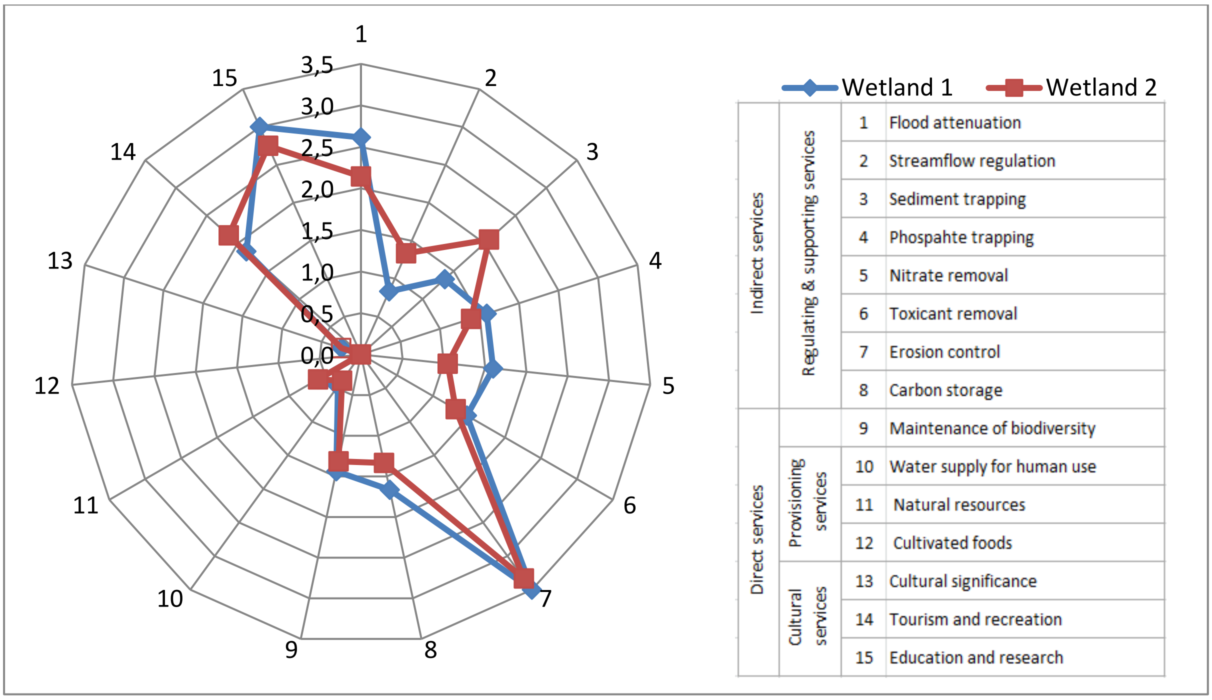
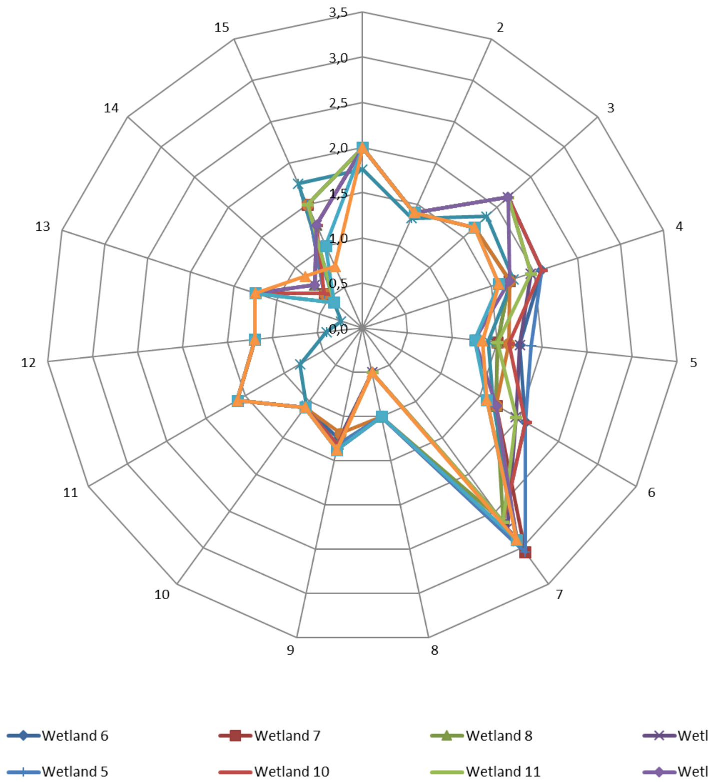


Figure 4.23: Radar graphs showing the scores given to each of the 15 wetland ecosystem services measured in each of the urban wetlands of the TMA.

4.5.1.2 Rural Wetlands

Rural wetlands are divided into two HGM types namely channelled and unchannelled valley bottom wetlands. Most of the wetlands were channelled valley bottom wetlands (wetlands 3, 4, 5, 10, 11, 12, 13 and 14), and only four rural wetlands were identified to be unchannelled valley bottom wetlands (wetlands 6, 7, 8 and 9). As previously mentioned in Section 4.3, no significant differences were evident between the rural channelled and unchannelled valley bottom wetlands, and were therefore grouped together.

All of the rural wetlands have similar scores for most of the ecosystem services being delivered (Figure 4.24). Regulating ecosystem services (flood attenuation, sediment trapping, phosphate trapping, nitrate removal, toxicant removal, and carbon storage) were scored the highest (average score > 2.1). Due to their rural setting, more provisioning services are delivered by these wetlands (water supply for human use, natural resources and cultivated foods) when compared to wetlands in the urban areas. Very low cultural significance is scored for these wetlands, and although they are surrounded by agricultural activities, still has a low value of cultivated foods. Wetland 3 is the only wetland with a lower delivery of natural resources and cultivated foods.



Indirect services	Regulating & supporting services	1	Flood attenuation
		2	Streamflow regulation
		3	Sediment trapping
		4	Phospahte trapping
		5	Nitrate removal
		6	Toxicant removal
		7	Erosion control
		8	Carbon storage
Direct services		9	Maintenance of biodiversity
	Provisioning services	10	Water supply for human use
		11	Natural resources
		12	Cultivated foods
	Cultural services	13	Cultural significance
		14	Tourism and recreation
		15	Education and research

Figure 4.24: Radar graph showing the scores given to each of the 15 wetland ecosystem services measured in each of the rural wetlands of the TMA.

4.5.1.3 Distribution of ecosystem services over the urban-rural gradient

Figures 4.25 to 4.28 show the IDW surface rasters of how the 15 types of ecosystem services delivered by the 14 wetland sites are distributed spatially. This indicates how the urban area of Potchefstroom might impact on the distribution of ecosystem services. Figure 4.25 and Figure 4.26 shows the scores of the different indirect ecosystems services (regulating and supporting ecosystem services) delivered by the wetlands. The IDW rasters in Figure 4.25 show that flow attenuation (Figure 4.25 a) is spatially scored lower the further it is from urban Potchefstroom. The highest scores for sediment trapping (Figure 4.25 c) and phosphate trapping (Figure 4.25 d) were found the furthest away from urban Potchefstroom.

The IDW rasters in Figure 4.26 show the spatial distribution of the remaining indirect ecosystem services (regulating and supporting ecosystem services) delivered by the wetlands. Nitrate (Figure 4.26 a) and toxicant removal (Figure 4.26 b) scored highest with increasing distance from urban Potchefstroom. Erosion control (Figure 4.26 c) and carbon storage (Figure 4.26 d) scored lower the further the wetland sites were located from urban Potchefstroom.

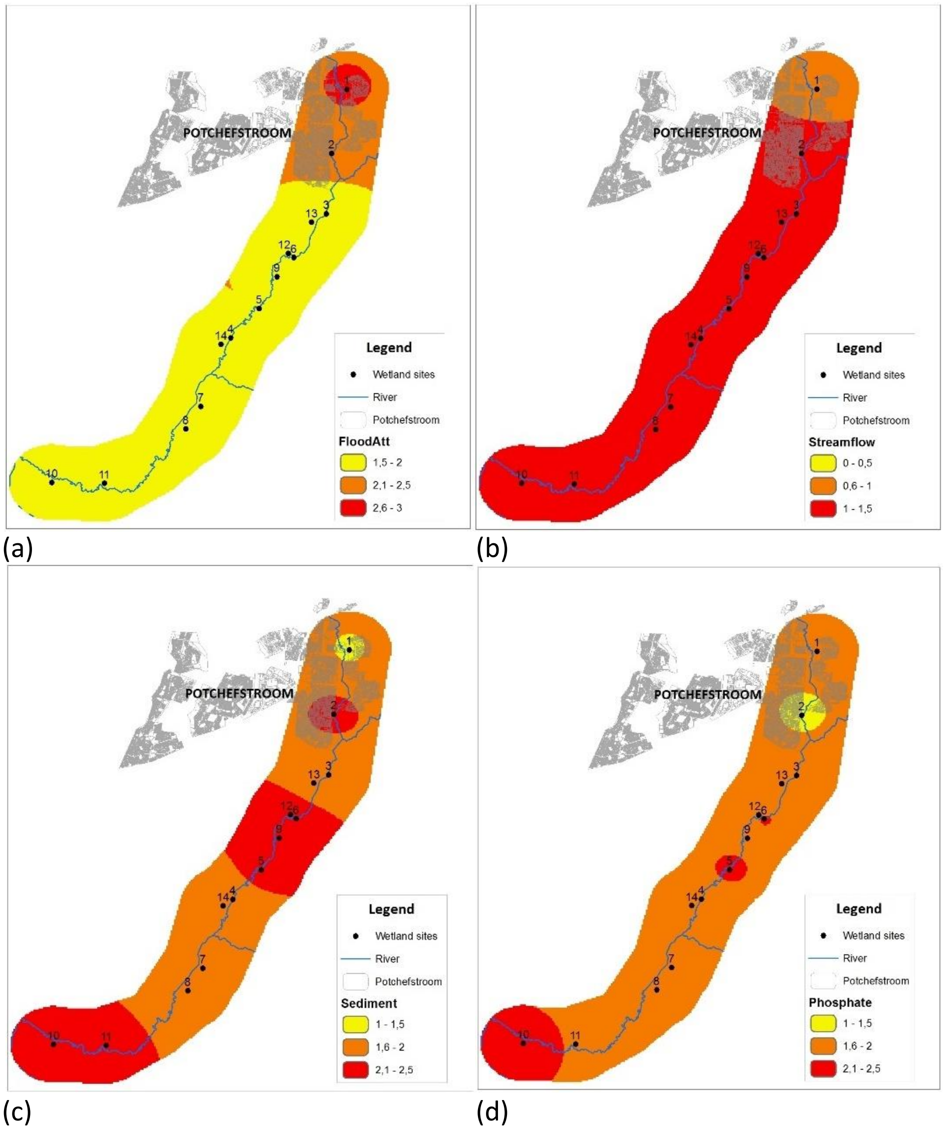


Figure 4.25: IDW surface rasters of the different regulating and supporting ecosystem services being delivered per wetland site in the TMA. These services include (a) flood attenuation, (b) streamflow regulation, (c) sediment trapping and (d) phosphate trapping.

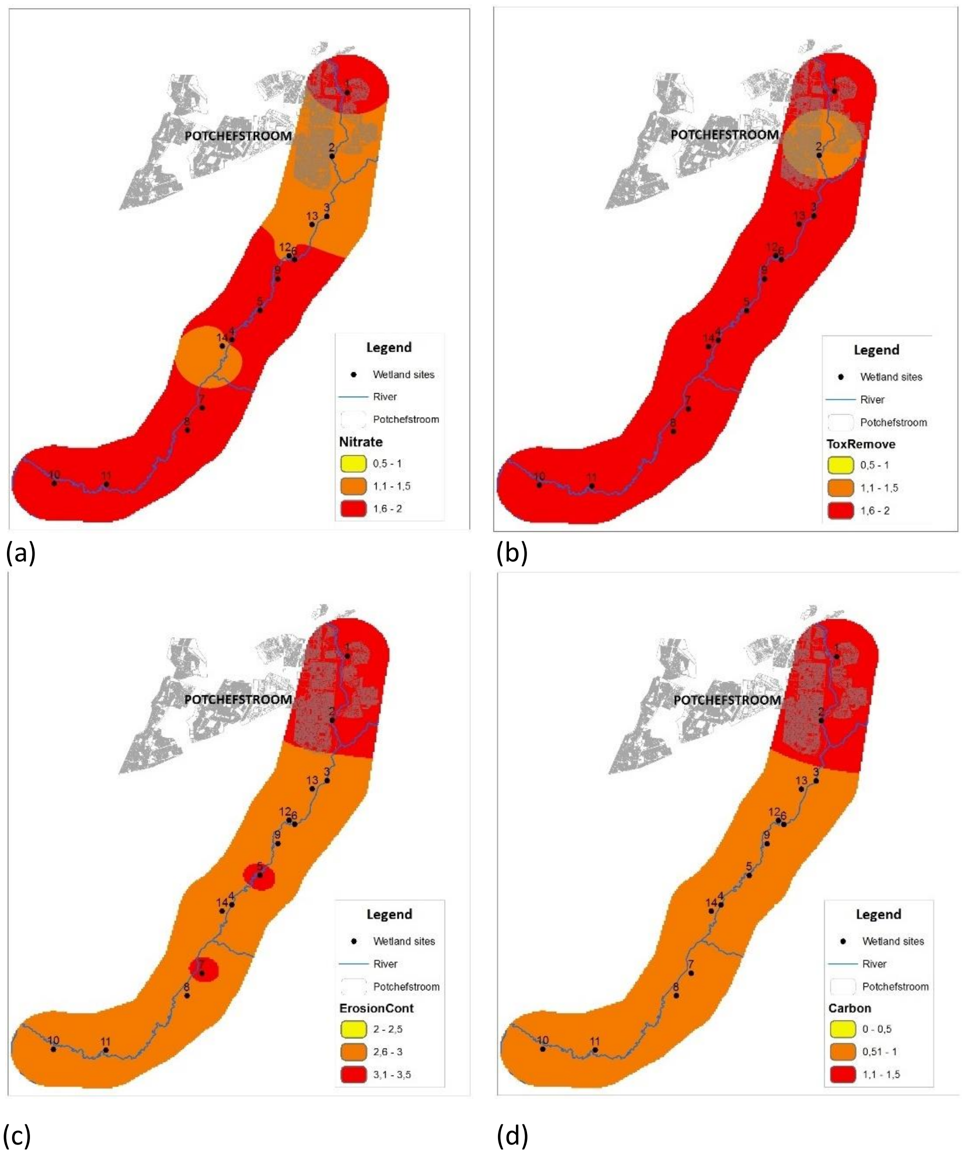
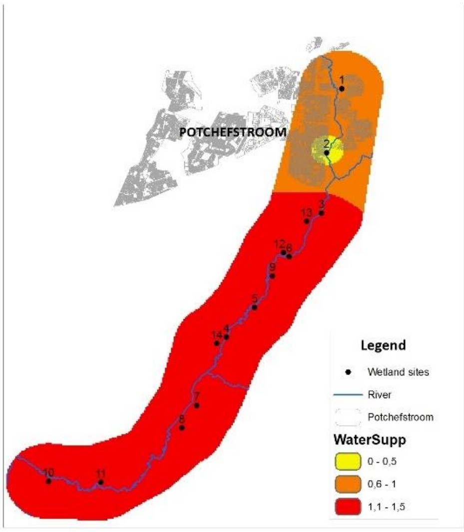
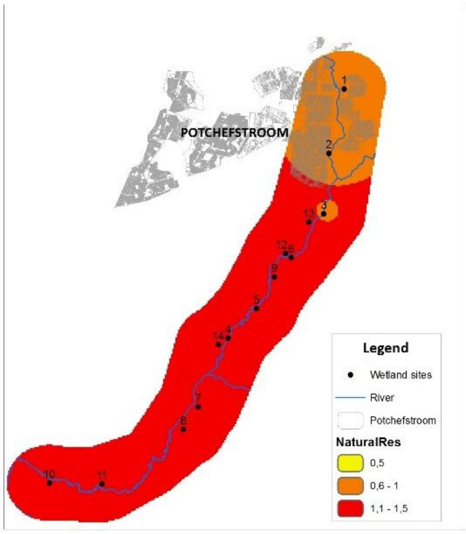


Figure 4.26: IDW surface rasters of the different regulating and supporting ecosystem services being delivered per wetland site in the TMA. These services include (a) nitrate removal, (b) toxicant removal, (c) erosion control and (d) carbon storage.

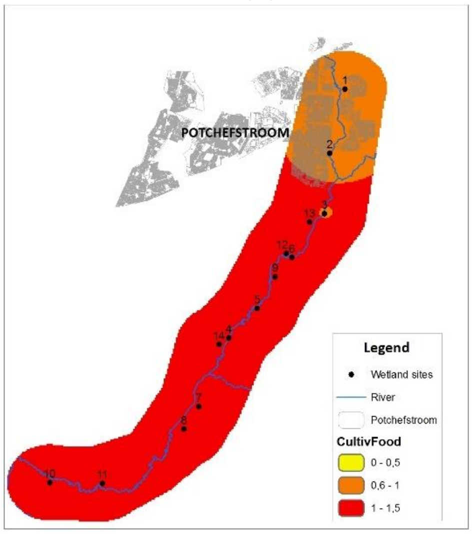
Figure 4.27 shows the spatial distribution of provisioning services. All the provisioning services were scored higher in the wetland sites that are further away from urban Potchefstroom.



