# Restoring the Kuils River:
# Understanding the Past to Inform the Future

by

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# Declaration

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# Abstract

This study examined the state of the Kuils River catchment in Cape Town, South Africa, in terms of land-use change, ecosystem services, and water quality, to identify potential drivers of degradation and pathways for river ecosystem repair, which will help guide future rehabilitation and restoration efforts. To begin, the status of the Kuils River was analysed using historical and present Land-Use/Land-Change (LULC) data to create a baseline for rehabilitation or restoration. This was accomplished by mapping LULC across time (from 1944 to 2021) inside a 2.5 km radius of the Kuils River using GIS technology. Statistical assessments found that urbanisation and related anthropogenic activities were the primary drivers of alterations to the Kuils River. The extent of alien plants has increased because of these changes, and the system has shifted from a seasonal to a permanent wet state. Some of the Kuils River's effects may be mitigated by creating more sustainable green places. Given the Kuils River's urban environment and severe disruption, rehabilitation rather than restoration is the most likely level of action. In addition to other management alternatives such as building more sustainable green spaces along the Kuils River, sections of the river should be prioritised for restoration, with a special emphasis on alien vegetation eradication.

The reasons for the Kuils River's water quality variations were investigated using long-term water quality trends (1974-2021) for numerous physical, chemical, and biological variables. River quality indicators all indicate a stressed urban river system, with major variables indicating deteriorating quality over time and across space, likely due to urbanisation-related land-use change. The Bottelary River, a very polluted tributary, as well as other anthropogenic influences including urban runoff, have been recognised as important drivers of these developments. The main contributors to poor water quality were organic contaminants, nutrients, pH, toxicants, and salts. As a result, concentrating on the primary point sources should be the first step in moderating or correcting such deteriorating water quality trends.

The WET-Health and WET-EcoServices programs were used to examine four wetlands along the Kuils River in the hopes of identifying river portions that could be rehabilitated. These techniques helped assess the existing (and future) health of wetlands and the ecosystem services they provide, and found that the wetlands along the Kuils River are deteriorating, with ecological thresholds likely surpassed. In the Kuils River, restorative activities should strive to increase ecosystem service delivery while maintaining some ecological integrity in these wetlands. Rehabilitating existing, larger wetlands, such as the Kuils River corridor, would be necessary. Controlling invasive species and improving hydrological and geomorphological conditions (e.g. by creating floodplain-like areas and wetlands to trap nutrients in sediment

and changing the hardened channels to mimic the natural hydrology of the system) are examples of specific efforts.

The final chapter covers possible remedial actions. Although the findings of this study call for immediate improvement in the status of the Kuils River, they also show that a typical restorative effort to return it to its historic state is unattainable. There are recommendations for restorative measures that could be implemented in the Kuils River, with an emphasis on alien vegetation management, ecosystem service delivery, and improved water quality.

# Opsomming

Hierdie studie ondersoek die stand van die Kuilsrivier-opvangsgebied in Kaapstad, Suid-Afrika, ten opsigte van verandering in grondgebruik, ekosisteemdienste en waterkwaliteit, ten einde potensiële drywers van agteruitgang en paaie vir die herstel van die rivierekosisteem te identifiseer, wat sal help lei tot toekomstige rehabilitasie- en herstelpogings. Om mee te begin, word die status van die Kuilsrivier geanaliseer met behulp van historiese en huidige data oor grondgebruik / grondverandering (LULC) om 'n basis vir rehabilitasie of herstel te skep. Dit is bewerkstellig deur LULC oor tyd (van 1944 tot 2021) binne 'n radius van 2,5 km van die Kuilsrivier met behulp van GIS-tegnologie te karteer. Statistiese beoordelings het bevind dat verstedeliking en verwante antropogeniese aktiwiteite die belangrikste dryfveer was vir veranderinge aan die Kuilsrivier. Die gebied van uitheemse plante het toegeneem as gevolg van hierdie veranderinge, en die stelsel het van 'n seisoenale na 'n permanente nat toestand verskuif. Sommige van die gevolge van die Kuilsrivier kan versag word deur meer volhoubare groen plekke te skep. Gegewe die stedelike omgewing van die Kuilsrivier en ernstige ontwrigting, is rehabilitasie eerder as herstel die waarskynlikste aksie. Benewens ander bestuursalternatiewe, soos die bou van meer volhoubare groen ruimtes langs die Kuilsrivier, moet gedeeltes van die rivier geprioritiseer word vir herstel, met 'n spesiale klem op die uitwissing van uitheemse plantegroei.

Die redes vir die Kuilsrivier se variasies in waterkwaliteit is ondersoek met behulp van langtermyn waterkwaliteitstendense (1974-2021) vir talle fisiese, chemiese en biologiese veranderlikes. Aanwysers vir rivierkwaliteit dui almal aan op 'n beklemtoonde stedelike rivierstelsel, met groot veranderlikes wat die kwaliteit van die tyd en oor die ruimte vererger, waarskynlik as gevolg van verstedeliking-verwante verandering in grondgebruik. Die Bottelary rivier, 'n baie besoedelde sytak, sowel as ander antropogene invloede, insluitend stedelike afloop, is erken as belangrike drywers van hierdie ontwikkelings. Die belangrikste bydraers tot die swak watergehalte was organiese kontaminante, voedingstowwe, pH, toksiese middels en soute. As gevolg hiervan, moet konsentrasie op die primêre puntbronne die eerste stap wees om sulke verslegtende neigings te modereer of reg te stel.

Die WET-Health- en WET-EcoServices-programme is gebruik om vier vleilande langs die Kuilsrivier te ondersoek in die hoop om riviergedeeltes te identifiseer wat gerehabiliteer kon word. Hierdie tegnieke het gehelp om die bestaande (en toekomstige) gesondheid van vleilande en die ekosisteemdienste wat gelewer word, te beoordeel, en hulle het gevind dat die vleilande langs die Kuilsrivier agteruitgaan, met ekologiese drempels waarskynlik oortref. In die Kuilsrivier moet herstellende aktiwiteite daarna streef om ekosisteemdienste te verhoog, terwyl die ekologiese integriteit in hierdie vleilande gehandhaaf word. Dit sou nodig wees om

bestaande, groter vleilande, soos die Kuilsriviergang, te rehabiliteer. Die beheer van indringerspesies en die verbetering van hidrologiese en geomorfologiese toestande (deur byvoorbeeld vloedvlakteagtige gebiede en vleilande te skep om voedingstowwe in sediment vas te vang en die verharde kanale te verander om die natuurlike hidrologie van die stelsel na te boots) is voorbeelde van spesifieke pogings.

Die laaste hoofstuk dek moontlike regstellende aksies. Ondanks die feit dat die bevindinge van hierdie studie onmiddellike verbetering in die status van die Kuilsrivier vereis, toon hulle ook dat tipiese herstellende poging om dit na sy historiese toestand terug te keer, onbereikbaar is. Daar is aanbevelings vir herstellende maatreëls wat in die Kuilsrivier geïmplementeer kan word, met die klem op uitheemse plantegroei, lewering van ekosisteemdienste en verbeterde waterkwaliteit.

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# Acronyms and Abbreviations

| | |
|---|---|
| ANOVA | Analysis of Variance |
| AU | Assessment Unit |
| CD:NGI | Chief Directorate: National Geospatial Information |
| COD | Chemical Oxygen Demand |
| DO | Dissolved Oxygen |
| E. coli | Escherichia coliform |
| EC | Electrical Conductivity |
| F. coli | Faecal coliform |
| FRESH | Fountain River Environmental Sanctuary Hennops |
| GIS | Geospatial Information System |
| KR | Kuils River |
| KRB | Bellville WWTF discharge into the Kuils River |
| KRZ | Zandvliet WWTF discharge into the Kuils River |
| LULC | Land-Use/Land-Cover |
| $NH_3$ | Ammonia |
| $NH_4$ | Ammonium ion |
| NWQG | South African National Water Quality Guidelines |
| OP | Orthophosphates |
| PES | Present Ecological State |
| POS | Percentage Oxygen Saturation |
| SAEON | South African Environment Observation Network |
| SDGs | Sustainable Development Goals |
| TDS | Total Dissolved Solids |
| The Standards | The International Principles and Standards for the Practice of Ecological Restoration |
| TP | Total Phosphorous |
| TSS | Total Suspended Solids |
| USA | United States of America |
| WRC | Water Research Commission |
| WWTF | Wastewater Treatment Facility |

# Definitions

| | |
|---|---|
| Absolute change | A computational measure is used to describe the difference in the indicator over only two time periods (refer to Chapter Three). |
| Environment | "The complex physical, chemical, and biotic factors (such as climate, soil, and living things) that act upon an organism or an ecological community and ultimately determine its form and survival" (Merriam-Webster, 2021). |
| Rehabilitation | The reinstatement of a level of ecosystem functioning for the renewed and ongoing provision of ecosystem services potentially derived from non-native ecosystems as well (Gann et al., 2019, p. S35). |
| Relative change | Absolute change as a percentage (refer to "Absolute change" above). |
| Restoration (or restore) | The process of assisting the recovery of an ecosystem that has been degraded, damaged or destroyed to its original state (Gann et al., 2019, p. S35). |
| Restorative or rehabilitative action | Activities or management actions used in the restoration, rehabilitation or enhancement of ecosystems and could include passive restoration, active restoration or management. |
| River continuum concept | The idea that a river is a series of physical gradients and a continuum of associated biotic adjustments that continuously changes with the flow of the river from the headwaters to the river mouth. |
| Watercourses (sometimes used interchangeably with "waterbodies") | "A river or spring, a natural channel in which water flows regularly or intermittently, a wetland, lake or dam into which, or from which, water flows, and any collection of water which the Minister may, by notice in the Gazette, declare to be a watercourse, and a reference to a watercourse includes, where relevant, its bed and banks" (Republic of South Africa, 1998a, p. 17). |
| Wetlands | "Land which is transitional between terrestrial and aquatic systems where the water table is usually at or near the surface, or the land is periodically covered with shallow water, and which land in normal circumstances supports or would support vegetation |

| | |
|---|---|
| | typically adapted to life in saturated soil" (Republic of South Africa, 1998a, p. 18). |

# Chapter 1
# Introduction

Human activities continuously affect the functioning of the earth to such a degree that its ability to bounce back may be restricted (Steffen, Richardson, Rockström and Cornell, 2015). Nine key earth system processes, or planetary boundaries, have been identified as processes (and limits) that can result in unacceptable environmental change and catastrophic impacts on human development (Rockström, Steffen, Noone, Persson, Chapin et al., 2009). The development of the planetary boundary concept has assisted in the recognition of the need for sustainable living and – development (Griggs et al., 2013; Randers et al., 2019). One of the planetary boundaries is 'freshwater use'; it is closely related to the sustainable development goal 'water and sanitation for all' (Rockström et al., 2009; Cole, Bailey, Cullis and New, 2018). Even though the planetary boundary for freshwater use is far from being crossed, research suggests that only 47% of all waterbodies in South Africa are of good ambient quality in comparison to the global 72% (Ritchie, Roser, Mispy, Ortiz-Ospina 2018; StatsSA, 2019). While it is widely recognised that water is crucial for all development on earth, water systems have experienced a decline in water quality and biodiversity (Bunn, Davies and Mosisch, 1999; Bernhardt and Palmer, 2011). Understanding the past causes of ecological degradation of water systems is key to uncover the future restorative and rehabilitative options for these systems, while also reducing the impacts that these systems experience. This is particularly true for urban water systems where additional human-induced pressures contribute to the further degradation of these water systems (Mwangi, 2014).

As most of the world is now influenced by humans and human activities (Vitousek, Mooney, Lubchenco and Melillo, 1997; Ellis, Goldewijk, Siebert, Lightman and Ramankutty, 2010), the need to focus on ecosystem impacts, and their mitigation has also increased. In many cases, drivers of change have resulted in the creation of novel ecosystems. Such ecosystems are typically characterised by firstly new species combinations, together with the potential change in ecosystem functioning and secondly the presence of humans or human activity, which can occur either because of the degradation of 'wild' ecosystems and/or the introduction of invasive species, or the abandonment of intensely modified areas, such as croplands (Hobbs et al., 2006). Many examples of novel ecosystems exist; they are especially common in the urban environment associated with urban rivers and wetlands, where extensive human alteration and the introduction of alien species has resulted in a complete change of the structure and functioning of such ecosystems.

Large scale urbanisation along river catchments has resulted in extensive ecological damage (Chin, 2006; Findlay and Taylor, 2006; Adámek, Orendt, Wolfram and Sychra, 2010; Meyer, 2010; Bagalwa and Majaliwa, 2013). Due to the complexity of and the impacts on urban water systems, the emerging novel systems often require novel interventions. However, these interventions are often focused on enhancing societal benefit rather than ecosystem integrity. This is problematic as the ability of an ecosystem to provide the necessary services is dependent on the proper ecological functioning of that system (Zhang et al., 2018; Shackleton, Cilliers, Davoren and du Toit, 2021). As river systems are highly interconnected with other terrestrial and aquatic ecosystems, the need for better management and an improved understanding of the pressures that these rivers face is critical. Reducing the effects of urbanisation on urban rivers and rehabilitating or restoring these rivers can have profound ecological, social and economic impacts and benefits on the environment and the people living along or close to the river.

There is therefore an increasing need to assess the current condition of urban systems, especially urban water systems such as rivers and wetlands, to suggest appropriate management and restorative actions. Elsewhere in South Africa, such initiatives have resulted in positive recovery. For example, in Centurion, South Africa, an urban river restoration project was established to clean the Hennops River that is polluted with tons of litter, agricultural and industrial chemicals, and sewage (ARMOUR, 2018; Maako, 2018). A programme initiated in 2018, Fountain River Environmental Sanctuary Hennops (FRESH) aimed to clean up the Hennops River and surrounding wetlands while protecting and maintaining the natural ecosystem (FRESH, 2020). While the pollution has resulted in extensive knock-on impacts to the river (Bartl, 2018; Maako, 2018), continuous river clean-up efforts, such as that in the Kaalspruit, a tributary of the Hennops River, has resulted in the partial passive recovery of the Kaalspruit (Bega, 2020). Riparian vegetation along the Hennops riverbanks has also been re-established through the planting of trees, removal of sludge and reintroduction of fish and other aquatic species (FRESH, 2020). Attempts to rehabilitate the river have, therefore, resulted in clearer water and a reduction in the amount of litter that enters the river (ARMOUR, 2018; FRESH, 2020). While ecosystem services of the Hennops River have not been fully realised yet, public input into its restoration suggests that communities value the river for its aesthetics.

## 1.1 Problem Statement

The Kuils River has been impacted by alien vegetation, agricultural, urban and rural development, land-use and flow modification for many years (Wiseman and Sowman, 1992; RHP, 2005). As a result, it has been subject to degradation, such as changes to river morphology and structure, and declines in water quality and native biodiversity. The Kuils River has received much scientific (e.g. Fourie, 2005; Swart and Pool, 2007; Ayuk, 2008; Pool, 2008; Thomas, Chingombe, Ayuk and Scheepers, 2010; Chingombe, 2012; Mwangi, 2014; Dube et al., 2017) and public attention (e.g. Green, Solomon, Barnes and Petrik, 2019; GroundUp, 2019; Kretzmann, 2019).

Research on the Kuils River has been ongoing and ranges from the effects of land-use change on the water quality of the river, to the assessment of water pollution on the river. Land-use changes were previously mapped for the Kuils River (Thomas et al., 2010; Chingombe, 2012; Mwangi, 2014). While extensive research has been conducted on the river, this thesis aimed to compile this information and to update it where needed. There are new and more innovative techniques that can be used to assess the various aspects of the state of rivers and their ecosystem functioning. These include better techniques to assess land-use change as well as assessing the provision of ecosystem services and disservices, both included in this thesis.

## 1.2 Goals

### *1.2.1 Study Aim*

This study forms part of an interdisciplinary research approach assessing the ecological (this study), governance and social context of the Kuils River to develop a feasible governance framework for the restoration of the Kuils River. Focusing on the environmental aspect, this research aimed to compile and update a spatio-temporal understanding of the current and historic state of the Kuils River in terms of land-use change, ecosystem services and water quality to inform prospects for future restorative actions.

### *1.2.2 Research Objectives and Significance of the Study*

The study assesses how spatial and temporal changes in land-use have influenced the river and its associated ecosystem services, thereby explaining which land-use changes contributed the most to the degradation of the river. In addition, water quality assessments of the river were compiled, focusing on selected chemical, physical and biological properties. Lastly, the study presents an assessment of the ecosystem services provided by the Kuils River and suggests potential avenues for restoration or rehabilitation of the areas assessed. By linking land-cover, ecosystem services and water quality assessments, the research aimed to identify and suggest management tools that can be used for the Kuils River, while also serving as an example for future assessment of other urban rivers.

Chapter Two reviews the history of urban river restoration and urban novelty and introduces the three specific health assessment techniques used in the thesis – land-use/land-cover mapping, ecosystem service assessments and water quality. Data chapters address each of these techniques in more detail; Chapter Three analyses the impact and drivers of land-use change on the Kuils River, Chapter Four addresses the impact of land-use change on the water quality of the river over time and through space and Chapter Five assesses wetlands along the river to determine their ability to provide ecosystem services. Chapter Six then concludes with a summary, and potential recommendations, for the management of the Kuils River, given the findings of this study.

# Chapter 2
# Literature Review: Techniques Used to Assess River Health and Previous Work on the Kuils River

## 2.1 Introduction

Environmental degradation is receiving much global scientific and public attention as we grapple with the consequences to biodiversity and ecosystem services. Environmental degradation, defined as "…any change or disturbance to the environment perceived to be deleterious or undesirable" (Johnson et al., 1997), is recognised as a major disaster risk driver according to The Global Assessment Report on Disaster Risk Reduction (UNDRR, 2019). Attempts to limit environmental degradation have therefore been on the increase (IPBES, 2020). For example, many conventions, policies and intergovernmental platforms now address the problem of environmental degradation and its consideration in decision-making processes (Alexander, Aronson, Whaley and Lamb, 2016). This includes the Sustainable Development Goals (SDGs), designed to address urgent global ecological, economic and political challenges (United Nations General Assembly, 2012). As a signatory to the United Nations Agenda 2030 since 2015, the Republic of South Africa is committed to achieving the SDGs.

Section 24 of the Constitution of the Republic of South Africa together with the National Environmental Management Act 107 of 1998 (Republic of South Africa, 1996, 1998b) requires that ecological degradation and pollution are either avoided, minimised or remedied. This further emphasises the need for restoration, the prevention of degradation, and the need for ecosystem service assessments. As a developing country, many of the SDGs are applicable in South Africa. These relate to poverty and hunger reduction, health and well-being, education, inequality (including gender), access to clean water and sanitation, and climate action, among others. In addition to the SDGs developed by the UN, the Western Cape Government (2014a, 2014b) has also set goals that are aligned with that of the UN to improve human health and well-being.

As mentioned in Chapter One, urban watercourses are experiencing rapid degradation. According to the South African National Water Act (Act 36 of 1998; Republic of South Africa, 1998a), watercourses are defined as "(a) a river or spring, (b) a natural channel in which water flows regularly or intermittently, (c) a wetland, lake or dam into which, or from which, water flows, and (d) any collection of water which the Minister may, by notice in the Gazette, declare to be a watercourse, and a reference to a watercourse includes, where relevant, its bed and banks". This thesis focuses on urban rivers and their restoration or rehabilitation. To determine goals for such activities, the systems require an assessment of their ecological state. To this end, various techniques have been used to assess the ecological state, in particular river heath. These are discussed in Section 2.3 below.

## 2.2 Urban River Restoration

Ecological restoration, as a way to mitigate degradation, has received escalating attention, with the number of related publications increasing tremendously since the 1980s (Wortley, Hero and Howes, 2013; van Wilgen et al., 2020). Ecological restoration refers to "the process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed" (Gann et al., 2019) and aims to re-establish ecosystem characteristics such as biodiversity and ecosystem function. Research on ecological restoration has provided many insights for improving ecosystems and their sustainable management.

The Society for Ecological Restoration has developed International Principles and Standards for the Practice of Ecological Restoration (hereon, 'the Standards'), designed to provide the tools to ensure the proper investment in restoration projects globally (Gann et al., 2019). The Standards established a set of eight principles, one of which describes the various restoration activities that form part of the restorative continuum (Figure 2.1, Gann et al., 2019). The restorative continuum includes six categories of restorative activities (defined as actions, treatments and interventions that are intended to promote the recovery of the ecosystem(s) or components thereof; and includes 1) reducing societal impacts, 2) improving ecosystem management, 3) repairing ecosystem function, 4) initiating native recovery, 5) partially recovering native ecosystems, and 6) fully recovering native ecosystems (as seen in Figure 2**.1**, Gann et al., 2019).

For management to be effective, it is important to understand the difference between the various activities of restoration along the restorative continuum (Figure 2.1). While ecological restoration is defined as "the process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed", ecological rehabilitation is the "[reinstatement of] a level of ecosystem functioning for the renewed and ongoing provision of ecosystem services potentially derived from non-native ecosystems as well" (Gann et al., 2019, p. S35). Remediation, on the other hand, is "a management activity that aims to remove sources of degradation, including the removal or detoxification of contaminates or excess nutrients from soil and water" (Gann et al., 2019, p. S37), and lastly, reduced societal impacts relate to the societal actions that can be taken to reduce the negative impacts on the environment (Gann et al., 2019). A similar explanation of the different actions along this restorative gradient was also drawn up by Blignaut and Aronson (2020).

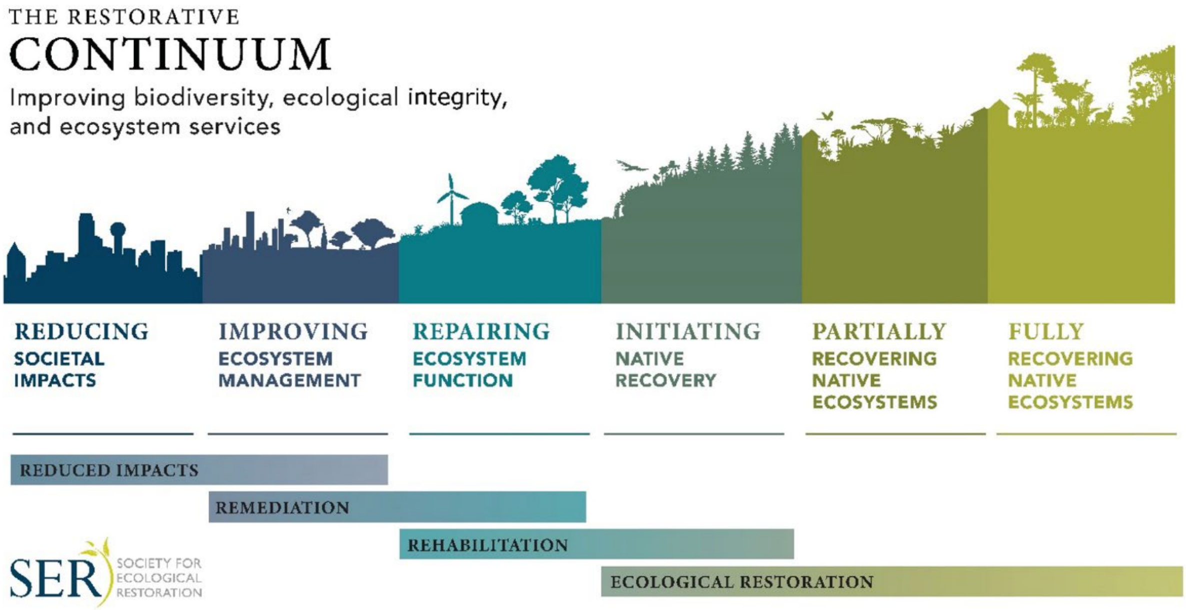



***Figure 2.1*** A graphic representation of the restorative continuum showing a range of activities and interventions that can improve environmental conditions and reverse ecosystem degradation and landscape fragmentation. Ecological health and biodiversity outcomes, and quantity and quality of ecosystem services, increase as one moves from left to right on the continuum. Ecological restoration can therefore occur in all kinds of environments, including urban, agricultural, and industrial landscapes. Figure from Gann et al. (2019).

Urban rivers provide many services to the communities living around them including wastewater attenuation, flood control, urban temperature regulation, recreation and aesthetic services (Palmer, Berold and Muller, 2004; Anderson and Elmqvist, 2012; Rebelo, Morris, Meire and Esler, 2019). However, due to large-scale urbanisation within their catchments, these rivers have an increased risk of potential environmental damage caused by the concentration of human activities, such as industrial discharge, treated and untreated effluent discharge (Martín-Vide, 2001; Forman, 2014) and river modifications such as the construction of channels, berms and infilled floodplains (Luger and Brown, 1999; Oberholster and Ashton, 2008; Pool, 2008; CSIR, 2010; Mwangi, 2014). This, in turn, results in changes in river and channel morphology, sediment deposition regime, bank erosion, loss of floodplains and stream biodiversity and a decline in the overall water quality and quantity due to various sources of pollution and water abstraction (Kleynhans, 1996; Withers, 2003; Findlay and Taylor, 2006; Everard and Moggridge, 2012; Mwangi, 2014). This is further influenced by the very nature of a river – the physical variables in a river system follows a continuous gradient of conditions and biotic interactions (known as the river continuum concept; Vannote, Minshall, Cummins, Sedell and Cushing, 1980). In other words, should the condition of the headwaters of the river change, it will most likely result in a change in the conditions of the river downstream.

Restorative activities, whether in the form of rehabilitation, remediation, or a full restoration, are thus necessary to combat the above-mentioned impacts. Based on the framings of Gann et al. (2019) and Blignaut and Aronson (2020), urban rivers are less likely to be fully recovered due to the widespread, ongoing degraded nature of the catchment, resource constraints, and mismanagement amongst other reasons. It is thus safe to assume that the best plan of action would be to rehabilitate or enhance (rather than restore) urban rivers to retain some ecological function and to reinstate the provision of a diversity of ecosystem services.

Reviewing restoration studies of aquatic systems has revealed the benefit of restoring or rehabilitating such systems. Elmqvist et al. (2015) found that economic investment into urban green spaces and restoring and rehabilitating urban aquatic environments is economically, socially and ecologically beneficial. An analysis of river restoration studies conducted by Vermaat et al. (2016) in Europe, showed that restoring river reaches increased their ability to provide ecosystem services, while simultaneously increasing the monetary ecosystem service value of these river reaches through increased willingness-to-pay. Similarly, for more than 75% of 13 South African wetlands assessed, restoration activities resulted in a 10-30% improvement in ecological condition from its pre-restoration baseline (Kotze, Tererai and Grundling, 2019). Additionally, ecosystem service provision increased substantially post-restoration (Kotze, Tererai and Grundling, 2019). Focussing specifically on the restoration of riparian vegetation, Richardson, Holmes, Esler and Galatowitsch (2007) reported that restoration activities would however, only be beneficial if context-specific restoration techniques are used.

Restoring urban rivers is highly complex due to the dependence on these systems to provide necessary services as well as the extensive impact that these systems have already experienced. Furthermore, urban rivers are embedded in social-ecological systems where the feedbacks are multiple; sometimes synergistic, sometimes antagonistic. As a result, urban rivers cannot be grouped under the same categories as rivers occurring in natural or rural settings. A focus on ecology alone is thus unlikely to be sufficient to undertake urban river restoration projects. However, the ecological context of urban river restoration is key to understanding these social-ecological systems. Studies have, for example, suggested the need for urban green spaces, and more particularly for urban rivers, to be recognised as novel ecosystems (Hobbs et al., 2006; Richardson et al., 2007; Francis, 2014). Novel ecosystems are characterised by changes in the composition of species assemblies and ecosystem functioning, as well as the involvement of humans (Hobbs et al., 2006) and require novel restoration techniques. It should be noted that many novel ecosystems cannot be restored to pre-historic conditions due to the substantial changes that they may have already faced (Hobbs et al., 2006; Francis, 2014). This is particularly true for urban rivers. Therefore, to understand which restoration technique(s) to apply, the condition of the system and drivers causing the change in its condition need to be assessed. In this study, novelty is referred to as anthropogenically impacted ecosystems, as defined by Hobbs et al. (2006).

## 2.3 River Health Assessment Techniques

Several methods are available to assess the condition of the environment and whether it is deteriorating. These include assessing changes in river hydrology and geomorphology, river water quality, riparian vegetation, bioindicators such as invertebrates and fish (amongst others), and habitat integrity. These proxies allow one to infer the state of the river, to assess changes in land-use surrounding the river, to assess which ecosystem services the river provides and to detect possible environmental disturbances (Angelier, 2003; RHP, 2006; Elmqvist et al., 2015). For example, using fish, Andres, Ribeyre, Tourencq and Boudou (2000) assessed the impact of cadmium and zinc discharges on the Lot River in France and found that pollution from an old zinc ore treatment facility had led to cadmium bioaccumulation in the sampled fish species. Direct water quality assessments through water sampling are therefore not the only technique that can be used to assess the water quality of polluted rivers; in this example, bioaccumulation in invertebrate and fish species was indicative of the current water quality state of the river.

In South Africa, the River Health Programme was developed in 1994, by the Department of Water Affairs and Forestry to assess the state of South Africa's aquatic ecosystems. For the programme, five indices were used to evaluate river health (RHP, 2006) and these included the assessment of water quality; the Index of Habitat Integrity (which assessed the impact of human disturbance factors on instream and riparian habitats); the Riparian Vegetation Index (assessing the degree of change in function and structure of riparian vegetation); the Fish Assemblage Integrity Index (assessing fish assemblages in river reaches and their range of sensitivity to environmental conditions); and lastly the South African Scoring System (which assesses aquatic invertebrate fauna and their sensitivity to water quality changes; RHP, 2006). The ecological state of various rivers across the country has been assessed using these indices. The River Health Programme does not, however, assess the provision of ecosystem services of each of these rivers, which is a gap if there is a need to justify management actions.

The remainder of this section unpacks the use of three specific techniques used to assess the status of rivers, with the emphasis on the uses of each of these techniques and with a specific focus on examples in South Africa. These techniques provide the basis for Chapters Three, Four and Five of this thesis.

### 2.3.1 *Land-Use/Land-Cover Mapping*

Land-use/land-cover (LULC) mapping is a well-developed technique that visually displays information about the current land-use of specific land units (Kraak and Ormeling, 2010; Govender, 2012). The applications for LULC mapping vary extensively – from urban planning (Tapiador and Casanova, 2003; Culshaw et al., 2006), mapping current transportation and other servitude lines, and using zoning maps for the economic evaluation of land resources (Kalogirou, 2002; Van Niekerk, 2008), to environmental management (including predicting stormwater runoff and drainage, planning for green space in urban areas and assessing fire risk – e.g. Lee, Woddy and Thompson, 2001). While land cover maps are used to describe the biophysical cover of the surface of the Earth, land-use maps are also used to describe how the biophysical environment has been altered, modified and used by humans (Cihlar and Jansen, 2001; Yang et al., 2017). Cihlar and Jansen (2001) reported a strong relationship between land-use and land change, with land-use being the result of the interaction between the natural environment and society. Understanding the land-use of a specific land unit can therefore improve the understanding of, and response to, human-induced environmental and global change. As a result, LULC maps are vital for various applications in planning, legislative, management and scientific domains (Cihlar and Jansen, 2001). The usefulness of LULC mapping has also been shown in environmental management, where, for example, land-use change of the Bot River catchment in South Africa was modelled to show changes in the catchment's rainfall-runoff quantity (Stipinovich, 2005). The model highlighted the impact of land-use change on the environmental integrity of the catchment, which has implications for the management of the catchment (Stipinovich, 2005).

Recently, with the development of new, innovative remote sensing technology, image processing and GIS software, it is now possible to analyse and compare various landscape features, including the analysis of LULC over time and space (Schneeberger, Bürgi and Kienast, 2007; Wilson and Schröder, 2008; Picuno, Tortora and Capobianco, 2011; Statuto, Cillis and Picuno, 2017). Understanding the drivers and causes of land-use change can inform future decision-making by, for example, identifying areas where protection is needed and where it can be more effective (Seto and Kaufmann, 2003; Franch-Pardo, Cancer-Pomar and Napoletano, 2017; Statuto, Cillis and Picuno, 2017). As in the case of the study by Franch-Pardo and colleagues (2017), GIS technology was used to identify the biophysical units of the Martin River in Aragon, Spain; these were further analysed to assess their fragility, quality and potential for protection. The maps generated could thus be used to identify the biophysical units needing prioritised protection.

In South Africa, LULC mapping at a large scale only officially started in 1994 through the development of various national and provincial LULC maps and the collaboration of private and governmental sectors (Ngcofe and Thompson, 2015). As international and national policies require frequent reporting of the state of the environment, LULC mapping needs to be detailed, accurate and should be updated regularly (Wilson and Schröder, 2008). Even though national LULC maps are produced regularly in South Africa, they are generated at national and provincial scales and are therefore too generalised to be useful on a local (such as a municipal) scale (Jantz and Goetz, 2005; Matsika, 2007). Local-scale land-use change maps are therefore needed to assess land-use change at a much finer scale.

Fine-scale assessments of land-use change of rivers and catchments are less common, although some good examples exist. Gajbhiye, Sharma and Jabalpur (2012), for example, assessed the impact of LULC change on the Indra River watershed in the Hoshangabad district, India over 14 years (from 1992 to 2006). Using both GIS and remote sensing data, they found that while forested land remained the dominant land-use class, a substantial increase in built-area was also observed (Gajbhiye, Sharma and Jabalpur, 2012). A similar study was conducted by Yorke and Margai (2007) where land-use change was mapped and analysed between 1990 and 2000 in the Densu River basin in Ghana. Here, it was found that one of the main drivers of land-use change was urban residential development. These studies emphasise the importance of using GIS and remote sensing technology in addressing the land-use problems and management of our growing global population.

In South Africa, Rebelo, Esler, Le Maitre and Cowling (2013) highlighted the need for land-use change assessments using the Kromme River as a case study. With historical imagery, the authors mapped river geomorphology over time, while also assessing the drivers of river change based on the assessment of temporal changes in land-use classes. It was found that agricultural development and alien invasion in the riparian areas drove the changes in river geomorphology (Rebelo et al., 2013). In a follow-up study, it was found that the degradation of the wetlands along the river and invasion of alien plants have reduced water-related ecosystem services provided by the catchment (Rebelo, Esler, Le Maitre and Cowling, 2015). These results were then used to suggest management actions based on catchment-specific problems identified in the study.

### *2.3.2 Water Quality*

One of the original ways to assess the state of waterbodies, including rivers, is the assessment of water quality. Water quality assessments involve the comparison between the physical, chemical and biological characteristics of a given water sample and a set of water quality standards or guidelines to determine the usability of the water (Western Cape Department of Environmental Affairs & Development Planning, 2011). Typical water quality characteristics include temperature, pH, nutrients, metals, toxicants and bacteria such as *E. coli*. Water quality can change in time and space and can be influenced by a variety of environmental and anthropogenic factors (Western Cape Department of Environmental Affairs & Development Planning, 2011; Odume, 2017). The traditional assessment of water quality focussed on analysing the physical and chemical characteristics of water to manage and control water pollution (Odume, 2017), and was mainly done to assess whether the water was suitable for specific uses (UNESCO, WHO and UNEP, 1996). It was only a few decades later (i.e. 1980's) when water quality assessment and monitoring programmes were put in place to assess the quality of aquatic environments by analysing trends and assessing the impacts of contaminants, land-use change and wastewater on the system (UNESCO, WHO and UNEP, 1996).

Initially, water quality assessments were conducted by hand where containers, nets and other gadgets were used to trap and collect samples (WHO, 1978). With the help of technology, the assessment of water quality for various water quality characteristics has improved substantially. Today, in addition to the physical collection of data, water quality parameters are also modelled by using baseline data specific to the area in question (e.g. Tong and Chen, 2002; Ayuk, 2008; Mustapha and Abdu, 2012; Din, 2020).

The concerning degradation of the Earth's waters has led to the need for the continuous assessment of water quality. To do so, various monitoring programmes have been developed to ensure quantitative and qualitative assessments. These include the National Water-Quality Assessment Project of the United States of America (Hirsh, Alley and Wilberg, 1990) and the River Health Programme in South Africa. Starting in 1991, the National Water-Quality Assessment Project was one of the first national monitoring systems designed to assess the status and trends of the water quality of American rivers, lakes and wetlands and how anthropogenic and natural features have affected the water (Hirsh, Alley and Wilberg, 1990). Not only do these programmes set out to assess water quality trends, but such programmes have been proven to be useful in terms of understanding the sources of pollution and informing management strategies that would be most suitable for those specific waterways.

With most water quality assessments generally only assessing the physical and chemical aspects of water quality, trend analyses, as discussed above, are becoming more common. In South Africa, an extensive history of water quality data is available for most rivers (Palmer, Berold and Muller, 2004), which simplifies the assessment of water quality trends. As an example, Ngwenya (2006) assessed water quality trends of the Eerste River, South Africa between 1990 and 2005. Assessing changes in both time and space, it was found that not only was water quality decreasing over time, but that downstream sites were worse than upstream sites (Ngwenya, 2006). It was concluded that anthropogenic activities such as fish farming, a wastewater treatment plant and a polluted river were the cause of a deteriorating state of the river (Ngwenya, 2006). Management strategies should thus be focussed on such activities to ultimately address the water quality problem of the Eerste River. Similar studies have been conducted worldwide and have shown the power of trend analysis in water resource management (e.g. Antonopoulos, Papamichail and Mitsiou, 2001; Khan, Husain and Lumb, 2003; Ballantine and Davies-Colley, 2014; Mei et al., 2014).

### *2.3.3 Ecosystem Service Provision*

Considering the global move toward sustainability, cities, communities and companies are becoming more environmentally cautious. Driven by international conventions (e.g. TEEB, 2010; CBD, 2011; United Nations, 2017), policies (e.g. United Nations Sustainable Development, 1992) and intergovernmental platforms (e.g. IPBES, 2013), the ecosystem service concept is now being incorporated into decision-making at multiple levels (IUCN, 2014). The ecosystem services concept is a way to assess people’s dependence on the environment (Müller, Fohrer and Chicharo, 2015) and can build on the understanding of ecosystem change and governance (Haase et al., 2014). This is rooted in the examination of the benefits that people obtain from ecosystems, and also the processes that produce or support the production of ecosystem goods (Millennium Ecosystem Assessment, 2005a).

The Millennium Ecosystem Assessment (2002) framing divides ecosystem services into four categories, namely: provisioning – “the products people obtain from ecosystems”, regulating – “the benefits people obtain from the regulation of ecosystem processes”, supporting – “the services necessary for the production of all other ecosystem services”, and cultural services – “the nonmaterial benefits people obtain from ecosystems” (Millennium Ecosystem Assessment, 2002, p. 8, 2005c). Each of these constituents of human well-being can therefore, be assessed either independently or in combination with other services. Research has shown that quantifying and valuing ecosystem services and dis-services have profound effects on the success of the management of ecosystems, especially in urban environments (Lyytimäki, Petersen, Normander and Bezák, 2008; Lyytimäki and Sipilä, 2009; Cilliers, Cilliers, Lubbe and Siebert, 2013; Everard and Waters, 2013; Lyytimäki, 2014; Elmqvist et al., 2015), as ecosystem services are more easily quantified and understood (Everard and Waters, 2013).

The assessment of ecosystem services and dis-services in general, has taken various forms over the years, from biophysical valuation methods to economic assessments, to GIS-based assessments, and qualitative, empirical, multiscale and multi-institutional assessments (Costanza et al., 1997, 2014; Daily et al., 2000; Millennium Ecosystem Assessment, 2002, 2005b; Larondelle and Haase, 2013; Haase, Frantzeskaki and Elmqvist, 2014; Häyhä and Franzese, 2014; Elmqvist et al., 2015; Neugarten et al., 2018). Multiple assessments exist on the use, availability and success of ecosystem service assessment tools that are utilised all over the world (TEEB, 2011; Häyhä and Franzese, 2014; Harrison et al., 2018; Neugarten et al., 2018). While the majority of ecosystem service assessments are designed for natural systems, the assessment of ecosystem services in urban environments dates back to the early 1970s to 1980s (Haase, Frantzeskaki and Elmqvist, 2014). Fabos and Hendrix (1981) conducted an ecosystem service assessment on an agricultural and urban community and found that there were marked differences in the ecological profiles of these communities. For example, the agricultural community had less productive land and a smaller area that could be traded off for development in comparison to the urban community. The results of this study showed how ecosystem services can be used to inform where development could potentially occur. Since then, the number of studies on urban ecosystem services has increased substantially (Haase et al., 2014). A recent study in Cape Town assessed the impacts of urban vegetation in providing ecosystem services and disservices using remote sensing techniques and found that large invasive trees and urban open space areas were some of the areas providing the most ecosystem services (Potgieter, Gaertner, O'Farrell and Richardson, 2019a). Many other studies describing ecosystem services and disservices in urban areas have been done globally (e.g. Davison and Ridder, 2006; Van Riper and Kyle, 2014; Du Toit, Cilliers, Dallimer and Goddard, 2018; including Cape Town – Anderson and O'Farrell, 2012; Cilliers and Siebert, 2012; Cohen et al., 2012; Kain, Larondelle, Haase and Kaczorowska, 2016; Brill, Anderson and Farrell, 2017).

While there are many ways to conduct ecosystem service assessments, one method is to link ecosystem service change to land-use change (as done by Fabos and Hendrix, 1981; Costanza et al., 1997, 2014; Millennium Ecosystem Assessment, 2005; Reyers, O'Farrell, Cowling and Egoh, 2009; O'Farrell, Anderson, Le Maitre and Holmes, 2012; Leh, Matlock, Cummings and Nalley, 2013; Potgieter et al., 2019). Studying the relationship between historical land-use change and the provision of ecosystem services in the Leipzig Region of Germany, Lautenbach, Kugel, Lausch and Seppelt (2011) showed that with a slight decrease in land-use change - 11% between 1964 and 2004, ecosystem service provision - based on the services of water quality enhancement, provision of food, recreational space and pollination, decreased by 23%. These results, therefore, revealed that ecosystem service provision is impacted by land-use change.

Two similar studies have been conducted in South Africa, one at a local, city-wide scale (see O'Farrell et al., 2012) and one at a catchment scale (see Rebelo et al., 2015; discussed in the next paragraph). In the city-wide study, O'Farrell et al. (2012) used a rapid ecosystem service assessment method to show how remnants of natural vegetation contribute to the delivery of ecosystem services. This was done by mapping past, current and future land cover of the entire City of Cape Town and mapping changes in ecosystem services associated with the remnant natural vegetation patches. The results showed that the ecosystem services provided by remnant natural areas had been compromised and were expected to degrade even further in the future, emphasizing the need for the conservation of natural remnants within urban landscapes (O'Farrell et al., 2012). To better manage the wetlands of South Africa, the Water Research Commission (WRC) developed a series of tools (i.e. the WET-Management tools) specifically designed to assess, rehabilitate and manage South African wetlands and riparian areas (Dada et al., 2007; Ellery et al., 2009). These tools can be used at any spatial or jurisdictional scale, whether nationally, provincially or on a wetland-specific scale (Dada et al., 2007). Not only are they designed to inform wetland rehabilitation decision-making, but some of the tools can also be used for more specific purposes such as assessing the functions, values and conditions of specific wetlands (Dada et al., 2007). Combined with spatial analysis and hydrological modelling, the WET-Management tool was used to assess the impact of LULC change on ecosystem service provision in the Kromme catchment (Rebelo et al., 2015). This study showed how valley-bottom wetlands have declined substantially over time due to land-use change, and that rehabilitating the wetland system will simultaneously result in the restoration of various water-based ecosystem services such as flood attenuation, water provision and water flow regulation (Rebelo et al., 2015). It was noted that decision-makers should not only take provisioning ecosystem services into account but also consider other services like supporting and regulating services (Rebelo et al., 2015).

## 2.4 Study Site and Previous Work on the Kuils River

### *2.4.1 Description of the Study Site*

The Kuils River ("the Kuils") is an urban river that forms part of the Eerste-Kuils River catchment tributary, within the Berg Catchment Management Area, located in Cape Town, Western Cape Province, South Africa (Figure 2.2). The Kuils River covers an area of 261 km$^2$ from its source in Durbanville (a town in the northern suburbs of Cape Town) to its confluence with the Eerste River in Macassar, from where the Eerste-Kuils River then drains into the ocean in Strand (Ninham Shand, 1979). The Kuils River itself is 30 km long and winds along the N2 highway, through an area of the Cape Flats and through various towns including Durbanville, Bellville, Kraaifontein and Khayelitsha, ending in Strand. Two tributaries connect to the Kuils-– the Bottelary River, connecting in the upper reaches of the river in Brackenfell, and the Eerste River, in the lower reaches in the Cape Flats (Figure 2.2). Large portions of the river (from Bellville to the Cape Flats) have been channelised and various road bridges have been constructed. This has affected the natural flow and water levels of the river, although historically seasonal, the Kuils River now has a perennial flow due to additional water sources from the sewage treatment works and urban runoff (Ninham Shand, 1986; Shand et al., 1994; Fourie, 2005; City of Cape Town, 2011). This has led to major impacts on both the water quality, biological diversity and ecosystem function of the river (Ninham Shand, 1986; Shand et al., 1994; Petersen, 2002; Fourie, 2005; Chingombe, 2012).

#### *2.4.1.1 Climate, Geology and Soils*

The Kuils River catchment has a Mediterranean climate, with a mean annual precipitation of the area being between 500 and 600 mm per year, mainly occurring between April and September (Ninham Shand, 1986). The average temperature of the region is 16.8°C. While summers are hot and dry with temperatures reaching 31°C, winters are cold and wet, with lows of 7°C (Ninham Shand, 1986; Chingombe, 2012).

The geology of the catchment can be broken up into two sections: one being the upper Kuils River and another below the Kuils-Bottelary River confluence. The upper Kuils River is characterised by rocks of the Malmesbury Group, which consist of phyllite, quartzites, shale, siltstone and greywacke (Ninham Shand, 1979; Heydorn and Grindley, 1982; Chingombe, 2012; Mwangi, 2014). Thin deposits of turf and loam cover this area. The lower reaches of the low-lying coastal plain of the Cape Flats consist of more sandy soils with alluvial deposits underlain by an extensive layer of clay (Ninham Shand, 1979; Heydorn and Grindley, 1982; Chingombe, 2012; Mwangi, 2014). Geological features drive surface flow patterns - while the upper reaches of the Kuils River experience high surface water run-off and low subsurface flow, the lower reaches have lower surface water run-off (Shand et al., 1994). Annual surface run-off is estimated at 22 000 000 $m^3$ (excluding effluent discharge; Mwangi, 2014; Cole et al., 2018).

#### *2.4.1.2 Faunal and Floral Characteristics*

The Kuils River catchment was home to a plethora of faunal and floral species, especially at the wetlands in the area of the Driftsands Nature Reserve (Ninham Shand, 1979, 1986; Shand et al., 1994). Historically, vegetation in the catchment used to be Swartland Shale

Renosterveld and Cape Flats Dune Strandveld with seasonal inundation of grasslands and some pockets of Dune Thicket Strandveld also being present (Shand et al., 1994). However, the extensive disturbance has led to the establishment of many non-native, invasive weeds, shrubs, insects and fish, displacing native biodiversity (Shand et al., 1994; Brown and Magoba, 2009). Large mammals, other than those used for farming purposes, are locally extinct.

The Kuils River is also known for its historic extensive wetlands, hence the name ‘Kuils’, or ‘pools’ (Brown and Magoba, 2009). According to the National Wetland Map of South Africa (Van Deventer et al., 2018), there are at least 25 wetlands along the Kuils River alone, this includes a large floodplain in the Cape Flats, and many smaller channel-valley bottom wetlands, seeps and depressions. These areas supported many bird species. Now, with the increase in nutrient loads in the river, some species of native aquatic vegetation have flourished, further negatively affecting the degrading ecological diversity (Shand et al., 1994; Brown and Magoba, 2009). Shand et al. (1994) reported that increases in dense *Typha capensis* stands and urban development had probably reduced the extent of the available habitat of birds, but a wide diversity of birdlife remained.

#### *2.4.1.3 Land-Uses and Human-Induced Impacts*

The Kuils River is considered an urban river, with many land-use classes driven by urban development. While the River is mainly surrounded by formal urban development, there are also various informal settlements along the river, especially in the region of the Cape Flats. Some industrial and commercial areas are also located along the river, including in the areas of Bellville, Brackenfell, Kuils River, Blackheath and in and around the Cape Flats (Ninham Shand, 1986; Brown and Magoba, 2009; Chingombe, 2012). Pockets of nature reserves are also found in the catchment and include the Durbanville Nature Reserve, Brackenfell Nature Reserve and Driftsands Nature Reserve (WCBSP, 2017). Vegetation types of these reserves include Swartland Shale Renosterveld, Cape Flats Sand Fynbos, Swartland Granite Renosterveld, and Cape Flats Dune Strandveld (City of Cape Town, 2010; Cape Nature, 2015). Many recreational areas considered “green spaces” are also scattered around the catchment and include golf courses, horse race tracks, and dams.

The Kuils River is therefore influenced by various land-use types, including agricultural activities, residential, industrial, and commercial areas. Sources of pollution to the Kuils River based on these land-use types include agricultural and industrial runoff, and domestic waste from dumping in open spaces, laundry, ablution, and sanitation (Ninham Shand, 1986; Shand et al., 1994; Brown and Magoba, 2009). Due to extensive urban run-off, sedimentation and erosion along the Kuils River are also evident (Ninham Shand, 1986). Three wastewater treatment facilities (WWTF), situated along the Kuils- and Bottelary Rivers discharge treated and untreated wastewater into the Kuils River (Figure 2.2). These are the Scottsdene, Bellville, and Zandvliet WWTF.

### *2.4.2. Previous Research on the Kuils River*

Research conducted on the Kuils River is extensive and ongoing. Starting in the early 1970s, the first studies conducted on the river focussed on flood management and the ongoing development along the river (e.g. studies by Ninham Shand Consulting Inc. – see Ninham Shand, 1973, 1977a, 1977b). A report titled 'Flow in the Kuils River and drainage of the catchment' was published in 1979, describing the characteristics of the Kuils River catchment, and flood management of the entire Kuils River (Ninham Shand, 1979). In 1986, the structure and characteristics of the river, flow regime and pollution and sedimentation were discussed (Ninham Shand, 1986). The sources of the point- and non-point pollution were also discussed and historical characteristics described. A summary of the findings of the report follows. By 1986, the flow and structure of the Kuils River had been highly impacted by urbanisation due to the extensive canalisation of the upper reaches of the river. Most observed land-use changes were in the upper reaches of the river, which lead to increases in flood occurrences downstream, especially in the Khayelitsha and Baden Powell Road areas. As a result, the report discussed possible management actions for the river intending to control water flow through further canalisation and/ or conservation (Ninham Shand, 1986). Expected increases in water flow (see Drawings 5.1 to 5.6 in Ninham Shand, 1986) led to the suggestion to construct a retention dam - now seen in the Driftsands Nature Reserve. Flood lines were extensively mapped and, management strategies for different regions along the Kuils River were discussed, taking into consideration the river characteristics and physical obstructions along the river (Ninham Shand, 1986).

The 1986 report further described the ecological sensitivity of some of the areas along the Kuils River and the need for its protection (Ninham Shand, 1986). Areas potentially affected by water pollution were also identified and mitigation measures proposed. With this, they also requested that the area of the Kuils River above the Kuilsriver Municipality be considered as 'urban'. To combat this, they suggested the need for a river reserve along the entire river reach to conserve and manage the remainder of the river reach. They also mentioned the need for vegetation overgrowth management and erosion control. In terms of the river ecology, the faunal and floral characteristics of the Kuils River wetland were also discussed (see Appendix F in Ninham Shand, 1986). Alien plants found along the river, such as Port Jackson (*Acacia saligna*) and water hyacinths (*Eichhornia crassipes*), were also termed problematic. Lastly, the authors discuss the sources of pollution to the river, which are mainly from agricultural land, stormwater, industrial and factory outlets, and sewage effluent discharge (see Appendix C of Ninham Shand, 1986).

Over the same period various other reports were published on flood management, the development of detention dams and the canalisation of the Kuils River (refer to Wessels and Greeff, 1980; Heydorn and Grindley, 1982; Ninham Shand, 1983). The state of the Kuils and Eerste Rivers were also categorised as ‘poor’ and ‘degraded’ (Heydorn, 1986). In the early 1990s and 2000s, studies were focussed around water quality and vegetation management (Ninham Shand, Cape Metropolitan Council and Southern Waters, 1999) and studies regarding proposed new developments along the Kuils River (e.g. Ninham Shand, 1987, 1988, 1994). However, studies on more detailed water quality, floodplain and river channel habitat and its importance in the natural functioning of the river, vegetation characteristics and amenity potential had not yet been discussed, until the publication of the ‘Kuils River Environmental Management Study’ by Shand et al. (1994).

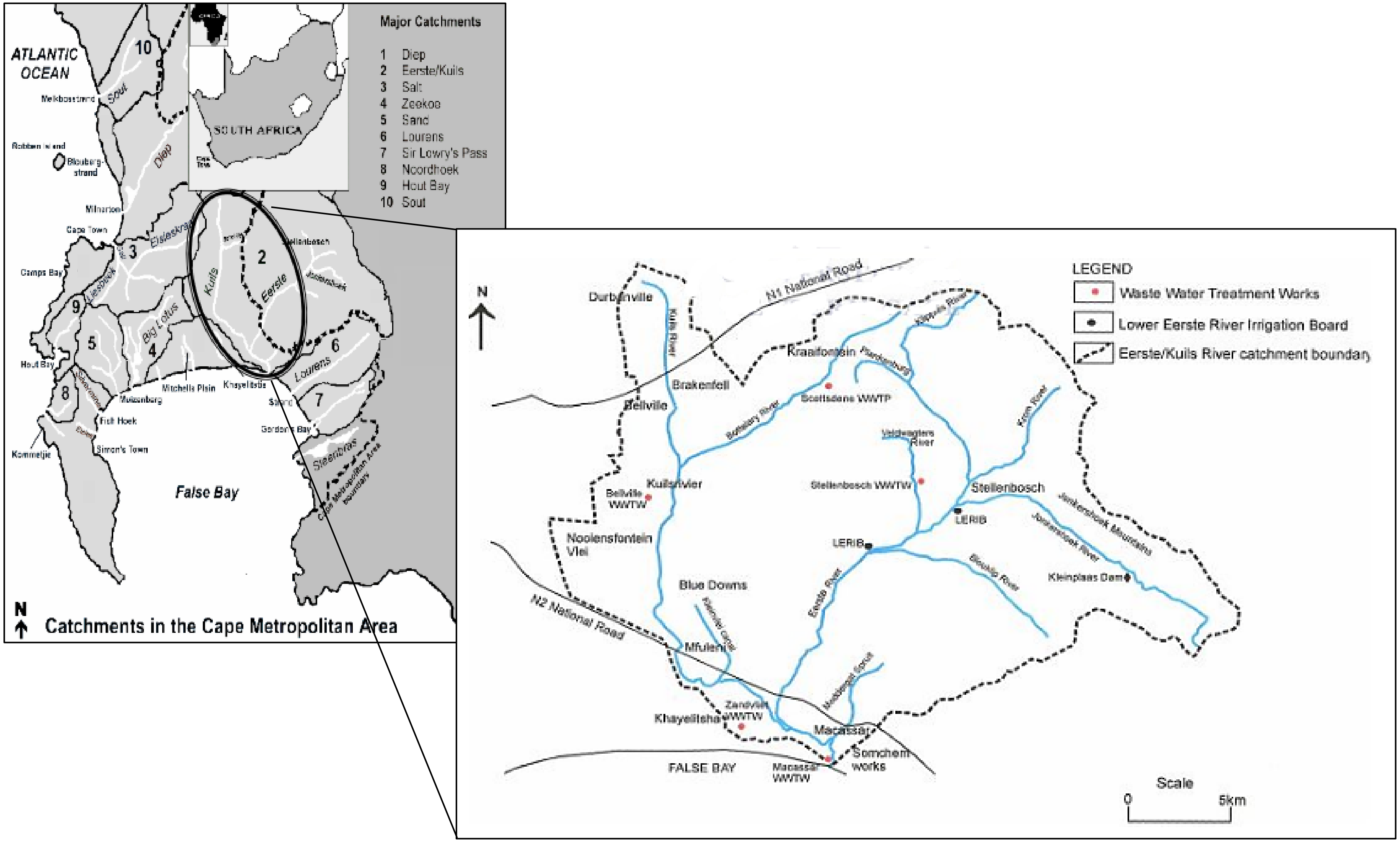



*Figure 2.2* The Kuils River concerning the different catchments in the Cape Metropolitan Area and the wastewater treatment plants around the Kuils and Bottelary Rivers. Figure adapted from Fourie (2005) and Chingombe (2012).

Shand et al. (1994) identified problem areas in the Kuils River catchment affecting the river downstream of the Kuils River Municipality, intending to suggest management solutions to the aforementioned problems. Similar to the 1986 study, this report started by describing the general characteristics of the Kuils River in terms of its climate, geology and hydrology while also including faunal and floral characteristics, land-use and function and its water quality. Whilst recognising the impact of urbanisation on the Kuils River system, the study focussed on the specific problems that the river was facing at the time – more particularly, water quality, aquatic and terrestrial weed control and flood control (Shand et al., 1994). To ensure that these problems were addressed, the authors held public workshops during which the concerned public voiced their opinion (Shand et al., 1994). As a quick summary, in terms of faunal and floral diversity, the Kuils River floodplain has been recognised as being "a biological corridor of metropolitan significance" (Shand et al., 1994, p. 12), with a large diversity of faunal and floral species. While many of the indigenous plant species in the Kuils River wetlands were considered of high conservation importance, there was invasive alien vegetation as well (Shand et al., 1994). Some other indigenous grasses and weeds were also problematic, including the indigenous bulrush (*Typha capensis;* Shand et al., 1994). These, along with the alien vegetation, are discussed in the sections below. The extent of these weeds occurring in the Nooiensfontein Vlei, Driftsands, and Old National Road to the Khayelitsha Stormwater Outlet areas have also been mapped over time (see Figures 3.6.1; 6.3.2-6.3.4; 6.4.2-6.4.4; and 6.6.2-6.6.4 of Shand et al. (1994) for more detail). An example of one of the maps is shown in Figure 2.3 below.

In terms of water quality of the Kuils River, data provided by the Department of Water Affairs and Forestry, Western Cape Regional Service Council, Bellville Municipality, and Zandvliet WWTF were gathered at various points along the river. The authors reported that, in general, some of the water quality parameters assessed exceeded the guideline concentrations (Shand et al., 1994), which had implications for the downstream users of the river.