(a)



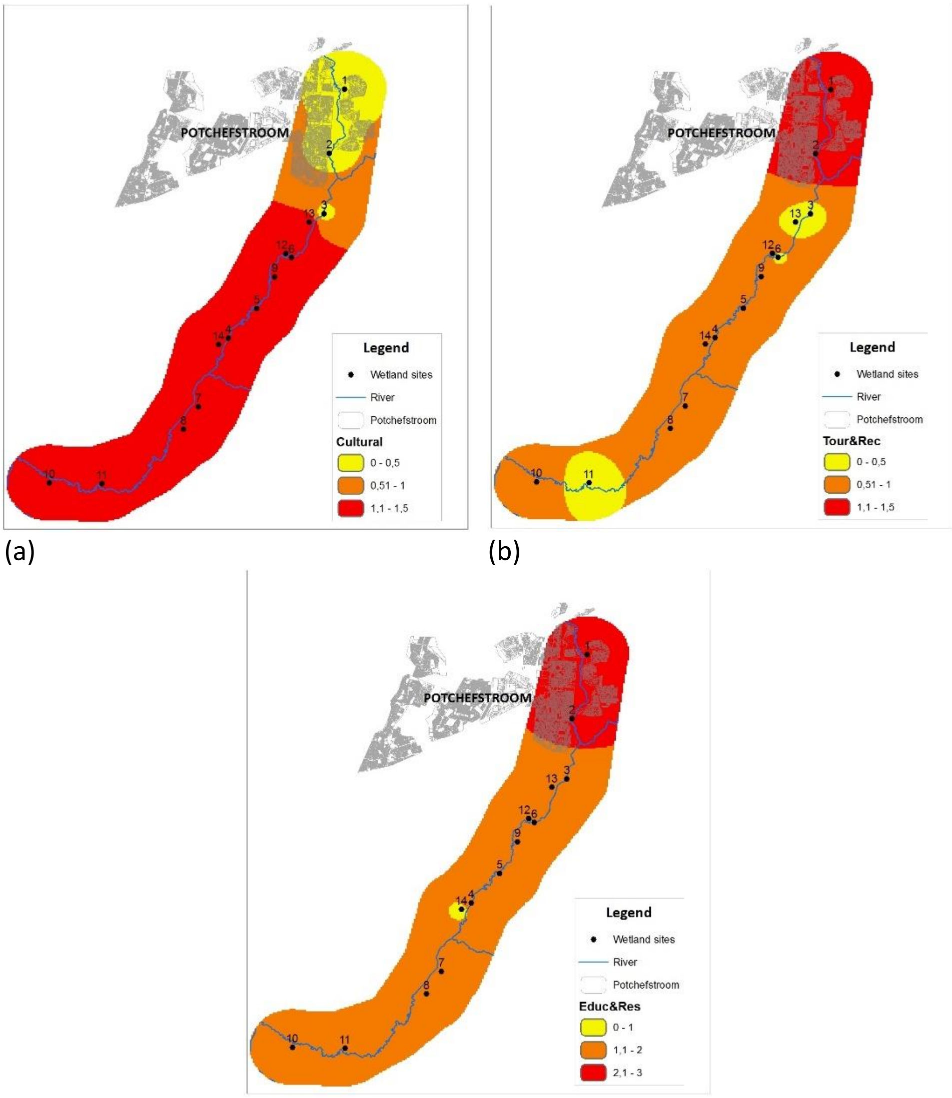
(b)



(c)

Figure 4.27: IDW surface rasters of the different provisioning ecosystem services being delivered per wetland site in the TMA. These services include (a) water supply for human use, (b) natural resources and (c) cultivated foods.

The IDW surface rasters of the cultural ecosystem services being delivered by the urban and rural wetland, are presented in Figure 4.28. Overall, all wetlands have very little cultural significance to the surrounding habitants (Figure 4.28). Urban wetlands 1 and 2 had ecosystem services scoring higher when located closer to urban Potchefstroom.



(a) (b)

(c)

Figure 4.28: IDW surface rasters of the different cultural ecosystem services being delivered per wetland site in the TMA. These services include (a) cultural significance, (b) tourism and recreation and (c) education and research.

Final comparisons between the urban and rural wetlands are indicated in Figures 4.29 and 4.30, and are presented irrespective of their geomorphological setting. From Figure 4.30 it is evident that the urban wetlands scored overall higher for some of the cultural ecosystem services (tourism and recreation and education and research) than rural wetlands, as also indicated in Figure 4.29. The average provisioning ecosystem scores (water supply for human use, natural resources and cultivated foods) of the rural wetlands are higher than the average provisioning scores delivered by urban wetlands (Figure 4.29 and 4.30).

The average scores of most of the regulating and supporting ecosystem services of the rural wetlands are higher than those of the urban wetlands, with the exception of flood attenuation, erosion control and carbon storage (Figure 4.28 and 4.29). The average scores for the maintenance of biodiversity were almost similar for rural and urban wetlands.

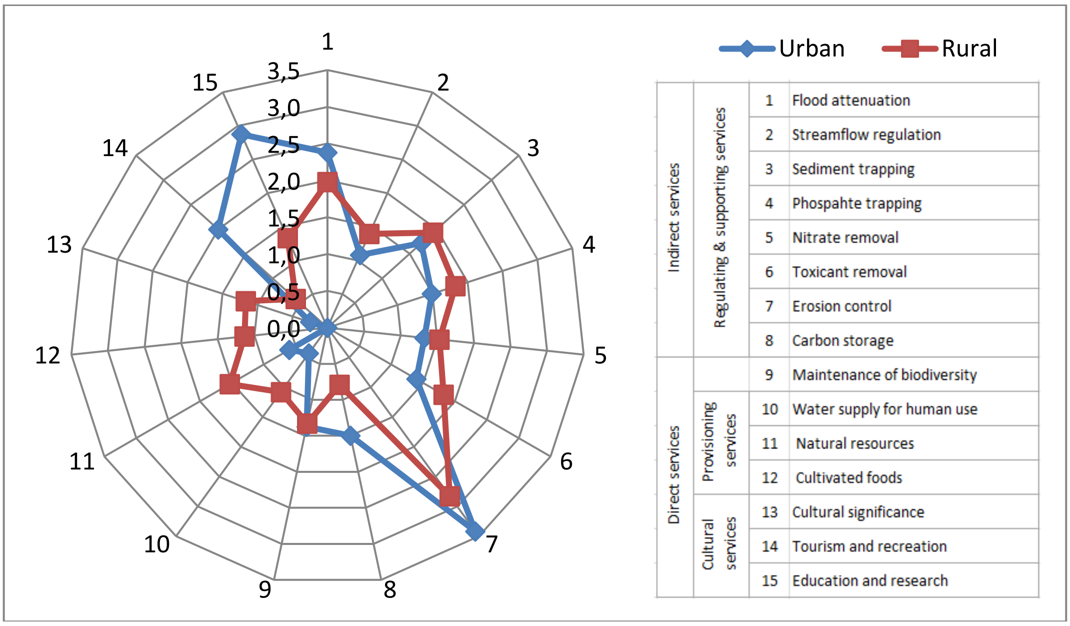


Figure 4.29: Radar graphs showing the average ecosystem scores calculated of the urban and rural wetlands of the TMA, to ultimately compare the ecosystem services delivered by each.

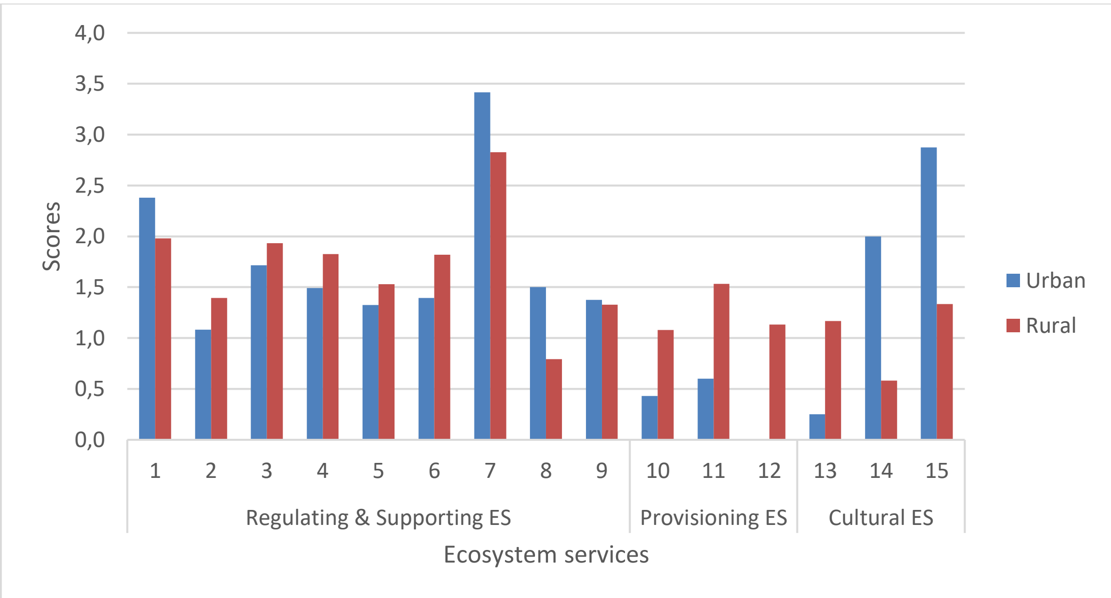


Figure 4.30: Histogram to indicate the average scores of the 15 different ecosystem services delivered by the urban and rural wetlands in the TMA (numbers of ecosystem services explained in Figure 4.29).

4.5.2 Discussion

4.5.2.1 Urban wetlands of the TMA

The flood attenuation regulating ecosystem services (indirect ecosystem services), tourism and recreation, education and research, and cultural services scored the highest in urban wetlands (Figure 4.23). This is also possibly the most impacted on wetlands in terms of the ecosystem services being delivered by these wetlands. Since these wetlands (wetland 1 and 2) are in the closest proximity to human influence, human activities will always first influence the wetlands' characteristics (i.e. wetland vegetation, geomorphology) responsible for delivering regulating ecosystem services. Thus, the regulating services of urban wetlands are being impacted on more than those of the rural wetlands, because of anthropogenic influences (i.e. residential developments surrounding the wetlands). This is also the case for the provisioning services of wetlands 1 and 2, as the use of these wetlands' provisioning services, depending upon an activity, its intensity and the duration thereof, may also contribute to the overall degradation of the urban wetlands (Kotze *et al.,* 2008).

At present, the effectiveness of wetlands 1 and 2 in terms of streamflow regulation (average score of 1,3), has been greatly altered in comparison to the valley bottom wetlands studied by Sinchembe and Ellery (2010) in Grahamstown. The valley bottom wetlands studied by Sinchembe and Ellery (2010) have an adequate amount of vegetation cover to effectively regulate streamflow which allows water to flow across the wetland from the surrounding catchment (Sinchembe and Ellerly, 2010). However, in wetland 1 and 2, the presence of roads adjacent to the wetlands account for a decline in hydrologic health and reduced the effectiveness of trapping sediments.

Possible pollutants (nitrates and phosphates) (Figure 4.23) entering the wetlands from adjacent residential gardens and swimming pools may cause disruption in an ecosystem and can contribute to algal blooms if it occurs in sufficient concentrations. Since phosphates tend to attach to sediments, the greater amount of sediment trapping within these urban wetlands allows for more phosphates to be removed from the system (Fennessy *et al.,* 1994). Nitrates are removed via plant uptake and the high amounts of vegetation cover within these urban wetlands probably allows for more nitrates to be removed within them (Fennessy *et al.,* 1994).

Characteristics contributing to erosion control are; (1) abundance of vegetation cover, (2) surface roughness and (3) physical disturbance (Kotze *et al.,* 2008). These characteristics allow both the urban and rural wetlands to have high scores for erosion control (Figure 4.26 c). As already mentioned, high vegetation cover reduces erosion by reducing water velocity, and the vegetation also bind the soil with their roots (Kotze *et al.,* 2008). In the urban wetlands, trampling by horses (Cilliers *et al.,* 1998) and previous agricultural activities, and in rural wetlands, trampling by livestock, ensure surface roughness which slows the velocity of water flow across the surface of the wetland and prevents soil from being washed away (Kotze *et al.,* 2008). A reduction in water velocity is also, generally, accompanied by an increase in water infiltration.

The close proximity and accessibility of the urban wetlands to urban residents in the city of Potchefstroom, allow them to have high scores for most of the cultural ecosystem services (Figure 4.28), when compared to the low scores of the other wetlands in this study. Recreational activities such as horse riding, walking with dogs and running are enjoyed by local residents. There are very few studies that have investigated the aesthetic, recreational or tourism value of inland wetlands in southern Africa. The studies that have considered urban wetlands, were to determine if the proximity of wetlands close to residential areas has an effect on property values (Van Zyl and Leiman, 2002). According to Bolund and Hunhammer (1999) these types of cultural ecosystem services are perhaps the highest valued ecosystem services in cities, as they contribute largely to the attractiveness of a broader urban area (Turpie and Malan, 2010).

4.5.2.2 Rural wetlands of the TMA

From the results of this study, it was found that overall the rural wetlands had higher average scores for regulating and supporting ecosystem services than those of urban wetlands. Despite small differences in ecosystem service delivery between all the rural wetlands, all received similar scores for ecosystem services than the wetlands in uMdloti catchment, as they are predominantly surrounded by commercial agriculture and near Le Mercy, Umhlanga and Verulam urban areas (Ramburan, 2013).

Another difference between rural and urban wetlands in this study is that these rural wetlands are all in close proximity to agricultural practices. This gives these wetlands the ability to deliver natural resources and cultivated foods to surrounding communities (Figure 4.27 c). These activities also had an influence on the wetlands (i.e. toxicant runoff), but due to the vegetation cover within the wetlands limiting runoff, detrimental effects are lower than they would have been given the lower levels of vegetation cover.

The effectiveness of the rural wetlands to assimilate nitrates, phosphates and toxicants has been reduced through the cultivation practices surrounding these wetlands (Figure 4.25), but is likely to still be relatively effective (Kotze *et al.*, 2008). Kotze *et al.* (2008) suggest that "valley bottom wetlands (channelled and unchannelled) are generally characterised by less active deposition of sediment". This is evident in the wetlands studied by Ramburan (2013), but in the current study sediment trapping scored high. These scores can be attributed to the intact vegetation cover found in the wetlands that remains saturated for prolonged periods diffusing the water flow through the wetlands (Kotze *et al.*, 2008).

In all 12 rural wetlands identified within the current study, very little to no cultural ecosystem services were delivered. Turpie and Malan (2010) stated that outside of urban areas, wetlands should be more attractive, especially for tourism, as it should contain a richer diversity of fauna and flora, as was the case of the Nylsvley floodplain wetland in Limpopo Province. This was not evident in the rural wetlands of the TMA. The main difference between the current studies' rural wetlands and the Nylsvley wetlands situated within a nature reserve, is the relative accessibility of the Nylsvley wetlands. Very low scores of cultural ecosystem services were observed for the rural wetlands in the TMA, because there were few humans in close proximity to gain any direct benefit.

4.6 Conclusions

Based on the plant composition of the wetlands of the TMA, the wetlands had similar plant species composition and hydrogeological characteristics as the Eastern Temperate Freshwater Wetlands as described by Mucina and Rutherford (2006). The azonal vegetation type found in the study area is visually different from the surrounding grassland, and can be used to successfully identify and delineate wetlands. However, it was noticed that overall the diversity of species declined from the species noted in the study of Louw (1951), over 60 years prior.

Urban wetlands had higher gamma and beta diversity than rural wetlands, but lower species richness (alpha diversity) than some of the rural wetlands- accompanied by an unexpectedly higher number of indigenous species. This was not the expected outcome which was predicted by the hypothesis in Section 1.2. The urban wetlands proved to be more diverse, despite the rural wetlands having a higher overall mean species richness. There was also no difference in the vegetation composition of unchannelled and channelled valley bottom wetlands, indicating that the vegetation composition is determined mainly by the surrounding environmental impacts on the wetlands and not by their geomorphological setting. Although it was not the aim of this study to classify plant communities as in Cilliers *et al.* (1998) and Van Wyk *et al.* (2000), it was evident that different plant communities occur in the 14 wetlands studied in the TMA, based on the ordinations and cluster analysis. The main differences between urban and rural wetlands were based on their plant species composition and diversity.

This was evidenced by determining the WIV of each wetland, where the assemblages of hydric vegetation of rural wetlands varied from those of urban wetlands. Overall, lower scores were revealed for the rural wetlands (with the exception of wetlands 5 and 6) indicating that these wetlands have more characteristic wetland vegetation than the urban wetlands. This is based on the abundance of the hydric species which occurred in the wetlands. However, the presence of a larger abundance of facultative dryland species within all of the wetlands and their low ranked coefficients of conservatism has decreased the floristic quality of these wetlands, implying a lower degree of 'natural' wetland condition, if it should be compared to a reference wetland.

Despite the identification of four declared plant invaders in some of the wetlands, the vegetation in the wetlands is considered to be in a relatively good state as less than 1% of the terrestrial cover was occupied by aliens with monocultures absent. The plant species richness was also expected to be lower for the urban wetlands, however the general conclusion can be made that the different impacts that all of these wetlands have already experienced, ultimately determines the presence and abundance of certain plant species which could provide an indication of the wetlands condition.

There is a difference in the delivery of ecosystem services in urban and rural wetlands. Urban wetlands were scored higher for more direct ecosystem services (cultural ecosystem services), that is services such as tourism and opportunities for education and research, whereas rural wetlands were scored higher for indirect ecosystem services, such as sediment and phosphate trapping, nitrate and toxicant removal and erosion control. It was expected, as stated in the hypothesis, that the rural wetlands would have a higher delivery rate of ecosystem services than that or the urban wetlands, however, from all the wetlands, it is evident that the presence and cover of vegetation can be considered a major driving force for delivering all other ecosystem services.

With regard to the ecosystem services being delivered by the wetlands of the TMA, it is clear that it is difficult to assign a score to an ecosystem service as we cannot encompass the true significance thereof with a single number. However, the WET-EcoServices technique did aid in this process and delivered potential scores for the delivery of ecosystem services amongst the different wetlands. This technique also allows for the comparison between wetlands in different urban and rural settings, and also to indicate the spatial distribution of these services.