Eight aquatic weeds were identified as problematic, of which three are indigenous - bulrush (*T. capensis*), water hornwort (*Ceratophyllum demersum*) and fennel-leafed pondweed (*Potamogeton pectinatus;* Shand et al., 1994). Even though they provide many ecosystem services, these weeds interfere with the recreational use of the waterbodies (Shand et al., 1994). Bulrush also has other natural and human impacts by, for example, reducing the ecological diversity of waterbodies, changing the nature of the waterbodies, and its seeds are considered a respiratory irritant (Quick (1987) cited in Shand et al., 1994). Exotic aquatic weeds listed as problematic in Shand et al. (1994) included *E. crassipes*, parrot's feather (*Myriophyllum aquaticum*), wild watercress (*Rorippa nasturtium-aquaticum*), duckweed

(*Azolla filiculoides*) and water fern (*Lemna gibba*). Management strategies for all the above-mentioned aquatic weeds were also described in the report. In addition to the aquatic weeds, terrestrial weeds such as various *Acacia* (currently known as *Vachellia)* species (*A. saligna, A. cyclops, A. longifolia, and A. mearnsii*), Kikuyu grass (*Pennisetum cladestinum*) and Spanish gold/ rattlebox/ scarlet sesban (*Sesbania punicea)* were also present along the Kuils River (Shand et al., 1994).

To manage the abovementioned problems, the report suggested management strategies for four different reaches of the Kuils River, as each of these reaches requires particular attention to address these problems (Shand et al., 1994). These areas were the Nooiensfontein Vlei, the area downstream of the Driftsands Detention Dam, the Baden Powell Drive wetlands upstream of Baden Powell Drive as well as the area below Baden Powell Drive, the Sandvlei area (Shand et al., 1994). The management strategies for each reach were discussed based on economic, ecological and social grounds (Shand et al., 1994). Management recommendations for the Kuils River (and Eerste River) catchments included, amongst others the need for a single river management authority (rather than multiple management parties) and a River Management Board to make decisions on behalf of the public, developing and implementing management programmes for specific problem weeds, installing litter traps, and implementing stricter regulations with regards to water pollution (Shand et al., 1994).

The Kuils River provides many amenity services to the communities living along the river. During Shand et al.'s (1994) study, the potential for the Kuils River as an ecosystem service provider was discussed. While the amenities that the Kuils River provided at the time of the study were somewhat limited (i.e. harvestable resources, livestock grazing, fishing and soil harvesting, and recreation), the authors recognised the potential that the Kuils River could have if more recreational areas were developed, especially in the lower reaches (Shand et al., 1994). They, therefore, suggested the need for nature reserves, ecological corridors and other recreational spaces. They also recognised the Kuils River for its educational, scientific and recreational significance to society (Shand et al., 1994).

Since the publication of Shand et al. (1994), studies on the Kuils River and its surroundings have increased. Reports on the impacts of the development of urban structures, such as housing, shopping centres, industrial areas, sewer lines, roads, etc. for environmental and development authorisation purposes in the vicinity of the Kuils River are widely available (see for example Ninham Shand, 1988, 1994, 1999b; Ninham Shand, Cape Metropolitan Council and Southern Waters, 1999; Belcher and Snyman, 2014; Kilian, 2018). Research by students from various tertiary learning institutions and other private and governmental institutions is also vast.

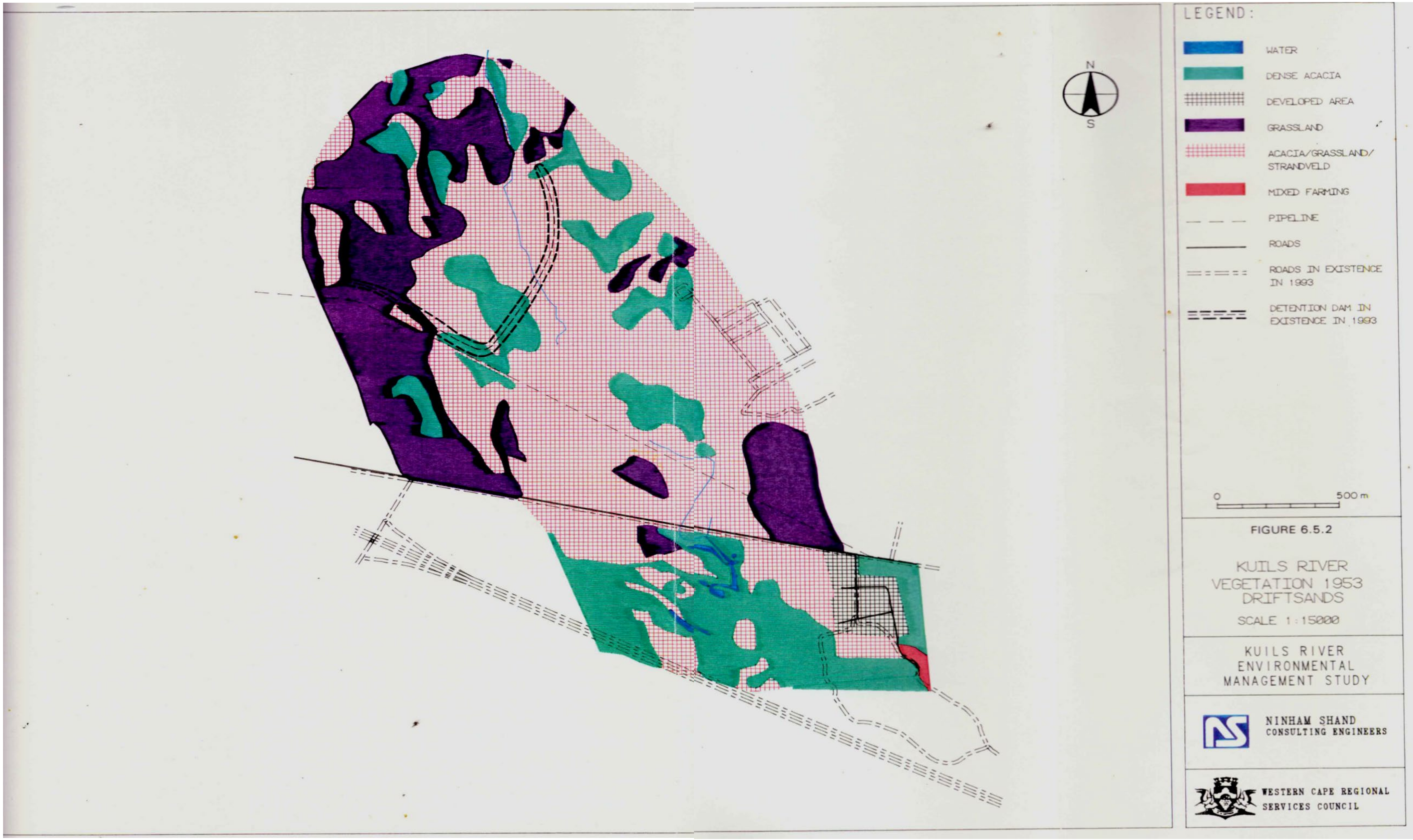



***Figure 2.3*** An example of one of the vegetation maps (Figure 6.5.2) drawn by Shand et al. (1994) in their report titled ‘Kuils River Environmental Management Study’. The map shows the extent of various ‘problem’ vegetation in the Driftsands Detention Dam area in 1953.

For the remainder of this section, the focus is on publications relevant to the current study, focusing on LULC change, wetland assessments and water quality. Most publications are concentrated around the water quality of the Kuils River. The River Health Programme, since 2016 known as the River Eco-status Monitoring Programme, is a South African National government programme that was developed to assess the status of South African rivers, and thereby identify areas where unacceptable ecological degradation is taking place (Resource Quality Information Services, 2019). This programme was, therefore, an improvement of the conventional water resource quality information program (DWS, 2016), as it expanded the assessment of only water quality to include the assessment of other river health indices such as invertebrate diversity, fish diversity, habitat integrity, and riparian vegetation (RHP, 2006). To assess the ecological state of rivers, river indices were categorised into one of five classes – natural, good, fair, poor and artificial (Table 2.1; RHP, 2006). For this specific report, another river health category was included, namely 'unacceptable' (RHP, 2005). From an ecological perspective, this category is characterised by an "almost complete loss of natural habitat and species" (RHP, 2005, p. 7), with a severe alien invasion, while from a management perspective it is highly urbanised and resource exploitation is extensive (RHP, 2005).

Results from the study revealed that the general condition of the river was 'fair' to 'poor' which means that habitat diversity and availability of the Kuils River had declined and that the river had been invaded by alien species (RHP, 2005). Comparing the three reaches (upper, middle and Eerste/Kuils) of the Kuils River, it was found that the upper reach of the river was in a much better condition than the middle and the Eerste/Kuils River reaches, with many of the health indices having 'unacceptable' levels (Figure 2.4; RHP, 2005). Even though both the middle and Eerste/Kuils River reaches had unacceptable water quality statuses, the Eerste/Kuils River reach was in a better overall condition than the middle reach. Therefore, based on the assessed indices for all three reaches of the Kuils River, it was clear that better management strategies needed to be implemented to improve the overall condition of the river.

#### *2.4.2.1 Studies related to Land-Use Change*

Since the 2005 publication of the River Health Programme report, there has been a growing need to uncover the relationship between land-use change and the water quality of South African river systems in general. Five publications on the Kuils River deal specifically with the impact of different land-uses on the water quality of the river. Wright, Kloppers and Fricke (1993), for example, assessed the quality of stormwater runoff in the Khayelitsha catchment for five different land-use classes. Interestingly, sub-catchments that predominantly had 'serviced sites' had the worst quality of stormwater whereas stormwater in formal developed areas had the best quality. In another study, Ayuk (2008), Thomas et al. (2010) and Chingombe (2012) mapped the extent of the Kuils/ Eerste River catchment using approximately 36 land-use classes. It was found that while vineyards contribute the most, 35% to the land-use cover of the catchment, natural vegetation (which included shrubs, fynbos and densely vegetated sections) covered 27.1% and residential developments and roads covered 17.4% of the total area (Chingombe, 2012). It should be noted that at the time, the occurrence of alien vegetation was not considered. Based on the map generated, they also modelled the occurrence and concentration of non-point source pollution in the system. While the map gave a more detailed understanding of the general land cover classes of the two catchments and provided a basis for describing the process of runoff, pollutant loading and infiltration in an urban catchment (Ayuk, 2008; Thomas et al., 2010; Chingombe, 2012), it was found that different land-use classes contributed differently to increased nutrients causing non-point pollution (Chingombe, 2012). Similar results were also found by Dhlembeu (2011). In comparison to the abovementioned studies, Mwangi (2014) focussed specifically on the upper reaches of the Kuils River, with sites carefully selected to represent different land-use classes. It was found that the water quality of the upper Kuils River was poor, which had contributed to a loss of invertebrate diversity in this area, showing that the upper reaches of the Kuils River were highly modified and disturbed (Mwangi, 2014).

Studies such as those mentioned above indicate the current uses and status of the Kuils River and how it has changed over time, while simultaneously exploring and explaining the link between land-use and water quality, which will be important for describing the results in Chapter Five.

#### *2.4.2.2 Studies related to Water Quality*

The Kuils River is an urban river, and as a result, is facing many urban challenges that are affecting the quality of the water. Various point- and non-point sources of pollution are being released into the catchment. Some of the major contributors are the massive water inputs from wastewater treatment plants and stormwater runoff and the release of industrial and agricultural chemicals into the system (Ninham Shand, 1986).

In addition to the studies mentioned above in Section 0, studies on water quality have been undertaken (see e.g. Ayuk, 2008; Mwangi, 2014). Mwangi (2014) specifically conducted studies in the upper reaches of the Kuils River and compared the results to that of the RHP (2005), as shown in Table 2.1. Results showed degrading trends between 2005 ('poor' water quality) and 2012 ('unacceptable' water quality) and all the essential nutrients sampled and analysed exceeded South Africa's ecosystem health criteria recommended limits (Mwangi, 2014). In more recent studies, water quality tests are including assessments for the presence of metals (e.g. Somerset, Hernandez and Iwuoha, 2011; Olujimi et al., 2015); hormones such as the endocrine disruptors estroids and steroids (e.g. Fourie, 2005; Swart and Pool, 2007; Pool, 2008); and synthetic compounds such as polybrominated diphenyl ethers and hexabromobiphenyl (e.g. Daso, Fatoki and Odendaal, 2016). All of these studies have confirmed the presence of these substances in high quantities, which can have serious effects on the human body.

In addition to scientific publications, the Kuils River has also received attention in the media regarding its deteriorating state. While Eyewitness News (2014) reported that residents and their livestock fell ill due to the polluted river, (Green et al., 2019) stressed the need for the City of Cape Town to take action to clean up and better manage the river after a recent announcement of the bacterial and chemical count in the river, just below the Zandvliet Wastewater Treatment Works. Various other reports encouraging residents to stay away from the river due to its polluted state have also been published (Charles, 2019; Knoetze, 2019; Kretzmann, 2019). In 2019, a 13-minute news report was published by the television programme, Carte Blanche, examining the problem of water pollution in the lower Kuils River and the effects on human health (Carte Blanche, 2019). In the last year, the area of Driftsands Nature Reserve has also been impacted by a recent land invasion (Fisher, 2021).

Water quality data for the Kuils River is available from various governmental and private institutions, which allows for the evaluation of water quality over time (see Chapter Four for more information). This will aid in determining whether the river has degraded even further from the first assessments conducted and will determine the possible restorative actions that could be applied to the Kuils River (discussed in Chapter Six).

#### *2.4.2.3 Studies related to Wetland Assessments*

As discussed above (in Section 0), assessing wetlands using the WET-Health and WET-EcoServices tools allows for the assessment of the health and the provision of ecosystem services over time (Macfarlane, Ollis and Kotze, 2020). As many facets contribute to wetland health, publications of differing topics are applicable, including water quality, vegetation characteristics, land-use and land-use change, hydrology, etc. These facets all contribute to the generation of a Present Ecological State (PES) score. One study in particular (see Govender, 2004), used the Khayelitsha wetlands as a case study to assess the impact of development on the wetland. In addition to mapping land-use change over time, Govender (2004) also mapped the extent of *T. capensis* invasion and open water, and found an increase in both these parameters over time (Govender, 2004). Using five parameters to assess PES, Govender (2004) found that the Khayelitsha wetlands are classified as a Category D wetland at the time which means that most of the ecosystem function and natural habitat of the wetland has been lost, also confirmed by the land-use change maps.

Five other studies have been conducted on the Kuils River that relate to different aspects of the wetland assessment used in Chapter Five. The first is a study by Ninham Shand (1999a), which is focused on the management of the Khayelitsha wetlands. More particularly, data were gathered on the flood management, water quality, environmental aspects and planning aspects of the area (Ninham Shand, 1999a). From the research conducted for the report, it was clear that water quality had declined and excessive sedimentation in the wetlands was evident (Ninham Shand, 1999a). The river (and the wetlands) had also been altered significantly, from a seasonal, nutrient-poor system, to a permanent, nutrient-rich system and had thus suffered an immense faunal and floral loss (Ninham Shand, 1999a). Suggestions were made to incorporate the findings into the future planning and management of the wetlands. To ensure that the management objectives were on par with the community needs, some public participation meetings were held where public concerns and the benefits of the wetland were discussed (Ninham Shand, 1999a).

In a follow-up study on the Kuils River, Mathenjwa (2017) explored the social and ecological ecosystem services of the Khayelitsha Wetlands Park. Not only were vegetation structure and water quality assessments conducted, but the community's perception of the open spaces was assessed during interviews. The assessment revealed that the Park is used by many community members for a variety of different activities (Mathenjwa, 2017). Concerns raised about the state of the wetland, e.g. water quality, were also confirmed in the water quality and vegetation assessments, where water quality was not fit for human or agricultural use and invasion by acacias and typha reeds were evident (Mathenjwa, 2017).

Similar to the report by Mathenjwa (2017), Greeff (2014) and Myeza and Mdunyelwa (2020) conducted social studies on the entire Kuils River to gather information about the water quality concerns of the river as well as uncover the uses and problems of the river. Meetings held with community members revealed that there was a mismatch in terms of community needs and service delivery (i.e. a management problem), but there was also a consensus that the river was highly polluted, which had deleterious impacts on human health and use for recreational purposes (i.e. an ecological and social problem; Greeff, 2014; Myeza and Mdunyelwa, 2020). However, the river also brings joy to some community members as it is regularly used for fishing, cultural activities, farming, and other recreational activities (Greeff, 2014). To address the problems and concerns, various recommendations were made. It was suggested that the river be incorporated into the 2020/2021 Integrated Development Plan for the City of Cape Town, while also hosting educational programmes for community members such as snake awareness courses and gardening courses, and organising regular clean-up events and monitoring services in the catchment (Greeff, 2014; Myeza and Mdunyelwa, 2020).

**Table 2.1** Five categories describe the ecological state of South African rivers from both an ecological and management perspective, according to the River Health Programme. Table from RHP (2006, p. 23).

| River Health Category | Ecological Perspective | Management Perspective |
|---|---|---|
| Natural (N) | No or negligible modification of in-stream and riparian habitats and biota. | Protected rivers; relatively untouched by human activities; no discharges or impoundments allowed. |
| Good (G) | Ecosystems are essentially in a good state; biodiversity largely intact. | Some human-related disturbance but mostly of low impact potential. |
| Fair (F) | A few sensitive species may be lost; lower diversity and abundances of biological populations are likely to occur, or sometimes, higher abundances of tolerant or opportunistic species occur. | Multiple disturbances are associated with the need for socio-economic development, e.g. impoundment, habitat modification and water quality degradation. |
| Poor (P) | Habitat diversity and availability have declined; mostly only tolerant species are present; species present are often diseased; population dynamics have been disrupted (e.g. biota can no longer reproduce or alien species have invaded the ecosystem). | Often characterised by high human densities or extensive resource exploitation. Management intervention is needed to improve river health – e.g. to restore flow patterns, river habitats or water quality. |
| Artificial | Transformed to such an extent that their habitat types, biological communities and ecosystem processes bear no or little resemblance to those that would occur under natural conditions. | Modified beyond rehabilitation to anything approaching a natural condition. Example: canalised rivers in urban environments |

Even though most of these studies did not specifically deal with the different facets of wetland assessment, as described in Section 0, they provide a baseline for information regarding ecosystem services and disservices that the river provides. In addition, it also gives an overview of the current and previous conditions of the wetlands along the river.

The review above provided the basis for understanding the techniques available to conduct river assessments and was used to formulate the aims and methods of this research. The aims were; 1) to gain a spatio-temporal understanding of the land-use change of the Kuils River, 2) to assess which ecosystem services the Kuils River provides, and 3) to acquire a comprehensive understanding of the current state of the Kuils River.

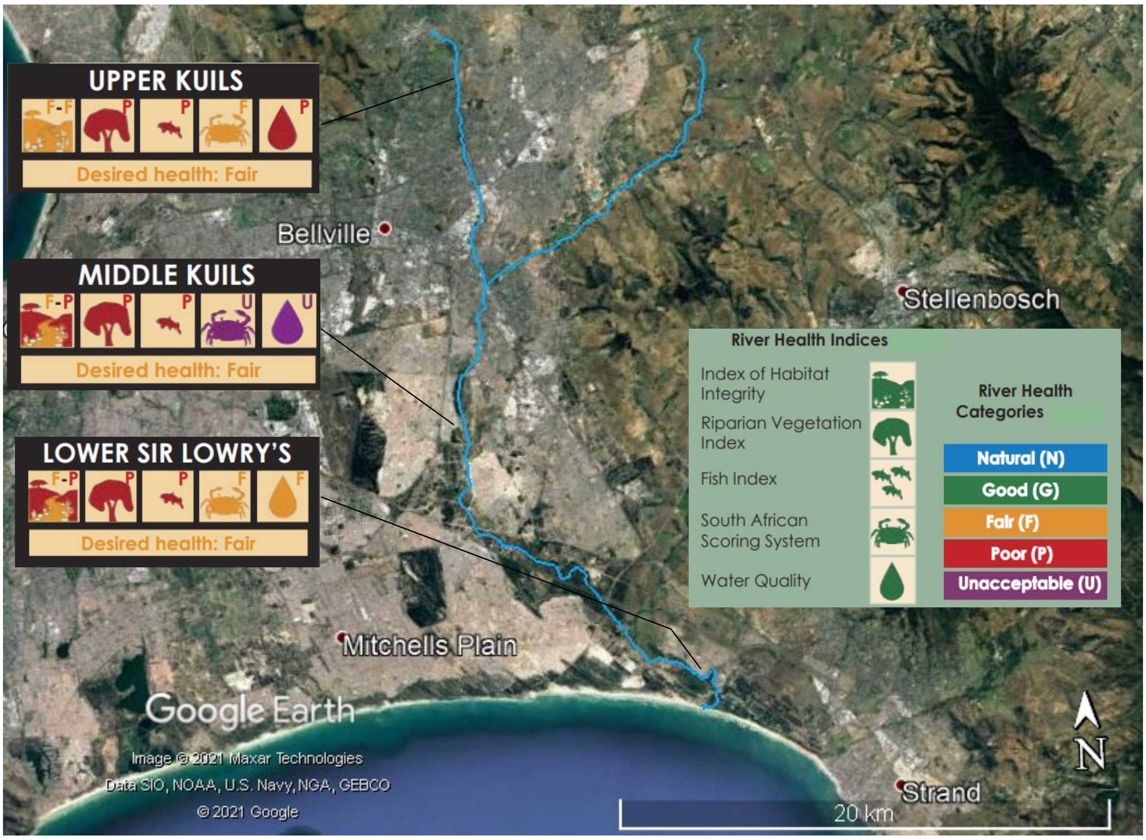



***Figure 2.4*** The ecological state of the three reaches of the Kuils River, for each of the river health indices. Figure adapted from RHP (2005).

This involved addressing the following questions and objectives:

0. What land-use changes have contributed the most to the degradation of the Kuils River?
1. What ecosystem services are currently generated by the Kuils River and how would ecosystem service provision change with the implementation of appropriate restorative measures?
2. What is the current status (in terms of water quality) of the river and how has water quality changed since the first assessments conducted on the river?
3. From an ecological perspective, recommend best practices and management actions that could contribute to marked improvements in the overall condition of the Kuils River.

# Chapter 3
# Land-Use Change of the Kuils River

## 3.1 Introduction

The worlds urban population numbers increased more than four-fold between 1950 and 2018 (UNDESA, 2018) and by 2028 is expected to reach the five billion mark (UNDESA, 2018). This places pressure on the remaining patches of ecosystems to provide the necessary goods and services (Schneider, Logan and Kucharik, 2015), with urban river ecosystems being disproportionately affected (Costanza et al., 2014).

As a consequence of large-scale urbanisation along river catchments, urban rivers have an increased risk of environmental damage due to the concentration of human activities, the resulting treated and untreated effluent discharge (Martín-Vide, 2001; Chin, 2006; Forman, 2014) and river modifications, such as the construction of channels, berms and infilled floodplains (Luger and Brown, 1999; Oberholster and Ashton, 2008; Pool, 2008; CSIR, 2010; Mwangi, 2014). These perturbations change the hydrology, geomorphology and water quality of these systems, which ultimately impacts biodiversity (Kleynhans, 1996; Withers, 2003; Findlay and Taylor, 2006; Everard and Moggridge, 2012; Mwangi, 2014). Consequently, many, if not all, urban rivers are degraded, transformed and invaded, resulting in novel species combinations and a reduction in ecosystem function (Findlay and Taylor, 2006; Everard and Moggridge, 2012).

Due to the transformed and disturbed state of urban ecosystems, 'novel ecosystems' have replaced natural ecosystems (Hobbs et al., 2006; Francis, 2014; Blignaut and Aronson, 2020). These novel systems 1) feature new species combinations that were generally not present in the historical state of the ecosystem, 2) have experienced changes in ecosystem function, and 3) have been altered by human activities (Hobbs et al., 2006). As it is likely too costly and too difficult to return novel ecosystems to their pre-anthropogenic disturbance state, special attention needs to be given to design management strategies that are appropriate for such ecosystem types (Hobbs et al., 2006; Hobbs, Higgs and Harris, 2009; Francis, 2014; Blignaut and Aronson, 2020). Hobbs et al. (2009) argue that the management of novel systems should be focused on maximising the services that they provide, including nutrient assimilation, flood control and carbon sequestration. However, to enhance ecosystem services, the drivers of change affecting these ecosystems should be understood and targeted, which would ultimately require some form of restorative effort.

It is widely recognised that restorative measures are needed for urban rivers (Holmes, Richardson, Esler, Witkowski and Fourie, 2005; Aronson et al., 2010; Everard and Moggridge, 2012; Aronson and Alexander, 2013; Elmqvist et al., 2015). However, selecting the appropriate strategy and treatment requires careful planning, adequate financing and the development of appropriate restoration or rehabilitation goals (Blignaut and Aronson, 2020). The Society for Ecological Restoration has developed a set of tools to ensure aligned investment into restoration projects (Gann et al., 2019). While ecological restoration aims tofully recover the targeted ecosystem to its reference model, rehabilitation sets out to reinstate the ecological function of the ecosystem (Gann et al., 2019). Since the scale of restoration of urban rivers is dependent on the condition of the catchment area (Richardson et al., 2007), ecological restoration of entire urban river systems is unlikely to be successful due to the general and ongoing degradation of the catchment, resource constraints, and inappropriate management. A more appropriate goal would thus be to rehabilitate degraded urban rivers (Holmes, Esler, van Wilgen and Richardson, 2020).

To describe the condition of the catchment area and to choose an appropriate reference model upon which to base management strategies, temporal and spatial analyses could be used (Wissmar and Beschta, 1998; Richardson et al., 2007). One such technique is to use land-use/land-cover (LULC) change mapping using Geospatial Information Systems which allows for the understanding of spatial and temporal patterns of change caused by both anthropogenic and natural processes (Lambin, 1997; Zhang et al., 2011). For example, LULC change analyses can be conducted in catchment areas, river basins and urban systems (Seto and Kaufmann, 2003; Wilson and Schröder, 2008; Zhang et al., 2011; Grundling, van den Berg and Price, 2013; Rebelo, Scheunders, Esler and Meire, 2017; ). This technique can be used to assess ecosystem change as well as potential environmental implications, while also guiding management decisions (e.g. Yorke and Margai, 2007; Rebelo et al., 2013).

The Kuils River is an urban river situated in the northern suburbs of Cape Town, that provides valuable ecosystem services to many of Cape Town's residents. As a result, it has become degraded and has received much scientific and public attention due to its deteriorating state (see for example Ninham Shand, Cape Metropolitan Council and Southern Waters, 1999; Greeff, 2014; Carte Blanche, 2019; Kretzmann, 2020). Despite such attention, the Kuils River remains degraded.

The purpose of this study was to assess the state of the Kuils River based on historic and current LULC to determine a baseline for restoration. To achieve this, the following questions were addressed:

1) How has the Kuils River and its surroundings changed over the last 80 years (based on the earliest available aerial photography) in terms of anthropogenic and vegetational characteristics?
2) What are the main drivers of this change?
3) Based on the findings of questions 1 and 2, how should the Kuils River be rehabilitated and managed?

These findings will aid in the development of appropriate restoration and management actions for the Kuils River.

## 3.2 Methods

### *3.2.1 Study Site Description*

The Kuils River forms part of the Kuils-Eerste River catchment, located in Cape Town, Western Cape (Figure 3.1), with the geological extent of the study area lying between the latitudes 33°50' and 34°07'S and longitudes 18°38' and 18°45' E.

### *3.2.2 Land-Use Change Mapping*

***Aerial Imagery***

To spatially understand the river, LULC maps were constructed (as described in Rebelo et al., 2015). In South Africa, aerial photography has been used for analysing mapping extent and change over time due to the extensive historical photographic record, with high-resolution photographs (Rebelo et al., 2017). Historical aerial imagery and orthorectified maps of the Kuils River were gathered from the Chief Directorate: National Geospatial Information (CD:NGI) portal of the Department of Rural Development and Land Reform and used to map land-use at a high resolution for five decades. The time slices 1944, 1968, 1989, 2008 and 2020 were chosen based on the aerial photography made available by the CD:NGI. Time slices between the oldest available aerial photographs (1944) and the most recent layer (2020) were selected to portray changes over 15-to-20-year time gaps. The earlier years' resolution was the lowest (at 25 m), while the most recent year (2020) has a 30 cm–1 m resolution. For the 2020 map, an ArcGIS's World Imagery Layer was used. This layer uses both aerial and satellite imagery in various resolutions to generate a global map (ESRI, 2020).

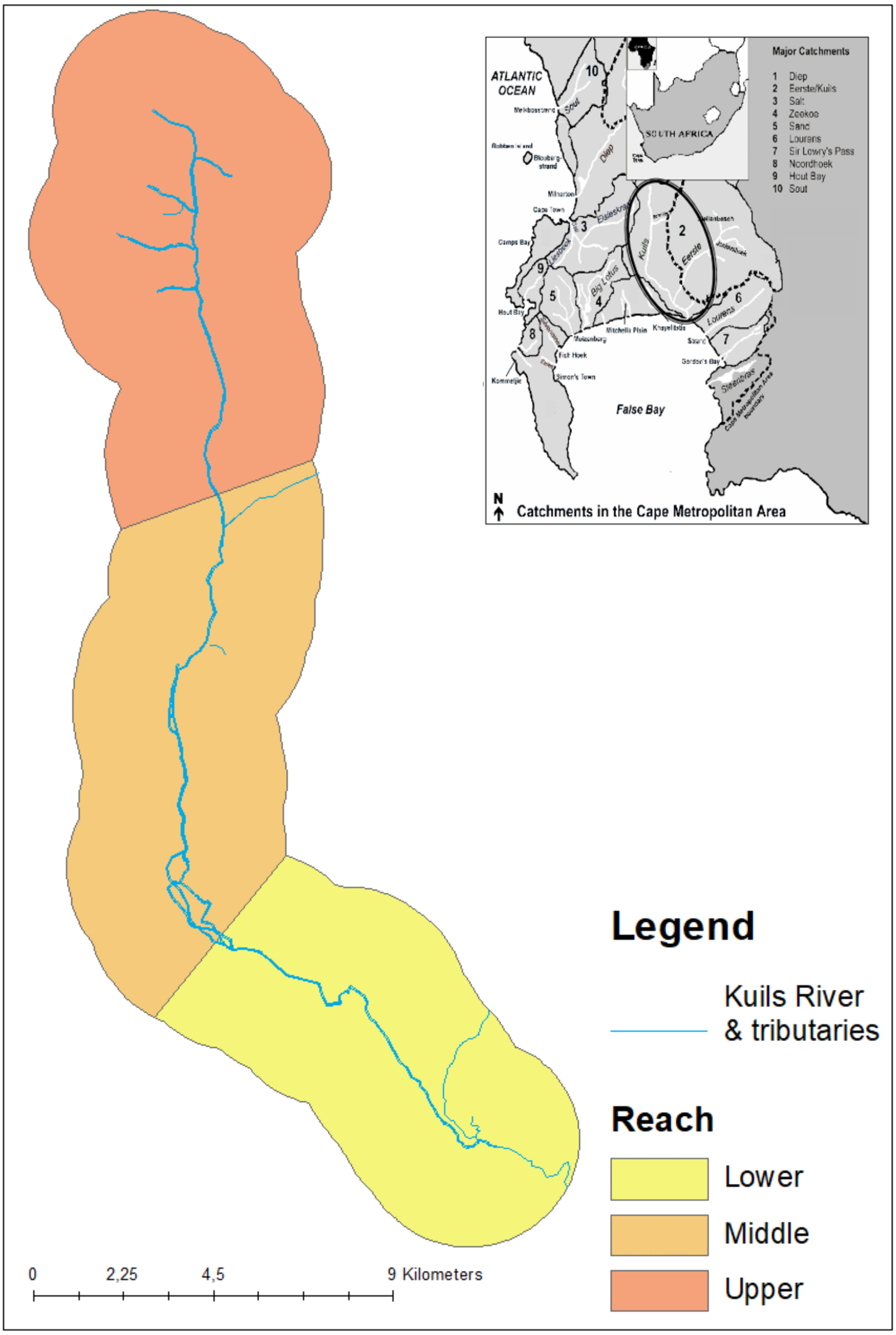



***Figure 3.1*** The Kuils River and surrounding mapped area (within a 2.5 km radius of the river), with Cape Town shown in relation to the different catchments in the Cape Metropolitan Area and its reaches.

### *Land-Use/Land-Cover Classes and Mapping*

The aerial photographs (ranging in scale from 1:18 000 to 1:50 000) and orthorectified images were used in the ArcGIS Geospatial Information System (GIS), (published by ESRI, 2016) to map land-use change into fourteen LULC classes (see Table 3.1), within a 2.5 km radius around the Kuils River. This 2.5 km radius was selected to incorporate a wide enough area to account for all possible land cover types found in an urban setting. The already georeferenced and rectified 2008 aerial photographs were used to georeference the maps for the years 1944 to 1989 in ArcMap (version 10.5). LULC classes included the spread of alien invasive trees such as *Acacia* species, *Hakeas* and various pines, urban development and waterbodies. These LULC classes were digitised using the aerial images for each time slice. Most of the LULC classes were selected based on vegetation characteristics except for three – urban built, waterbodies and wastewater treatment facilities. In most cases, roads were included in the urban LULC class, but in the years before 2008, it was included in the other LULC classes. This is because of the time constraints in mapping individual roads between the different LULC classes.

Some assumptions were made while mapping the LULC classes. Dense bulrush (*Typha capensis*) stands were categorised under the ‘alien plants’ category as this species is a key indicator of the presence of water and is highly invasive in many parts of South Africa. Due to the urban nature of the river and catchment, and previous mentions of alien plant invasion dating back to the early 1900s (Heydorn and Grindley, 1982; Ninham Shand, 1986; O’Callaghan, 1990; Shand et al., 1994; Brown and Magoba, 2009), it was assumed that most of the vegetation occurring throughout the site contained some alien vegetation.

Accurate LULC mapping is often difficult when using scanned, black and white maps of low resolution. There is thus a small degree of error in the mapping of the LULC classes due to potential misinterpretation of visual characters (Rebelo et al., 2013). Therefore, the final LULC categorisation depends on the researcher’s interpretation. The most difficult LULC classes to map were differentiating between natural vegetation and alien tree invasion due to their similar appearance as dense pockets of vegetation on the maps. For example, for the Cape Flats, it was difficult to distinguish between Cape Flats Dune Fynbos and alien trees. Where possible, ground-truthing was used or previous research was consulted to map the extent of an alien invasion (such as maps found in Shand et al. (1994) and Ninham Shand (1999) and research conducted by the Agricultural Research Council (ARC, 2011)). The remainder of the vegetation that could not be verified through records or ground-truthing, was characterised as ‘natural (open)’, with the assumption that some alien vegetation is present.

The maps generated for 2008 and 2020 were ground-truthed by using photographs that were taken on various field trips during the study period as well as land-use maps that were generated by previous studies (Thomas et al., 2010; Chingombe, 2012). Data layers from Google Earth (Pro) (version 7.3.3.7786) and Cape Farm Mapper (version 2.3.4) were also consulted. Google Earth was used to confirm historical change using three tools: the time slider tool, public photos stored in the database and street view; whilst Cape Farm Mapper confirmed the general LULC classes (based on the National Land Cover from the Department of Environmental Affairs (2018)), the extent of waterbodies (CD:NGI, no date; CSIR, no date; DWS, no date; Van Deventer et al., 2018), the presence of alien trees (ARC, 2011), areas of conservation significance (WCBSP, 2017) and other geographic features. Other than the vegetation maps generated by Shand et al. (1994) and Ninham Shand (1999a), aerial photographs before 2008 could not be verified.

***Land-Use/Land-Cover Change Analysis and Statistics***

To understand how land-use has changed over time, percentage change was calculated based on area and relative perimeter (perimeter/area) change over the 76 years. This was done by summing the total percentage area and relative perimeter data of the various LULC classes for each of the time slices and comparing them between the five years. Relative and absolute change of the LULC classes was also calculated. Spearman's rank-order correlation test (Spearman, 1904) was used to assess the statistical significance of the occurrence of constant increasing or decreasing monotonic trends over time using non-parametric data, with the assumption that there was no trend (or no correlation) between the data points. Using RStudio (2019; version 1.3.1093) and its *trend* and *Kendall* packages (McLeod, 2011; Pohlert, 2020), significant trends in percentage area and perimeter data between 1944 and 2020 were assessed. Spearman's test uses two parameters as statistics: $r_s$, and p. While the $r_s$ statistic signifies the strength and direction of the trend, the p statistic denotes the significance of the trend. The closer the $r_s$ value is to one (or -1), the stronger the trend. It should be noted that for the statistical analysis, the LULC class 'Wastewater Treatment Facilities (WWTF)' was removed and split into three other LULC classes: waterbodies, urban built and urban open space. This was to reduce the distortion of the statistical analyses.

To analyse differences in land-use change for different reaches of the river, the river was split into three sections, the upper, middle and lower reaches (Figure 3.1). Reaches were split based on water accumulation points along the Kuils River, as well as hydrogeomorphic characteristics of each river reach. The upper and middle reaches were split just above the Bottelary confluence and the middle and lower reaches were split just below the Driftsands Nature Reserve. To get a better understanding of the drivers of land-use change within each reach, the same tests mentioned above were used when comparing land-use change for the different reaches of the Kuils River.

***Table 3.1*** Categories used to describe the land-use/land-cover (LULC) classes of the Kuils River, modified from Rebelo et al. (2015) and Du Plessis (2020).

| **LULC Class** | **Description** |
|---|---|
| Agriculture | Crop fields and pastures |
| Alien (riparian) | Alien trees occur in and/ or along with waterbodies, in the riparian zone. Trees include *Acacia* species, *Hakea* species and pines. |
| Alien (terrestrial) | Alien trees occur in terrestrial areas. Trees include *Acacia* (currently known as *Vachellia*) species, *Hakea* species and pines. |
| Alien (windbreak) | Alien trees are planted in a single row; usually used to separate crops or planted along roads. Trees include *Acacia* species, *Hakea* species and pines. |
| Alien plants | Alien weedy species are generally found in/ along with waterbodies. Species include reeds like *Typha capensis, Eichhornia crassipes*, among others. |
| Barren land | Land with exposed soil, sand or rocks with less than 10% vegetation cover (vegetation cover does not include grass). This also includes the beach of the False Bay coast and the dunes of the Cape Flats. |
| Fallow | Land with exposed soil, sand or rocks with more than 10% vegetation cover. |
| Natural (open) | Terrestrial areas are covered in mainly naturally occurring vegetation, including areas of natural Cape Flats Dune Strandveld, Cape Flats Sand Fynbos and Swartland Shale Renosterveld. This LULC class could also include alien vegetation such as acacias. |
| Ocean | A section of the False Bay coast. |
| Riparian vegetation | Naturally occurring vegetation occurring in and/or along with waterbodies. |
| Urban (built) | Land covered by buildings and other man-made structures (including residential, industrial and commercial areas). |
| Urban (open) | The land between built urban structures is covered in grass, and/ or used for recreational purposes (including golf courses, sports fields, etc.). Can include invasive weeds and grasses such as kikuyu grass. |
| Waterbodies | Streams, rivers, channels, dams, wetlands, reservoirs |
| WWTF | Wastewater treatment facilities |

## 3.3 Results

### *3.3.1 Land-Use/Land-Cover Change Analysis*

Land-use/land-cover (LULC) were assessed for a 2.5 km radius around the entire extent of the Kuils River, equating to an area of approximately 187 $km^2$. In 1944 and 1968, agricultural and natural (open) land covered the largest total area (86% and 74%, respectively). This, however, changed from 1989 onwards as built urban development increased (Figure 3.2). These three LULC classes made up approximately 70-80% of the land cover for each time slice. Natural open vegetation remained relatively constant for the first two time slices but decreased from 51% to 10% over the eight decades. Alien (windbreak) also showed the same decreasing trend. Barren land and riparian vegetation, however, remained relatively constant over the entire 76 years, changing approximately by between 0 % and 1 % area cover (Figure 3.**2**). The remainder of the LULC classes showed an increasing trend in percentage area cover from 1944 to 2020. These included alien vegetation in riparian areas ('alien (riparian)'), alien weeds ('alien plants'), fallow land and urban (open) space (Figure 3.2).

Interestingly, only two of the LULC classes showed a significant trend over the entire 80 years – while agricultural development decreased from 1944 to 2020 ($r_s$ = -1, p-value = 0.017), built urban structures increased ($r_s$ = 1, p-value = 0.017). Many of the other LULC classes showed increasing and decreasing trends that were very close to being significant (see Table 3.2 for more information). Even though not graphically represented, Wastewater treatment facilities increased in percentage area by 0.18% since their construction in 1968.

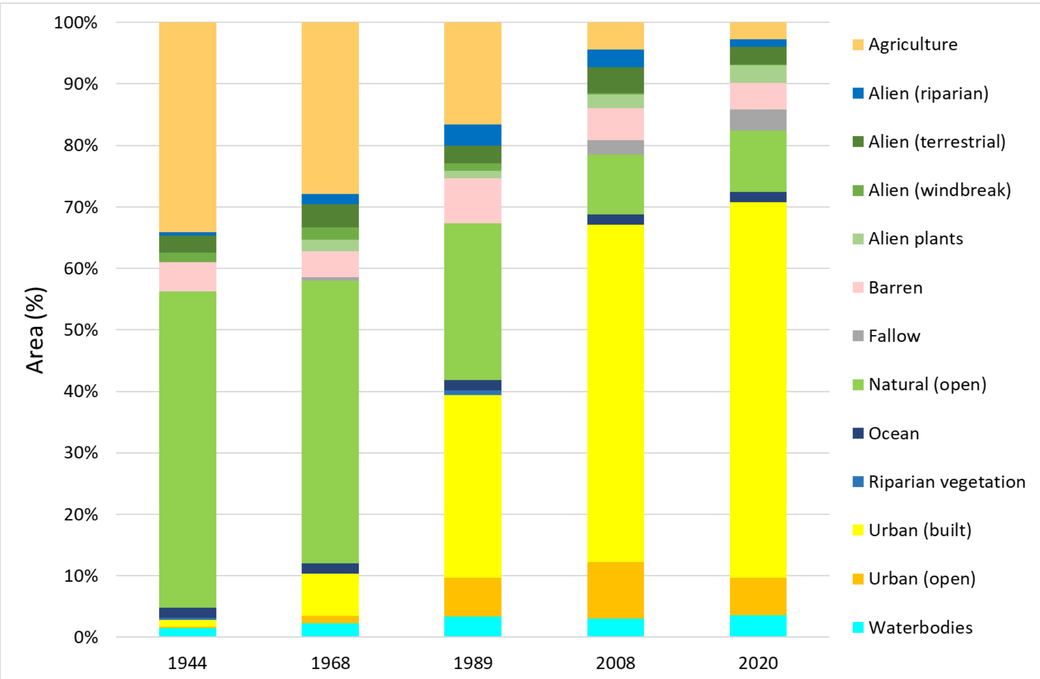



***Figure 3.2*** Land-use/land-cover (LULC) percentage area change in the Kuils River over the last 80 years for five-time slices (1944, 1968, 1989, 2008 and 2020).

Relative perimeter, on the other hand, varied over time (Figure 3.3). Between 1989 and 2008, perimeter change was at its largest; increasing from 1571 km to 1910 km. In comparison to the percentage area data, three LULC classes showed a significant trend for the relative perimeter data, which could indicate an increase in fragmentation of LULC class over time. These were agricultural land ($r_s = -1$, $p = 0.017$), alien plants ($r_s = 1$, $p = 0.017$) and waterbodies ($r_s = 1$, $p = 0.017$). Many other LULC classes also had 'indicated strong trends, albeit not significant (as seen in Table 3.2). Barren land, urban built structures and urban open space showed increasing trends in the total perimeter. These trends were supported by Figure 3.3, which shows the change in the perimeter for various LULC classes of the Kuils River over the last 80 years. However, alien windbreak, natural open areas and riparian vegetation decreased in perimeter over time. In general, however, there was no significant change in relative perimeter between 1944 and 2020.

***Table 3.2*** Statistical values for the percentage area and perimeter change for various LULC classes. The statistics that are reported only includes p values that are either significant (indicated with a *) or almost significant. The p-values are above 0.1 where 'Not Applicable (NA)' is reported.

| **LULC Class** | **Percentage Area ($m^2$) Change** | | **Perimeter (m) Change** | |
|---|---|---|---|---|
| | $r_s$ | p | $r_s$ | p |
| Agriculture | -1 | 0.017* | -1 | 0.017* |
| Alien (windbreak) | -0.9 | 0.08 | -0.9 | 0.08 |
| Alien plants | 0.9 | 0.08 | 1 | 0.017* |
| Barren | NA | NA | 0.9 | 0.08 |
| Natural (open) | -0.9 | 0.08 | -0.9 | 0.08 |
| Urban (built) | 1 | 0.017* | 0.9 | 0.08 |
| Urban (open) | NA | NA | 0.9 | 0.08 |
| Waterbodies | 0.9 | 0.08 | 1 | 0.017* |

Interestingly, both alien vegetation occurring in terrestrial and riparian areas show non-monotonic trends. For example, the perimeter of alien vegetation occurring in riparian areas increased markedly from just over 18 600 to just over 88 000 m between 1944 and 2008 and decreased to approximately 48 500 m in 2020 (Figure 3.5A). The perimeter of alien vegetation occurring in terrestrial areas was noticeably more and the total perimeter continuously increased and decreased (Figure 3.5B).

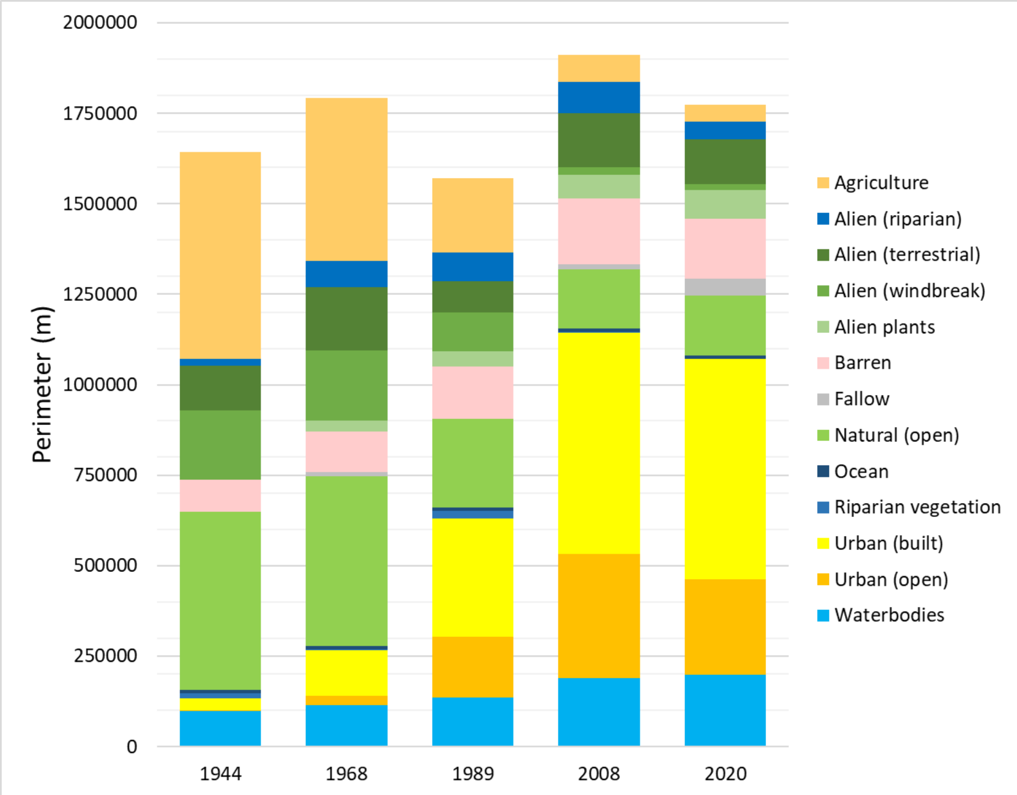



***Figure 3.3*** Perimeter (m) change of 14 different land-use/land-cover (LULC) classes of the Kuils River over 80 years, as captured in five-time slices (1944, 1968, 1989, 2008 and 2020).

Relative and absolute change calculations showed how LULC have changed over the 80 years (Figure 3.4). The two LULC classes that showed the greatest increase in area between 1944 and 2020 were urban built structures ('urban (built)') and urban open spaces, showing a 54-fold and 30-fold increase of its initial extent in 1944respectively, . These exponential increases indicate extensive urbanisation occurring along the Kuils River. Other increases in relative area change were that of waterbodies (132%) and alien (riparian) vegetation (122%). The remainder of the LULC classes showed a decrease in area cover, ranging from 3% to 100% (Figure 3.4). It should be noted that alien plants and fallow land were excluded from the relative change assessments (see Figure 3.4) as they had no area cover in 1944. However, they increased more than one, and six times respectively since their initial extent in 1968.

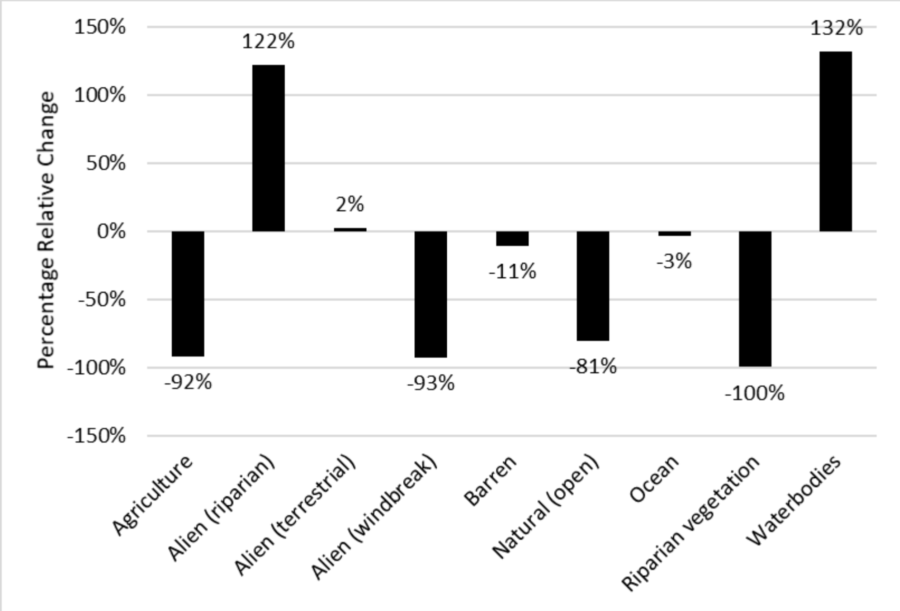
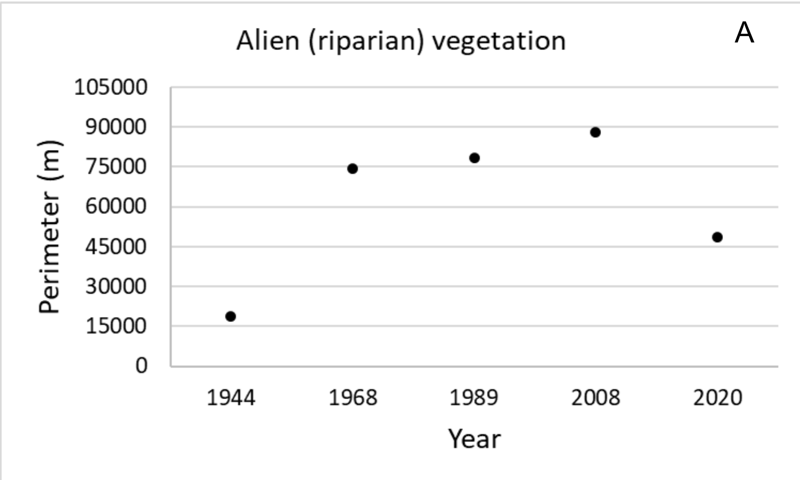



***Figure 3.4*** Relative change in the area of nine land-use/land-cover (LULC) classes between 1944 and 2020, represented as 2020-1944/1944, for the Kuils River. To prevent distortion, urban built (5355%) and urban open (3223%) have been removed. Alien plants and fallow land have also been removed due to the lack of area extent in respective years.



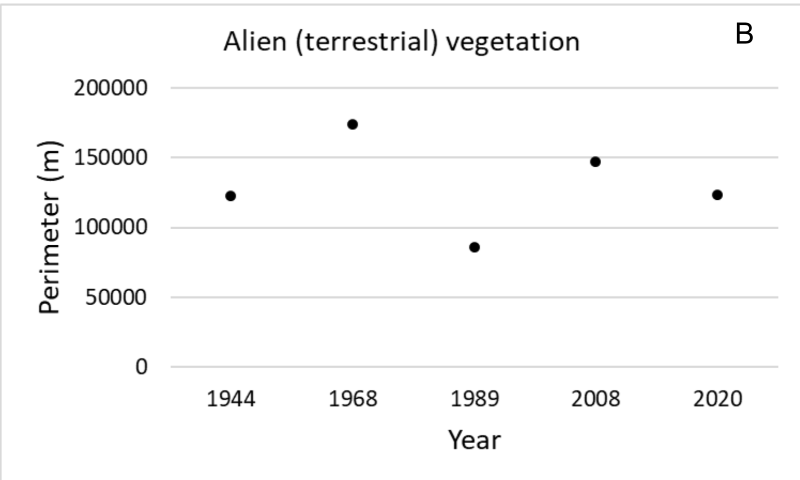




***Figure 3.5*** Graphic representation of the change in total perimeter (in meters) over the five decades for A) alien (riparian) and B) alien (terrestrial) vegetation in the entire Kuils River.

### *3.3.2 Visual Analysis of Land-Use/Land-Cover Change*

Comparing the visual representation of the upper reaches (as seen in Figure 3.6), it is clear that urban built structures and urban open spaces have increased substantially. Agricultural land and natural open spaces have experienced the opposite trend, with a marked reduction in land used for farming and areas containing natural vegetation (see light brown and light green changes between Figure 3.6A and Figure 3.6C). It should be noted that Figure 3.6 only represents a fragment of the land-use change in the entire Kuils River system, but is the area where the most marked changes have occurred. For the full LULC change maps, refer to Addendum A.

### *3.3.3 Comparing Land-Use/Land-Cover Changes between the Lower, Middle and Upper Reaches*

To detect context-specific trends in LULC classes over time and therefore attempt to isolate key drivers of change, the percentage proportion of each LULC class was examined for the three separate reaches of the Kuils River (upper, middle and lower) between 1944 and 2020. All three reaches have experienced remarkable change since 1944 (Table 3.3). The upper reaches of the Kuils River were mostly driven by increases in built urban structures (77.35%) and urban open areas (8.84%), and reductions in agricultural land (- 57.43%) and natural open space (- 24.81%), with agricultural land and urban built structures showing a monotonic decrease and increase respectively over time (agriculture: $r_s$ = -1, p = 0.0017; urban built structures: $r_s$ = 1, p = 0.017). Natural (open) space also showed a negative but strong trend (Table 3.3; $r_s$ = -0.9, p = 0.08).

Similar trends were also found in the middle and lower reaches where change was driven by agricultural development, urban built structures and natural (open) space (Table 3.3). Other LULC classes that were driving the changes in the middle and lower reaches were urban open spaces and fallow land. In the middle reaches, urban open space increased by 5.11% between 1944 and 2020, while fallow land increased in the lower reaches by 7.15%. Similar to the entire river, many of the other LULC classes for all the reaches also showed strong trends even though non-significant (Table 3.3).

Visual analysis of the upper reaches further show a change from agricultural land-use practices to residential and industrial land-use, with little natural open space. Apart from urbanisation, the lower reaches have become wetter over time (as indicated by the change from 'alien (riparian) vegetation to 'alien plants'). The middle reaches on the other hand have experienced changes from urban open space to fallow land as well as a reduction in alien riparian vegetation to more natural open space, which could indicate that the area has become drier.

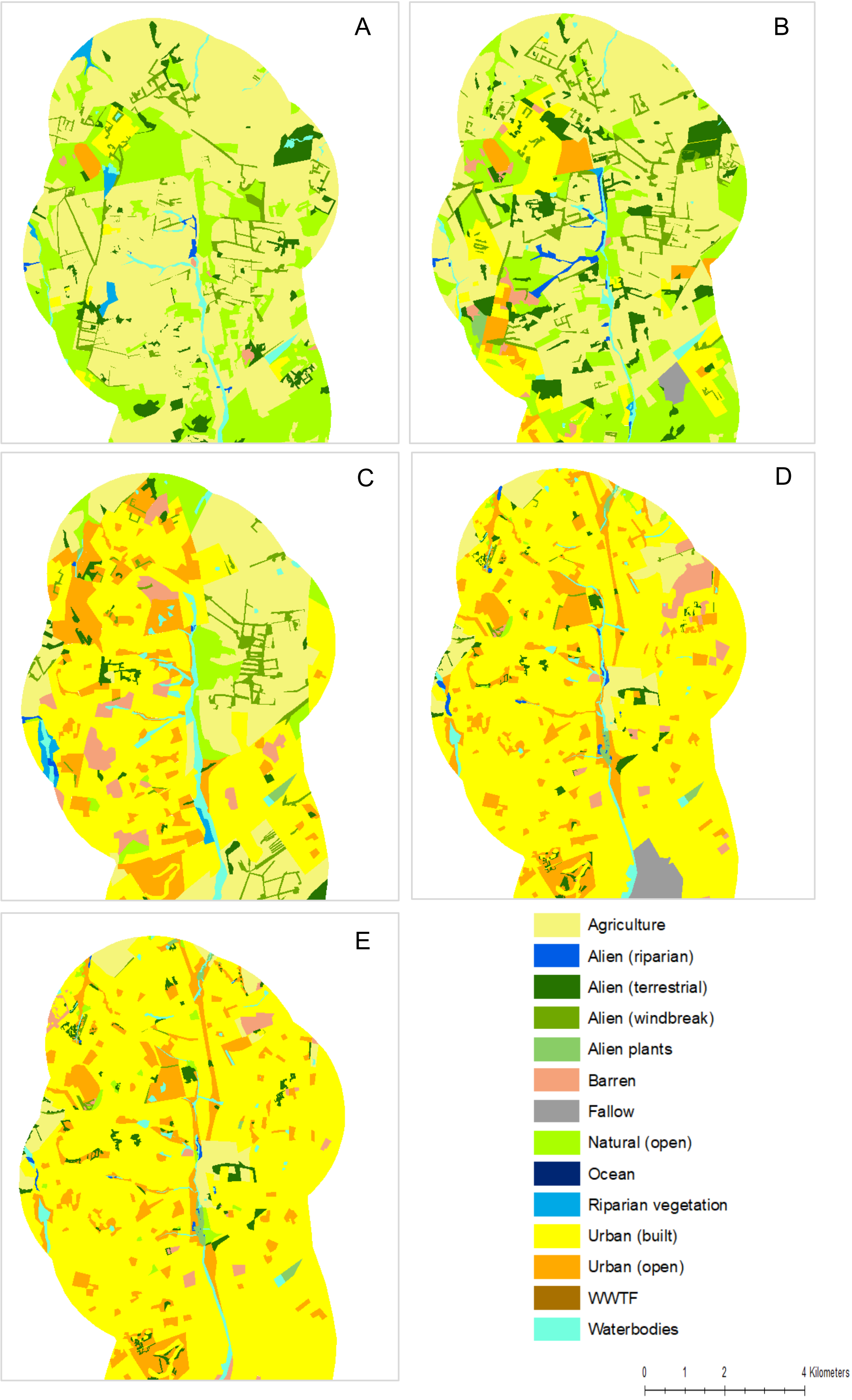



***Figure 3.6*** Land-use change within 2.5 km of the upper Kuils River, Cape Town. Different maps represent different time slices: A) 1944, B) 1968, C) 1989, D) 2008 and E) 2020.