The first part of the hypothesis in Section 1.2 could be rejected as it was clear that the urban wetlands had a higher total vegetation cover and plant species richness than rural wetlands, as well as a different species composition. The second part of the hypothesis was difficult to test as urban wetlands scored higher in certain ecosystem services than the rural wetlands. However, as stated above, the difficulty to determine the delivery of ecosystem services could not be encompassed, as certain ecosystem services had higher scores in rural wetlands than it was scored in urban wetlands, and vice versa. Therefore, the importance of which ecosystem services dominated in order to ultimately decide which wetlands (urban or rural) delivers more ecosystem services, could not be made.

5 CONCLUDING REMARKS: CONSERVATION AND MANAGEMENT OF WETLANDS ALONG AN URBANISATION GRADIENT

5.1 Introduction

The identification of ecosystem services being delivered by natural areas has become “a policy tool to protect biodiversity” (CBD, 2013). Since scientific consensus about the mutual relationship between ecosystem services and biodiversity has not been well established, this policy tool was instated as part of the global strategic plan (2011−2020) of the Convention on Biological Diversity (Aichi biodiversity targets) (Liquete *et al.,* 2016), to aid in the conservation of natural areas where there is a lack of legislation to protect the natural resources.

The importance of evaluating plant biodiversity and condition, and ecosystem services are being promoted by both science and legislation policies by means of ecosystem assessments, or environmental impact assessments (Mace *et al*., 2012). Biodiversity assessments are intended to provide guidance for ecosystem management and the realisation of estimating the importance of the function of various vegetation community assemblages.

Determining the plant species composition, species and functional diversity and condition (Section 4.4) and assessing the ecosystem services (Section 4.3) of urban and rural wetlands of the TMA, formed the main themes of this dissertation. To be able to address the main themes, the urbanisation gradient was quantified (Section 4.2) and the wetlands were classified (Section 4.3). Concluding remarks on section 4.4 and 4.5 are given in this chapter, as well as a discussion on wetland conservation and management strategies being applied in a South African context. This chapter will conclude with recommendations for future wetland research in and around urban areas.

5.2 Plant species composition and diversity of the wetlands in the TMA

Cilliers *et al*. (1998) were first to study the wetlands within an urban context in the TMA. From this study, they found a lack of knowledge of the wetlands in this area, especially in terms of their current floristic condition and composition. Following extensive field vegetation assessments of the biodiversity within the selected wetland sites, it was found that some of the urban and rural wetlands did not differ from each other in terms of their plant species composition. Urban wetlands were also found to have higher gamma and beta diversity (Figure 4.9 and Table 4.6) and have a higher overall number of indigenous species (Figure 4.13), which was not expected as previous studies found that urban areas are more susceptible to alien plant invasions (Alston and Richardson, 2006). The rural wetlands were found to have an overall higher species richness than the urban wetlands, indicating that despite the environment surrounding the rural wetlands being transformed (from natural areas to agricultural fields), the state of these rural wetlands could be considered intact to some degree. Some of the species assemblages found in the study of Cilliers *et a*l. (1998) and Louw (1951) could still be identified in the current study but not necessarily with the same abundance. Only remnants of identified plant communities in the study of Cilliers *et al*. (1998) persists in the urban wetlands (i.e. clusters of *Typha* species).

Little to no tree and shrub species have invaded the wetlands from the adjacent riparian vegetation and herbaceous and graminoid growth forms dominate the wetlands. Despite these fourteen study sites being identified and confirmed as true wetlands (all sites had WIVs below 2.5; Figure 4.19), a higher frequency of facultative and facultative wetland species was recorded, with a lower frequency of obligate wetland species. Some of these facultative dryland species identified (i.e. *Eragrostis* spp.), indicates a transition zone between dry grassland surroundings (uplands) and the more saturated soils in the central zone of the wetland where facultative wetland and obligate wetland species persist. This invasion of dryland species into wetlands is also an indication of the effect that the transformation of surrounding land-use has on the wetlands. Without any historical or current anthropogenic influences, it is most likely that the wetland sites would have had high frequencies of typical wetland species (thus have lower WIV and higher FQAI), but land transformations caused the invasion of dryland species into wetlands (Zedler and Kercher, 2004), resulting in a mixture of facultative dryland and facultative wetland species being found in all of the urban and rural wetland sites. The condition of the overall wetland habitats would also have been expected to be higher, if not for the urbanisation activities surrounding the urban wetlands, and the agricultural activities surrounding the rural wetlands.

5.3 Ecosystem services of the wetlands in the TMA

Very little published literature has focussed on evaluating wetlands based on the ecosystem services they deliver using the WET-EcoServices method (Kotze *et al.,* 2008). Most environmental practitioners used the WET-EcoServices manual to describe the functionality of wetlands to ultimately aid in compiling environmental reports for mining industries or infrastructure development. Even though the manual was created for industry professionals, it is still simple and accurate enough to be used by anyone with little experience of wetland assessments. Thus, it was considered sufficient enough to be used as a method for this study as it delivered an adequate evaluation of the ecosystem services delivered by the wetlands in the TMA. Even though this is a subjective method of assessing ecosystem services, it is currently the advocated method of evaluation of wetland ecosystem services in South Africa by the Department of Water and Sanitation (DWS).

Differences in the delivery of ecosystem services by urban and rural wetlands were noticeable. It was found that urban wetlands scored higher for their direct ecosystem services (regulating ecosystem services), whereas the scores of the rural wetlands indirect ecosystem services (provisioning services) were negligibly different. From all the wetlands, it is evident that the presence and cover of vegetation could be considered the major driving force for delivering all ecosystem services, whereas their hydrogeomorphic setting (i.e., HGM unit) has a negligible effect on the delivery thereof.

5.4 Wetland biodiversity and the delivery of ecosystem services in the TMA

It is important to know how biodiversity fits into the concept and practical application of ecosystem services, their values, and what it means for biodiversity, ecosystem science and the conservation and management thereof (Mace *et al*., 2012). Knowing the factors which influence the provision of ecosystem services in wetlands, these can then be protected and managed to be more sufficient and sustainable. Although determining the species composition and functional attributes of wetland vegetation will provide a good estimation of the condition of a wetland, it can be concluded that the overall presence of vegetation and the amount of cover in wetlands is a much more important factor to consider (rather than the HGM setting or surrounding topography of the wetlands) when evaluating the ecosystem services being delivered, as plants are the primary producers in most ecosystems (Quijas *et al.,* 2010). Hook (2003) also obtained results implicating the value of vegetation cover (independent of the type of vegetation present) in delivering greater ecosystem services and noted that dense vegetation of moist and wet riparian sites generally produced more ecosystem services (more specifically, retained sediment effectively). In any type of ecosystem, Loreau (2001) states that the more primary producers (vegetation) there are in the ecosystem, the greater the delivery of ecosystem services will be.

Maintenance of all other biodiversity is often most influenced by the vegetation present within the wetland, as this type of vegetation (facultative wetland and facultative dryland, which had higher coverage than obligate wetland species in the case of this current study) provides a habitat as well as resources for survival. The ability of wetlands to sustain biodiversity is negatively influenced by the removal of vegetation for cultivation (rural wetlands of this study), infrastructure (urban wetlands of this study), or through the flooding of dams and other transformations. In both the urban and rural wetlands in the TMA, a low degree of biodiversity maintenance was evident, since some transformation of the wetland and surrounding areas (in both urban and rural wetlands) has decreased the provision of habitat and maintenance of natural processes of the wetlands (Kotze *et al.,* 2008). Urban wetlands are constantly influenced by the surrounding road infrastructure and inflow of low quality water from upland residential areas (areas from where water would flow from) through drainage outlets into the wetland (wetland 2), whereas rural wetlands' size are reduced to accompany larger agricultural fields, reducing their potential for increasing biodiversity.

There are also benefits to a diversity of primary producers. However, considering the diversity of species sampled by Louw (1951), currently the TMA has a relatively low diversity of plant species. "A higher diversity of primary producers contributes to a higher diversity of primary consumers, which is a supporting service" (Balvanera *et al.,* 2006). This allows for a more biodiverse habitat, which increases the species richness of fauna and flora in wetlands. A strong emphasis is therefore placed on the need to identify and quantify the plant characteristics (such as functional traits) of wetlands that may possibly promote the delivery of ecosystem services (i.e. robust reed species such as *Phragmites australis* and *Typha capensis* that keep soil intact and prevent soil erosion), and to assess service provision relative to the demands of human beneficiaries. Monitoring these changes in traits will provide insight as to how key processes in cycling resources in ecosystems are affected, since these physical vegetative traits also influence most biogeochemical cycling processes (Chapin, 2003).

In terms of the overall delivery of ecosystem services over the urban-rural gradient, cultural benefits to humans are the most relevant service. Cultural ecosystem services are mostly correlated to the vegetation present in the wetlands. The scenic beauty of a wetland site is a key element of its tourism potential, and many wetlands are well recognised for their high aesthetic value (Roggeri, 1995). Wetlands may have high scenic beauty depending on features such as the diversity of colours and textures, contrast with the surrounding landscape, presence of attractive flowers or open water, and absence of litter (Ammann and Lindley-Stone, 1991). Wetlands therefore add aesthetical value to urban areas (wetland 1 and 2), more so if vegetation cover is substantial. The value of residential property in urban areas in close proximity to wetlands is found to be higher than a similar property not located near a wetland (Boyer and Polasky, 2004). Nature appreciation by urban residents often relates to the presence of conspicuous 'charismatic' animal species within wetlands, such as the Marsh owl (*Asio capensis*) occurring in wetland 1.

Understanding how biodiversity, and especially the plant diversity, as in the case of this study, is linked to ecosystem services could aid in the development of more sustainable environmental policies and landscape planning (Balvanera *et al.,* 2015). Understanding this connection will help to improve the quality of life of the surrounding inhabitants. Biodiversity is closely linked to many ecosystem services, but generally in a complex way. The nature and functioning of the relationship between biodiversity and ecosystem services delivery still remain unknown for most ecosystem services. For those that are known, relationships are highly variable. The species present within a wetland and their physical structure may all have an influence on the type of ecosystem services being delivered. For this, the Wet-EcoServices Methodology (however subjective) (Kotze *et al.,* 2008) and other generic methods of assessments internationally suggested (i.e. Diaz *et al.,* 2007) is acceptable, but none of these techniques could explain the extent of the influence of biodiversity on ecosystem services. Further research is needed to develop trait-based approaches and the understanding of the mechanisms of linking biodiversity to ecosystem services to advance our understanding of these interlinkages.

5.5 Preservation of wetlands in the TMA to promote ecosystem services

The implementation of the policies to protect the remaining natural environment (and especially that of wetlands within the TMA) within and surrounding urban areas is frought with challenges. Municipalities all over South Africa are finding it difficult to maintain and provide resources for even the most basic of services and infrastructures within the municipality, let alone provide resources for the protection of the environment.

Within Mandlanzini in Richards Bay (RSA), public participation and non-formal education of local community members, allows for wetland management and conservation to be successfully subsidised in this rural setting. This is because of active community participation to ensure sufficient usage and protection of the wetlands, which are regarded as the only income resource for many in the community (Mthuyane, 2009).

It is difficult to grasp that urban dwellers seem not to recognise their influence on the wetlands within the TMA and that the rural community over utilise the available resources in their quest for survival. These groups are usually not concerned with the conservation of biodiversity or ecosystem services of these wetlands unless they realise their inherent value. Public awareness of the value of natural open spaces (which includes wetlands) within urban and surrounding rural areas need to be raised “in order to develop an appreciation for their role and to develop a conservation ethic amongst the local community” (Mthuyane, 2009).

Implying the economic value of wetlands is an effective approach to create awareness in a world driven by economic growth, surrounding the value of wetlands. However, certain wetland attributes are intangible and it is difficult to assign a monetary value to them. Yet, some studies have already considered this to be a viable option. Faber *et al.* (2002) elucidated the concept of value and the methods of valuation that will assist in guiding decision making based on the calculated value of natural resources.

However, based on the local scale of the TMA, it would be more viable to take a public participatory approach than providing the residents with monetary value data. Du Plessis (2008) explained the value of the linkages between the public participation approach and the fulfilment of environmental rights by the relative government parties. Du Plessis (2008) stated that public participation will inform those most affected, of any planned activities. Local residents could also provide insight to the decision makers regarding the condition of wetlands, and another advantage of a public participatory approach is that communities directly affected by any proposed activities could provide direct feedback to the authorities, which would be a more streamlined way to communicate environmental issues (Du Plessis, 2008). However, the issue (mostly regarding to the environmental authorities) with the involvement of the public, is that they would make the political and administrative decision-makers accountable if the necessary environmentally relevant processes are not reinforced. Thus, a solution to this would be to inform the local residents about all environmental issues, which would include the community in the wider decision-making process, and therefore community ownership of decisions and resultant outcomes would be enhanced (Du Plessis, 2008). The decisions made by the stakeholder and government implementers are more readily accepted by the public participators, if they can see the evidence of their inputs and views being implemented in the environmentally-relevant issues being addressed (Du Plessis, 2008).

With the public being aware of their impacts on wetlands and the processes and implementation of the environmental policy to manage these wetlands, the question arises: What could be done to enhance the ecosystem services of the urban and rural wetlands in the TMA, to limit the deterioration thereof? Additionally, what could those most reliant on these ecosystem services do to ensure the enhancement thereof? Unfortunately, no current step-by-step procedure exists as to provide protection of these services, since all wetlands are unique in their functioning, and no universal method could be implemented to guarantee successful conservation. However, some basic approaches are recommended to be implemented on a local scale, which could preserve what is present. These includes:

- Baseline data on the plant species composition, species and functional diversity, condition and potential ecosystem services provided by the wetlands should be collected, as in this current study. This will allow for long-term monitoring to evaluate the condition of the vegetation within the wetlands, in order to determine the ecosystem services being delivered by the wetlands;
- Restoration, conservation and management actions should be implemented based on site-specific physical features, management and landscape context, aesthetic value, and political and economic influences. For example, actions to enhance the urban wetland ecosystem services should probably not exclude the presence of horses (within wetland 1 and 2), as this already provides recreational value (cultural ecosystem services), but managing the size of the herd of horses and the areas being grazed or trampled, and their resultant effect on the wetlands, could be decreased. The study of Tesauro (2001) examined this principle by using livestock (cows, goats and sheep) to restore wetland habitats. The main factor of using grazing to the advantage of the wetland (for restoration purposes), is controlling the density of the livestock herd and also taking into account the size of the livestock and available habitat (Tesauro, 2001). However, in the report of Reeves and Campion (2004), who examined the effects of livestock grazing on wetlands in New Zealand, found that the effects of grazing are inconsistent due to the variability of grazing effects and the purpose of grazing in wetlands. It is therefore recommended that for grazing to work for conservation or management objectives, grazing should be carried out on a site-specific basis based on observations and collected data (Reeves and Campion, 2004);
- With specific relevance to the rural wetlands, no indiscriminate transformation of the wetland buffer zone should be further converted to agricultural fields. This should be controlled by the relevant authorities under the Conservation of Agricultural Resources Act (CARA) (Act 43 of 1967). The intention of this Act is to limit any intersection of agricultural activities within freshwater resources (wetlands and rivers), and to promote the conservation thereof. Thus, this would entail authorization for a range of impacts associated with cultivation of wetland areas. If unauthorized activities do commence, a reduction in the size of the wetlands surrounding the Mooi River could ultimately influence the channel competency of the river, influence the sedimentation of the remaining wetland, and decrease habitat for flora and fauna, ultimately decreasing the available entities which deliver the regulating ecosystem services (MacFarlane *et al.*, 2009);
- Conservation of individual species should not be the only focus of wetland management or conservation. Rather, the conservation of umbrella species (species whose protection serve

to protect many co-occurring species (Fleishman *et al*., 2000)) and their habitat, within the urban and rural setting, should be emphasised as conservation goals, since this will conserve not only all species but also the ecological processes of the entire wetland ecosystem. Fleishman *et al.* (2000) proved that using umbrella species (which could be fauna or flora) in the Great Basin (USA), is a useful tool in meeting conservation challenges as it could be used to prioritize habitat remnants for conservation or other land uses. In the case of the TMA, possible umbrella species could be a variety of avifaunal species (including the Marsh owl (*Asio capensis*)) since an existing bird sanctuary (Wetland 2 – Prozesky Bird Sanctuary) is located within the TMA;

- Within these wetlands, regular and consistent monitoring to determine the presence of invasive alien species should occur. Even though no large abundances of invasive species were present within the wetlands of the TMA, awareness of their abundance and regular maintenance would limit the spread of opportunistic alien species during their initial growth stages. If such species invasion has been noticed, it should be reported to the relevant authorities and the correct procedure for alien plant species control should be implemented. Campbell (2000) provides a protocol for establishing an Alien and Invasive Plant (AIP) species control program, in which basic principles (regarding removal, control of regrowth, and maintenance and aftercare) for the eradication and control of such species are provided; and
- Surrounding communities should report any misuse or municipal defects (inflow of sewage, stormwater system blockages) to the relevant municipal department, to create awareness of any anthropogenic influences which could be detrimental to wetland environments.

5.6 Recommendations for future research

It is recommended that further studies on urban and rural wetlands and their potential ecosystem service delivery, especially in developing areas, and a full comprehension of the value of wetlands, should be prioritized. Also, it is recommended that wetlands should be conserved so as to limit any further loss of the delivery of ecosystem services on a local scale, but also that best practice management be implemented by the local municipalities and surrounding urban residences or farm owners. Studying the various aspects of wetlands can provide advances in the information on the socio-cultural and economic benefits of ecosystem services. "These cultural ecosystem services demonstrate the influence of wetlands on the local and national economy (and thus aid in building local and political support for their conservation and sustainable use of wetlands)" (WWF, 2016). It can also aid in convincing decision-makers in the local municipality, of the benefits of conservation and sustainable use of wetlands (WWF, 2016). In addition to raising awareness about the benefits of wetlands in decision-making, assessing wetlands may help to improve how local institutions (i.e municipalities) manage resources and identify better markets and resource management for wetlands and their possible products.

Future studies could benefit from including additional urban wetland sites to verify whether the results of this study are also similar in other urban wetlands in the North West Provinces. The surveying of the entire wetland site (not just the permanent and temporary wetland zone, but also surrounding upland areas) could also yield additional information on the structures affecting the delivery of ecosystem services within the wetlands.