***Table 3.3*** Proportional change in and strength of trends ($r_s$ values) for the percentage area of each LULC class for the different reaches of the Kuils River between 1944 and 2020. Not Applicable (NA) was denoted when p-values were invalid or if no LULC class was found in the reach. Asterisks (*) denote significant p-values.

| | 1944 - 2020 | | | | | |
|---|---|---|---|---|---|---|
| | **Upper** | | **Middle** | | **Lower** | |
| **LULC Class** | **% Change** | **$r_s$** | **% Change** | **$r_s$** | **% Change** | **$r_s$** |
| Agriculture | -57,43 | -1 | -18,79 | -0.9 | -11,55 | -0.9 |
| Alien (riparian) | -0,06 | -0.6 | -0,08 | -0.1 | 2,47 | 0.6 |
| Alien (terrestrial) | -2,93 | -0.6 | 1,49 | 0 | 0,81 | 0.6 |
| Alien (windbreak) | -2,80 | -0.7 | -0,34 | -0.9 | -0,99 | -1* |
| Alien plants | 0,37 | 0.8 | 2,92 | 0.7 | 5,93 | 0.9 |
| Barren | 2,61 | 0.7 | -2,26 | -0.2 | -2,72 | -0.3 |
| Fallow | 0,00 | NA | 4,04 | NA | 7,15 | NA |
| Natural (open) | -24,81 | -0.9 | -56,14 | -0.9 | -44,33 | -1* |
| Ocean | 0,00 | NA | 0,00 | NA | -0,21 | -0.7 |
| Riparian vegetation | -0,73 | NA | -0,02 | NA | -0,02 | 0.1 |
| Urban (built) | 77,35 | 1 | 62,02 | 1 | 35,13 | 1* |
| Urban (open) | 8,84 | 0.6 | 5,11 | 0.9 | 3,19 | NA |
| Waterbodies | -0,40 | -0.1 | 2,05 | 0.3 | 5,14 | 1* |

## 3.4 Discussion

Recognizing which activities drive land-use/land-cover (LULC) change over time forms a vital part in determining restorative goals and associated actions. Such maps not only visually show the extent of change that has occurred over time, but can also be useful for urban planning.

This study aimed to understand how LULC has changed over time along the Kuils River, to guide rehabilitation efforts in a 2.5 km radius of this urban river. Aerial imagery and GIS technology were used to assess land-use change over the last 80 years. Changes in LULC were mainly driven by human development, farming, the presence of waterbodies and, to some extent, the presence of alien vegetation. Similar changes were found by Liu et al. (2019) to have occurred in the Pearl River Delta. All analysis methods (percentage area, perimeter, relative percentage change from 1944 to 2020 and visual analyses) revealed the same trend – LULC change in the Kuils River is driven by urbanisation. Therefore, to make way for urban development, it was expected that agricultural development (and alien windbreak vegetation) and naturally occurring vegetation ('natural (open)') along the Kuils River would decrease. Various other researchers have also constructed general LULC maps for different sections of the Kuils River and these compare favourably to the findings of this study (Ninham Shand, 1994, 1999a; Brown and Magoba, 2009; Chingombe, 2012). Similar decreasing trends in agricultural land and natural open space were also found while conducting LULC assessments on the Berg River catchment in South Africa (Stuckenberg, 2012) between 1986 and 2000, as with the Dwars River (Du Plessis, 2020).

Although not universally the case in Africa, increases in urbanisation go hand in hand with increases in the need for ecosystem service provision, including access to water (United Nations, 2017). The occurrence of water bodies in the Kuils River has increased significantly, which can be attributed to increased water storage for human use, the resultant channelisation and canalisation of this river and the development of more permanent wetlands over time to accommodate for the increase in surface water flow and effluent discharge. Liu et al. (2019) also reported similar increases in area cover for waterbodies in the Pearl River Delta. Closer to home, Du Plessis (2020) also observed increases in waterbodies in the Dwars River due to an increasing need for water storage as a drought mitigation measure. Reports on the channelisation of the Kuils River, for example, date back to the 1980's which describe the plans for channelisation of the Nooiensfontien vlei, as well as a large area from the Bottelary River toward Kuilsriver (Ninham Shand, 1973, 1977a&b, 1979, 1983, 1994; Brown and Magoba, 2009) and the construction of the Driftsands Detention Dam (Ninham Shand, 1987). The construction of these permanent features has, however, resulted in the loss of seasonally inundated wetlands especially in the region of the Khayelitsha wetlands (see the bottom sections of the land-use maps in Addendum A for a visual illustration – refer to the presence of alien riparian vegetation and alien plants). Channelisation of the Kuils River has been shown to profoundly affect the diversity of native flora and macro-invertebrate species (Fisher, 2003) as only cosmopolitan weed species and hardy macro-invertebrate fauna were found.

Across the Western and Eastern Cape Provinces of South Africa, researchers have reported an increase in alien vegetation along river reaches over time. Along the Kromme River in the Eastern Cape Province, *A. mearnsii* density increased from approximately 20% to 30% area cover between 1954 and 2007 (Rebelo et al., 2015). Increases along the Kuils River were not as pronounced, with combined alien terrestrial vegetation, alien riparian vegetation and alien plants only increasing by 2.14% from 1944 to 2020. Alien species include *Acacia* species, *Hakea* species, willow trees, many invasive grass species and some invasive reeds (such as *Typha capensis*, water hyacinths and duckweed). The low increase in alien vegetation could be because alien and natural vegetation were underestimated due to their similarity in structure to native vegetation on aerial photographs. The occurrence of alien vegetation in the Kuils River has, nevertheless been reported on extensively, again dating back to the reports by Ninham Shand (1977, 1979, 1986, 1999a). The extent of many of the alien *Acacia* species (including *A. cyclops*, *A. saligna* and *A. longifolia*) has been recorded in the Cape Flats area since the 1960s (Roux, 1961; Roux and Middlemiss, 1963) yet records show that they were planted in the area as a dune stabiliser since the 1850s (Taylor, 1972). Their spread in the Nooiensfontein vlei, Khayelitsha wetlands and Baden Powell Drive area have also been mapped (Shand et al., 1994; Ninham Shand, 1999a). The above-mentioned invasive species were also recorded in the 1980s and 1990s by Ninham Shand and colleagues. Hence, the extent of LULC of alien terrestrial and alien riparian vegetation may be underestimated in this study.

Various authors have suggested the need to retain open space (or “green” space) as it provides many ecosystem services to the urbanising world (Botzat, Fischer and Kowarik, 2016) and suggestions to increase the extent of urban open space in response to the increase in urbanisation around the Kuils River have been made (Ninham Shand, 1987). As such, the occurrence of pockets of urban open space around the Kuils River have increased substantially since 1944 (Figure 3.6; compare 1968 to 2020 in Addendum A). However, without appropriate management, and due to the continuous disturbance that these areas face, these areas are considered either degraded or highly invaded and may generate ecosystem disservices rather than ecosystem services (Chapter Five). Similar results were also found downstream of this site, between the Bottelary River confluence and the N1 highway (Fisher, 2003) as well as in other parts of South Africa (see Musil, Milton and Davis, 2005). This particular wetland is also used as a temporary home for the homeless in the drier months, which was evident due to the presence of make-shift shelters and litter around the area.

The trend in land-use change of the upper, middle and lower reaches of the Kuils River, reflected similar results to that of the entire river. Major drivers of change in the reaches were increases in urbanisation, and decreases in natural vegetation (“natural open space”) and agricultural land). These results confirm that the Kuils River is becoming more urbanised and the biotic systems they support are novel, and this supports previous findings on the Kuils River (Ninham Shand, 1999a; Brown and Magoba, 2009; Chingombe, 2012).

As urban rivers are considered novel systems (Hobbs, Higgs and Hall, 2013), the management options for such ecosystems differ from that of natural rivers possibly due to the addition of novel drivers of change and the need for social, ecological and economic integration into their management. In urban systems, it is often said that management and restorative actions are less successful when public interest and concerns are not considered (Ractliffe and Day, 2001; Eden and Tunstall, 2006; Aronson et al., 2010). As a result, management actions should be designed to include the potential social and economic benefits of the restoration or rehabilitation activities. Learning from past restoration and rehabilitation projects can be useful for informing future management plans of other studies (Bernhardt and Palmer, 2011). Two urban river restoration projects provide lessons that can be applied to the Kuils River. The Bronx River in New York City is an urban river that has been impacted by urbanisation, effluent and stormwater discharge, littering, and alien vegetation since the early 1900s, which have ultimately resulted in biodiversity loss and water pollution (Marshall, 2001; Byron and Greenfield, 2006; Crimmens and Larson, 2006). The extensive damage that the Bronx has faced prevents it from being restored to its historic reference state. Various restorative activities have taken place in the Bronx River, including the development of multiple urban parks, improvement of stormwater management surrounding the river, planting of naturally occurring vegetation, removal of alien vegetation, stabilisation of river banks and removal of litter from the river (Perini and Sabbion, 2016; Bronx River Alliance, 2020).

Similar to both the Kuils and the Bronx Rivers, the Liesbeek River in the southern suburbs of Cape Town has also been impacted by urbanisation and effluent and stormwater discharge which have resulted in the pollution and degradation of the river (Brown and Magoba, 2009; Brill, Anderson and Farrell, 2017a) with canalisation of large sections of the river. Restorative activities in the Liesbeek River have now resulted in the river being recognised as one of the cleanest rivers in South Africa and some of the measures taken include removing litter weekly, creating various recreational parks and constructing artificial wetlands (Kotzé, 2020). To date, the Liesbeek River is home to many indigenous amphibian, fish and plant species and is enjoyed by many Capetonians.

Based on the lessons learnt from the above two river systems, it would not be feasible to restore the Kuils River to its historic state due to the extensive and multifaceted history of the river as well as financial and resource limitations. Holmes et al. (2020) have recommended that the first step in ensuring successful restoration is to act early to prevent biotic and abiotic thresholds from being crossed. Unfortunately, biotic and abiotic thresholds in the Kuils River have most likely already been crossed. Therefore, sections along the river should be targeted for rehabilitation or the enhancement of ecosystem services rather than restoration to its historical state (Gann et al., 2019). This could include reconstructing sections of the river to 1) mimic natural processes such as creating river sinuosity (Booth, 2005), thereby improving flood control management, 2) allow for the development of self-sustaining habitats for biotic communities (Booth, 2005), which will also provide safe spaces for recreational and cultural services, and 3) remove alien invasive vegetation.

Furthermore, previous work recommending management action along the Kuils River has been focused on alien vegetation control (Shand et al., 1994; Ninham Shand, 1999a; Fisher, 2003; Dube et al., 2017). Holmes et al. (2020) have suggested that active restoration (involving the removal of alien vegetation and actively collecting, planting and monitoring of native vegetation) in lowland riparian areas should be targeted in sites with either low-density or recent, dense alien tree invasions. In the Kuils River, the increase of alien weed and tree vegetation over time is evident in the middle and lower reaches of the river. However, if management options were to be focussed on these reaches, it could waste valuable investment (both in time and resources). This is because rivers are known to act as conduits for propagule dispersal (Richardson et al., 2007) so propagules from the upper and middle reaches would make the lower reaches more prone to invasion, as predicted by the river continuum concept (Vannote et al., 1980).Therefore, if alien clearing is included in the management plans, the upper and middle regions of the river should be prioritised. Similar alien invasive vegetation management strategies have been implemented in the Bronx River, where the river has been divided into multiple reaches based on the hydrogeomorphic setting of each river reach (Crimmens and Larson, 2006) as well as different parks (Yau et al., 2012). The result is that invasive species management is targeted from the upper watershed of the river downward (Crimmens and Larson, 2006).

The removal of alien vegetation in the Kuils River alone will however, not be sufficient to prevent further impacts on the Kuils River. Many other drivers of change, together with alien vegetation removal, will have to be addressed simultaneously, to improve the structure and function of the entire Kuils River. While mostly a by-product of urbanisation, these drivers include controlling water quality through stormwater runoff and effluent released into the river, limiting further urban development in and around the Kuils River and ensuring that farming activities in and around the river are sustainable and not detrimental to the river. It is therefore, suggested that the focus should be shifted towards creating more sustainable green spaces along the Kuils River, as done for the Liesbeek and Bronx Rivers, to help combat some of the impacts the river is experiencing while also educating the public on the benefits of rivers. In both the cases, the development of urban green spaces along the rivers has drawn more people to the rivers while also generating more public awareness about the state of the respective rivers (Friends of The Liesbeek, no date; Perini and Sabbion, 2016). As many cities, including in South Africa are now faced with the challenge of reducing their environmental footprint, ensuring that urban green spaces are efficiently managed can play a significant role in the sustainability of cities (World Health Organization, 2017; StatsSA, 2019). The focus of restorative action in the Kuils River catchment should thus be on the wetlands of the upper reaches of the river, as restoration of the upper reaches could have a positive effect on the river downstream (Kotze, Macfarlane and Edwards, 2020). This is discussed in more detail in Chapter Five.

While the results of this study described the current drivers of LULC change in the Kuils River, some limitations should be acknowledged. One of these is the uncertainty associated with the mapping of alien vegetation. This limitation was however, accounted for by recognising that the areas marked as native fynbos vegetation could contain alien vegetation. Secondly, land-use change of the Kuils River was only assessed in five-time slices over the 80 years, which resulted in a statistical power problem due to small sample size (Norton and Strube, 2001) when trend analyses were run on these data. Even though only a few LULC classes showed statistically significant trends, the majority of the LULC classes only had a 10% probability that the trends observed were by chance. It was therefore, safe to assume that these trends were likely to be significant. It is suggested that future studies should increase the sample size by creating more land-use maps and reducing the timeframe between the assessment periods, while also conducting finer scale LULC assessments to address the statistical power problem. Lastly, to further analyse the effect of fragmentation on the remaining natural spaces, road networks, sewage lines and other man-made structures should be included in the land-use mapping. For this study, however, the majority of roads were accounted for by their inclusion in the urban LULC classes.

## 3.5 Conclusions and Recommendations

In this study, the aim was to understand the drivers of land-use change in the Kuils River. Urban rivers, like the Kuils River, have experienced extensive land-use change due to the increase in human activities around these waterways. Mapping land-use change using aerial photography and GIS showed how land-uses along the Kuils River have changed between 1944 and 2020. Mainly driven by urbanisation and the loss of native vegetation, this study showed how the Kuils River has changed both across time and space. This study paints a general picture of what can be expected for other urban rivers. Degradation of urban rivers, as in the case of the Kuils River, is often caused by urbanisation, agricultural development, native ecosystems losses and, to some extent, alien invasion. When managing and rehabilitating urban rivers, the drivers of change within the river system should be considered before goals and action plans are strategised and implemented (Grobicki et al., 2001). In the case of the Kuils River, rehabilitation efforts should thus focus on selecting individual wetlands for rehabilitation in the upper and middle reaches rather than opting for entire river system restoration.

# Chapter 4
# Water Quality of the Kuils River and its Link to Land-Use Change

## 4.1 Introduction

It has been widely recognised that land-use practices have had major impacts on both the quality and integrity of freshwater resources (Carpenter et al., 2011; El-Khoury, Seidou, Lapen, Que, Mohammadian, Sunohara and Bahram, 2015; Giri and Qiu, 2016). Together with the high demand for freshwater, urbanisation and agriculture are two of the main contributors to degrading water quality, especially in urban rivers (Everard and Moggridge, 2012). Eden and Tunstall (2006:662) explain the relationship between humans and urban rivers as “bury(ing) them, turn(ing) them into canals, lin(ing) them with concrete, and build(ing) upon” them. Research on temporal and spatial water quality analyses have shown that land-use practices can have severe impacts on biological community structure, nutrient balances, hydrology and ecosystem services (Scott, Lucas and Wilson, 2005; Siriwardena, Finlayson and McMahon, 2006; Gao et al., 2017; Wang et al., 2020). Anthropogenic activities related to land-use change have, for example, resulted in changes to the geology, morphology and hydrology of many urban rivers, and have contributed to their pollution and habitat loss (Everard and Moggridge, 2012). A major consequence of urbanisation on urban river systems is water pollution (e.g. Mei et al., 2014), it has the potential to not only affect drinking water and human use availability, but can also affect water-based recreational activities and aquatic life.

Target 6.1 of the 2030 Agenda Sustainable Development Goals Targets is to “improve water quality,” while Target 6.3 emphasises the need for continuous water quality monitoring (United Nations General Assembly, 2015). Water pollution (or water quality) assessments are some of the most commonly used measurements to determine river ecosystem health (Stromberg et al., 2006; Hopkins, Marcarelli and Bechtold, 2011; Jayawardana et al., 2017). While the majority of water quality assessments are used to assess the suitability of water for specific uses, increasingly more studies focus on the importance of water quality assessments in terms of ecosystem health and the quality of aquatic environments (UNESCO, WHO and UNEP, 1996; Miserendino, Casaux, Archangelsky, Di Prinzio, Brand and Kutschker, 2011). These studies are important to ensure the availability of water for human use and to conserve and protect the natural terrestrial and aquatic environment. However, the water quality of urban rivers can be challenging to manage and monitor due to the presence of point and non-point sources of pollution. In urban systems, in particular, research has recently focussed on identifying and assessing sources of water pollution and spatio-temporal variations to inform sustainable water management strategies (Kannel et al., 2007; Tripathi and Singal, 2019; Maina and Yunusa, 2020). For example, Zhou, Wu and Peng (2012) have found that water quality deterioration of the Dongjiang River watershed in China is driven by increasing concentrations of nutrients and biological pollutants, which suggests that both point and non-point sources of pollution in this urban river are responsible for its degradation. It was thus recommended that the control of pollution sources should receive particular attention when considering management strategies for this river (Zhou, Wu and Peng, 2012).

However, without understanding the sources of water quality deterioration, the land-uses related to them and the complexity of the interactions between sites and water quality variables, addressing water quality problems can be challenging (Vrebos, Beauchard and Meire, 2017). To address this, Ren et al. (2018) used multivariate statistical techniques (including cluster analysis, Spearman's correlation, factor analysis and principal component analysis) to analyse spatio-temporal water quality trends and pollution sources of the Tongzhou Section of the Beiyun River. It was found that depending on the season, water quality was driven by different land-uses, with urbanisation being the primary driver during the dry season. Similar studies also highlight the power of multivariate analyses in analysing large, complex datasets and identifying indicators responsible for deteriorating water quality (Pekey, Karakaş and Bakoğlu, 2004; Yidana and Yidana, 2010; Zhao et al., 2011; Hajigholizadeh and Melesse, 2017).

The Kuils River, like many other urban rivers, has been impacted by urbanisation (Chapter Three; Chapter Four; Chingombe, 2012). Previous research on the state of the Kuils River has revealed that water quality has declined over time (Ninham Shand, 1983; Shand et al., 1994; Nel, Parker and Silbernagl, 2013; Mwangi, 2014). While most of these studies provide shortterm snapshots, long term trends in water quality provide better insight into the causes of water quality changes over time. Many assessments incorporate long term trend analyses in South Africa (e.g. De Villiers and Thiart, 2007; Dabrowski, Oberholster and Dabrowski, 2014; Griffin, Palmer and Scherman, 2014) that rely on coarse-scale national data provided by governmental authorities, and only a few private institutions conduct large-scale assessments. Studies combining both private, institutional and governmental data are thus rare. Furthermore, linking water quality degradation with land-use change can provide insight into potential causes or sources of pollution. In response to this, this study combined governmental, institutional and privately gathered data to assess how the water quality of the Kuils River has changed, spatially and temporally, over the last fifty years and what land-use activities have been driving this change.

With the hypothesis that the water quality of the Kuils River has degraded over time and space, the following questions were addressed:

1) How does water quality change over time?
2) How does water quality change over space?
3) How does the water quality of the Kuils River differ before and after wastewater treatment facilities (WWTFs)?
4) What are the main drivers of water quality change in the Kuils River?

## 4.2 Methods

### *4.2.1 Study Site Description*

The length of the Kuils River was divided into 12 sections, each representing different land-use impacts, including different towns and the location of WWTFs (see Figure 4.1 and Table 4**.1**).

### *4.2.2 Data Collection and Methodology Followed*

#### *4.2.2.1 Data Preparation*

As the water quality of urban aquatic systems is possibly one of the best measures to assess the effectiveness of urban river management in cities, basing water quality assessments solely on data gathered by government officials can be problematic, especially because most water quality samples are taken only at sites of particular concern (Day et al., 2020). This study, therefore, combined information from various sources to describe the ambient water quality of the Kuils River. Water quality data of the Kuils River from the 1970s to 2021 were collated from six governmental, private and tertiary educational institutions, including the City of Cape Town, the Department of Water and Sanitation, Ninham Shand, and the Universities of Cape Town, Stellenbosch and the Western Cape (see Addendum B, Table S4.1 for more information). Across the different studies, water sampling collection points and variables assessed differed over the years and between the different institutions. To address this, the river was divided into different sections (refer to Figure 4.1 and Table 4.1 below) and monthly water quality averages were then calculated per section. For statistical analyses, more frequent collection of data (e.g. daily monitoring) can be problematic as the data can lose its independence (McBride and Loftis, 1994; Cox, Moss and Smyth, 2005; Ngwenya, 2006). To address this, all daily data were averaged into monthly data points. Where two observations were recorded in the same month (either by the same data collector or different sources), the average was used, and when the number of monthly observations exceeded two, the median was used, similar to the approach used by Ngwenya (2006). To convert to absolute values, any data containing smaller than (“<”) symbols had a percentage of the value deducted from

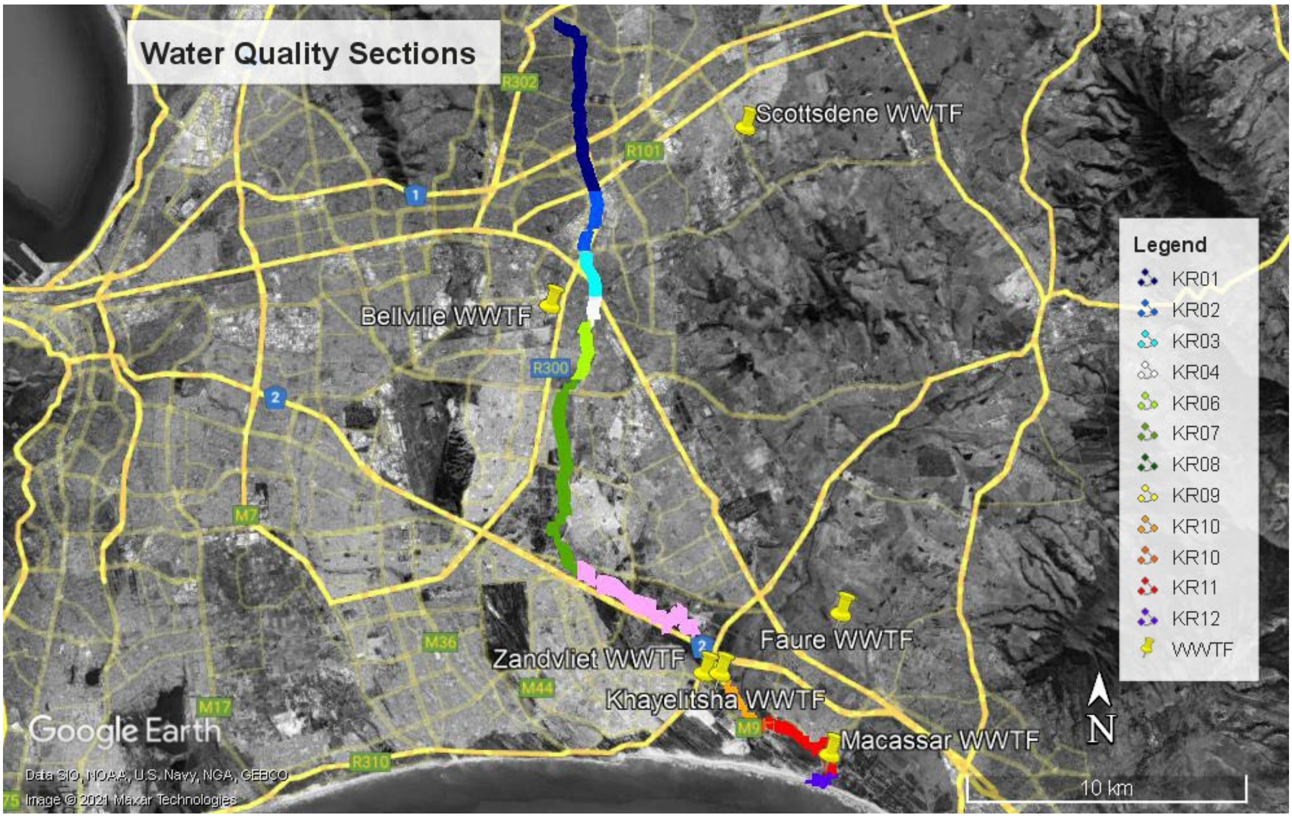



***Figure 4.1*** Map of the Kuils River sections (KR01-KR12) where water quality data, sampled since the 1970s, were spatially and temporally aggregated.

the observation. Before any analyses were conducted, variables were tested for normality and other statistical analysis assumptions, including homoscedasticity. All statistical analyses were conducted using Statistica (TIBCO Software, 2020; version 14.0.0.15).

#### *4.2.2.2 Temporal and Spatial Analyses and Water Quality Limits*

For the temporal analysis, data were averaged to generate yearly data points. Temporal changes of the physical, chemical and biological variables were assessed using a non-parametric Mann-Kendall test and a one-way analysis of variance (ANOVA), and significant differences between groups were determined using the appropriate posthoc tests. The Mann-Kendall test was used to describe monotonic trends of various water quality variables over time, to identify particular variables that should be monitored more closely. Data were then also split into two time periods of 24 years each (1972 - 1996 and 1997 - 2021) to assess changes between decades using an ANOVA.

For the spatial analysis, data were averaged to create a single variable value per river section. Similar to the temporal analysis, ANOVAs and posthoc tests were used to identify potential sites or sections of concern. The assumptions of an ANOVA (normality, independence and homogeneity of variance) were also assessed for both temporal and spatial analyses. A Levene's test was run to test the assumption of homogeneity of variance. If the variables conformed to the homogeneity assumption (p>0.05), the least-squares means were used for the ANOVA, otherwise, the weighted means were used.

For both temporal and spatial trends, water quality variables were compared to the South African national water quality guidelines (DWS, 1996) for domestic use and aquatic systems to determine whether the water quality conformed to national standards. Some of the variables were also compared to the health risk rating ranges of Day, Ollis, Ngobela & Rivers-Moore (2020). The health risk ratings used by Day et al. (2020) describe extreme exceedance of human health risk thresholds for various chemicals and biological variables as interpreted from the City of Cape Town's water quality categories, the Ecological Categories used for Present Ecological State Categories and the Department of Water and Forestry's water quality categories (see Day et al., 2020 for more information about the thresholds described). These risk ratings are categorised into 'Good', 'Fair', 'Poor' and 'Unacceptable' and are shown in Table 4.2.

**Wastewater Treatment Facilities and Drivers of Water Quality Change**

To assess the difference in water quality before and after WWTFs, ANOVAs and appropriate posthoc tests were used. This was done by comparing water quality variables before and after the discharge points and at the discharge point of both Bellville WWTF and Zandvliet WWTF. A correlation matrix determined relationships between all the variables assessed which was then used to highlight key variables to discuss in further detail. Lastly, a principal component analysis was conducted to explain the variance of larger inter-correlated variables (such as water quality variables) and to group the variables into smaller sets of uncorrelated data in order to recognise patterns (Pekey, Karakaş and Bakoğlu, 2004; Singh et al., 2004; Ren et al., 2018). A principal component analysis and a factor analysis (and related eigenvalues and factor loadings) were thus used to identify factors that drive the changes in water quality.