Although the IDW rasters proved sufficient in displaying the distribution of certain measured attributes over the urban-rural gradient in this study, finer scale rasters of each patch visualising the entire wetland area and the specific measures attributed at different points in the patch could also potentially lead to a better understanding of plant species patterns and the processes influencing these patterns.

Other functional traits should also be considered in determining the functional diversity of the vegetation in the wetland sites. Plant height and leaf traits (such as dry leaf matter content and leaf nitrogen or phosphorus concentration) are considered by Lavorel *et al.* (2011) to be strongly influenced by the surrounding land-use and abiotic environment.

Using coefficients of conservatism to determine the FQAI is considered to be subjective, and therefore more local knowledge of the tendency of species to occur and survive within different environments should be a factor to consider when assigning a coefficient to a species.

Evaluating the ecosystem services of the wetlands in this study using the WET EcoService approach was highly subjective. During evaluation, questionnaires should be directed to residents in close proximity to the wetlands, to gain additional perspective on what their views are concerning wetlands. Even though the WET-EcoServices method for evaluating ecosystem services was sufficient to compare the ecosystem services being delivered by urban wetlands to those of rural wetlands, more objective methods should be investigated.

Lastly, the findings of the study should also be tested in additional urban areas in South Africa to indicate general trends and strengthen the basis of ecosystem service delivery research needed to adequately understand and ultimately limit further wetland loss in South Africa.

6 REFERENCES

Abila, R., Salzburger, W., Ndonga, M.F., Owiti, D.O., Barluenga, M. and Meyer, A. 2008. The role of the Yala swamp lakes in the conservation of Lake Victoria region haplochromine cichlids: Evidence from genetic and trophic ecology studies. *Lakes and Reservoirs: Research and Management*. 13:95-104.

Albert, D. and Minc, L. 2004. Plants as regional indicators of Great Lakes coastal wetland health. *Aquatic Ecosystem Health and Management*. 7:233-247.

Alston, K.P. and Richardson, D.M. 2006. The roles of habitat features, disturbance, and distance from putative source populations in structuring alien plant invasions at the urban/wildland interface on the Cape Peninsula, South Africa. *Biological Conservation*. 132(2):183-198.

Ammann, A.P. and Lindley-Stone, A. 1991. Method for the comparative evaluation of nontidal wetlands in New Hampshire. New Hampshire Department of Environmental Services, Concord, NH.

Anderson, P.M.L., Avlonitis, G. and Ernstson, H. 2014. Ecological outcomes of civic and expert-led urban greening projects using indigenous plant species in Cape Town, South Africa. *Landscape and Urban Planning*. 127:104-113.

Andreas, B.K. and Lichvar, R.W. 1995. Floristic assessment standards: A case study for northern Ohio. Wetlands Research Program: Technical Report W RP-DE-8. U.S. Army Corps of Engineers Waterways Experiment Station, Vicksburg, Mississippi, USA.

Aslaksen, I., Nybø, S., Framstad, E., Garnåsjordet, P.A. and Skarpaas, O. 2015. Biodiversity and ecosystem services: The Nature Index for Norway. *Ecosystem Services*. 12:108-116.

Atkinson, R.B., Perry, J.E., Smith, E. and Cairns, J. 1993. Use of created wetland delineation and weighted averages as a component of assessment. *Wetlands*. 13(3):185-193.

Audet, J., Baattrup-Pedersen, A., Andersen, H.E., Andersen, P.M., Hoffmann, C.C., Kjaergaard, C. and Kronvang, B. 2015. Environmental controls of plant species richness in riparian wetlands: Implications for restoration. *Basic and Applied Ecology*. 16(6):480-489.

Baldwin, A.H. 2004. Restoring complex vegetation in urban settings: The case of freshwater tidal marshes. *Urban Ecosystems*. 7:125–137.

Balvanera, P., Pfisterer, A.B., Buchmann, N., He, J.S., Nakashizuka, T., Raffaelli, D. and Schmid, B. 2006. Quantifying the evidence for biodiversity effects on ecosystem functioning and services. *Ecology Letters*. 9(10):1146-1156.

Balvanera, P., Quijas, S., Martin-Lopez, B., Barrios, E., Dee, L., Isbell, F., Durance, I., White, P., Blanchard, R. and de Groot, R.S. 2015. The links between biodiversity and ecosystem services. In *Handbook of Ecosystems Services*. Earthscan. http://www.unep.org/about/sgb/Portals/50153/green%20room/Balvanera%20et%20al%202016_Chapter%204_Routledge%20Handbook%20of%20Ecosystem%20Services.pdf? Date of access: 10 March 2016

Barnard, S., Venter, A. and van Ginkel, C.E. 2013. Overview of the influences of mining-related pollution on the water quality of the Mooi River system's reservoirs, using basic statistical analyses and self organised mapping. *Water South Africa*. 39(5):655-662.

Barthel, S. and Isendahl, C. 2013. Urban gardens, agriculture, and water management: Sources of resilience for long-term food security in cities. *Ecological Economics*. 86:224-234.

Bateganya, N.L., Nakalanzi, D., Babu, M. and Hein, T. 2015. Buffering municipal wastewater pollution using urban wetlands in sub-Saharan Africa: a case of Masaka municipality, Uganda. *Environmental technology*. 36(17):2149-2160

Baveye, P.C., Baveye, J. and Gowdy, J. 2013. Monetary valuation of ecosystem services: it matters to get the timeline right. *Ecological Economics*. 95:231-235.

Bavrlic, K and Bowers, K. 2010. Credit Valley Conservation: Monitoring forest integrity within the Credit River Watershed. http://www.creditvalleyca.ca/wp-content/uploads/2011/10/Chapter-3-Forest-Tree-Health-FINAL.pdf Date of access: 25 November 2016

Baxter, B., 2013. Updated Wetland Delineation and Assessment for the Proposed Zonnebloem Greenfields Mining Area, Mpumalanga. *Final EIA and EMP for the proposed Zonnebloem Opencast Coal Mine. Golder Associates*.

Bezuidenhout, H. 1993. Syntaxonomy and synecology of western Transvaal grassland. Unpublished Ph.D. dissertation, University of Pretoria, Pretoria.

Bianchi, F. J. J. A., Booij, C. J. H., and Tscharntke, T. 2006. Sustainable pest regulation in agricultural landscapes: a review on landscape composition, biodiversity and natural pest control. *Proceedings of the Royal Society of London: Biological Sciences*. 273(1595):1715-1727.

Bibby, P. 2009. Land-use change in Britain. *Land-use Policy*. 26:2–13.

Biddle, M.A., Tyler, R.M., Barthelmeh, T.G. and Canfield, T.J. 2007. Effectiveness of Restored Wetlands for the Treatment of Agricultural Runoff. Delaware Department of Natural Resources and Environmental Control, Dover, Delaware.

Biggs, R., Bohensky, E., Desanker, P.V., Fabricius, C., Lymann, T. and Misselhorn, A.A. 2004. Nature Supporting People: The South African Millennium Ecosystem Assessment. Council for Scientific and Industrial Research, Pretoria, South Africa.

Birdlife South Africa. 2009. Tungsten mining threatens RAMSAR site. http://verlorenvlei.co.za/tungsten-mining-threatens-ramsar-site/ Date accsessed: 25 November 2015

Boers, A.M., Veltman, R.L. and Zedler, J.B. 2007. Typha× glauca dominance and extended hydroperiod constrain restoration of wetland diversity. *Ecological Engineering. 29*(3):232-244.

Bolund, P. and Hunhammer, S. 1999. Ecosystem Services In Urban Areas. *Ecological Economics*. 29(2):293-301.

Botta-Dukat. 2005. Rao's quadratic entropy as a measure of functional diversity based on multiple traits. *Journal of Vegetation Science .16*:533-540.

Bouchard, V., Frey, S.D., Gilbert, J.M. and Reed, S.E. 2007. Effects of macrophyte functional group richness on emergent freshwater wetland functions. *Ecology*. 88(11):2903-2914.

Bourdaghs, M., Johnston, C.A, and Regal, R.A. 2006. Properties and performance of the floristic quality index in Great Lakes coastal wetlands. *Wetlands.* 26:718-735.

Boyer, T. and Polasky, S. 2004. Valuing urban wetlands: a review of non-market valuation studies. *Wetlands*. 24(4):744-755.

Braack, A.M., Walters, D. and Kotze, D.C. 2000. Practical Wetland Management. South Africa: Rennies Wetlands Project.

Brand, R.F., Du Preez, P.J. and Brown, L.R. 2013. High altitude montane wetland vegetation classification of the Eastern Free State, South Africa. *South African Journal of Botany*. 88:223-236.

Bratton, S.P. 1997. Alternative models of ecosystem health. *In*: Costanza, R., Norton, B.G. and Haskell, B.D. (Eds.), Ecosystem Health: New Goals for Ecosystem Management, Island Press, Washington, DC.

Bried, J.T., Jog, S.K. and Matthews, J.W. 2013. Floristic quality assessment signals human disturbance over natural variability in a wetland system. *Ecological indicators*. 34:260-267.

Brooks, N. 2007. Imminent water crisis in India. World's Biggest Problems, Arlington Institute, August. http://www.arlingtoninstitute.org/wbp/global-water-crisis/606 Date of access: 22 March 2015.

Brown, T.C., Bergstrom, J.C. and Loomis, J.B. 2006. Ecosystem Goods and Services--definition, Valuation, and Provision. Rocky Mountain Research Station, US Forest Service.

Burgess, J., Harrison, C.M. and Limb, M. 1988. People, parks and the urban green: a study of popular meanings and values for open spaces in the city. *Urban Studies.* 25:455–473.

Burkhard, B., F. Kroll, F. Muller and W. Windhorst. 2009. Landscapes' capacities to provide ecosystem services: a concept for land-cover based assessments. *Landscape Online.* 15:1–22.

Burkhard, B., I. Petrosillo and R. Constanza. 2010. Ecosystem services: Bridging ecology, economy and social sciences. *Ecological Complexity.* 7:257–259.

Butchart, S.H., Walpole, M., Collen, B., Van Strien, A., Scharlemann, J.P., Almond, R.E., Baillie, J.E., Bomhard, B., Brown, C., Bruno, J. and Carpenter, K.E. 2010. Global biodiversity: indicators of recent declines. *Science.* 328(5982):1164-1168.

Cadotte, M.W., Carscadden, K. and Mirotchnick, N. 2011. Beyond species: functional diversity and the maintenance of ecological processes and services. *Journal of Applied Ecology*. 48(5):1079-1087.

Campbell P.J. 2000. Rehabilitation Recommendations after Alien Plant Control. Plant Protection Research Institute, Agricultural Research Council, Hilton.

Carter, V., M. K. Garrett and P. T. Gammon. 1988. Wetland boundary determination in the Great Dismal Swamp using weighted averages. *Water Resources Bulletin.* 24(2):297–306.

CBD (Convention on Biological Diversity). 2013. Quick guide to the Aichi Biodiversity Targets: protected areas increased and improved. https://www.cbd.int/doc/strategic-plan/targets/compilation-quick-guide-en.pdf Date of access: 25 November 2015

Chamberlain, S.J. and Brooks, R.P. 2016. Testing a rapid Floristic Quality Index on headwater wetlands in central Pennsylvania, USA. *Ecological Indicators*. 60:1142-1149.

Chang, J.J., Wu, S.Q., Dai, Y.R., Liang, W. and Wu, Z.B. 2014. Nitrogen removal from nitrate-laden wastewater by integrated vertical-flow constructed wetland systems. *Ecological Engineering.* 58:192-201.

Chapin, F.S. 2003. Effects of plant traits on ecosystem and regional processes: a conceptual framework for predicting the consequences of global change. *Annals of Botany*. 91(4):455-463.

Cilliers, S. S. and Siebert, S.J. 2011. Urban flora and vegetation: patterns and processes. (*In* Niëmela, J., Breuste, J., Elmqvist, T., Guntenspergen, G., James, P. and McIntyre, N.E., *eds*. Urban ecology: Patterns, Processes and Applications. Oxford University Press, Oxford.)

Cilliers, S. S., Cilliers, E. J., Lubbe, C. E., and Siebert, S. J. 2013. Ecosystem services of urban green spaces in African countries: Perspectives and challenges. *Urban Ecosystems.* 16(4):681–702.

Cilliers, S.S., Schoeman, L.L. and Bredenkamp, G.J. 1998. Wetland plant communities in the Potchefstroom Municipal area, North-west, South Africa. *Bothalia*. 28(2):213-229.

Cilliers, S.S., Williams, N.S.G., and Barnard, F.J. 2008. Patterns of exotic plant invasions in fragmented urban and rural grasslands across continents. *Landscape Ecology.* 23: 1243–1256.

Clarke, K.R. 1993. Non-parametric multivariate analyses of changes in community structure. *Australian Journal of Ecology.* 18:117-143.

Clarke, K.R. and Ainsworth, M. 1993. A method of linking multivariate community structure to environmental variables. *Marine Ecology Progress Series*. 92:205-219.

Clarke, K.R. and Gorley, R.N. 2001. 'PRIMER v.5: User manual/Tutorial.' PRIMERE Ltd.:Plymouth, UK.

CoGTA (Cooperative Governance and Traditional Affairs). Integrated Urban Development Framework (IUDF). 2014. http://www.cogta.gov.za/sites/cogtapub/B2BDOCS/The%20Intergrated%20Urban%20Development%20Framework(IUDF).pdf#search=integrated%20urban%20development%20framework Date of access: 22 November 2015.

Cohen, M.J., Carstenn, S. and Lane, C.R. 2004. Floristic quality indices for biotic assessment of depressional marsh condition in Florida. *Ecological Applications.* 14:784-794.

Colding, J. 2011. The role of ecosystem services in contemporary urban planning. *In*: Niemelä, J. (Ed.), Urban Ecology. Patterns, Processes, and Applications. Oxford University Press, New York.

Coles-Richie, M.C., Roberts, D.W., Kershner, J.L. and Henderson, R.C. 2007. Use of a Wetland Index to Evaluate Changes in Riparian Vegetation After Livestock Exclusion. American Water Resources Association. 43(3):731-743.

Collins, N.B. 2005. Wetlands: The basics and some more. Free State Department of Tourism, Environmental and Economic Affairs.

Conservation of Agricultural Resources Act (CARA) 43 of 1983.

Cools, J., Diallo, M., Boelee, E., Liersch, S., Coertjens, D., Vandenberghe, V. and Kone, B. 2013. Integrating human health into wetland management for the Inner Niger Delta, Mali. *Environmental Science and Policy*. 34:34-43.

Cornelissen, J.H.C., Lavorel, S., Garnier, E., Diaz, S., Buchmann, N., Gurvich, D.E., Reich, P.B., Ter Steege, H., Morgan, H.D., Van Der Heijden, M.G.A. and Pausas, J.G. 2003. A handbook of protocols for standardised and easy measurement of plant functional traits worldwide. *Australian journal of Botany*. 51(4):335-380.

Costanza, R., d'Arge, R., de Groot, R., Farbek, S., Grasso, M., Hannon, B., Limburg, K., Naeem, S., O'Neill, R., Paruelo, J., Raskin, R., Suttonkk, P. and van den Belt, M. 1997. The value of the world's ecosystem services and natural capital. *Nature.* 387:253-260.

Cowan G.I. 1999. Wetlands of South Africa. Pretoria: Department of Environmental Affairs and Tourism.

Cowden, C and Kotze, D.C. 2009. WET-RehabEvaluate: guidelines for the monitoring and evaluation of wetland rehabilitation projects. WRC Report No. TT 342/08. Pretoria: Water Research Commission.

Cowden, C., Kotze, D.C., Ellery, W.N. and Sieben, E.J.J. 2014. Assessment of the long-term response to rehabilitation of two wetlands in KwaZulu-Natal, South Africa. *African Journal of Aquatic Science*. 39(3):237-247.

Crist, T.O., Veech, J.A., Gering, J.C. and Summerville, K.S. 2003. Partitioning species diversity across landscapes and regions: a hierarchical analysis of α, β, and γ diversity. *The American Naturalist*. 162(6):734-743.

Cronk, C.K. and Fennessy, M.S. 2001. Wetland Plants: Biology and Ecology. Lewis Publishers, New York.

Cuevas-Reyes, P., Gilberti, L., González-Rodríguez, A. and Fernandes, G.W. 2013. Patterns of herbivory and fluctuating asymmetry in Solanum lycocarpum St. Hill (Solanaceae) along an urban gradient in Brazil. *Ecological Indicators*. 24:557-561.

Daily, G.C. 1997. Nature's Services: Societal Dependence on Natural Ecosystems. Island Press.

Dallimer, M., Rouquette, J. R., Skinner, A. M., Armsworth, P. R., Maltby, L. M., Warren, P. H., and Gaston, K. J. 2012. Contrasting patterns in species richness of birds, butterflies and plants along riparian corridors in an urban landscape. *Diversity and Distributions*. 18(8):742-753.

Davies, R.G., Barbosa, O., Fuller, R.A., Tratalos, J., Burke, N., Lewis, D., Warren, P.H. *and* Gaston, K.J. 2008. City-wide relationships between green spaces, urban land-use and topography. *Urban Ecosystems.* 11:269–287.

De Bello, F., Lavorel, S., Díaz, S., Harrington, R., Cornelissen, J.H., Bardgett, R.D., Berg, M.P., Cipriotti, P., Feld, C.K., Hering, D. and da Silva, P.M. 2010. Towards an assessment of multiple ecosystem processes and services via functional traits. *Biodiversity and Conservation*. 19(10):2873-2893.

De Groot, R.S. 1992. Functions of nature: evaluation of nature in environmental planning, management and decision making. Wolters-Noordhoff BV.

De Groot, R.S., Wilson, M.A. and Boumans, R.M. 2002. A typology for the classification, description and valuation of ecosystem functions, goods and services. *Ecological Economics*. 41(3):393-408.

DEAT (Department of Environmental Affairs and Tourism). 2005. The State of the Environment of South Africa Report. Government Publication: Pretoria.

DEAT (Department of Environmental Affairs and Tourism). 2012a. Classification of wetlands: The wetland inventory. www.deat.co.za/documents Date of access: 15 February 2015.

DEAT (Department of Environmental Affairs and Tourism). 2012b. State of the ENVIRONMENT Theme Articles: State of Wetlands in South Africa. http://soer.deat.gov.za/dm_documents/WWD_Final_2Feb20123_TQdyf.pdf Date of access: 16 July 2016

Department of Water and Sanitation. 2014. A Desktop Assessment of the Present Ecological State, Ecological Importance and Ecological Sensitivity per Sub Quaternary Reaches for Secondary Catchments in South Africa. Secondary: [C2]. Compiled by RQIS-RDM: https://www.dwa.gov.za/iwqs/rhp/eco/peseismodel.aspx

Dethier, M.N. and Steneck, R.S. 1994. A functional group approach to the structure of algal-dominated communities. *Oikos.* 476-498.

Deutschewitz, K., Lausch, A., Kűhn, I. and Klotz, S. 2003. Native and alien plant species richness in relation to spatial heterogeneity on a regional scale in Germany. *Global Ecology and Biogeography*. 12:299-311.

Díaz, S. and Cabido, M. 2001. Vive la différence: plant functional diversity matters to ecosystem processes. *Trends in Ecology and Evolution.* 16(11):646-655.

Díaz, S., Lavorel, S., De Bello, F., Quétier, F., Grigulis, K. and Robson, M. 2007. Incorporating plant functional diversity effects in ecosystem service assessments. *Proceedings to the National Academy of Sciences of the United States of America.* 104(52):20684-20689.

Dini, J. 2012. Restoring wetlands and healing a nation: South Africa's Working for Wetlands programme. http://www.bwa.co.za/Articles/Restoring%20wetlands%20and%20healing%20a%20nation.pdf Date of access: 30 July 2015.

DMA (Durban Metropolitan Area). 1998. DMA Spatial Development Plan. http://www.durban.gov.za/Documents/City_Government/IDP_Policy/18%20SDF(original).pdf Date of access: 3 March 2014.

Dolan, R.W., Moore, M.E. and Stephens, J.D. 2011. Documenting effects of urbanization on flora using herbarium records. *Journal of Ecology.* 99:1055–1062.

DRDLR (Department of Rural Development and Land Reform). 2011. Guidelines for the development of Municipal Spatial Development Frameworks.

Driver, A., Maze, K., Rouge, M., Lombard, A.T., Nel, J., Turpie, J.K. Cowling, R.M., Goodman, R.J., Harris, J., Jonas, Z., Reyers, B., Sink, K. and Strauss, T. 2005. National Spatial Biodiversity Assessment. South African National Biodiversity Institute, Pretoria.

Driver, A., Sink, K.J., Nel, J.N., Holness, S., Van Niekerk, L., Daniels, F., Jonas, Z., Majiedt, P.A., Harris, L. and Maze, K. 2012. National Biodiversity Assessment 2011: An assessment of South Africa's biodiversity and ecosystems. Synthesis Report. South African National Biodiversity Institute and Department of Environmental Affairs, Pretoria, RSA.

Du Plessis, A. 2008. Public participation, good environmental governance and fulfilment of environmental rights. *PER: Potchefstroomse Elektroniese Regsblad*. 11(2):170-201.

Du Toit, M.J. 2009. Grassland ecology along an urban-rural gradient using GIS techniques in Klerksdorp, South Africa. Potchefstroom: NWU. (Thesis – MSc). http://hdl.handle.net/10394/4197

Du Toit, M.J. and Cilliers, S.S. 2011. Aspects influencing the selection of representative urbanisation measures to quantify urban-rural gradients. *Landscape Ecology*. 26:169-181.

Duan, N., Liu, X.D., Dai, J., Lin, C., Xia, X.H., Gao, R.Y., Wang, Y., Chen, S.Q., Yang, J. and Qi, J. 2011. Evaluating the environmental impacts of an urban wetland park based on emergy accounting and life cycle assessment: a case study in Beijing. *Ecological Modelling*. 222(2):351-359.

Dugan, P.J. 1990. Wetland conservation: A review of current issues and required action. IUCN.

DWAF (Department of Water Affairs and Forestry). 1999. South Africa Version 1.0 of Resource Directed Measures for Protection of Water Resources.

DWAF (Department of Water Affairs and Forestry). 2001. The legal regime applicable to wetlands and wetland rehabilitation. www.dwaf.gov.za Date of access: 20 October 2015.

DWAF (Department of Water and Forestry) 2005. A practical field procedure for identification and delineation of wetland riparian areas. DWAF. Pretoria.

DWS (Department of Water and Sanitation). 2014. A Desktop Assessment of the Present Ecological State, Ecological Importance and Ecological Sensitivity per Sub Quaternary Reaches for Secondary Catchments in South Africa. Secondary: [W5]. Compiled by RQIS-RDM: https://www.dwa.gov.za/iwqs/rhp/eco/peseismodel.aspx

Dyssel, M. 2013. An overview of urban-ecological challenges and changes in Bellville. *Life in a Changing Urban Landscape*. 1:318.

Egoh, B., Reyers, B., Rouget, M., Bode, M. and Richardson, D.M. 2009. Spatial congruence between biodiversity and ecosystem services in South Africa. *Biological Conservation*. 142(3):553-562.

Egoh, B., Rouget, M., Reyers, B., Knight, A.T., Cowling, R.M., van Jaarsveld, A.S. and Welz, A. 2007. Integrating ecosystem services into conservation assessments: a review. *Ecological Economics*. 63(4):714-721.

Ehrenfeld, J. G. 2000. Evaluating wetlands within an urban context. *Ecological Engineering*. 15(3):253-265.

Ellery, W., Grenfell, M., Grenfell, S., Kotze, D., McCarthy, T.S., Tooth, S., Grundling, P.L., Beckedahl, H., Maitre, D.L. and Ramsay, L. 2011. WET-origins: controls on the distribution and dynamics of wetlands in South Africa. *Water Research Commision Report* (334/09).

Ellis, J.B., Shutes, R.B., Revitt, D.M. and Zhang, T.T. 1994. Use of macrophytes for pollution treatment in urban wetlands. *Resources, Conservation and Recycling*. 11(1-4):1-12.

Elmqvist, T., Maltby, E., Barker, T., Mortimer, M., Perrings, C., Aronson, J., De Groot, R., Fitter, A., Mace, G., Norberg, J. and Pinto, I.S. 2010. Biodiversity, ecosystems and ecosystem services. (*In* Kumar, P., *ed.* The Economics of Ecosystems and Biodiversity (TEEB). Earthscan. London and Washington.

Emerton, L., Iyango, L., Luwum, P. and Malinga, A. 1999. The present economic value of Nakivubo urban wetland, Uganda. IUCN—The World Conservation Union, Eastern Africa Regional Office, Nairobi and National Wetlands Programme. Wetlands Inspectorate Division, Ministry of Water, Land and Enviornment, Kampala.

Ernstson, H. and Sörlin, S. 2013. Ecosystem services as technology of globalization: On articulating values in urban nature. *Ecological Economics*. 86:274–284.

ESRI (Environmental System Research Institute). 2015. ArcGIS desktop 10.3.1 help. Implementing Inverse Distance Weighted (IDW). www.esri.com Redlands, CA:USA.

eThekwini Municipality Environmental Management Department. 2009. Durban Metropolitan Open Space System. http://www.durban.gov.za/City_Services/development_planning_management/environmental_planning_climate_protection/Durban_Open_Space/Documents/Forms/DispForm.aspx?ID=1 Date of access: 23 November 2015.

Faber, S.C., Costanza, R. and Wilson, M.A. 2002. Economic and ecological concepts for valuing ecosystem services. *Ecological Economics. 41*(3):375-392.

Fennessy, M. S., Brueske, C. C. and Mitsch, W. J. 1994. Sediment deposition patterns in restored freshwater wetlands using sediment traps. *Ecological Engineering*. 3(4):409-428.

Fennessy, M.S. 2002. Methods for evaluating wetland condition: using vegetation to assess environmental conditions in wetlands. EPA/843/B-00/0002j. US Environmental Protection Agency, Washington, DC.

Fennessy, M.S., Jacobs, A.D. and Kentula, M.E. 2004. *Review of rapid methods for assessing wetland condition*. EPA/620/R-04/009. US Environmental Protection Agency, Washington, DC.

Fernandez-Juricic, E. 2004. Spatial and temporal analysis of the distribution of forest specialists in an urban-fragmented landscape (Madrid, Spain) – implications for local and regional bird conservation. *Landscape and Urban Planning*. 69:17–32.

Fitzhugh, T.W. and Richter, B.D. 2004. Quenching urban thirst: growing cities and their impacts on freshwater ecosystems. *BioScience.* 54:741–754.

Fleishman, E., Murphy, D.D. & Brussard, P.F. 2000. A new method for selection of umbrella species for conservation planning. *Ecological Applications*. 10(2):569-579.

Flinn, K.M., M.J. Lechowicz and M.J. Waterway. 2008. Plant species diversity and composition of wetlands within an upland forest. *American Journal of Botany.* 95(10):1216-1224.

Flynn, D.F., Mirotchnick, N., Jain, M., Palmer, M.I. and Naeem, S. 2011. Functional and phylogenetic diversity as predictors of biodiversity-ecosystem-function relationships. *Ecology*. 92(8):1573-1581.

Francis, C.M., Austen, M.J., Bowles, J.M. and Draper, W.B. 2000. Assessing floristic quality in southern Ontario woodlands. *Natural Areas Journal.* 20:66-77.

Franklin, J.F. 1988. Structural and functional diversity in temperate forests. *Biodiversity. National Academy Press, Washington, DC.*

Ganlin, W., Minyi, H., Renyan, D. and Kai, Z. 2006. Effects of different travelling disturbances on the species diversity in *Pinus talwanensis* communities. *Acta Ecologica Sinica.* 26:3924-3930.

Garden, S. 2008. Wetland geomorphology and floodplain dynamics on the hydrologically variable Mfolozi River, KwaZulu-Natal, South Africa. Durban: University of KwaZulu-Natal (Doctoral dissertation, PhD thesis).

Gibbs, J.P., 2000. Wetland loss and biodiversity conservation. *Conservation Biology*. 14(1):314-317.

Gichuki, J.F., Dahdouh, G.F., Mugo, J., Rabuor, C.O., Triest, L. and Dehairs, F. 2001. Species Inventory and the Local uses of the Plants and the Fisheries of the Lower Sondu Miriu Wetland of Swamp Victoria, Kenya. *Hydrobiologia*. 458: 99-106.

Goddard, M.A., Dougill, A.J., Benton, T.G. 2009. Scaling up from gardens: biodiversity conservation in urban environments. *Trends in Ecology and Evolution*. 25:90–98.

Govender-Ragubeer, Y., Meeuwis, J. and McKay, T. 2014. A series of unfortunate events: how the battle to save an urban wetland was both won and lost. *The Journal for Transdisciplinary Research in Southern Africa*. 10(1):149-168.

Grime, J.P. 1998. Benefits of plant diversity to ecosystems: Immediate, filter and founder effects. *Journal of Ecology*. 86:902-910.

Grime. J.P. 1979. Plant Strategies and Vegetation Processes. Bath: John Wiley and Sons.

Gusewell, S. and Klitzli, F. 1998. Abundance of common reed (*Phragmites australis*), site conditions and conservation value of fen meadows in Switzerland. *Acta Botanica Neerlandica.* 47:113-129.

Hahs, A.K. and McDonnell, M.J. 2006. Selecting independent measures to quantify Melbourne's urban-rural gradient. *Landscape and Urban Planning.* 78: 435-448.

Haines-Young, R. and Potschin, M. 2010. The links between biodiversity, ecosystem services and human well-being. *Ecosystem Ecology: A New Synthesis.* 1:110-139.

Hanley, T.A., 1978. A comparison of the line-interception and quadrat estimation methods of determining shrub canopy coverage. *Journal of Range Management*. 60-62

Harper, D.M., Adams, C. and Mavuti, K.M. 1999. The aquatic plant communities of the Lake Naivasha wetland, Kenya: pattern, dynamics and conservation. *Wetlands Ecology and Management*. 3(6):111-123.

Harrison, P.A., Berry, P.M., Simpson, G., Haslett, J.R., Blicharska, M., Bucur, M., Dunford, R., Egoh, B., Garcia-Llorente, M., Geamănă, N. and Geertsema, W. 2014. Linkages between biodiversity attributes and ecosystem services: a systematic review. *Ecosystem Services*. 9:191-203.

Hayasaka, D., Akasaka, M., Miyauchi, D., Box, E.O. and Uchida, T. 2012. Qualitative variation in roadside weed vegetation along an urban–rural road gradient. *Flora-Morphology, Distribution, Functional Ecology of Plants*. 207(2):126-132.

Hennessy, K. 2000. Towards a metropolitan open space system for the city of Cape Town. http://www.ierm.org.za/library/congress2001/towards%20a%20metropolitan%20open%20space.pdf Date of access: 3 March 2014.

Heydorn, A.F. 1996. Human population growth, land-use planning and wise wetland management –a challenge for the future. South Africa: World Wildlife Fund.

Hook. P.B. 2003. Sediment retention in rangeland riparian buffers. *Journal of Environmental Quality*. 32:1130-1137.

Hooper, D.U., Chapin Iii, F.S., Ewel, J.J., Hector, A., Inchausti, P., Lavorel, S., Lawton, J.H., Lodge, D.M., Loreau, M., Naeem, S. and Schmid, B. 2005. Effects of biodiversity on ecosystem functioning: a consensus of current knowledge. *Ecological monographs.* 75(1):3-35.

Hooper, D.U., Solan, M., Symstad, A.J., Diaz. S. and Gessner, M.O. 2002. Species diversity, functional diversity, and ecosystem functioning. In: M. L, Naeem S, Inchausti P, editors. Biodiversity and Ecosystem Functioning: Synthesis and Perspectives. Oxford, U.K.: Oxford University Press.

Hope, D., Gries, C., Zhu, W.X., Fagan, W.F., Redman, C.L., Grimm, N.B., Nelson, A.L., Martin, C. and Kinzig, A. 2003. Socioeconomics drive urban plant diversity. *Proceedings of the National Academy of Sciences.* 100:8788-8792.

Horwitz, P. and Finlayson, C. M. 2011. Wetlands as settings for human health: incorporating ecosystem services and health impact assessment into water resource management. *BioScience*. 61(9):678-688.

Isbell, F., Calcagno, V., Hector, A., Connolly, J., Harpole, W.S., Reich, P.B., Scherer-Lorenzen, M., Schmid, B., Tilman, D., van Ruijven, J. and Weigelt, A. 2011. High plant diversity is needed to maintain ecosystem services. *Nature.* 477(7363):199-202.

ISCW (Institute for Soil, Climate and Water). 2003. Comprehensive Report: Brakspruit AGR. Agricultural Research Council, Agromet Section, Pretoria.

Ishiyama, N., Akasaka, T. and Nakamura, F. 2014. Mobility-dependent response of aquatic animal species richness to a wetland network in an agricultural landscape. *Aquatic Sciences*. 76(3):437-449.

Jackson, L. E. 2003. The relationship of urban design to human health and condition. *Landscape and Urban Planning.* 64(4):191-200.

Janousek, C.N. 2009. Taxonomic composition and diversity of microphytobenthos in Southern California Marine Wetland habitats. *Wetlands*. 29:163-175.

Jog, S., Kindscher, K., Questad, E., Foster, B. and Loring, H. 2006. Floristic quality as an indicator of native species diversity in managed grasslands. *Natural Areas Journal*. 26(2):149-167.

Johnston, C.A., Zedler, J.B., Tulbure, M.G., Frieswyk, C.B., Bedford, B.L. and Vaccaro, L. 2009. A unifying approach for evaluating the condition of wetland plant communities and identifying related Stressors. *Ecological Applications.* 19:1739-1757.

Kent, M. 2011. Vegetation description and analysis: a practical approach. Chichester: Wiley.

Kingsford, R.T. 2000. Ecological impacts of dams, water diversions and river management on floodplain wetlands in Australia. *Australian Ecology*. 25(2):109-127.

Konijnendijk, C.C., Ricard, R.M., Kenney, A. and Randrup, T.B. 2006. Defining urban forestry - A comparative perspective of North America and Europe. *Urban Forestry and Urban Planning*. 4(3-4):93-103.

Kotze D.C, Breen C.M. and Quinn N. 1995. Wetland losses in South Africa. (*In*. Wetlands of South Africa. Cowan G.I. *ed.* Department of Environmental Affairs and Tourism, Pretoria.)

Kotze, D.C., Marneweck, G.C., Batchelor, A.L., Lindley, D.S. and Collins, N.B. 2008. WET-EcoServices: A technique for rapidly assessing ecosystem services supplied by wetlands. WRC Report No TT 339/08, Water Research Commission, Pretoria.

Land Type Survey Staff. 1984. Land types of the maps 2626 West Rand and 2726 Kroonstad. *Memoirs of Agricultural Natural Resources of South Africa*. 4:1-441.

Lavorel, S., Grigulis, K., Lamarque, P., Colace, M.P., Garden, D., Girel, J., Pellet, G. and Douzet, R. 2011. Using plant functional traits to understand the landscape distribution of multiple ecosystem services. *Journal of Ecology.* 99:135-147.

Lavorel, S., McIntyre, S., Landsberg, J. and Forbes, T.D.A. 1997. Plant functional classifications: from general groups to specific groups based on response to disturbance. *Trends in Ecology and Evolution.* 12:474-478.

Le Maitre, D.C., Milton, S.J., Jarmain, C., Colvin, C.A., Saayman, I. and Vlok, J.H. 2007. Linking ecosystem services and water resources: landscape-scale hydrology of the Little Karoo. *Frontiers in Ecology and the Environment*. 5(5):261-270.

Lélé, S.M., 1991. Sustainable development: a critical review. *World Development*. 19(6):607-621.