***Table 4.1*** Description of the water quality sections of the Kuils River.

| Section Name | Description | Section Coordinates Start | Section Coordinates End | (Average) Approx. Distance from the River Headwaters (km) | Land-Use & Sources of Pollution |
|---|---|---|---|---|---|
| KR01 | Upper Kuils River, to Durbanville area | 33°49'45.51"S 18°39'27.16"E | 33°53'15.45"S 18°40'30.69"E | 3,43 | Residential runoff, golf course |
| KR02 | Upper Kuils River, to Stikland Industrial | 33°53'15.45"S 18°40'30.69"E | 33°54'30.51"S 18°40'17.97"E | 7,34 | Residential runoff, industrial discharge |
| KR03 | Above Bottelary River to confluence | 33°54'30.51"S 18°40'17.97"E | 33°55'24.36"S 18°40'33.76"E | 9,79 | Residential runoff, industrial discharge |
| KR04 | Bottelary River confluence, to area above Bellville WWTF discharge point | 33°55'24.36"S 18°40'33.76"E | 33°55'54.55"S 18°40'27.54"E | 11,99 | Residential runoff, Bottelary tributary |
| KR06 | Below Bellville WWTF discharge point, to Stellenbosch Arterial | 33°55'54.55"S 18°40'27.54"E | 33°57'3.38"S 18°40'10.55"E | 15,41 | Bellville WWTF discharge, residential runoff, industrial discharge |
| KR07 | Upstream of Stellenbosch Arterial to Old Faure Road bridge | 33°57'3.38"S 18°40'10.55"E | 34° 0'48.87"S 18°40'14.01"E | 23,39 | Residential runoff, degraded land, protected area (wetlands) |
| KR08 | Area of Khayelitsha wetlands before Zandvliet WWTF discharge | 34° 0'48.87"S 18°40'14.01"E | 34° 2'7.63"S 18°43'1.62"E | 29,53 | Natural wetlands, urban park |
| KR09 | Area of Baden Powell Road to the Eerste River | 34° 2'7.63"S 18°43'1.62"E | 34° 2'55.04"S 18°43'43.29"E | 31,39 | Zandvliet WWTF discharge |
| KR10 | Upstream of the Eerste River confluence | 34° 2'55.04"S 18°43'43.29"E | 34° 3'42.68"S 18°44'41.88"E | 33,91 | Mining and agricultural land |
| KR11 | After the Eerste River confluence to upstream of the Eerste River Estuary (including Macassar WWTF discharge) | 34° 3'42.68"S 18°44'41.88"E | 34° 4'41.87"S 18°46'10.48"E | 38,11 | Eerste River confluence, residential runoff, Macassar WWTF discharge |
| KR12 | Eerste River Estuary | 34° 4'41.87"S 18°46'10.48"E | 34° 4'51.60"S 18°45'43.19"E | 39,11 | Eerste River estuary, Macassar WWTF overflow |

***Table 4.2*** Relationship between the Present Ecological State (PES) water quality categories and the City of Cape Town's water quality categories. Table adapted from Day et al. (2020).

| Department of Water Affairs and Forestry's water quality categories | Department of Water and Sanitation PES Categories | PES % score range | City of Cape Town Water Quality Categories | Interpretation of City of Cape Town Water Quality Categories |
|---|---|---|---|---|
| Ideal | A: Natural, unmodified | 90-100% | Good | Target |
| | B: Small change/ largely natural with few modifications | 80-89% | | |
| Acceptable | C: Moderately modified | 60-79% | Fair | |
| Tolerable | D: Largely modified | 40-59% | Poor | Below Target |
| Unacceptable | E: Seriously modified | 20-39% | Unacceptable | Unacceptable |
| | F: Critically modified | 0-19% | | |

## 4.3 Results

Six water quality variables were selected to report on further, based on previous studies assessing the water quality of urban rivers (e.g. Dallas and Day, 2004; Ngwenya, 2006; Mwangi, 2014) and the results of a correlation matrix assessing the relationship between all the variables (see Figure 4.5). The variables selected were 1) pH, which describes the natural variability of the waterbody and is an indication of acidification; 2) electrical conductivity (EC), which indicates the presence of metals such as chloride, potassium and calcium; 3) chemical oxygen demand (COD), describes the occurrence of organic pollution in waterbodies; 4) ammonia ($NH_3$), which is a measure of toxicity; 5) orthophosphates (OP), which is an indicator of nutrient levels and eutrophication; and 6) *Escherichia coli* (*E. coli*), which indicates the presence of bacterial pollution. To assess the drivers of water quality change, the full set of chemical and physical variables were included in the analysis; the additional variables are shown in the supporting Addendum B2.

### *4.3.1 Temporal and Spatial Variation of Variables along the Kuils River*

#### *4.3.1.1 Temporal Trends*

Comparing the selected variables to the National Water Quality Guidelines (NWQG) and the health risk rating ranges, it can be seen that the water quality of the Kuils River is poor. Five out of the six variables (electrical conductivity – EC, chemical oxygen demand – COD, ammonia – $NH_3$, orthophosphates – OP and *Escherichia coli – E coli*) were either beyond the NWQG's (Table 4.2) for domestic use or beyond the rating ranges of Day et al. (2020). COD of the Kuils River was more than 6 times higher than the NWQG for domestic use, while the OP average was approximately 17 times higher than the health risk rating ranges (Table 4.3). *E. coli* levels averaged at approximately 176 000 cfu/ 100 ml. pH was ~10% higher than the

NWQG for aquatic systems (which should not vary by more than a 5% or 0.5 pH from the background levels, either which is more conservative: DWAF, 1996a). Most of the six variables also showed increasing trends over time, even though these trends were not strong. The strongest increasing trends were found for pH and $NH_3$ (Table 4.3; pH: Kendall's tau = 0.41, p <0.01, $NH_3$: Kendall's tau = 0.43, p <0.01). The only variable showing a decreasing trend was COD (Table 4.3; Kendall's tau = -0.20, p = 0.04).

Three of the six water quality variables experienced significant changes between the first 24 and second 24 years of the analysis period. While both pH and EC showed significant increases (Figure 4.2; pH: Welch test F-statistic (1, 31.5) = 12.86, p <0.01; EC: Welch test F statistic (1, 28.7) = 7.97, p <0.01), COD decreased significantly (Figure 4.2; Current effect F (1, 40) = 5.37, p = 0.03).

#### *4.3.1.2 Spatial Trends*

The spatial analysis revealed that the water quality of the Kuils River decreased with distance from headwaters, particularly for COD, $NH_3$ and OP. COD decreased significantly over space from the headwaters to the Eerste River mouth (Figure 4.3A; Welch test F (10, 820.9) = 9.09, p< 0.01), but varied between sites. No clear differences were found between reaches (Table S4.4). A general increasing trend from KR3 to KR10 was observed for $NH_3$ (Figure 4.3B; Welch test F (10, 804.8) = 43.55, p <0.01), with no clear differences between reaches. OP concentration increased from the headwaters to the Eerste River mouth (Figure 4.3C, Welch test F (10, 577.8) = 283.29, p<0.01), with the upper reaches of the river (KR01-KR04) being significantly lower than the rest of the sites. For COD, $NH_3$ and OP, marked increases were seen after the Bottelary River confluence (after KR03; Figure 4.3). Comparing pH between sites, KR04 was significantly different from the other sites (Table S4.4), which is the reach after the Bottelary River confluence.

Most of the sites assessed were either above the NWQR for domestic use or above the health risk rating ranges for all the variables. For EC, COD, and *E. coli,* levels were above the NWQR domestic use limits for all sites. The health risk rating thresholds also showed OP and *E. coli* as 'unacceptable'. $NH_3$ concentrations for all sites (especially those in the middle and lower reaches) were below the NWQR limits for domestic use and aquatic systems, except for KR02-KR04, which were considered 'unacceptable' as per the health risk rating thresholds of Day et al. (2020).

pH and EC levels of treated wastewater discharged from both Bellville (KRB) and Zandvliet (KRZ) WWTFs into the Kuils River were significantly different from above and below the discharge points (Table 4.4). Additionally, *E. coli* levels of wastewater discharged at Zandvliet were also lower than that of KR08 and KR10. The *E. coli* count of wastewater discharged from Bellville WWTF was however higher than that of KR04 (Table 4.4). $NH_3$ and OP concentrations of the treated wastewater discharged from Bellville and Zandvliet were significantly higher than before and after the discharge points, with OP concentrations being considerably higher than $NH_3$ concentrations (Table 4.4).

***Table 4.3*** Summary statistics of the temporal changes in water quality of the Kuils River with statistical analyses (Mann-Kendall tests) of trends. S.D. – standard deviation, EC - electrical conductivity, COD –chemical oxygen demand, $NH_3$ – ammonia, OP – orthophosphates, *E. coli – Escherichia coli.*

| Variable | Unit | Summary Statistics of Variables | | | Kendall's tau | p value[1] | National Water Quality Guidelines (NWQG) | | Health Risk Rating Ranges[2] |
|---|---|---|---|---|---|---|---|---|---|
| | | Min | Max | Mean (S.D.) | | | Domestic Use | Aquatic Life | |
| pH | | 6.8 | 7.8 | 7.5 (0.2) | 0.413 | *<0.01* | 6-9 | * | - |
| EC | mS/m | 60.5 | 223.6 | 106.9 (30) | 0.27 | *<0.01* | <70 | - | - |
| COD | mg/l O | 0.001 | 158.7 | 60.9 (29.7) | -0.201 | *0.04* | 0-5 | - | - |
| $NH_3$ | mg/l N | 0.003 | 4.5 | 0.2 (0.7) | 0.426 | *<0.01* | <1 | <0.007 | >0.1 = Unacceptable |
| OP | mg/l P | 0.849 | 3.9 | 2.1 (0.7) | 0.085 | 0.44 | | <0.005 | >0.125 = Unacceptable |
| *E. coli* | cfu/ 100 ml | 65.0 | 902 871.7 | 175 788.9 (245 173.2) | 0.117 | 0.40 | <0 | - | > 4000 = Unacceptable |

[1] –P-values in *italics* show significant trends, [2] – Health risk rating ranges of Day et al (2020), * According to the National Water Quality Guidelines, pH should not vary by more than 5% or 0.5 pH from the background levels.

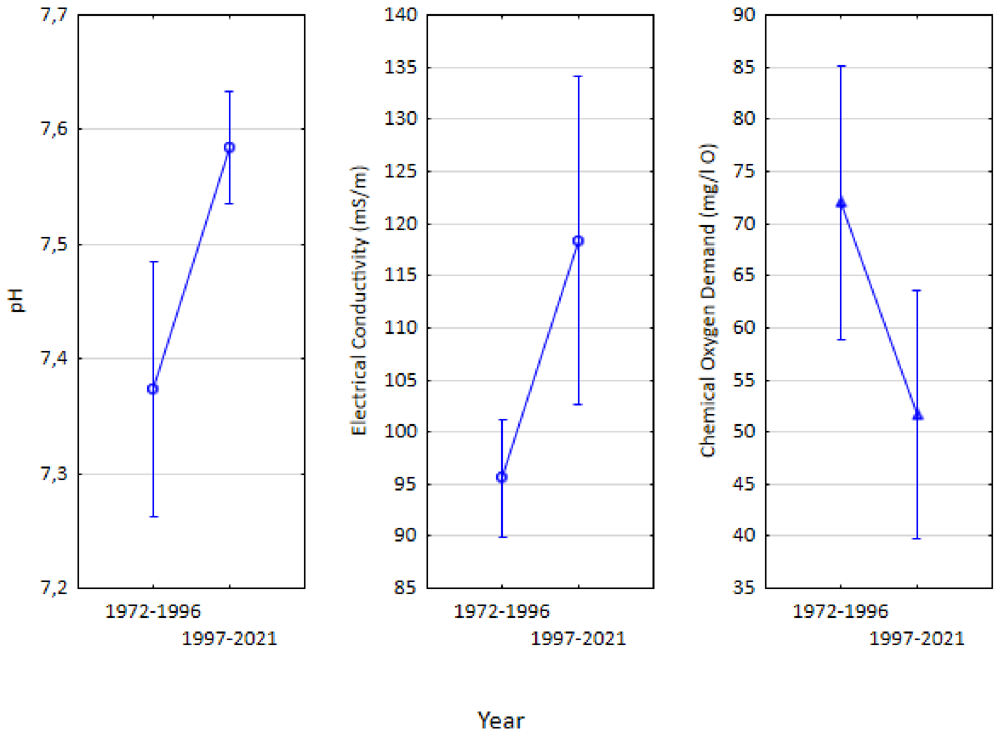



***Figure 4.2*** Comparison of temporal weighted means differences between two-time groups (1972-1996 and 1997-2021) of significant water quality variables in the Kuils River. Vertical bars denote 95% confidence intervals.

### *4.3.2 Drivers of Water Quality of the Kuils River*

To assess which variables drive the water quality of the Kuils River, a correlation matrix and principal component analysis were run. A strong positive correlation was found between dissolved oxygen (DO) and percentage oxygen saturation (POS; Figure 4.4). Strong positive correlations were also found between total phosphorous (TP), OP and total dissolved solids (TDS) and between *E. coli* and faecal coliform levels (Figure 4.4), confirming that these variables are interrelated. The first four principal components of the principal component analysis accounted for approximately 75% of the explained variance. The first principal component explained 37 % of the total variance and was positively correlated (loading > 0.75) with TP and OP and negatively with pH, DO and POS. The second principal component explained 16% of the variation and was correlated positively (loading > 0.55) with EC, COD, $NH_3$ and ammonium ion ($NH_4$). The third principal component explained 13% of the variance and correlated strongly (loading > 0.90) with temperature. A correlation biplot is shown in Figure 4.5 to show the principal component correlations. Variables related to nutrients and toxicants ($NH_3$, $NH_4$, TP and orthophosphates OP), organic pollution (DO, POS and COD) and salt concentration (EC) were the most important variables contributing to the variation in water quality of the Kuils River.

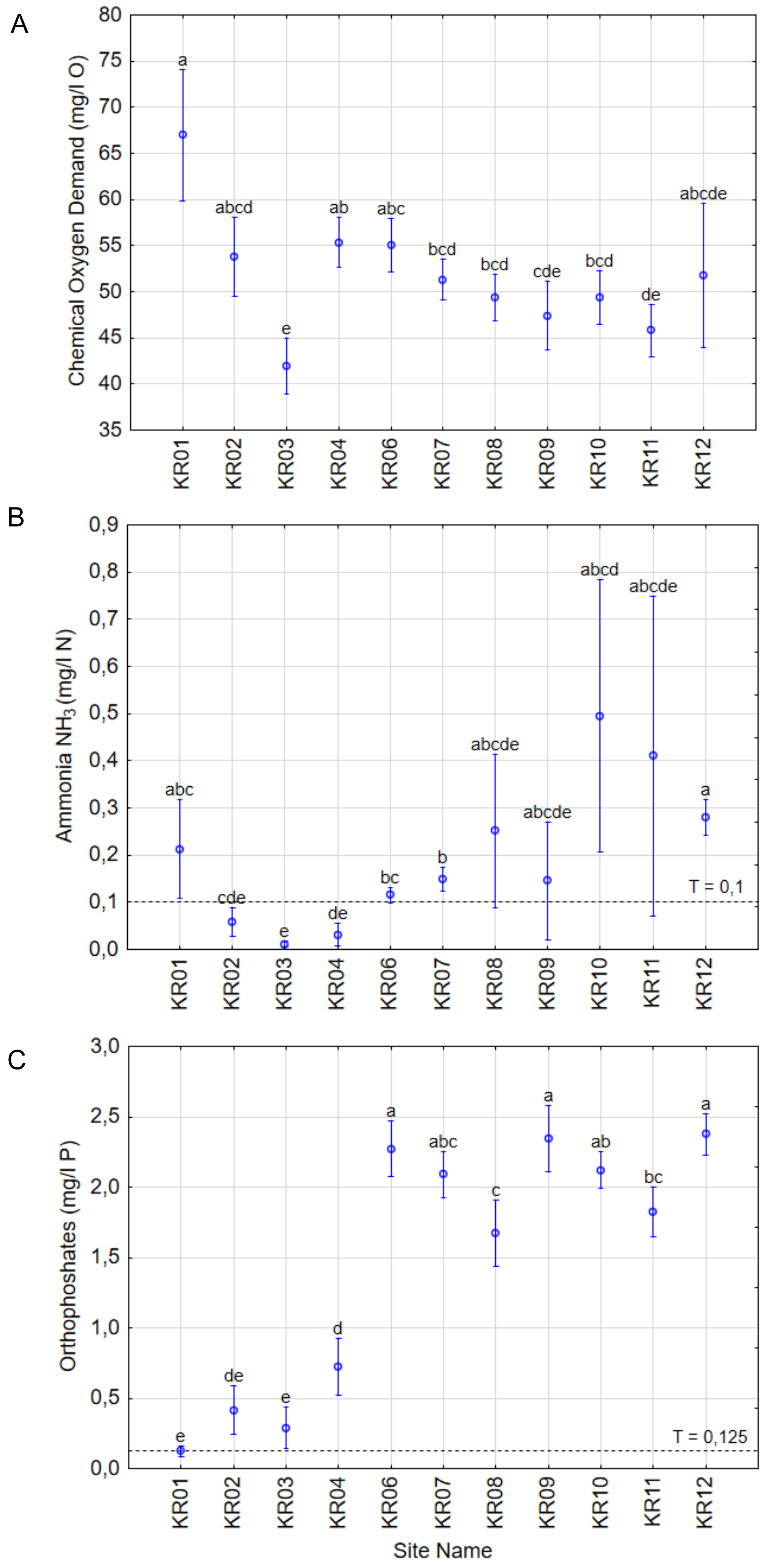



***Figure 4.3*** Change in water quality variables: a) chemical oxygen demand, b) ammonia and c) orthophosphates. The vertical lines indicate 95% confidence levels from the dot (the weighted mean) and the dashed black lines in B and C indicate the 'unacceptable' health risk threshold (T) according to Day et al. (2020). Significant differences between sites are indicated by different letters. Refer to Table 4.1 and Figure 4.1 for the location and description of the sites.

***Table 4.4*** ANOVA results of the comparison of average water quality before (KR04 and KR08), at (KRB and KRZ) and after (KR06 and KR10) wastewater discharge at the Bellville and Zandvliet Wastewater Treatment Facilities (WWTFs).

| | Bellville WWTF | | | | | Zandvliet WWTF | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| Variables | KR04 | KRB | KR06 | F statistic[1] | p-value | KR08[1] | KRZ[1] | KR10[1] | F statistic[1] | p-value |
| pH | 7.83 | 7.28 | 7.48 | 89.81[b] | *<0.001* | 7.56 | 7.12 | 7.57 | 56.34[b] | *<0.001* |
| Electrical Conductivity (mS/m) | 94.58 | 85.19 | 89.17 | 14.67[b] | *<0.001* | 89.31 | 67.13 | 93.34 | 93.24[a] | *<0.001* |
| Chemical Oxygen Demand (mg/l O) | 55.36 | 127.05 | 55.07 | 22.46[b] | *<0.001* | 49.38 | 71.08 | 49.36 | 7.34[b] | *0.026* |
| Ammonia ($NH_3$) (mg/l N) | 0.03 | 0.46 | 0.12 | 5.60[b] | *0.004* | 0.25 | 1.68 | 0.50 | 3.80[b] | *0.023* |
| Orthophosphate (mg/l P) | 0.72 | 4.23 | 2.27 | 92.49[b] | *<0.001* | 1.67 | 3.47 | 2.12 | 12.07[b] | *<0.001* |
| *E. coli* (cfu/ 100ml) | 22 839 | 117 419.8 | 631 665.1 | 3.09[b] | *0.046* | 98 565.05 | 1 900.63 | 13 668.34 | 9.59 | *<0.001* |

[1] – F-values annotated with an (a) were calculated using LS means (current effect) and (b) weighted means; p values in *italics* show significant differences.

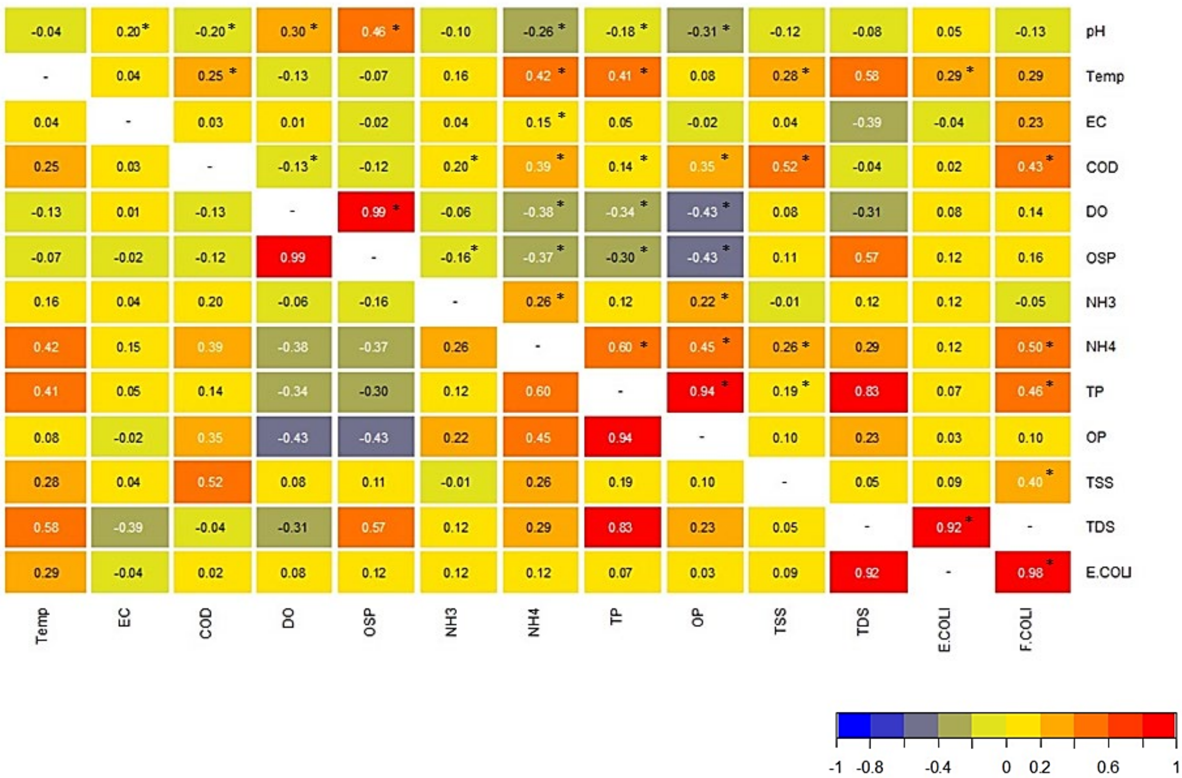



***Figure 4.4*** Correlation matrix showing the relationship between variables assessed along the Kuils River. DO – dissolved oxygen, OSP – percentage oxygen demand, pH, Temp – temperature, EC – electrical conductivity, COD – chemical oxygen demand, $NH_3$ – ammonia, $NH_4$ – ammonium ion, TP – total phosphates and OP – orthophosphates, TSS – total suspended solids, TDS – total dissolved solids, *E. coli* – *Escherichia coli*, F. coli – faecal coliforms.

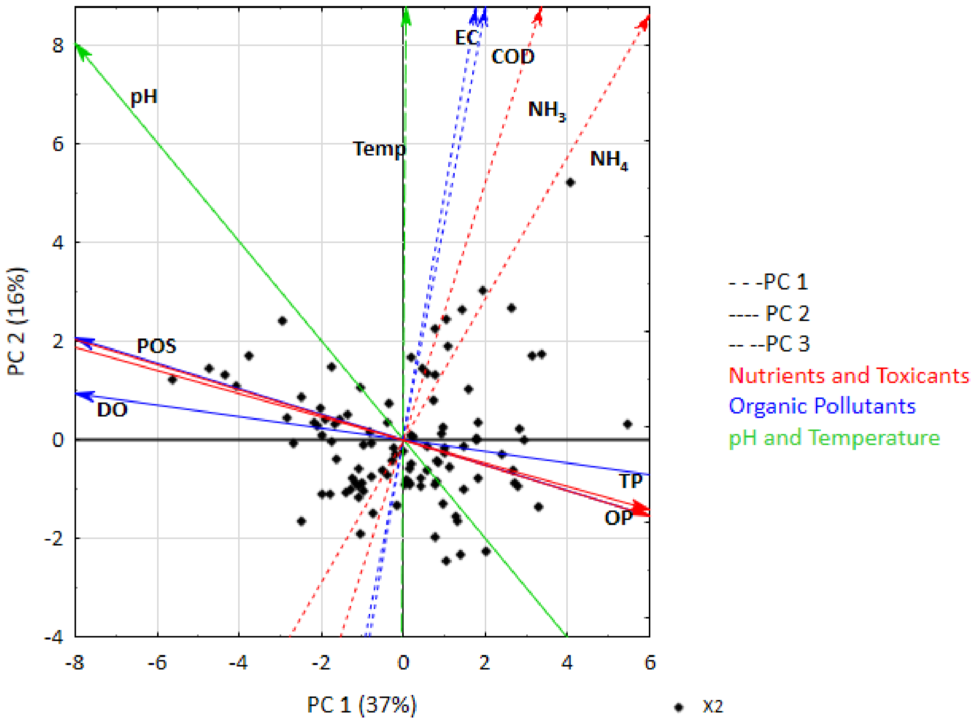



***Figure 4.5*** Correlation biplot of the principal component (PC) scores for the variables explaining the degrading water quality of the Kuils River. DO – dissolved oxygen, POS – percentage oxygen demand, pH, Temp – temperature, EC – electrical conductivity, COD – chemical oxygen demand, $NH_3$ – ammonia, $NH_4$ – ammonium ion, TP – total phosphates and OP – orthophosphates. Different dashed lines show which variables explain those PCs and colours explain the groupings of the PCs.

## 4.4 Discussion

In the Kuils River, indicators of river quality all point to an urban river system under stress, with key variables declining in quality over time and across space. Similar results have been found locally and internationally for other urban river systems (Khan, Husain and Lumb, 2003; Nel, Parker and Silbernagl, 2013; Ballantine and Davies-Colley, 2014; Mei et al., 2014; Mwangi, 2014; Glińska-Lewczuk et al., 2016; Ren et al., 2018; City of Cape Town, 2019; Wang et al., 2020). The findings of the current study suggest a degrading trend along the length of the Kuils River. Some of the most concerning trends were those indicators related to nutrients and toxicants, organic pollution and salts, and faecal contamination, all most likely contributing to the eutrophication of the Kuils River (Ngwenya, 2006; Mwangi, 2014; Dube et al., 2017; Day et al., 2020). The findings of Ren et al. (2018) point to the same indicators causing the decline in water quality in the Beiyun River. Comparing the water quality along the Chocancharava River in Codoba, Argentina, Gatica et al. (2012) found that urban and agricultural land-uses had elevated levels of pollution in comparison to their more natural counterparts due to a wider array of disturbance factors and pollution sources.

Correspondingly, the trends observed in the Kuils River are likely the result of land-use change associated with urbanisation, supported by the findings of Chapters Three and Five. The results indicate that urban expansion along the Kuils River has resulted in wetland deterioration over time, which includes water quality. Some possible land-uses and anthropogenic activities include canalisation, channelisation, municipal, industrial and sewerage discharge, agricultural development, eutrophication and the nutrient enrichment of gardens and recreational areas and an increase in impervious surfaces due to increased urbanisation. In a study by Singh, Bhardwaj and Verma (2020), similar trends were found. It was found that the extent of the Harike wetland in India decreased between 2006 and 2018 due to human activities, including urban expansion, which has ultimately led to a reduction in water quality of the wetland. Other cases of degradation of water quality due to urbanisation and related activities have been recorded in China (Huang et al., 2010; Li et al., 2019), the United States (Bengraïne and Marhaba, 2003; Hobbie et al., 2017) and Thailand (Bordalo, Nilsumranchit and Chalermwat, 2001). While water abstraction might have been a contributing factor of water quality degradation in the mid to late 1900's (e.g. as found in Coopenhagen, Denmark; Gejl, Rygaard, Henriksen, Rasmussen and Bjerg, 2019), the results of this study suggests that current causes of water quality degradation of the Kuils River relate to activities associated with an increase in urbanisation. Further research is however needed to assess the potential impact of groundwater water abstraction on water quality of the Kuils River.

Due to the large impact that land-use can have on a river's quality, it is recommended that catchment-scale impacts are considered when planning mitigation measures (Macfarlane et al., 2008; Lintern et al., 2018). This would include acknowledging pollution sources that might be more challenging to mitigate or control, such as urban runoff. Even though larger, catchment-scale changes are likely contributing to the overall degradation of the Kuils River quality, some key point sources can be highlighted; these are associated with 1) the tributary of the Kuils River – the highly polluted Bottelary River, and 2) WWTFs. According to the factor analysis, the three factors that have caused the degrading water quality of the Kuils River are organic pollutants, nutrients (phosphates) and pH, followed by toxicants, nutrients (nitrogen) and salts. While nutrients, toxicants and salt enrichment in the Kuils River are mainly a result of increased urban runoff, organic enrichment of the Kuils River is mainly due to sewage discharge (Dallas and Day, 2004; Mei et al., 2014). Furthermore, results indicate that the water quality of the Kuils River decreased after the Bottelary River confluence, particularly for organic pollutants, nutrients and pH. A spatio-temporal water quality analysis found that water quality of the Bottelary is worse than that of the Kuils, attributing the degradation of the water quality to urbanisation and the discharge of treated wastewater into the Bottelary system (Feng, 2005; Itoba Tombo, Thomas and Ed, 2012). Both studies point to Scottsdene WWTF as one of the main contributors to poor quality.

The Kuils River has been similarly impacted largely by discharge of treated effluent from the Bellville and Zandvliet WWTFs, to the extent that the Kuils River is now perennial (Brown and Magoba, 2009). Increased levels of *E. coli*, nutrients and organic pollutants, commonly associated with treated effluent (Dallas and Day, 2004), are observed after both the Bellville and Zandvliet WWTFs. In recent years, the release of untreated effluent into the Kuils River system has also caused public outrage, as communities that live along the Kuils River have become seriously ill (Carte Blanche, 2019; Knoetze, 2019). This is particularly concerning as water users downstream of these plants are also in the areas of the Cape where access to basic services such as water and sanitation is considerably lower. The degrading water quality of the Kuils and Bottelary rivers, as a result of WWTF discharge, is however a common issue across the world (e.g. Gannon and Busse, 1989; Edokpayi, Odiyo and Durowoju, 2017; Ozbay, Fan and Yang, 2017; do Nascimento et al., 2019). For example, organic pollutants, nutrients and salts were the major contributors to the variance in the water quality of the Wen-Rui Tang River in China, which was attributed to urban runoff and untreated sewage discharge (Mei et al., 2014). Schliemann, Grevstad and Brazeau (2021) also reported that urban runoff and WWTF discharge were the main contributors to the degrading water quality of the South Platte River in the United States of America (USA).