Li, F., Wang, R., Paulussen, J. and Liu, X. 2005. Comprehensive concept planning of urban greening based on ecological principles: a case study in Beijing, China. *Landscape and Urban Planning*. 72(4):325-336.

Li, W., Cheng, J.M., Yu, K.L., Epstein, H.E., Guo, L., Jing, G.H., Zhao, J. and Du, G.Z. 2015. Plant Functional Diversity Can Be Independent of Species Diversity: Observations Based on the Impact of 4-Yrs of Nitrogen and Phosphorus Additions in an Alpine Meadow. *PloS one.* 10(8).

Liquete, C., Cid, N., Lanzanova, D., Grizzetti, B. and Reynaud, A. 2016. Perspectives on the link between ecosystem services and biodiversity: The assessment of the nursery function. *Ecological Indicators*. 63:249-257.

Lockaby, B.G., Zhang, D., McDaniel, J., Tian, H. and Pan, S. 2005. Interdisciplinary research at the urban-rural interface: The WestGa project. *Urban Ecosystems*. 8:7-21.

Longcore, T. and Rich, C. 2004. Ecological light pollution. *Frontiers in Ecology and the Environment*. *2*(4):191-198.

Lopez, R. D. and Fennessy, M. S. 2002. Testing the floristic quality assessment index as an indicator of wetland condition. *Ecological Applications.* 12:487-497.

Loreau, M., Naeem, S., Inchausti, P., Bengtsson, J., Grime, J.P., Hector, A., Hooper, D.U., Huston, M.A., Raffaelli, D., Schmid, B. and Tilman, D. 2001. Biodiversity and ecosystem functioning: current knowledge and future challenges. *Science*. 294(5543):804-808.

Louw, W.J. 1951. An ecological account of the vegetation of the Potchefstroom area, Memoirs of the Botanical Survey of South Africa, No. 24

Lubbe, C.S. 2011. Comparison of the urban domestic garden flora along a socio-economic gradient in the Tlokwe City Municipality (Doctoral dissertation, North-West University).

Lubbe, C.S., Siebert, S.J. and Cilliers, S.S. 2010. Political legacy of South Africa affects the plant diversity patterns of urban domestic gardens along a socio-economic gradient. *Scientific and Research Essays*. 5:2900–2910.

Luck, G.W. 2007. A review of the relationships between human population density and biodiversity. *Biological Review.* 82:607-645.

Luck, M. and Wu, J. 2002. A gradient analysis of urban landscape pattern: a case study from the Phoenix metropolitan region, Arizona, USA. *Landscape Ecology*. 17:327-339.

Lundholm, J., Tran, S. and Gebert, L. 2015. Plant Functional Traits Predict Green Roof Ecosystem Services. *Environmental Science and Technology*. *49*(4):2366-2374.

Lyytimäki, J. and Sipilä, M. 2009. Hopping on one leg – The challenge of ecosystem disservices for urban green management. *Urban Forestry and Urban Greening.* 8:309–315.

Lyytimäki, J., Petersen, L.K., Normander, B. and Bezák, P. 2008. Nature as a nuisance? Ecosystem services and disservices to urban lifestyle. *Environmental Sciences*. 5(3):161-172.

Mace, G.M., Norris, K. and Fitter, A.H. 2012. Biodiversity and ecosystem services: a multilayered relationship. *Trends in Ecology and Evolution*. 27(1):19-26.

Macfarlane, D.M., Dieckens, J. & Von Hase, F. 2009. Development of a methodology to determine the appropriate buffer zone width and type for developments associated with wetlands, watercourses and estuaries. *INR Report* (400/09).

Macfarlane, D.M., Kotze, D.C., Ellery, W.N., Walters, D., Koopman, V., Goodman, P. and Goge, C., 2009. WET-Health: A technique for rapidly assessing wetland health. *Water Research Commission Report TT*, *340*(09).

Magurran, A.E. 2004. Measuring biological diversity. Victoria, Australia: Blackwell Science Ltd. 256 p.

Malan, H. L., Appleton, C. C., Day, J. A. and Dini, J. 2009. Wetlands and invertebrate disease hosts: Are we asking for trouble? *Water South Africa*. 35(5):753-768.

Mangold, S. Momberg, M. and Newbery, R. 2002. State of the Environmental Report, North West Province. http://soer.deat.gov.za/dm_documents/19_Full_Overview_Print_zWCET.pdf Date of access: July 2014.

Marzluff, J.M., Bowman, R. and Donnelly, R. 2001. A historical perspective on urban bird research: trends, terms, and approaches. In *Avian ecology and conservation in an urbanizing world*. Springer, US.

Mason, N. W., Mouillot, D., Lee, W. G., and Wilson, J. B. 2005. Functional richness, functional evenness and functional divergence: the primary components of functional diversity. *Oikos*. 111(1):112-118.

Mason, N.W., MacGillivray, K., Steel, J.B. and Wilson, J.B. 2003. An index of functional diversity. *Journal of Vegetation Science*. 14(4):571-578.

Masondo, S. 2011. Water crisis by 2020. *Sunday Times*. http://www.timeslive.co.za/local/2011/02/14/water-crisis-by-2020 Date of access: 20 November 2015.

Mayfield, M.M., Boni, M.F., Daily, G.C. and Ackerly, D. 2005. Species and functional diversity of native and human-dominated plant communities. *Ecology.* 86(9):2365-2372.

Mayfield, M.M., Bonser, S.P., Morgan, J.W., Aubin, I., McNamara, S. and Vesk, P.A. 2010. What does species richness tell us about functional trait diversity? Predictions and evidence for responses of species and functional trait diversity to land-use change. *Global Ecology and Biogeography*. 19(4):423-431.

McDonnell M.J. and Pickett, S.T.A. 1990. Ecosystem structure and function along urban–rural gradients: an unexploited opportunity for ecology. *Ecology.* 71(4):1232–1237.

McDonnell, M.J., Pickett, S.T., Groffman, P., Bohlen, P., Pouyat, R.V., Zipperer, W.C., Parmelee, R.W., Carreiro, M.M. and Medley, K. 1997. Ecosystem processes along an urban-to-rural gradient. *Urban Ecosystems*. 1(1):21-36.

McGarigal, K. and Marks, B. J. 1995. FRAGSTATS: spatial pattern analysis program for quantifying landscape structure. Gen. Tech. Rep. PNW-GTR-351. Portland: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station.

McGinley, M. 2014. Species richness. http://www.eoearth.org/view/article/156216 Date of access: 20 November 2015.

McInnes, R. 2010. Urban Development, Biodiversity and Wetland Management. Expert Workshop Report. Oxford. UK.

McIntyre, N.E., Knowles-Yánez, K. and Hope, D. 2000. Urban ecology as an interdisciplinary field: differences in the use of “urban” between the social and natural sciences. *Urban Ecosystems*. 4(1):5-24.

McKinney, M.L. 2008. Effects of urbanization on species richness: a review of plants and animals. *Urban Ecosystems.* 11:161–176.

MEA (Millennium Ecosystem Assessment). 2005. Ecosystems and Human Wellbeing: Synthesis. Island Press, Washington, DC. World Resources Institute.

Mertz, O., Ravnborg, H.M., Lövei, G.L., Nielsen, I. and Konijnendijk, C.C. 2007. Ecosystem services and biodiversity in developing countries. *Biodiversity and Conservation*. 16(10):2729-2737.

Miller, S.J. and Wardrop, D.H. 2006. Adapting the floristic quality assessment index to indicate anthropogenic disturbance in central Pennsylvania wetlands. *Ecological Indicators.* 6(2):313-326.

Mitsch, W.J. and Gosselink, J.G. 2007. Wetlands. 4th ed. Hoboken, NJ: John Wiley and Sons, Inc.

Moor, H., Hylander, K. and Norberg, J. 2015. Predicting climate change effects on wetland ecosystem services using species distribution modeling and plant functional traits. *Ambio*. 44(1):113-126.

Mthiyane, T.S. 2009. Small scale farming on Wetland resource *utilisation: a case study of Mandlanzini, Richards Bay* (Doctoral dissertation). http://uzspace.uzulu.ac.za/bitstream/handle/10530/550/small%20scale%20farming%20on%20wetlang%20resource.pdf?sequence=1 Date of access: 3 June 2016.

Mucina, L. and Rutherford, M.C. 2006. The Vegetation of South Africa, Lesotho and Swaziland. Strelitzia 19. Pretoria: South African National Biodiversity Institute (SANBI).

Mugisha, P., Kansiime, F., Mucunguzi, P. and Kateyo, E. 2007. Wetland vegetation and nutrient retention in Nakivubo and Kirinya wetlands in the Lake Victoria basin of Uganda. *Physics and Chemistry of the Earth, Parts A/B/C*. 32(15):1359-1365.

Mushet, D.M., Euliss, N.H. and Shaffer, T.L. 2002. Floristic quality assessment of one natural and three restored wetland complexes in North Dakota, USA. *Wetlands*. 22(1):126-138.

Naeem, S. and Wright, J.P. 2003. Disentangling biodiversity effects on ecosystem functioning: deriving solutions to a seemingly insurmountable problem. *Ecology letters*. 6(6):567-579.

Naeem, S., Bunker, D.E., Hector, A., Loreau, M. and Perrings, C. 2009. Biodiversity, ecosystem functioning and human wellbeing. An ecological and economic perspective. Oxford, UK: Oxford University Press.

Namaalwa, S., Funk, A., Ajie, G.S. and Kaggwa, R.C. 2013. A characterization of the drivers, pressures, ecosystem functions and services of Namatala wetland, Uganda. *Environmental Science and Policy*. 34:44-57.

Nel, J.L. and Driver, A. 2012. South African National Biodiversity Assessment 2011: Technical Report. Volume 2: Freshwater Component. CSIR Report Number CSIR/NRE/ECO/IR/2012/0022/A. Council for Scientific and Industrial Research, Stellenbosch.

Nel, L.T., Truter, F.C. and Willemse, J. 1939. The geology of the country around Potchefstroom and Klerksdorp. An explanation of sheet 61. Geology Survey. Union of South Africa. Dept. of Mines.

Nicolet, P., Biggs, J., Fox, G., Hodson, M.J., Reynolds, C., Whitfield, M. and Williams, P. 2004. The wetland plant and macroinvertebrate assemblages of temporary ponds in England and Wales. *Biological Conservation*. 120(2):261-278.

Niemelä, J. 1999a. Ecology and Urban Planning. *Biodiversity and Conservation*. 8(1):119-131.

Niemelä, J. 1999b. Is there a need for a theory of urban ecology? *Urban Ecosystems*. 3(1):57-65.

Niemelä, J., Breuste, J., Elmqvist, T., Guntenspergen, G., James, P. and MacIntyre, N. 2011. *Urban Ecology: Patterns, Processes and Applications*. Oxford Biology, Oxford, UK.

Niemelä, J., Kotze, J., Ashworth, A., Brandmayr, P., Desender, K., New, T., Penev, L., Samways, M. and Spence, J. 2000. The search for common anthropogenic impacts on biodiversity: a global network. *Journal of Insect Conservation.* 4(1):3-9.

Norberg, J., Swaney, D.P., Dushoff, J., Lin, J., Casagrandi, R. and Levin, S.A. 2001. Phenotypic diversity and ecosystem functioning in changing environments: a theoretical framework. *Proceedings of the National Academy of Sciences*. 98(20):11376-11381.

NRC (National Research Council). 1995. Wetlands: Characteristics and Boundaries. National Academy Press, Washington, DC, USA.

O'Connell, T.J., Jackson, L.E. and Brooks, R.P. 2000. Bird guilds as indicators of ecological condition in the central Appalachians. *Ecological Applications*. 10(6):1706–1721.

O'Farrell, P.J., De Lange, W.J., Le Maitre, D.C., Reyers, B., Blignaut, J.N., Milton, S.J., Atkinson, D., Egoh, B., Maherry, A., Colvin, C. and Cowling, R.M. 2011. The possibilities and pitfalls presented by a pragmatic approach to ecosystem service valuation in an arid biodiversity hotspot. *Journal of Arid Environments*. 75(6):612-623.

O'Farrell, P.J., Anderson, P.M., Le Maitre, D.C. and Holmes, P.M. 2012. Insights and opportunities offered by a rapid ecosystem service assessment in promoting a conservation agenda in an urban biodiversity hotspot. *Ecology and Society*. 17(3):27.

Ollis, D. J., Ewart-Smith, J. L., Day, J. A., Job, N. M., Macfarlane, D. M., Snaddon, C. D., Sieben, E. J. J., Dini, J. A. and Mbona, N. 2015. The development of a classification system for inland aquatic ecosystems in South Africa. *Water South Africa.* 41(5):727-745.

Ollis, D. J., Job, N., Macfarlane, D. and Sieben, E. J. 2009. Further Development of a Proposed National Wetland Classification System for South Africa. Supplementary Project Report: Application and Testing of Proposed National Wetland Classification System. Prepared for the South African National Biodiversity Institute, Pretoria.

Oneal, A.S. and Rotenberry, J.T. 2008. Riparian plant composition in an urbanizing landscape in southern California, USA. *Landscape Ecology*. 23:553–567.

Owino, A.O. and Ryan, P.G. 2007. Recent papyrus swamp habitat loss and conservation implications in western Kenya. *Wetlands Ecological Management*. 15:1-12.

Paker, Y., Yom-Tov, Y., Alon-Mozes, T. and Barnea, A. 2014. The effect of plant richness and urban garden structure on bird species richness, diversity and community structure. *Landscape and Urban Planning*. 122:186-195.

Pauw, A. and Louw, K. 2012. Urbanization drives a reduction in functional diversity in a guild of nectar-feeding birds. *Ecology and Society*. 17(2):27.

Pérez-Harguindeguy, N., Díaz, S., Garnier, E., Lavorel, S., Poorter, H., Jaureguiberry, P., Bret-Harte, M.S., Cornwell, W.K., Craine, J.M., Gurvich, D.E. and Urcelay, C. 2013. New handbook for standardised measurement of plant functional traits worldwide. *Australian Journal of Botany*. 61(3):167-234.

Petchey, O. L. 2003. Integrating methods that investigate how complementarity influences ecosystem functioning. *Oikos.* 101:323–330.

Petchey, O.L. and Gaston, K.J. 2006. Functional diversity (FD), species richness and community composition. *Ecological Letters.* 5:402–411.

Petchey, O.L., O'Gorman, E.J. and Flynn, D.F.B. 2009. A functional guide to functional diversity measures. (*In* Naeem *et al., eds.* Biodiversity, ecosystem functioning and human wellbeing. Oxford University Press.)

Pollard, S. R. and Cousins, T. 2008. Towards integrating community-based governance of water resources with the statutory frameworks for Integrated Water Resources Management: A review of community–based governance of freshwater resources in four southern African countries to inform governance arrangements of communal wetlands. *Water Research Commission Report TT328*(08). Pretoria.

Pretorius, C., Du Toit, M.J. and Cilliers, S.S. 2013. Abstract: Effect of land-cover change on the vegetation types and ecosystem services of the Tlokwe Municipal Area, North West Province. *South African Journal of Botany*. 86:165-166.

Primack, R.B. 2002. Essentials of Conservation Biology. 3rd ed. Sunderland: Sinauer Associates.

PRIMER-E. 2012. PRIMER software v6. PRIMER-E ltd.

Pugnaire, F.I. and Valladares, F. 2007. Handbook of Functional Plant Ecology. Marcel Dekker INC., New-York.

Quijas, S., Schmid, B. and Balvanera, P. 2010. Plant diversity enhances provision of ecosystem services: a new synthesis. *Basic and Applied Ecology*. 11(7):582-593.

Qureshi, S., Breuste, J.H. and Lindley, S.J. 2010. Green space functionality along an urban gradient in Karachi, Pakistan: a socio-ecological study. *Human Ecology*. 38(2):283-294.

Ramburan, S. 2013. Review and analysis of variety distribution trends in the South African sugar industry: A 2013 perspective. In *Proceedings of the South African Sugar Technologists' Association*. 86:261-272.

Ramsar Convention Secretariat. 2013. The Ramsar Convention Manual: a guide to the Convention on Wetlands (Ramsar, Iran, 1971), 6th ed. Ramsar Convention Secretariat, Gland, Switzerland.

Ramsar. 2000. What are wetlands? http://www.ramsar.org/about_infopack1e.htm Date of access: July 2015.

Ramsay, P.M. and Oxley, E.R.B. 1997. The growth form composition of plant communities in the Ecuadorian páramos. *Plant Ecology.* 131(2):173-192.

Ravit, B., Rohleder, L., Johnson, L., Ehrenfeld, J. and Kalin, P. 2006. A vegetation survey of Teaneck Creek wetlands. *Urban Habitat*. 5:199-214.

Rebelo, L.M., Johnston, R., Hein, T., Weigelhofer, G., D'Haeyer, T., Kone, B. and Cools, J. 2013. Challenges to the integration of wetlands into IWRM: The case of the Inner Niger Delta (Mali) and the Lobau Floodplain (Austria). *Environmental Science and Policy*. 34:58-68.

Reeves, P.N. & Champion, P.D. 2004. Effects of livestock grazing on wetlands: Literature review. Environment Waikato Regional Council. Environment Waikato Technical Report 2004/16

Reeves, P.N. and Champion, PD. 2004. Effects of livestock grazing on wetlands: Literature Review. Environment Waikato Technical Report 2004/16.

Republic of South Africa. 1983. Conservation of Agricultural Resources Act (CARA), 1983. Government Printer, Pretoria.

Richardson, D.M. and Van Wilgen, B.W. 2004. Invasive alien plants in South Africa: how well do we understand the ecological impacts? Working for Water. *South African Journal of Science*. 100(1and2):45.

Ridd, M.K. 1995. Exploring a V-I-S (vegetation-impervious surface-soil) model for urban ecosystem analysis through remote sensing: comparative anatomy for cities. *International Journal of Remote Sensing.* 16:2165-2185.

Ring, I., Hansjürgens, B., Elmqvist, T., Wittmer, H. and Sukhdev, P. 2010. Challenges in framing the economics of ecosystems and biodiversity: the TEEB initiative. *Current Opinion in Environmental Sustainability*. 2(1):15-26.

Roberts, D. C. and N. Diederichs. 2002. Durban's Local Agenda 21 programme: tackling sustainable development in a post-apartheid city. *Environment and Urbanization*. 14(1):189-201.

Roberts. D.C. 1993. The vegetation ecology of municipal Durban. Floristic classification. *Bothalia.* 23(2):271-326.

Roberts-Pichette, P. and Gillespie, L.J. 1999. *Terrestrial vegetation biodiversity monitoring protocols*. Ecological Monitoring Coordinating Office, Canada Centre for Inland Waters.

Rogers, P., de Silva, R. and Bathia, R. 2002. Water is an economic good: How to use prices to promote equity, efficiency, and sustainability. *Water Policy.* 4(1):1–17.

Roggeri. H. 1995. Tropical Freshwater Wetlands: A Guide to Current Knowledge and sustainable Management. Kluwer Academic Publishers, Dordrecht.

Rojo, C., Alvarez-Cobelas, M., Benavent-Corai, J., Barón-Rodríguez, M.M. and Rodrigo, M.A. 2012. Trade-offs in plankton species richness arising from drought: insights from long-term data of a National Park wetland (central Spain). *Biodiversity and Conservation*. 21(10):2453-2476.

Roscher, C., Schumacher, J., Gubsch, M., Lipowsky, A., Weigelt, A., Buchmann, N., Schmid, B. and Ernst-Detlef, S. 2012. Using plant functional traits to explain diversity–productivity relationships. *PLoS ONE*. 7:3660.

Rountree, M., Thompson, M., Kotze, D.C., Batchelor, A. and Marneweck, G. 2009. WET-Prioritze. Guidelines for prioritizing wetlands at national, regional and local scales. Wetland Management Series. Water Research Commission Report TT 337/09, Pretoria.

RSA (Republic of South Africa). 1998. National Water Act 36 of 1998. Government Printer, Pretoria.

Russi, D., ten Brink, P., Farmer, A., Badura, T., Coates D., Förster J., Kumar, R. and Davidson, N. 2013. The Economics of Ecosystems and Biodiversity for Water and Wetlands. IEEP, London and Brussels; Ramsar Secretariat, Gland.

Rutherford, M.C., Powrie, L.W. and Husted, L.B. 2012. Plant diversity consequences of a herbivore-driven biome switch from Grassland to Nama-Karoo shrub steppe in South Africa. *Applied Vegetation Science*. 15(1):14-25.

Ruto, W.K.S., Kinyamario, N.K., Akunda, E. and Mworia, J.K. 2012. Plant Species Diversity and Composition of Two Wetlands in the Nairobi National Park, Kenya. *Journal of Wetland Ecology.* 6:07-15.

Saarnio, S., Winiwarter, W. and Leitão, J. 2009. Methane release from wetlands and watercourses in Europe. *Atmospheric Environment*. 43(7):1421-1429.

SACS (South African Committee for Stratigraphy) 1980. Part 1: Lithostratigraphy of the Republic of South Africa, South West Africa/Namibia and the Republics of Bophuthatswana, Transkei and Venda. Government Printer, South Africa.

SAIRR (South African Institute of Race Relations). 2013. 2012 Survey: Demographics. http://irr.org.za/reports-and-publications/south-africa-survey/south-africa-survey-2012/downloads/2012%20Survey%20-%20Demographics%20-WEB.pdf/at_download/file Date of access: 20 October 2015.

Sajikumar, N. and Remya, R.S. 2015. Impact of land cover and land-use change on runoff characteristics. *Journal of Environmental Management*. 161:460-468.

SANBI (South African National Biodiversity Institute). 2014. Rehabilitation Plan for Gauteng North Wetland Project, Gauteng: Planning Year 2014. Prepared by Elmie Weideman and Johan Goosen, Aurecon South Africa (Pty) Ltd as part of the planning phase for the Working for Wetlands Rehabilitation Programme. SANBI Report No. 109664/8806.

SANBI. 2012. Further Development of a Proposed National Wetland Classification System for South Africa. Primary Project Report. Prepared by the Freshwater Consulting Group (FCG) for the South African National Biodiversity Institute (SANBI).

SANBI. 2012. South African National Biodiversity Institute: The National Wetland Map. http://bgis.sanbi.org/nwi/map.asp Date of access: 20 October 2015.

Sanesi, G. and Chiarello, F. 2006. Residents and urban green spaces: The case of Bari. *Urban Forestry and Urban Greening*. 4(3-4):125-134.

Schmidt, K. J., Poppendieck, H. H., and Jensen, K. 2014. Effects of urban structure on plant species richness in a large European city. *Urban Ecosystems.* 17(2):427-444.

Schuijt, K. 2005. Economic consequences of wetland degradation for local populations in Africa. *Ecological Economics*. 53:177-190.

Schwartz, M.W, Bringham, C.A, Hoeksema, J.D, Lyons, K.G, Mills, M.H. and van Mantgem, P.J. 2000. Linking biodiversity to ecosystem function: implications for conservation ecology. *Oecologia*. 12:297-305.

Secretariat of the CBD (Convention on Biological Diversity). 2012. Cities and Biodiversity Outlook—Executive Summary. Montreal.

Shepherd, P. 1994. A review of plant communities of derelict land in the city of Nottingham, England and their value for nature conservation. *Memorabilia Zoologica*. (49):129-137

Shulse, C.D., Semlitsch, R.D., Trauth, K.M. and Gardner, J.E. 2012. Testing wetland features to increase amphibian reproductive success and species richness for mitigation and restoration. *Ecological Applications*. 22(5):1675-1688.

Sieben, E.J.J. 2011. Compiling Vegetation Data in Wetlands in KwaZulu-Natal, Free State and Mpumalanga, Providing Minimum Data Requirements and a Sampling Protocol**.** Report to the Water Research Commission, Department of Plant Sciences, University of the Free State, Qwaqwa, SA.

Sieben, E.J.J. 2012. Plant functional composition and ecosystem properties: The case of peatlands in South Africa. *Plant Ecology*. 213(5):809-820.

Sieben, E.J.J., Mtshali, H. and Janks, M. 2013. National Wetland Vegetation Database: Classification and analysis of wetland vegetation types. Report to the WRC (Water Reseach Commission). Report No. K5-1980.

Sinchembe, M. and Ellery, W.N. 2010. Human impacts on hydrological health and the provision of ecosystem services: a case study of the eMthonjeni–Fairview Spring Wetland, Grahamstown, South Africa. *African Journal of Aquatic Science*. 35(3):227-239.

Smith, R.W., 1984. Department of Defense World Geodetic System 1984: its definition and relationships with local geodetic systems. Defense Mapping Agency.

SRK Consulting. 2012. Phase 1 Heritage Impact Assessment Report: Zulti South Mineral Lease Area, Umhlatuze Local Municipality, Uthungulu District Municipality, Kwazulu-Natal, South Africa. http://www.srk.co.za/files/File/SouthAfrica/publicDocuments/ZULTI/S2Z/Z/1_366069_Wetland_appendix_X_20140710.pdf Accessed: 14 April 2015

Ssegawa, P., Kakudidi, E., Muasya, M. and Kalema, J. 2004. Diversity and distribution of sedges on multivariate environmental gradients. *African Journal of Ecology*. 42(1):21-33.

Statsoft, Inc. 2013. STATISTICA (data analysis software system), version 12. www.statsoft.com

Stewart, W. 2006. Nelson Mandela Metropolitan Open Space System. Unpublished Report.

Strategic Environmental Focus. 2002. Johannesburg Open Space System. www.joburg.org.za/docs/jmossreport.doc Date of access: 23 November 2015.

Stuip, M.A.M., Baker, C.J. and Oosterberg, W. 2002. Socio-economics of wetlands. Wetlands International and RIZA. The Netherlands.

Stumm, L.H., Dennis, D., Whigham, J., Dos, T.A. and Verhoeven, C. 2009. Wetlands of the World: the next instalment. *Wetlands Ecological Management*. 17:167.

Suding, K.N., Lavorel, S., Chapin, F.S., Cornelissen, J.H., DIAz, S., Garnier, E., Goldberg, D., Hooper, D.U., Jackson, S.T. and NAVAS, M.L. 2008. Scaling environmental change through the community-level: a trait-based response-and-effect framework for plants. *Global Change Biology*. 14(5):1125-1140.

Summerford, S.E. 2009. Characterization of soil/vegetation on flood irrigated hayfields in Grand Teton National Park, Wyoming: a predictive evaluation tool for agricultural wetlands (Doctoral dissertation, Montana State University-Bozeman, College of Agriculture).

Sutton-Grier, A.E., Moore, A.K., Wiley, P.C. and Edwards, P.E. 2014. Incorporating ecosystem services into the implementation of existing US natural resource management regulations: operationalizing carbon sequestration and storage. *Marine Policy*. 43:246-253.

Swink, F. and Wilhelm, G.S. 1979. Plants of the Chicago Region, revised and expanded edition with keys. *The Morton Arboretum, Lisle, IL, USA*.

Taft, J.B., Wilhelm, G.S., Ladd, D.M. and Masters, L.A. 1997. *Floristic quality assessment for vegetation in Illinois, a method for assessing vegetation integrity*. Westville, Illinois: Illinois Native Plant Society.

Tesauro, J. 2001. Restoring wetland habitats with cows and other livestock. *Conservation*. 2(2): 26-31.

Thackway, R., Frakes, I. and Lesslie, R. 2006. Reporting trends in vegetation assets, states and transitions at the farm level–a southern tablelands case study. In *VegFutures 06: The conference in the field proceedings*.

Threlfall, C. G., Walker, K., Williams, N. S., Hahs, A. K., Mata, L., Stork, N., and Livesley, S. J. 2015. The conservation value of urban green space habitats for Australian native bee communities. *Biological Conservation*. 187:240-248.

Thullen, J. S., Sartoris, J. J. and Nelson, S. M. 2005. Managing vegetation in surface-flow wastewater-treatment wetlands for optimal treatment performance. *Ecological Engineering*. 25(5):583-593.

Tiner, R. W. 2003. Geographically isolated wetlands of the United States. *Wetlands.* 23:494-516.

Tiner, R.W. 1999. Wetland Indicators: A Guide to Wetland Identification, Delineation, Classification and Mapping. Lewis Publishers, Washington, D.C., USA.

Tlokwe City Council. 2013. Integrated Development Plan REVIEW of the Tlokwe City Council. http://www.potch.co.za/pdf2013/IDP%20%20ADOPTED%20ON%2028%20MAY%202013.pdf Date of access: 18 July 2016

Tlokwe Local Municipality. 2008. Tlokwe Spatial Development Framework – 2008. http://www.potch.co.za/pdf2012/Budget/budget%20blad/Tlokwe%20SDF%20Final%20-%20June%20(Reduced).pdf Date of access: 18 June 2016

Turpie, J. and Malan, H. 2010. Wetland valuation Vol II-wetland valuation case studies. *WRC Report*, (441/09). Pretoria.

Turpie, J.K., Marais, C. and Blignaut, J.N. 2008. The working for water programme: Evolution of a payments for ecosystem services mechanism that addresses both poverty and ecosystem service delivery in South Africa. *Ecological Economics*. 65(4):788-798.

Tzoulas, K., Korpela, K., Venn, S., Yli-Pelkonen, V., Kazmierczak, A., Niemelä, J., James, P. 2007. Promoting ecosystem and human health in urban areas using green infrastructure: A literature review. *Landscape and Urban Planning*. 81(3):167–178.

UKNEA (UK National Ecosystem Assessment). 2011. The UK National Ecosystem Assessment: Synthesis of the Key Findings. UNEP-WCMC, Cambridge.

UNDESA (United Nations, Department of Economic and Social Affairs, Population Division). 2015. World Urbanization Prospects: The 2014 Revision, (ST/ESA/SER.A/366).

United Nations. 2008. World Urbanization Prospects: The 2007 Revision. http://www.un.org/esa/population/publications/wup2007/2007WUP_Highlights_web.pdf Date of access: 22 October 2015.

USEPA (US Environmental Protection Agency). 2014. Green Infrastructure. http://www2.epa.gov/sites/production/files/2014-11/documents/gi_brochure_508_compliant.pdf Assessed: 22 November 2014.

Van der Walt, L. and Cilliers, S.S, Du Toit, M.J. and Kellner, K.K. 2014. Conservation of fragmented grasslands as part of the urban green infrastructure: how important are species diversity, functional diversity and landscape functionality? *Urban Ecosystems.* 18(1):87-113.

Van Deventer, R., Teixeira-Leite, A. and Macfarlane, D. 2013. Specialist Wetland Assessment Report: Lions River-Howick Bulk Water Pipeline, KwaZulu-Natal. Available at: http://www.sahra.org.za/sahris/sites/default/files/additionaldocs/LionsRiverWaterPipeline_Wetland%20Assessment%20Report%20for%20Synergistics_Eco-Pulse_20Mar2014.pdf Date of accessed: March 2016

Van Ginkel CE, Glen RP, Gordon-Gray KD, Cilliers CJ, Muasya M and van Deventer PP. 2010. Easy identification of some South African wetland plants. WRC Report No. TT 479/10. Pretoria: Water Research Commission.

Van Oudtshoorn, F. 2006. Guide to grasses of Southern Africa. Briza.

Van Rensburg, B.J., Peacock, D.S. and Robertson, M.P. 2009. Biotic homogenization and alien bird species along an urban gradient in South Africa. *Landscape and Urban Planning*. 92(3):233-241.

Van Wyk, E., Cilliers, S.S. and Bredenkamp, G.J. 2000. Vegetation analysis of wetlands in the Klerksdorp Municipal area, North West Province, South Africa. *South African Journal of Botany*. 66:52–62.

Van Zyl, H. and Leiman, A. 2002. Hedonic approaches to estimating the impacts of open spaces: A case study in the Cape. *South African Journal of Economic and Management Sciences*. 5(2):379-394.

Violle, C., Navas, M.L., Vile, D., Kazakou, E., Fortunel, C., Hummel, I. and Garnier, E. 2007. Let the concept of trait be functional! *Oikos*. 116(5):882-892.

Von Döhren, P. and Haase, D. 2015. Ecosystem disservices research: A review of the state of the art with a focus on cities. *Ecological Indicators*. 52:490-497.

Wang, H., Chen, Z.X., Zhang, X.Y., Zhu, S.X., Ge, Y., Chang, S.X., Zhang, C.B., Huang, C.C. and Chang, J. 2013. Plant species richness increased belowground plant biomass and substrate nitrogen removal in a constructed wetland. *CLEAN–Soil, Air, Water*. 41(7):657-664.

Wania, A., Kühn, I. and Klotz, S. 2006. Plant richness patterns in agricultural and urban landscapes in Central Germany—spatial gradients of species richness. *Landscape and Urban Planning*. 75(1):97-110.

Wear, D.N., Turner, M.G. and Naiman, R.J., 1998. Land cover along an urban–rural gradient: implications for water quality. *Ecological Applications*. 8(3):619-630.

Westoby, M. and Wright, I.J. 2006. Land-plant ecology on the basis of functional traits. *Trends in Ecology and Evolution*. 21(5):261-268.

White, J.M. and Stromberg, J.C. 2011. Resilience, restoration, and riparian ecosystems: case study of a dryland, urban river. *Restoration Ecology*. 19(1):101-111.

Whittaker, R.H. 2006. Ordination and Classification of Plant Communities. Longman, London.

Wilhelm, G.S. 1977. Ecological Assessment of Open Land Areas in Kane County, Illinois. Kane County Urban Development Division, Geneva, Illinois.

Wilhelm, G.S. and Ladd, D. 1988. Natural area assessment in Chicago region. *In:* Transactions of the 53rd North America Wildlife and Natural Resources Conference. Wildlife Management Institute, Washington, DC. 361-375.

Williams, N.S. 2005. The ecology of fragmented native grasslands in urban and rural landscapes. Melbourne: University of Melbourne. (Thesis: Ph.D).

Williams, N.S., Schwartz, M.W., Vesk, P.A., McCarthy, M.A., Hahs, A.K., Clemants, S.E., Corlett, R.T., Duncan, R.P., Norton, B.A., Thompson, K. and McDonnell, M.J. 2009. A conceptual framework for predicting the effects of urban environments on floras. *Journal of Ecology*. 97(1):4-9.

Wittmer, H., Schröter-Schlaack, C., Nesshöver, C., Bishop, J., ten Brink, P., Gundimeda, H., Kumar, P. and Simmons, B. 2010. The economics of ecosystems and biodiversity: mainstreaming the economics of nature: a synthesis of the approach, conclusions and recommendations of TEEB. TEEB.

Woods-Ballard B., Kellagher R., Martin P., Jefferies C., Bray R. and Shaffer P. 2007. The SUDS manual. CIRIA.

Working for Wetlands. 2015. Introduction into Working for Wetlands Programme. www.environment.gov.za/projectsprogrammes/workingfowetlands Date of access: 20 June 2015.

World Bank. 2012. http://www.worldbank.org/en/country/southafrica Date of access: 22 October 2015.

Wright, J.P., Jones, C.G. and Flecker, A.S. 2002. An ecosystem engineer, the beaver, increases species richness at the landscape scale. *Oecologia*. 132(1):96-101.

WWF (World Wildlife Fund). 2016. WWF Mondi Wetlands Program. http://www.wwf.org.za/what_we_do/freshwater/mwp/ Date of access: 19 September 2016

Yapp, G., Walker, J. and Thackway, R. 2010. Linking vegetation type and condition to ecosystem goods and services. *Ecological Complexity*. 7(3):292-301.

Zedler, J.B. and Kercher, S. 2004. Wetland resources: Status, trends, ecosystem services, and restorability. *Annual Review of Environment and Resources*. 30:39-74.

Zhang, C.B., Wang, J., Liu, W.L., Zhu, S.X., Ge, H.L., Chang, S.X., Chang, J. and Ge, Y. 2010. Effects of plant diversity on microbial biomass and community metabolic profiles in a full-scale constructed wetland. *Ecological Engineering*. 36(1):62-69.