Therefore, to mitigate the degradation trends, a first step would be to focus on the key point sources. As more than half of South Africa's WWTFs are failing (Edokpayi, Odiyo and Durowoju, 2017; Kretzmann et al., 2021), urgent attention to these sources of pollution is required. Effluent discharge of the Bellville WWTF should thus be monitored more closely to ensure that the wastewater discharged into the Kuils River conforms to the national water quality guidelines. Three cost-effective mitigation measures can be considered to assist in improving the quality of water released into the Kuils River by WWTFs, the Bottelary River and urban runoff. Firstly, problem areas (particularly after WWTF discharge points and the Bottelary River) along the Kuils River can be aerated using a nanobubble generator. Micro-nano bubble technology successfully increased the concentration of dissolved oxygen in the river and thereby increased the concentration of bacteria and microbes, which ultimately reduced the nutrients and toxicants in the river (Wu et al., 2019). Secondly, the use of floating wetlands has also been shown to be effective in improving the water quality of urban watercourses and WWTFs by removing pollutants, excess nutrients and heavy metals, while also increasing the potential for biodiversity and aesthetics (Headley and Tanner, 2008; Tanner and Headley, 2011; Mitchell et al., 2014; Benvenuti et al., 2018; Afzal et al., 2019). It should be noted that this is a more sustainable option, as the aeration technique mentioned above can be more labour- and cost intensive.

Another alternative is to rehabilitate existing wetlands along the Kuils River. Not only do urban wetlands reduce the impact of flooding, but they also filter pollutants from both the air and water (WWT Consulting, 2018). There are over twenty-five wetlands along the Kuils River alone (Van Deventer et al., 2018), all with the potential to trap sediments, nutrients and toxicants (also refer to Chapter Five). For example, water quality at the Driftsands detention dam and Khayelitsha wetlands shows some improvement in water quality variables, especially nutrients, indicating the positive impact of wetlands on water quality. However, it is clear that the ecological condition of the wetlands along the Kuils River is degrading (refer to Chapter Five). It is thus suggested that the restoration of wetlands along the Kuils River be considered as an important mitigation measure for water quality. Rehabilitating wetlands through the improvement of their ecological condition can have knock-on effects on the water quality of the wetland and the river downstream. Restorative actions include planting vegetation (such as restios, *Anthospermum* spp., *Protea* spp., *Leucadendron* spp. and *Rhus* spp.; Pretorius, Esler, Holmes and Prins, 2008), especially those involved in trapping nutrients and sediments and controlling contaminant loadings (Russel, 2009). Similar restorative action has been implemented for the Nanfeihe River in China, with a positive impact of vegetation on both biodiversity and water quality in merely one and a half years (Wu et al., 2013). A comprehensive list of plant species that are suitable biological filters in South Africa and more specifically in the Kuils River, are outlined in the WET-RehabMethods guideline (see Russel, 2009).

In conclusion, the results of this study confirmed the hypothesis that the water quality of the Kuils River has degraded over time and space. The business-as-usual approach currently adopted by the City of Cape Town in managing its watercourses urgently needs to be addressed to ensure that the Kuils River system does not continue to degrade. The current state of the Kuils River and its water quality, therefore, emphasise the need for continuous water quality assessment and urgent action for its improvement. Since the processes and water quality variables responsible for the degrading water quality of the Kuils River have been identified, restorative action can now be focussed on a more informed approach to address the water quality issue in the Kuils River and the greater Cape Town area.

# Chapter 5
# Ecosystem Services Assessment of Wetlands along the Kuils River

## 5.1 Introduction

Many ecosystems are in a degrading state due to anthropogenic activities, thus impacting their ability to support biodiversity and to provide necessary goods and services (Costanza et al., 1997, 2014). This challenge is receiving global attention with two sustainable development goals (SDGs) specifically focused on sustaining biodiversity and ecosystem processes (Griggs et al., 2013; United Nations, 2017). Many of the other SDGs are dependent on and even driven by, the provision of ecosystem services (Reid et al., 2017; Ramsar, 2018, p. 61; Wood et al., 2018), emphasising the need for strong ecosystem health goals.

Even though urban green spaces are often undervalued in urban planning and management (Bobbins and Culwick, 2015; Díaz et al., 2018; Farrell et al., 2019), they provide many benefits to those living around them. These spaces (or green infrastructure) include parks, gardens, wetlands, lakes, ponds, and urban forests (Bolund and Hunhammar, 1999). While the most common benefits of urban green spaces include health, economic, cultural, social and environmental values, the importance of these services is often neglected (Bolund and Hunhammar, 1999; McInnes, 2010; Haq, 2011; Vargas-Hernández, Pallagst and Zdunek-Wielgołaska, 2018). Various studies show that urban green spaces are highly valued by communities (Balram and Dragićević, 2005; Davenport, Shackleton and Gambiza, 2012; Cilliers et al., 2013; Schäffler and Swilling, 2013; Botzat, Fischer and Kowarik, 2016; Shackleton et al., 2018; Balbi et al., 2019), and yet they continue to be degraded.

Like many natural ecosystems, urban green spaces are not only reducing in extent but are also degrading, placing additional pressure on their ability to provide the necessary services and benefits (Cilliers et al., 2013; Colding, Gren and Barthel, 2020; Twumasi et al., 2020). Specifically, urban wetlands are particularly under pressure despite their disproportionate value in providing a range of ecosystem services (Russi et al., 2013), including flood attenuation, trapping sediment, toxicants and nutrients, carbon storage, access to water and harvestable resources, and cultural services. Not maintaining wetland ecosystem health can have serious economic, social and health implications (Russi et al., 2013). There is thus a need to evaluate the current conditions of urban wetlands to allow for planning and management actions to maintain the provided benefits and ecosystem services.

Ramsar (2018) highlights the importance of wetlands in the United Nations Agenda of 2030, stating that various aspects of wetland function directly contribute to the achievement of the 17 SDGs. For example, urban wetlands play a vital role in contributing to sustainable cities through carbon sequestration, nutrient and pollutant trapping, job opportunities and access to food and water (Ramsar, 2018). As a signatory to the United Nations Agenda of 2030 and the Conference of the Parties 21, South Africa is required to improve its sustainability practices. Target 15.1 of the SDG Agenda 2030 Agreement, for example, states that the "conservation, restoration and sustainable use of terrestrial and inland freshwater ecosystems and their services, in particular…wetlands" should be ensured by 2020 (United Nations General Assembly, 2015, p. 24). Yet, many urban wetlands in South Africa are either degraded or degrading (Nkosi, 2006; Dube et al., 2017; The Freshwater Consulting Group, 2019), stressing the need for continuous assessment of the state of urban systems. The increasing pressure on ecosystems, including wetlands, to provide ecosystem services in Cape Town (Rebelo et al., 2011; O'Farrell et al., 2012; Balbi et al., 2019), coupled with the degrading state of the Kuils River (Fisher, 2003; Chingombe, 2012), provided a perfect opportunity to assess the wetlands along the Kuils River. Incorporating such ecosystem service assessments into ecosystem health assessments, such as the River Health Programme's riparian vegetation, habitat integrity and faunal species assessments, have proven valuable to advocate for restorative actions (Villa et al., 2009; TEEB, 2011; Kotze, Macfarlane and Edwards, 2020).

In South Africa, the WET-Health and WET-EcoServices tools are the most common tools used by scientists and research practitioners to identify the state of health of wetlands and watercourses, while also assessing their ability to provide ecosystem services. Previous use of both these tools confirms its relevance in describing the current status and ecosystem service provision of wetlands across South Africa, including wetlands in the urban and rural setting (see Kotze, 2005; Sinchembe and Ellery, 2010; Ramburran, 2012; Rebelo et al., 2017; Khumalo, 2019; Libala, Palmer and Odume, 2021). Most of the wetlands assessed using these tools were found to be in a degrading or degraded condition, with a limited ability to provide ecosystem services mainly due to some human-induced (and related) land-use change impacts (see Malachite Specialist Services, 2016; The Freshwater Consulting Group, 2017, 2019; WaterMakers, 2018; Khumalo, 2019).

This study aimed to assess the health and potential ecosystem service provision of four of the wetlands along the Kuils River, to inform the rehabilitation potential of these wetlands. By assessing the current ecosystem health and determining the most prevalent ecosystem services of the selected wetlands along the Kuils River, the aim was to assess how the state and provision of ecosystem services would change if appropriate rehabilitation measures were implemented. The hypothesis was that wetlands situated in nature reserves would be in better condition than those surrounded by urban development.

The findings of this research are intended to aid the development of appropriate restorative and management actions for the Kuils River.

## 5.2 Methods

### *5.2.1 Study Site Description*

#### *5.2.1.1 Wetlands Selected*

Four wetlands along the Kuils River were selected for this study (see Table 5**.1** and Figure 5.1) based on their proximity to the Kuils River, and using the National Wetland Map 5 (Council for Scientific and Industrial Research, 2018). Three of the wetlands were located in the upper reaches of the river, while the remaining wetland was situated in the middle to lower reaches. Due to their size, wetlands two, three and four were subdivided into smaller assessment units (AUs) based on lateral water inputs to the assessment units and their hydrogeomorphic settings. A total of 13 AUs were evaluated in the four wetlands. The first wetland (KR 1) is located in Durbanville, opposite the Durbanville Sports club and golf course (see Figure 5.1). The main water input to this wetland is the river itself, fed through a single culvert. Vegetation in this area is dominated by grass and some larger, planted trees, such as pepper (*Schinus molle*) and water oak (*Quercus nigra*) trees. The second wetland (KR 2) is situated along Fairtrees Road, between the Durbanville Golf course and the National Road 1 bridge. The third wetland (KR 3) is located below KR 2 and stretches toward Frans Conradie Drive in Brackenfell. Both KR 2 and KR 3 are also fed by the main river channel with additional water input from stormwater canals and urban runoff. Vegetation in these wetlands is mostly semi-natural but invaded, with large areas of *Typha capensis* and other riparian vegetation as well as some alien acacias, willows, pines, grasses and native *Oxalis* species. The area of KR 1, 2 and 3 is broadly characterised by critically endangered Cape Flats Sand Fynbos with some areas of Swartland Shale Renosterveld, although little of these vegetation types remain intact. Lastly, KR 4 is located within the Driftsands Nature Reserve, in Blue Downs, in an area supporting endangered Cape Flats Dune Strandveld. Six assessment units were originally planned for evaluation in KR 4. However, due to recent land invasions, two assessment units were effectively eliminated as study sites. The main water input to KR 4 is stormwater and treated effluent from the surrounding WWTF's. Vegetation in the reserve includes some native Cape Flats Dune Strandveld vegetation and some alien vegetation such as *Acacia cyclops, A. saligna, Ricinus communis* and kikuyu grass (*Pennisetum clandestinum*). Unique anthropogenic impacts (as described in Table 5.1) affect the functioning and structure of each of the individual wetlands and their assessment units.

***Table 5.1*** Characteristics of the four selected wetlands along the Kuils River.

| Characteristics | | | | |
|---|---|---|---|---|
| Name | KR 1 | KR 2 | KR 3 | KR 4 |
| Location | 33°50'9.39"S 18°40'6.45"E | 33°51'10.07"S 18°40'6.52"E | 33°52'39.75"S 18°40'16.01"E | 33°59'39.29"S 18°39'55.99"E |
| | Durbanville | Stellenberg | Brackenfell | Blue Downs |
| District | Northern District | Northern District | Tygerberg District | Khayelitsha/ Mitchell's Plain Greater Blue Downs District |
| Type of wetland | Channel-valley bottom | Valley bottom wetland (some channelled, some unchanneled) | Floodplain | Floodplain |
| Size | 4.25 ha | 22.07 ha | 5.62 ha | 49.83 ha |
| Altitude | 108 ± 7 m | 88 ± 12 m | 75 ± 3 m | 35 ± 3 m |
| Number of assessment units | 1 | 6 | 2 | 4 |
| Description | Urban wetland; recreational activities; artificial birdhouses and bee hives | Semi-natural wetland; alien vegetation; potential WWF reserve | Semi-natural wetland; alien vegetation; public park | Semi-natural wetland; alien vegetation; nature reserve |
| Possible sources of pollution | Golf course and sports grounds, urban runoff | Golf course, adjacent farmland, urban runoff and stormwater discharge | Urban runoff and stormwater discharge | Treated wastewater; urban runoff, stormwater discharge |

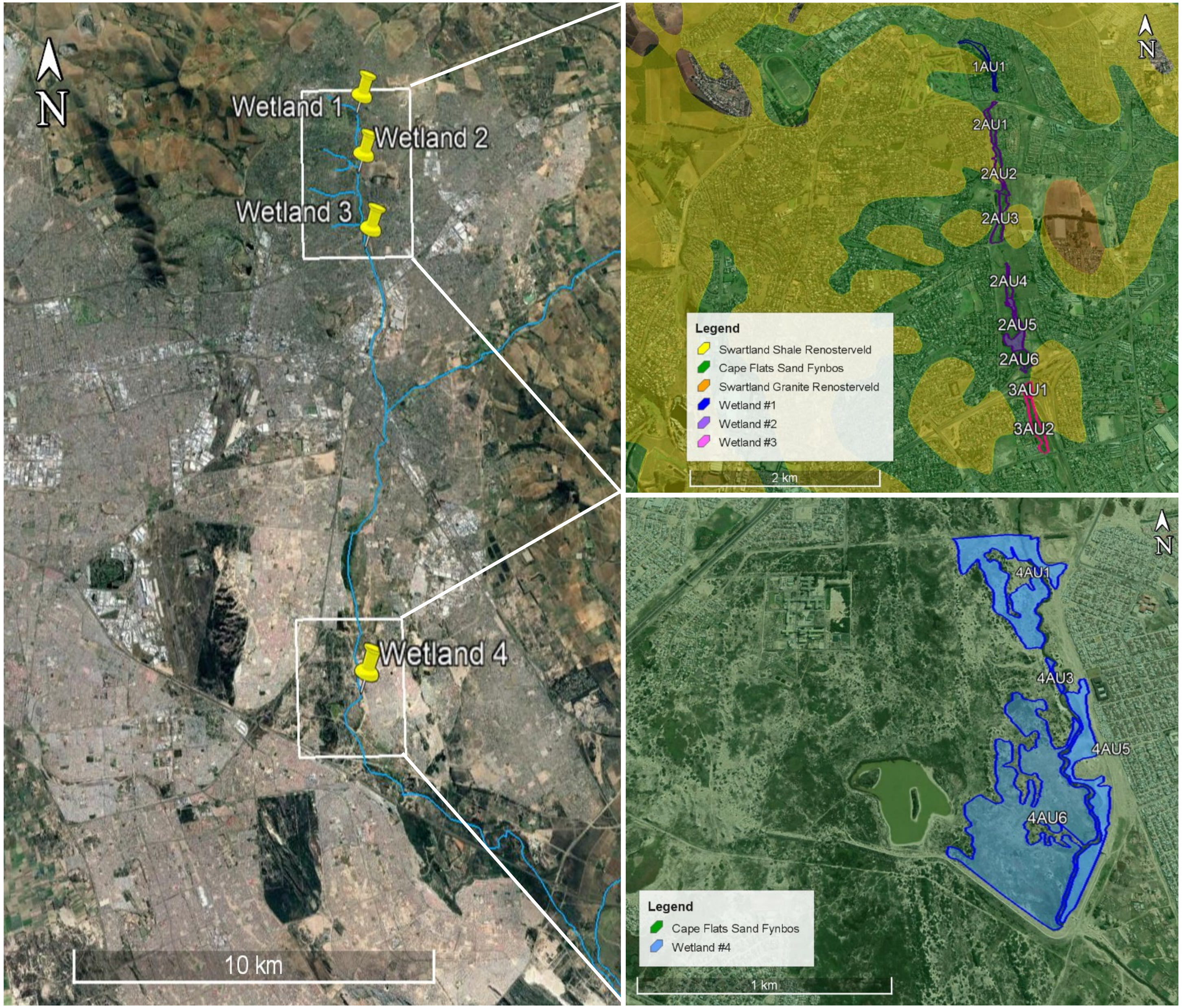



***Figure 5.1*** Sites selected for the wetland assessments along the Kuils River. Figures shown on the right indicate the various assessment units and the colour coding indicate different vegetation types.

### *5.2.2 Assessment of Wetland Health and Ecosystem Services*

#### *5.2.2.1 WET-Tools*

The Water Research Commission developed a series of tools (i.e. the WET-Series) specifically designed to assess, rehabilitate and manage South African wetlands and riparian areas (Dada et al., 2007; Ellery et al., 2009). These tools can be used at any spatial or institutional scale, be it for one wetland or a series of wetlands (Dada et al., 2007). Not only are they designed to inform wetland rehabilitation decision-making, but some of the tools can also be used for more specific purposes such as assessing the functions, values and conditions of specific wetlands (Dada et al., 2007). Refer to Addendum C for more information on the WET-Series (particularly the WET-Health and WET-EcoServices tools).

The purpose of the WET-Health tool is to monitor the state of the wetlands under assessment in comparison to a natural reference condition (Macfarlane, Ollis and Kotze, 2020). The first step involves defining the hydrogeomorphic setting of the wetland. Then, using a rapid assessment, users score the health of the wetland from zero to four based on pre-developed questionnaires of indicators that relate to the four health components of the wetland assessment - hydrology, geomorphology, vegetation and water quality (Dada et al., 2007; Macfarlane, Ollis and Kotze, 2020). These scores are based on assessments conducted in the field, reviewed literature and expert knowledge (Rebelo et al., 2019). An explanation of the assessment of each health component can be found in Table S5.5. Using an algorithm, the WET-Health application impact scores are combined to generate a Present Ecological State score for each of the health indicators which is then categorised into an Ecological Category to describe the extent of anthropogenic influence (Table 5.2; Dada et al., 2007; Macfarlane, Ollis and Kotze, 2020). The Ecological Categories range from A which describes a wetland as being intact, unmodified and natural, to an Ecological Category of F which describes a wetland that has been transformed to such an extent that ecosystem function and biota have been lost (Table 5.2; Kotze 2007).

The WET-EcoServices tool was developed to allow decision-makers, among others, to quickly assess the condition of wetlands based on the ecosystem services they provide and the demand for such services (Kotze, Macfarlane and Edwards, 2020). The WET-EcoServices tool follows the same procedure as the WET-Health tool, where the first step is to define the hydrogeomorphic setting of the wetland. Using 16 ecosystem services, this tool uses a five-point scale (0 to 4) to rate a set of indicators in terms of both the supply of the services and the demand therefor (more information in Table 5.3 and Table S5.6 in Addendum C5). Scoring of WET-EcoServices questions uses information boxes that provide the researcher with clear direction when choosing the appropriate impact or score (Kotze et al., 2007). While a score of four would indicate a high demand for or supply of an ecosystem service, a score of zero would suggest that there is no demand for the service or that the wetland is incapable of supplying the service (Kotze et al., 2007; Kotze, Macfarlane and Edwards, 2020). These scores are also based on field assessments, previewed literature and expert knowledge (Rebelo et al., 2019b). An algorithm then combines the indicator scores to generate overall supply and demand scores (Ecological Importance and Sensitivity scores).

More information about the process of each tool can be found in Addendum C1 and Addendum C4.

***Table 5.2*** Ecological Categories used for Present Ecological State assessments in the WET-Health assessment. Table adapted from Macfarlane, Ollis and Kotze (2020).

| **Ecological Category** | **Description** | **Impact Score** | **Present Ecological State (PES) range** |
|---|---|---|---|
| A | Natural, unmodified | 0 - 0.9 | 90 - 100% |
| B | Mostly natural with few modifications; slight change of ecosystem processes and a potential small loss of natural biota and habitats. | 1 - 1.9 | 80 - 89% |
| C | Moderately modified; intact natural habitat, but moderate change of ecosystem processes and a loss of natural habitat. | 2 - 3.9 | 60 - 79% |
| D | Largely modified; large change in ecosystem processes and loss of natural habitat and biota. | 4 - 5.9 | 40 - 59% |
| E | Seriously modified; presence of some remaining natural habitat features, but a great change in ecosystem processes and loss of habitat. | 6 - 7.9 | 20 - 39% |
| F | Critically modified; critical modification, almost complete loss of natural habitat and modification of ecosystem processes. | 8 - 10 | 0 - 19% |

***Table 5.3*** Ecosystem services considered in the WET-EcoServices tool.

| | **Ecosystem Services** |
|---|---|
| Supporting and Regulating Services | Flood attenuation |
| | Streamflow regulation |
| | Sediment trapping |
| | Erosion control |
| | Phosphate assimilation |
| | Nitrate assimilation |
| | Toxicant assimilation |
| | Carbon storage |
| | Biodiversity maintenance |
| Cultural Services | Cultural and spiritual experience |
| | Education and research |

| | |
|---|---|
| | Tourism and recreation |
| Provisioning Services | Provision of cultivated foods |
| | Food for livestock |
| | Provision of harvestable resources |
| | Provision of water for human use |

### *5.2.2.2 Wetland Assessments and Methodology*

The wetlands of the Kuils River were identified as being valley-bottom and floodplain wetlands. Due to the size of the wetlands assessed, they were split into AUs to simplify the assessment process. AUs were split when there was a physical disturbance, e.g. a road or a stormwater inlet into the AU. AUs and "external areas of influence[1]" were then delineated and zoned into pre-defined LULC classes through landcover mapping (as conducted in Chapter Three) using aerial photography. The areal extent (in hectares) of the LULC classes of the AUs and areas of influence were then mapped and recorded to account for changes in the natural functioning of the different hydrogeomorphic units (Macfarlane, Ollis and Kotze, 2020). The land cover maps were then refined through the assessment of indicators that impact wetland health and ecosystem services.

Desktop-based research included maps of geological features, rainfall patterns, etc. as well as previously published reports on the wetlands. Data layers from Google Earth (Pro) (version 7.3.3.7786) and Cape Farm Mapper (version 2.3.4) were consulted for information on land-use (Department of Environmental Affairs, 2018), wetland extent (CD:NGI, no date; DWS, no date; Van Deventer et al., 2018), alien vegetation (ARC, 2011), conservation (WCBSP, 2017) and other geographic features. Desktop assessments were then ground-truthed by conducting site visits at each wetland between the 10th and 12th of May 2021. Vegetation characteristics (presence of different species on site), the hydrological setting (flow of water), geomorphology (texture and presence of different soil types), and indicators of wetland water quality were assessed visually (e.g. for the presence of oil and litter). Further information noted while on-site included identifying different faunal and floral species, identifying sources of point- and non-point pollution, and finding evidence of recent human disturbance such as infilling. For the WET-EcoServices assessments, the human use of the AUs were noted while on site in order to account for cultural services supplied by the AUs and the demand therefor. Similarly,

[1] Mapping "areas of influence" involves delineating a 200m buffer around the wetland, a 200m buffer around all inflow and outflow streams of the wetland and the remainder of the wetland's catchment. The wetland catchment is defined as "all portions of the catchment beyond the 200m … buffer adjacent to the wetland", which is mapped using 100m contours from the mountaintop to the edge of the wetland (MacFarlane, Ollis and Kotze, 2020, p. 17).

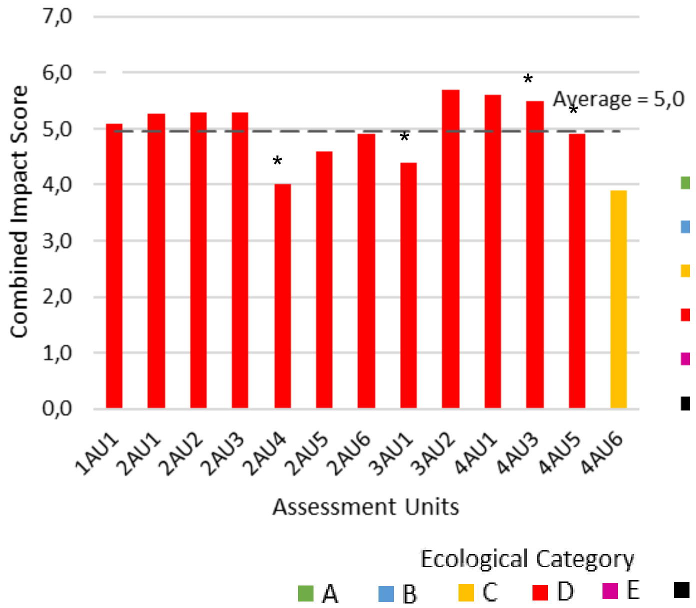
demand scores were generated based on the number of beneficiaries of the service and their level of dependency and the context of the wetland catchment (e.g. point-source pollution; Kotze, Macfarlane and Edwards, 2020).

Based on the findings of the desktop research and site verification, the WET-Health and WET-EcoServices applications were then used to describe the impacts on each AU, based on the scores given to each indicator. The tools then generate present ecological state and ecological importance and sensitivity scores, respectively. To graphically represent the ecological importance and sensitivity scores, radar diagrams were used. Radar diagrams use a circular display to show multiple variables (in this case various ecosystem services) simultaneously and therefore allow the comparison of the variables assessed. This allows the reader to compare the provision of ecosystem services for wetlands as a whole, while also highlighting aspects of the wetland system which could be targeted to enhance the provision of certain ecosystem services.

Prospects of the condition of the AUs were then considered to determine if restorative action would result in changes in the condition of the AUs (based on the present ecological state and ecological importance and sensitivity scores). The present state wetland assessments informed which theoretical restorative action could best enhance the provision of ecosystem services and improve the health of the wetlands. Following the same methodology as described above for the WET-Health and WET-EcoServices assessments, the future health state and ecosystem service delivery potential were estimated by incorporating the impact of various restorative and management actions on wetland health and ecosystem service provision into the allocated scores. Restorative actions that were incorporated into the improved scores were concrete-lined canal removal to improve wetland and river sinuosity and increase surface roughness, alien vegetation removal, the reintroduction of native riparian and fynbos vegetation, and infill material removal. Again, present ecological state and ecological importance and sensitivity scores were generated, which were then used to compare pre-restoration and post-restoration health and ecosystem service delivery. This also involved some assumptions. For example, it was assumed that rehabilitation measures such as improving the aesthetics of the wetlands would increase the cultural use of the wetlands. It was also assumed that it would be possible to restore wetlands 4AU3 and 4AU5 by removing illegal settlements from the area and replanting indigenous vegetation.

## 5.3 Results

Rapid wetland health assessments revealed that almost all wetlands assessed had an overall ecological category D (Largely modified, see Figure 5.2 and Figure 5.3). Of all the health components assessed, the geomorphology of each wetland was in a much better state than the remaining components (hydrology, water quality and vegetation), each scoring an ecological category of C. Variation in ecological category scores for the different health components were also highest for geomorphology (ranging from an A to a D), whereas the other health components scored between a C and F. Only four assessment units out of thirteen experienced a change in ecological category after the theoretical implementation of restorative measures (Figure 5.2).




| | Assessment Unit | Impact Score Change |
|---|---|---|
| Durbanville | 1AU1 | 0.4 |
| Fairtrees Park/ KR Corridor | 2AU1 | 0.9 |
| | 2AU2 | 0.4 |
| | 2AU3 | 0.8 |
| | 2AU4 | 0.1 |
| | 2AU5 | 0.5 |
| | 2AU6 | 0.5 |
| KR Corridor | 3AU1 | 0.9 |
| | 3AU2 | 0.6 |
| Driftsands NR | 4AU1 | 0.5 |
| | 4AU3 | 1.7 |
| | 4AU5 | 1.1 |
| | 4AU6 | 0.4 |

***Figure 5.2*** Combined impact score and its related ecological category of the 13 assessment units (AU) assessed. Asterisks (*) indicate AUs that experience a PES score improvement after theoretical rehabilitation. The table describes the change in the overall combined impact score after the implementation of theoretical restorative measures. The first number of the AU name refers to the wetland number.

The WET-EcoServices assessments indicated no clear difference in the current provision of ecosystem services between the wetlands in the urban greenbelts and those in Driftsands Nature Reserve (Figure 5.4). Supporting and regulating services were the most supplied services across all assessment units. Wetlands in Driftsands Nature Reserve provided higher cultural and biodiversity maintenance services in comparison to other wetlands (Figure 5.4). Assessment units 1AU1 and 2AU1 supplied the least number of services due to the channelisation of the Kuils River at that point. Similarly, 4AU3 and 4AU5 also supplied a low range of services due to the recent land invasion and related disturbances which includes erosion to the river banks, illegal dumping in the area and the release of effluent into the river. Assessment units 1AU1, 2AU1, 4AU3 and 4AU5 also had the lowest nutrient and toxicant assimilation services. The supply of food for livestock and cultivated food services was highest for assessment units 2AU1 and 2AU4.


***Figure 5.3*** Photos taken while conducting wetland assessments at 2AU1 on 12/05/2021. A) Sections of the river channel stabilised with gabions, B) Stormwater drains emptied into the wetland and its surrounds, with non-native weed overgrowth, C) Erosion of the river banks in the upper reaches of the wetland, and D) Channelisation of the headwaters of the wetland.

Wetlands in the Driftsands Nature Reserve were associated with the highest demand for a larger variety of supporting, regulating and provisioning services. Demand for services supplied by the other wetlands was almost non-existent other than for some supporting and regulating services. For all wetlands, the demand for toxicant and nutrient assimilation services were assessed as higher than the supply. The demand for provisioning services (e.g. water supply) was only found in the Driftsands Nature Reserve wetlands associated with the recent land invasion.

After applying theoretical rehabilitation measures, a general increase in the supply of cultural services was observed, when comparing the current to the future supply of ecosystem services (Table 5.4). Some ecosystem services such as flood attenuation, carbon storage and harvestable resources also increased after theoretical rehabilitation measures were implemented. The effectiveness of wetlands in performing ecosystem services in the future after theoretical rehabilitation measures was more pronounced for the assessment units in the

Driftsands Nature Reserve. Wetlands in the upper reaches of the Kuils River mostly only showed improvements in the delivery of cultural ecosystem services. However, there was no change in the demand for ecosystem services after the implementation of the theoretical rehabilitation measures.

Lastly, the ecological importance and sensitivity results based on the demand and supply of ecosystem services showed that nutrient and toxicant assimilation services are more valuable for most of the unchanneled valley bottom wetlands compared to the upstream channelled valley bottom wetland (wetland 1) and floodplain wetlands of wetland 4 (Table 5.4). Biodiversity maintenance services, and cultural and spiritual services, were however mostly confined to the assessment units in Driftsands Nature Reserve.