Zhu, S.X., Zhang, P., Wang, H., Ge, H.L., Chang, J., Chang, S., Qiu, Z., Shao, H. and Ge, Y., 2012. Plant Species Richness Affected Nitrogen Retention and Ecosystem Productivity in a Full-Scale Constructed Wetland. *CLEAN–Soil, Air, Water*. 40(4):341-347.

Ziter, C. 2015. The biodiversity–ecosystem service relationship in urban areas: a quantitative review. *Oikos*. 125(6):761-768.

Zorrilla-Miras, P., Palomo, I., Gómez-Baggethun, E., Martín-López, B., Lomas, P.L. and Montes, C. 2014. Effects of land-use change on wetland ecosystem services: A case study in the Donana marshes (SW Spain). *Landscape and Urban Planning*. 122:160-174.

APPENDIX A

Table A1: Complete list of all identified plant species identified within the TMA and their trait data

Species	Growth Form	Wetland Indicator Species	Classification	Status	Coefficient of conservatism	Ecological index
Agrostis continuata	Graminoid	Obligate wetland	Monocotyledons	Indigenous	6	1
Albuca setosa	Herb	Facultative dryland	Monocotyledons	Indigenous	6	4
Alternanthera sessilis	Herb	Facultative dryland	Dicotyledons	Alien	0	4
Ambrosia psilostachya	Herb	Facultative dryland	Dicotyledons	Alien	0	4
Ammi majus	Herb	Facultative dryland	Dicotyledons	Alien	0	4
Andropogon appendiculatus	Graminoid	Facultative wetland	Monocotyledons	Indigenous	10	2
Anthospermum herbaceum	Herb	Facultative dryland	Dicotyledons	Indigenous	4	4
Asparagus laricinus	Shrub	Facultative dryland	Monocotyledons	Indigenous	4	4
Aster squamatus	Herb	Facultative dryland	Dicotyledons	Alien	0	4
Berkheya setifera	Herb	Facultative wetland	Dicotyledons	Indigenous	1	2
Berula erecta	Herb	Facultative wetland	Dicotyledons	Indigenous	1	2
Bidens bipinnata	Herb	Facultative dryland	Dicotyledons	Alien	0	5
Bidens pilosa	Herb	Facultative dryland	Dicotyledons	Alien	0	5
Brachiaria eruciformis	Graminoid	Facultative dryland	Monocotyledons	Indigenous	1	4
Carex acutiformis	Herb	Obligate wetland	Monocotyledons	Indigenous	10	1
Carex glomerabilis	Herb	Obligate wetland	Monocotyledons	Indigenous	10	1
Chenopodium album	Herb	Obligate dryland	Dicotyledons	Alien	0	5
Cichorium intybus	Herb	Facultative dryland	Dicotyledons	Alien	0	4
Cirsium vulgare	Herb	Facultative dryland	Dicotyledons	Alien	0	4
Conyza bonariensis	Herb	Facultative dryland	Dicotyledons	Alien	0	4
Cotula australis	Herb	Facultative dryland	Dicotyledons	Indigenous	1	4
Crinum bulbispermum	Geophyte	Facultative wetland	Monocotyledons	Indigenous	10	2

Species	Growth Form	Wetland Indicator Species	Classification	Status	Coefficient of conservatism	Ecological index
Cynodon dactylon	Graminoid	Facultative dryland	Monocotyledons	Indigenous	1	4
Cynodon transvaalensis	Graminoid	Facultative dryland	Monocotyledons	Indigenous	1	4
Cyperus congestus	Graminoid	Facultative wetland	Monocotyledons	Indigenous	6	2
Cyperus eragrostis	Graminoid	Facultative wetland	Monocotyledons	Alien	0	2
Cyperus esculentus	Graminoid	Facultative wetland	Monocotyledons	Indigenous	1	2
Cyperus fastigiatus	Graminoid	Obligate wetland	Monocotyledons	Indigenous	10	1
Cyperus laevigatus	Graminoid	Obligate wetland	Monocotyledons	Indigenous	10	1
Cyperus longus	Graminoid	Obligate wetland	Monocotyledons	Indigenous	10	1
Cyperus marginatus	Graminoid	Facultative wetland	Monocotyledons	Indigenous	6	2
Echinochloa colona	Graminoid	Facultative wetland	Monocotyledons	Indigenous	4	2
Echinochloa holubii	Graminoid	Facultative wetland	Monocotyledons	Indigenous	4	2
Echinochloa pyramidalis	Graminoid	Facultative wetland	Monocotyledons	Indigenous	4	2
Eleocharis dregeana	Herb	Facultative wetland	Monocotyledons	Indigenous	10	2
Eragrostis curvula	Graminoid	Facultative dryland	Monocotyledons	Indigenous	10	4
Eragrostis heteromera	Graminoid	Facultative dryland	Monocotyledons	Indigenous	4	4
Eragrostis micrantha	Graminoid	Facultative dryland	Monocotyledons	Indigenous	4	4
Eragrostis plana	Graminoid	Facultative	Monocotyledons	Indigenous	4	3
Euphorbia helioscopia	Herb	Facultative dryland	Dicotyledons	Alien	0	4
Falckia oblonga	Herb	Facultative wetland	Dicotyledons	Indigenous	1	2
Festuca caprina	Graminoid	Facultative dryland	Monocotyledons	Indigenous	10	4
Fuirena pachyrrhiza	Herb	Obligate wetland	Monocotyledons	Indigenous	10	1
Fuirena pubescens	Herb	Obligate wetland	Monocotyledons	Indigenous	10	1
Gleditsia triacanthos	Shrub	Obligate dryland	Dicotyledons	Alien	0	5
Gomphrena celosiodes	Herb	Obligate dryland	Dicotyledons	Alien	0	5
Haplocarpha lyrata	Herb	Facultative dryland	Dicotyledons	Indigenous	10	4
Helichrysum krausii	Herb	Facultative dryland	Dicotyledons	Indigenous	10	4

Species	Growth Form	Wetland Indicator Species	Classification	Status	Coefficient of conservatism	Ecological index
Hemarthria altissima	Graminoid	Facultative wetland	Monocotyledons	Indigenous	10	2
Hibiscus trionum	Herb	Facultative	Dicotyledons	Indigenous	1	3
Hydrocotyle verticillata	Herb	Facultative	Dicotyledons	Indigenous	6	3
Juncus punctorius	Herb	Obligate wetland	Monocotyledons	Indigenous	10	1
Juncus rigidus	Herb	Obligate wetland	Monocotyledons	Indigenous	10	1
Kniphofia ensifolia	Herb	Facultative	Monocotyledons	Indigenous	10	3
Lactuca capensis	Herb	Facultative dryland	Dicotyledons	Indigenous	4	4
Lactuca inermis	Herb	Facultative dryland	Dicotyledons	Indigenous	4	4
Leersia hexandra	Graminoid	Obligate wetland	Monocotyledons	Indigenous	4	1
Lobelia thermalis	Herb	Facultative dryland	Dicotyledons	Indigenous	1	4
Marsilea capensis	Herb	Facultative wetland	Monocotyledons	Indigenous	1	2
Medicago sativa	Herb	Obligate dryland	Dicotyledons	Alien	0	5
Melilotus alba	Herb	Facultative	Dicotyledons	Alien	0	3
Mentha aquatica	Herb	Facultative wetland	Dicotyledons	Indigenous	10	2
Modiola caroliniana	Herb	Facultative dryland	Dicotyledons	Alien	0	4
Moraea thomsonii	Herb	Facultative dryland	Monocotyledons	Indigenous	1	4
Oenothera rosea	Herb	Facultative dryland	Dicotyledons	Alien	0	4
Oxalis corniculata	Herb	Facultative dryland	Dicotyledons	Alien	0	4
Panicum coloratum	Graminoid	Facultative dryland	Monocotyledons	Indigenous	10	4
Paspalum dilatatum	Graminoid	Facultative wetland	Monocotyledons	Alien	0	2
Paspalum distichum	Graminoid	Obligate wetland	Monocotyledons	Indigenous	10	1
Pennisetum clandestinum	Graminoid	Facultative wetland	Monocotyledons	Alien	0	2
Persicaria decipiens	Herb	Facultative	Dicotyledons	Indigenous	1	3
Persicaria lapathifolia	Herb	Facultative	Dicotyledons	Alien	0	3
Persicaria senegalensis	Herb	Facultative	Dicotyledons	Indigenous	1	3
Phragmites australis	Graminoid	Obligate wetland	Monocotyledons	Indigenous	10	1
Plantago lanceolata	Herb	Facultative dryland	Dicotyledons	Indigenous	6	4

Species	Growth Form	Wetland Indicator Species	Classification	Status	Coefficient of conservatism	Ecological index
Plantago major	Herb	Facultative dryland	Dicotyledons	Indigenous	6	4
Pycreus macranthus	Herb	Obligate wetland	Monocotyledons	Indigenous	10	1
Pycreus macrostachyos	Herb	Obligate wetland	Monocotyledons	Indigenous	10	1
Pycreus nitidus	Herb	Obligate wetland	Monocotyledons	Indigenous	10	1
Rumex crispus	Herb	Facultative wetland	Dicotyledons	Alien	0	2
Rumex lanceolatus	Herb	Facultative	Dicotyledons	Indigenous	1	3
Salix babylonica	Tree	Facultative wetland	Dicotyledons	Alien	0	2
Schoenoplectus brachyceras	Herb	Obligate wetland	Monocotyledons	Indigenous	10	1
Schoenoplectus corymbosus	Herb	Obligate wetland	Monocotyledons	Indigenous	10	1
Sesbania bispinosa	Herb	Facultative dryland	Dicotyledons	Alien	0	4
Sesbania transvaalensis	Herb	Facultative dryland	Dicotyledons	Indigenous	1	4
Setaria incrassata	Graminoid	Facultative dryland	Monocotyledons	Indigenous	10	4
Setaria pallide-fusca	Graminoid	Facultative	Monocotyledons	Indigenous	1	3
Sisymbrium thellungii	Herb	Facultative dryland	Dicotyledons	Alien	0	4
Solanum retroflexum	Herb	Facultative dryland	Dicotyledons	Indigenous	4	4
Sonchus oleraceus	Herb	Facultative dryland	Dicotyledons	Alien	0	4
Sonchus wilmsii	Herb	Facultative dryland	Dicotyledons	Indigenous	4	4
Sporobolus fimbriatus	Graminoid	Facultative dryland	Monocotyledons	Indigenous	10	4
Taraxacum officinale	Herb	Facultative dryland	Dicotyledons	Alien	0	4
Trifolium africanum	Herb	Facultative	Dicotyledons	Indigenous	4	3
Trifolium repens	Herb	Facultative	Dicotyledons	Alien	0	3
Typha capensis	Herb	Obligate wetland	Monocotyledons	Indigenous	10	1
Verbena bonariensis	Herb	Facultative dryland	Dicotyledons	Alien	0	4
Verbena brasiliensis	Herb	Facultative dryland	Dicotyledons	Alien	0	4
Verbena officinalis	Herb	Facultative dryland	Dicotyledons	Alien	0	4
Veronica anagalis-aquatica	Herb	Facultative	Dicotyledons	Alien	0	3
Xanthium strumarium	Herb	Obligate dryland	Dicotyledons	Alien	0	5

Table B1: Score sheet of the ecosystem services being delivered by all the wetland within the TMA

Condensed summary sheet	Wetland unit 1		Wetland unit 2		Wetland unit 3		Wetland unit 4		Wetland unit 5	
Hydro-geomorphic setting	v		v		vc		vc		vc	
Size	31,58		17,01		2,85		4,24		1,76	
	Overall score	Confidence rating	Overall score	Confidence rating	Overall score	Confidence rating	Overall score	Confidence rating	Overall score	Confidence rating
Flood attenuation	2,6	2,8	2,1	2,5	1,8	2,5	2,0	1,2	2,0	1,1
Streamflow regulation	0,8	2,8	1,3	2,8	1,3	3,0	1,4	2,7	1,4	2,7
Sediment trapping	1,4	3,0	2,1	2,7	1,8	2,9	1,7	2,2	2,2	2,2
Phosphate trapping	1,6	2,6	1,4	2,8	1,8	2,8	1,7	2,7	2,1	2,6
Nitrate removal	1,6	2,6	1,1	2,6	1,4	2,7	1,6	2,4	1,9	2,3
Toxicant removal	1,5	2,9	1,3	2,7	1,7	2,7	1,7	2,4	2,1	2,3
Erosion control	3,5	2,4	3,3	2,4	2,9	2,3	2,9	1,6	3,1	1,6
Carbon storage	1,7	2,7	1,3	2,7	1,0	2,3	1,0	1,7	1,0	1,7
Maintenance of biodiversity	1,4	2,8	1,3	3,1	1,4	2,9	1,2	2,8	1,3	2,8
Water supply for human use	0,5	3,1	0,4	1,5	1,1	2,5	1,1	2,3	1,1	2,3
Natural resources	0,6	3,6	0,6	3,6	0,8	3,0	1,6	3,0	1,6	3,0
Cultivated foods	0,0	3,6	0,0	3,0	0,4	3,0	1,2	3,0	1,2	3,0
Cultural significance	0,3	3,0	0,3	3,3	0,3	3,3	1,3	3,3	1,3	3,3
Tourism and recreation	1,9	2,4	2,1	4,0	0,4	3,4	0,6	3,4	0,6	3,4
Education and research	3,0	3,3	2,8	4,0	1,8	2,3	1,5	3,0	1,5	3,0
Threats	1,0	3,0	1,0	2,0	1,0	3,0	1,0	3,0	1,0	3,0
Opportunities	1,0	3,0	1,0	2,0	1,0	3,0	1,0	3,0	1,0	3,0

Condensed summary sheet	Wetland unit 6		Wetland unit 7		Wetland unit 8		Wetland unit 9		Wetland unit 10	
Hydro-geomorphic setting	v		v		v		v		vc	
Size	2,95		3,21		3,41		1,14		3,79	
	Overall score	Confidence rating	Overall score	Confidence rating	Overall score	Confidence rating	Overall score	Confidence rating	Overall score	Confidence rating
Flood attenuation	2,0	1,2	2,0	1,1	2,0	1,2	2,0	1,2	2,0	1,1
Streamflow regulation	1,4	2,7	1,4	2,7	1,4	2,7	1,4	2,7	1,4	2,7
Sediment trapping	2,2	2,4	1,7	2,2	1,7	2,4	2,2	2,2	2,2	2,2
Phosphate trapping	2,1	2,8	1,7	2,7	1,7	2,8	2,0	2,6	2,1	2,6
Nitrate removal	1,8	2,6	1,5	2,6	1,5	2,6	1,8	2,4	1,6	2,4
Toxicant removal	2,1	2,4	1,7	2,4	1,7	2,4	2,0	2,3	2,1	2,3
Erosion control	2,6	1,6	3,1	1,6	2,6	1,7	2,7	1,6	2,6	1,6
Carbon storage	0,5	1,7	1,0	1,7	1,0	1,7	0,5	1,7	0,5	1,7
Maintenance of biodiversity	1,3	2,9	1,4	2,8	1,3	2,8	1,3	2,9	1,3	2,8
Water supply for human use	1,1	2,3	1,1	2,3	1,1	2,3	1,1	2,3	1,1	2,3
Natural resources	1,6	3,0	1,6	3,0	1,6	3,0	1,6	3,0	1,6	3,0
Cultivated foods	1,2	3,0	1,2	3,0	1,2	3,0	1,2	3,0	1,2	3,0
Cultural significance	1,3	3,3	1,3	3,3	1,3	3,3	1,3	3,3	1,3	3,3
Tourism and recreation	0,4	3,4	0,6	3,4	0,7	3,4	0,7	3,4	0,6	3,4
Education and research	1,5	3,0	1,5	3,0	1,3	3,0	1,3	3,0	1,3	3,0
Threats	1,0	3,0	1,0	3,0	1,0	3,0	1,0	3,0	1,0	3,0
Opportunities	1,0	3,0	1,0	3,0	1,0	3,0	1,0	3,0	1,0	3,0

Condensed summary sheet	Wetland unit 11		Wetland unit 12		Wetland unit 13		Wetland unit 14	
Hydro-geomorphic setting	vc		vc		vc		vc	
Size	2,58		4,79		1,47		1,47	
	Overall score	Confidence rating	Overall score	Confidence rating	Overall score	Confidence rating	Overall score	Confidence rating
Flood attenuation	2,0	1,2	2,0	1,1	2,0	1,1	2,0	1,1
Streamflow regulation	1,4	2,7	1,4	2,7	1,4	2,7	1,4	2,7
Sediment trapping	2,2	2,4	2,2	2,4	1,7	2,4	1,7	2,2
Phosphate trapping	2,0	2,6	1,7	2,6	1,6	2,6	1,6	2,6
Nitrate removal	1,5	2,4	1,3	2,4	1,3	2,4	1,3	2,0
Toxicant removal	2,0	2,3	1,7	2,3	1,6	2,3	1,6	2,3
Erosion control	2,6	1,6	2,9	1,6	2,9	1,6	2,9	1,6
Carbon storage	0,5	1,7	1,0	1,7	1,0	1,7	0,5	1,7
Maintenance of biodiversity	1,4	2,8	1,4	2,8	1,4	2,8	1,4	2,8
Water supply for human use	1,1	2,3	1,1	2,3	1,1	2,3	1,1	2,3
Natural resources	1,6	3,0	1,6	3,0	1,6	3,0	1,6	3,0
Cultivated foods	1,2	3,0	1,2	3,0	1,2	3,0	1,2	3,0
Cultural significance	1,3	3,3	1,3	3,3	1,3	3,3	1,3	3,3
Tourism and recreation	0,4	3,4	0,7	3,4	0,4	3,4	0,9	3,4
Education and research	1,5	3,0	1,3	3,0	1,0	3,0	0,8	3,0
Threats	1,0	3,0	1,0	3,0	1,0	3,0	1,0	3,0
Opportunities	1,0	3,0	1,0	3,0	1,0	3,0	1,0	3,0