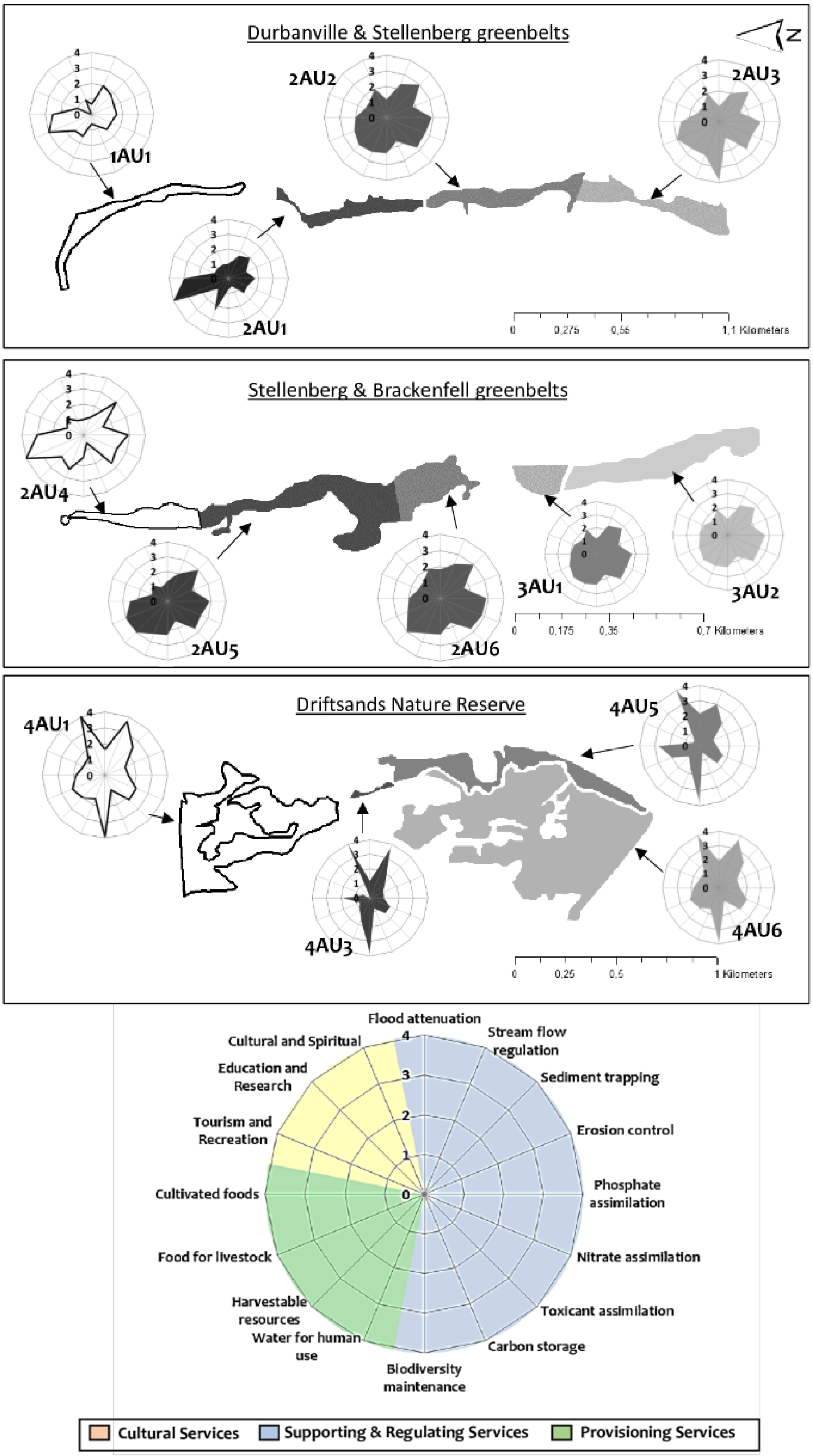



***Figure 5.4*** The ecosystem service provision of four wetlands in the radar diagrams. The location of the wetlands and assessment units are shown in Table 5.1 and Figure 5.1. The 16 ecosystem services are scored from 0 to 4. The shading of the radar diagrams corresponds to the assessment unit in the corresponding block. For scores for each ecosystem service, refer to Table S5.3 in Addendum C7.

***Table 5.4*** Present Ecological Importance and Sensitivity of 13 wetlands along the Kuils River. Different gradients of green represent different importance categories from 'Very Low' (white) to 'Very High' (dark green). Numerical values of the Ecological Importance and Sensitivity scores linked to the categories are 'Very Low' (0-0.79), 'Low' (0.8-1.29), 'Moderately-Low' (1.3-1.69), 'Moderate' (1.7-2.29), 'Moderately-High' (2.3-2.69), 'High' (2.7-3.19) and 'Very High' (3.2-4.0). As an example, a 'Very Low' category is where the importance of services supplied is very low relative to that supplied by other wetlands.

| | | ASSESSMENT UNITS | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| ECOSYSTEM SERVICE | | 1AU1 | 2AU1 | 2AU2 | 2AU3 | 2AU4 | 2AU5 | 2AU6 | 3AU1 | 3AU2 | 4AU1 | 4AU3 | 4AU5 | 4AU6 |
| REGULATING AND SUPPORTING SERVICES | Flood attenuation | 0,0 | 0,0 | 0,0 | 0,0 | 0,0 | 0,0 | 0,5 | 0,0 | 0,1 | 1,1 | 0,6 | 1,6 | 0,9 |
| | Stream flow regulation | 0,8 | 0,5 | 1,2 | 0,8 | 0,4 | 0,8 | 1,2 | 1,2 | 1,2 | 3,7 | 3,7 | 3,0 | 3,7 |
| | Sediment trapping | 0,8 | 1,0 | 2,0 | 1,8 | 2,0 | 2,0 | 2,0 | 1,8 | 2,0 | 1,5 | 0,5 | 1,1 | 1,5 |
| | Erosion control | 0,1 | 0,0 | 1,2 | 1,7 | 0,4 | 1,2 | 1,0 | 1,2 | 1,0 | 1,9 | 1,5 | 1,5 | 1,9 |
| | Phosphate assimilation | 1,6 | 1,8 | 2,9 | 2,7 | 2,9 | 2,9 | 2,9 | 2,7 | 2,9 | 2,0 | 1,4 | 1,7 | 2,0 |
| | Nitrate assimilation | 1,3 | 1,3 | 2,7 | 2,3 | 2,3 | 2,5 | 2,8 | 2,5 | 2,8 | 2,7 | 2,0 | 2,2 | 2,7 |
| | Toxicant assimilation | 0,9 | 0,9 | 2,3 | 2,0 | 2,1 | 2,2 | 2,3 | 2,1 | 2,3 | 2,2 | 1,5 | 1,6 | 2,2 |
| | Carbon storage | 0,6 | 0,4 | 1,5 | 0,9 | 0,4 | 1,3 | 1,5 | 1,5 | 1,5 | 0,8 | 0,6 | 0,3 | 0,8 |
| | Biodiversity maintenance | 0,1 | 0,3 | 1,8 | 3,4 | 1,0 | 1,8 | 1,8 | 1,8 | 1,8 | 4,0 | 4,0 | 4,0 | 4,0 |
| PROVISIONING SERVICES | Water for human use | 0,0 | 0,9 | 1,1 | 1,2 | 1,7 | 1,2 | 1,1 | 1,1 | 1,1 | 1,6 | 1,3 | 1,6 | 1,4 |
| | Harvestable resources | 0,0 | 0,0 | 1,2 | 1,2 | 0,7 | 1,7 | 1,7 | 1,0 | 0,7 | 1,5 | 0,0 | 0,0 | 1,5 |
| | Food for livestock | 1,5 | 2,5 | 0,8 | 1,5 | 2,7 | 1,5 | 0,8 | 0,8 | 0,8 | 1,3 | 0,0 | 0,5 | 1,3 |
| | Cultivated foods | 1,0 | 1,5 | 0,5 | 1,0 | 1,7 | 1,0 | 0,5 | 0,5 | 0,5 | 0,8 | 0,3 | 1,3 | 0,7 |
| CULTURAL SERVICES | Tourism and Recreation | 0,2 | 0,0 | 0,3 | 0,3 | 0,0 | 0,0 | 0,3 | 0,4 | 0,2 | 0,2 | 0,0 | 0,0 | 0,2 |
| | Education and Research | 0,0 | 0,0 | 0,2 | 0,2 | 0,2 | 0,2 | 0,2 | 0,2 | 0,2 | 0,2 | 0,0 | 0,0 | 0,2 |
| | Cultural and Spiritual | 0,0 | 0,0 | 0,7 | 0,5 | 0,0 | 0,0 | 0,5 | 0,5 | 0,5 | 3,5 | 2,5 | 2,5 | 3,5 |

## 5.4 Discussion

This study assessed the health of the wetlands along the Kuils River as well as the ecosystem services they provide, with the intention to recommend future restorative actions. Understanding the condition of wetlands is an important step in determining the drivers of wetland change and the restorative actions that can be implemented. Using the WET-Health and WET-EcoServices tools (Kotze, Macfarlane and Edwards, 2020; Macfarlane, Ollis and Kotze, 2020) proved useful in determining the condition of the wetlands along the Kuils River and helped to identify potential anthropogenic impacts responsible for their condition.

The findings of this research suggest that urban wetlands along the Kuils River have experienced a system regime shift away from their historic state (Hobbs et al., 2006; Gann et al., 2019). This shift is due to the ongoing and increasing human impact on these systems and related changes to the biotic species composition (the spread of alien vegetation), hydrological processes (erosion, water quality and quantity changes), historical disturbance (wetland infilling, channelisation and excavations) and nutrient inputs (Chapter Three; Chapter Four; Fisher, 2003; Hobbs, Higgs and Harris, 2009; Chingombe, 2012; Francis, 2014; Morse et al., 2014). As the land-use of the area surrounding the Kuils River has become more urban over the last 80 years (refer to Chapter Three), a degrading condition of the wetlands was expected. While one would predict “protected” systems to be in a better state, however, this was not the case for the wetlands along the Kuils River as all wetlands were in a similar ecological condition (i.e. Category D), and all either form part of an urban park (or “greenbelt”) or are in a protected area, i.e. Driftsands Nature Reserve. The history of development within Driftsands over the last 50 years could explain its low ecological condition (Cape Nature, 2015). Additionally, since the Driftsands wetlands are in the lower end of the catchment, it is also subject to cumulative impacts from the upper reaches of the Kuils River, as predicted by the river continuum concept (Vannote et al., 1980).

Even though the wetlands along the Kuils River are degrading, they still provide many ecosystem services. Results show that erosion control, carbon storage, tourism and recreation are the most positively impacted by restorative action; indicating the importance of the wetlands’ attenuation services in urban settings, especially in Cape Town. Natural hazard regulation, improved water quality and waste assimilation, tourism and recreation are considered the most important services contributing to the City’s economy (with an approximate net present value of between R 77.5 and R 148 billion annually based on the current 2021 rand value; De Wit et al., 2009). Thus, the importance of restorative action and appropriate management of the wetlands along the Kuils River and elsewhere in Cape Town cannot be underestimated.

After theoretical rehabilitation, the health of all wetlands improved but only some showed an improvement in present ecological state scores. This implies that restorative actions could improve wetland health and to some extent the provision of ecosystem services (Kotze, Tererai and Grundling, 2019), although this study did not consider the cost implications or the trade-offs between restoration, hard-engineering and soft engineering approaches. Ecological infrastructure investments may well be more affordable than hard engineering solutions, for example, Kotze, Tererai and Grundling (2019) and Black and Turpie (2016) found that hard-engineering solutions such as hard-structural surfaces were found to be generally more costly than invasive alien species clearing (Kotze, Tererai and Grundling, 2019; Black and Turpie, 2016). Costs should thus be considered when identifying and prioritising wetlands for restoration. For example, the cost of assessing the impact of restorative action in the urban Colbyn wetland, in Pretoria (which included the construction of gabion weirs, loosening soils, removing invasive alien Kikuyu and re-vegetating the area with wetland grasses) was high (approximately R 2.4 million), but when assessed using the WET-Health and WET-EcoServices tools, it was found that restoration with these interventions greatly improved both the condition of the Colbyn and its ability to provide ecosystem services regardless of the high costs of restoration (Delport, 2012; Sherwill, 2015; Nemutamvuni, 2018; Kotze, Tererai and Grundling, 2019). The cost of implementing restorative action along the Kuils River could therefore most likely also be justified, depending on the restorative measures implemented.

Gann et al. (2019) recognised a range of actions that can be applied to improve habitat conditions and achieve the highest possible recovery. These actions include ecological restoration (i.e., the full recovery of a system to its prehistoric state) and rehabilitation (i.e. restoring ecosystem functioning of the system without reference to its prehistoric condition). Ecological restoration of the wetlands along the Kuils River to a historic state is however not feasible due to the long and complex history of the Kuils River and its surroundings, the limited availability of resources and funds and the changes to the wetlands' hydrology, water quality and vegetation structure. Similar conclusions were made for the Eerste, Dwars, and Bottelary Rivers, as well as studies conducted on other urban rivers (Fisher, 2003; Richardson et al., 2007; Hobbs, Higgs and Harris, 2009; Meek, Richardson and Mucina, 2010, 2013; Du Plessis, 2020). Gann et al. (2019) thus suggest that in the urban space, if ecosystems only have the potential to partially recover, ecological restoration should at least aim to manage and mitigate potential threats, while ensuring that some ecosystem function is retained, and some native species persist.

The focus of restorative measures of the Kuils River and its wetlands should thus be to enhance ecosystem service provision whilst also maintaining some ecological integrity

(Grayson, Chapman and Underwood, 1999; Hobbs et al., 2006; Hobbs, Higgs and Harris, 2009; Gobster, 2010; Nilsson and Aradóttir, 2013; Aronson, Blignaut and Aronson, 2017; Ravit et al., 2017), and rehabilitating where required. Restorative measures could include removing man-made canals, channels, and excess sand from illegal dumping and infilling, selectively enhancing impoundment and stabilizing soil banks (USEPA, 2000). As some of these interventions can be costly, environmental managers and decision-makers should act before ecological thresholds have been crossed and also prioritise sites for rehabilitation (Gann et al., 2019; Holmes et al., 2020). For the wetlands along the Kuils River, this would mean that restorative action should be targeted based on the ecosystem services that specific wetlands provide.

Due to the urban nature of the wetlands in the upper reaches of the river, these wetlands are mostly associated with assimilation services (sediment trapping, nutrient assimilation and toxicant assimilation services). According to Tong et al. (2007), the enhancement of assimilation services is mostly a function of the vegetation structure of an area. In the Kuils River, the majority of the vegetation in the wetlands are invasive plants. Therefore, to enhance the assimilation services of the wetlands in the upper reaches of the Kuils River, it is suggested that invasive alien vegetation be removed and replaced with naturally occurring riparian and fynbos vegetation (e.g. indigenous grasses, *Protea* spp., *Leucadendron* spp., *Rhus* spp. and restios). The presence of invasive species is known to constrain the recovery of native vegetation in wetlands of South Africa, and thus more focus should be put on actively improving wetland vegetation structure during restoration rather than assuming vegetation recovery will follow after hydrology and geomorphology improvements have been implemented (Holmes, 2007; Tererai, Gaertner, Jacobs and Richardson, 2013; Cowden, Kotze, Ellery and Sieben, 2016; Kotze, Tererai and Grundling, 2019). Re-vegetation of indigenous vegetation can however be costly and should therefore be carefully considered.

In a study by Ruwanza, Gaertner, Esler and Richardson (2018), post-restoration success of alien removal and active restoration was measured by surveying riparian sites along the Berg River in the Western Cape. Although native vegetation recovery was successful after aliens were cleared, the recovery process was slowed by reinvasion. This is also likely to occur in urban riparian zones as urban areas are often more prone to disturbance. Richardson et al. (2007) argue that the uniqueness of cities in terms of their vegetation structure should be embraced. This would mean that communities should be allowed to self-assemble. The problem, however, is that although the majority of alien plants in riparian zones are non-disruptive and could potentially contribute to the provision of ecosystem services (such as the nutrient assimilation services of poplar trees; Potgieter et al., 2020), some invasive species require continuous intervention due to their fast regeneration times (Holmes et al., 2005; Pretorius et al., 2008). In the Kuils River, this includes giant reeds (*Arundo donax*), *Acacia* species and *Sesbania* species (Holmes et al., 2005) that should be prioritised for clearing. Furthermore, regenerating aliens should be controlled to limit their further spread (Reinecke et al., 2007; Richardson et al., 2007; Potgieter et al., 2020). Future studies could investigate the relationship between invasive vegetation along the river and the ecosystem services they provide to further aid in management decisions regarding invasive species control.

Assessing the value of ecosystem services by ecosystem type, it was found that wetlands are among the most valuable ecosystems per hectare (Costanza et al., 1997, 2014). Based on this, the focus should be to enhance and rehabilitate existing, larger, functioning wetlands along the river, such as the Kuils River Corridor in the Stellenberg/ Vredekloof area, the detention dam and its surrounds in Driftsands Nature Reserve and the Khayelitsha Wetlands. As the provision of supporting and regulating services supplied to users are affected by the condition of upstream wetlands (Kotze, Macfarlane and Edwards, 2020), restorative action should be focussed on the upstream sites such as the Kuils River Corridor, which will have knock-on effects on the state of the downstream wetlands (e.g. those in the Driftsands Nature Reserve). Increasing the recreational use value of the Kuils River corridor, clearing the alien acacia seedlings in the wetland and managing stormwater and urban runoff input into the individual assessment units could be considered as options for wetland rehabilitation.

From the above, it is clear that appropriate management would be needed to ensure the successful rehabilitation of wetlands along the Kuils River. In the Kuils River, wetlands are degrading due to human-induced activities such as land-use change and stormwater and urban runoff, which have resulted in changes to the wetlands function and their ability to supply ecosystem services. Due to the continuous disturbance these systems face, as well as their novelty, restoration to a historic state will not be feasible. It is thus recommended that focus should be placed on enhancing and maintaining ecosystem services, while also aiming to promote ecological integrity. It should be noted that if current conditions remain unchanged, these systems could struggle to keep up with the demand to supply the necessary ecosystem services. Planning restorative action is thus urgently needed, with the implementation of an adaptive management approach focused on enhancing ecosystem service delivery, invasive species control, hydrology and geomorphology improvements and involving the communities living around these wetlands.

### *5.4.1 A Reflection on the Use of the WET-Health and WET-EcoServices Tools*

Both the WET-Health and WET-EcoServices tools proved useful in describing the ecological condition and provision of ecosystem services of the wetlands along the Kuils River. However, some limitations of the tools need to be recognised. Firstly, this study and others (Cowden et al., 2016; Kotze, Tererai and Grundling, 2019) found that the WET-EcoServices tool does not incorporate the extent of the wetlands to quantify ecosystem service provision more accurately and to enable an ecosystem service supply comparison between wetlands. For example, the areal extent of 4AU3 and 4AU6 differed by more than 28 hectares, yet ecosystem service provision scores were similar. It is unlikely that both these assessment units can provide the same level or quantity of ecosystem services. Quantifying the provision of services through the incorporation of the size of the wetland can prove useful in determining which wetlands should be prioritised for rehabilitation. Furthermore, because restorative action has not been implemented in the wetlands along the Kuils, the future scores had to be estimated based on proposed rehabilitation action, which means that the future scores could be over or underestimated. It is therefore recommended that future studies should consider reassessing wetland conditions and ecosystem service provision after rehabilitation measures are implemented as part of an adaptive management process and to generate best-practice evidence.

Secondly, temperature amelioration is an ecosystem service that is not considered by the WET-EcoServices tool, which is generally provided by urban green spaces. Modelling of the near-surface air temperature in Beijing City, China, showed that urban wetlands reduce the atmospheric temperature of their surroundings, also known as the urban cooling effect (Sun and Chen, 2012; Hou et al., 2013). Therefore, considering the current climate change trends and cities' contribution thereto (Satterthwaite, 2008), the benefit of urban waterbodies in climate regulation should be acknowledged as an important ecosystem service, which is currently not accounted for in the WET-EcoServices tool. Thus, when rehabilitating urban systems, the actions that enhance the cooling effect of waterbodies should be considered, such as creating large vegetated buffer areas (Sun and Chen, 2012).

In conclusion, even though the wetlands along the Kuils River are in a deteriorating state, they provide valuable ecosystem services to society. However, if these wetlands continue to degrade, ecosystem service delivery could be jeopordised. Wetland rehabilitation is thus needed to halt and reverse the degradation of the wetlands, while also ensuring that wetlands continue to play a vital role in achieving the Sustainable Development Goals toward a better future for society.

# Chapter 6
# Conclusion and Recommendation

Professor Johan Rockström et al. (2009) published an article describing nine planetary boundaries that provide the basis for a “safe operating space for humanity”, contributing to the motivation for the Millennium Ecosystem Goals and Sustainable Development Goals (Millennium Ecosystem Assessment, 2002; Sachs, 2012) and other international policies, plans and treaties which are based on human well-being, economics, governance and environmental protection. Using the urban Kuils River as a case study, an interdisciplinary research approach was employed to develop a feasible governance framework for the restoration of the Kuils River by considering the ecological, governance and social context of the Kuils. This thesis (the ecological aspect of the research) focuses on land-use and water quality changes over time, and an assessment of ecosystem service provision, as well as drawing on the literature, to provide recommendations for restorative actions of the Kuils River.

In response to widespread degradation and the crossing of planetary boundaries, many global assessments have been conducted to assess the current trends of natural systems in urban settings. These include the 2005 Millennium Ecosystem Assessments, the 2013 Economics of Ecosystems and Biodiversity Assessments (Russi et al., 2013), the 2014 Convention of Biological Diversity Assessments, the 2018 Land Degradation and Restoration Assessment (Butchart et al., 2019) and the 2018 Ramsar Wetlands Assessment (Ramsar, 2018), all highlighting the interconnectivity between water and human well-being. Water forms the basis of at least five of the seventeen SDGs. These include goal three: good health and well-being; goal six: clean water and sanitation; goal eleven: sustainable cities and communities; goal thirteen: climate action; and goal fifteen: life on land (United Nations, 2017). Many other SGDs also incorporate water to some extent (e.g. Cole et al., 2018; Ramsar, 2018), indicative of how much waterbodies play an important role in human societies (Ho and Goethals, 2019). In Africa, waterbodies account for approximately 20% of the annual gross domestic product (Opperman et al., 2018). Yet, in many cases, waterbodies are continuously undervalued (Opperman et al., 2018); this is particularly true for urban rivers.

Not only do the findings of this study contribute to the body of knowledge of the impacts on urban water systems (e.g. Dawson, 2003; Millennium Ecosystem Assessment, 2005d; Richardson et al., 2007; Cilliers et al., 2013; Grundling, van den Berg and Price, 2013; Larondelle and Haase, 2013; Mander and Van Niekerk, 2013; Rebelo, 2019), the outcomes of this research also show how human activities and urbanisation have resulted in the long-term degradation of the Kuils River, an urban river in Cape Town, South Africa (refer to

Chapters Three, Four and Five). Yet, studies on Cape Town's natural assets show the importance of natural open space, including wetlands, to the city's economy (De Wit, Van Zyl, et al., 2009; TEEB, 2011; de Wit et al., 2012) and, therefore, the need for its protection and management.

Even though the results of this study call for the urgent improvement in the condition of the Kuils River, supporting the findings of research previously conducted on the Kuils River (e.g. Ninham Shand, 1986; King, Scheepers, Fisher, Reinecke and Smith, 2003; Feng, 2005; RHP, 2006; Ayuk, 2008; Nel, Parker and Silbernagl, 2013; Greeff, 2014; Mathenjwa, 2017), it also indicates that traditional restorative action such as those described in USEPA (2000) is not achievable in the case of the Kuils River, due to the complexity and extensive change that the Kuils River has experienced. For example, while the land-use change assessments revealed that urbanisation is the major contributor to the change in structure and function of the river, the wetland and water quality assessments also revealed that land-use has resulted in the degradation of the Kuils River and a shift to a potentially different functioning system. Restorative action in degraded urban environments should thus steer away from the preconceived idea of restoring 'what once was' (Luger, 1998; King et al., 2003; Uys, 2003; Hobbs, Higgs and Harris, 2009; Ravit, Gallagher, Doolittle, Shaw, Muñiz, Alomar, Hoefer, Berg and Doss, 2017), to a more appropriate 'what could be'.

From the land-use (Chapter Three), water quality (Chapter Four) and wetland assessments (Chapter Five), it is thus clear that the Kuils River is highly impacted by urbanisation. Anthropogenic impacts are regularly putting more strain on the river and its wetlands to provide the necessary ecosystem services. Some of the major stressors to the Kuils River and its wetlands include increased water input from stormwater runoff (from impervious surfaces such as roads and buildings) and wastewater discharge (Chapter Three). This has knock-on effects, for example, more disturbances to the wetland systems due to the increase in the occurrence of flash floods and changed hydrological patterns, resulting in changes to native species habitats (Pataki, Carreiro, Cherrier, Grulke, Jennings, Pincetl, Pouyat, Whitlow and Zipperer, 2011; Moccia and Palestino, 2013; Day et al., 2020). This can result in a greater opportunity for alien vegetation to establish (Richardson et al., 2007; Miller and Bestelmeyer, 2016). I found evidence of this in some of the wetlands in the upper reaches of the Kuils River where canalisation in the headwaters has resulted in increased velocity, which in turn resulted in erosion, a destabilised river bank and many non-native weeds (see Figure 5.3). Urban runoff and wastewater discharge into the river have also resulted in a decrease in water quality since the 1970s, with the discharge from the Bottelary River and from Bellville and Zandvliet WWTFs being the main drivers of the poor water quality (Chapter Four). It is thus safe to assume that the wetlands along the Kuils River are most likely novel in nature.

While restoration, in the sense of restoring 'what once was' is not possible in this urban context, restoring remnant patches of a biologically diverse urban nature with the inclusion of elements of Swartland Shale Renosterveld and Cape Flats Dune Strandveld is particularly important in biodiversity hotspots such as Cape Town because of the high risk of experiencing extinction debt (Rebelo et al., 2011; Standish, Hobbs and Miller, 2013). It has been suggested that areas along the Kuils River must be set out for conservation or management to ensure the provisioning of some ecosystem services and to conserve the remaining remnant patches of endemic biodiversity (Shand et al., 1994; Ninham Shand, 1999a; Snaddon and Day, 2009; Greeff, 2014; Mucina, Rutherford and Powrie, 2018; Myeza and Mdunyelwa, 2020). While attempts have been made to conserve some areas of nature along the Kuils River, few are formally protected or actively managed. In many cases, reports on biodiversity in Cape Town and the Western Cape in general, are focussed around national parks and nature reserves (e.g. WCBSP, 2017; City of Cape Town, 2018b, 2018a; Department of Environmental Affairs & Development Planning, 2019), yet many public open space areas such as wetlands and urban parks are not actively acknowledged, even though many of these corridors along the Kuils River are considered ecologically important or sensitive (City of Cape Town, 2012a-d, 2018b).

This research points to the need for improved management and rehabilitation of the wetlands along the Kuils River. These wetlands continue to degrade (Chapter Five) but "theoretical rehabilitation" indicates the potential for improvement to both wetland condition and wetland ecosystem service provision even though overall health scores did not necessarily improve (Chapter Five). Economic valuations of the City of Cape Town's natural assets highlight natural hazard reduction, water purification, waste treatment, recreation, aesthetics and a sense of place as key services provided by wetlands (De Wit et al., 2009; de Wit and van Zyl, 2011). Chapter Five highlights that the most important ecosystem services that the Kuils River and its wetlands provide are an assimilation of phosphates, nitrates and toxicants and sediment trapping services. If these urban green spaces and their wetlands are made more attractive for recreational and other low-impact uses (e.g. bird watching) there is the potential to provide opportunities for further provision of ecosystem services, including cultural services, which are currently not supplied. The majority of the wetlands assessed in this study were situated in informally recognised urban parks, yet these parks are not used for what they are intended and end up providing disservices (e.g. crime in the Khayelitsha Wetlands Park area; Mathenjwa, 2017). This may be due to lack of access (e.g. parking space to utilise the parks), lack of amenities in these parks, or lack of security (e.g. dense alien vegetation providing cover for crime). The Caldes River in Barcelona can be used as an example highlighting the success of the human use of urban green spaces to improve habitat quality and management (Vall-Casas et al., 2019). An increase in urban green space development and use over time has facilitated access to the Caldes River, and thereby, contributed to the rehabilitation of the river (Vall-Casas et al., 2019).

The present state of the Kuils River suggests that the current management strategy employed by the City of Cape Town is not sufficient to address the problems that the Kuils River is currently facing. If urgent action is not adopted soon, the wetlands along the Kuils River will continue to degrade. It is however not too late – there is an opportunity to do things differently. The Kuils River and its wetlands provide many ecosystem services, that are currently highly valued. These include its wastewater management services, flood management services and nutrient assimilation services. Incorporating these ecosystem services into urban planning will not only ensure that the necessary ecosystem services are supplied but also that the health of the Kuils River is improved. Additionally, when planning how the Kuils River will be managed, the interrelated ecological, social and governance dimensions should be considered to fully understand the restorative potential of the Kuils River.

One wetland in particular – the Kuils River Corridor – could serve as a pilot rehabilitation project (Chapter Five). The Kuils River Corridor is an urban greenbelt in the Stellenberg/ Vredekloof area, which is approximately 90 hectares in extent. With such a large area formally recognised as an urban park, it is surprising that it is not fully utilised. Not only does the Corridor contain at least four of the wetlands assessed in this study, but there is ample space to improve the park and make it more accessible. The goals for rehabilitation of the Kuils River Corridor could thus be to enhance ecosystem services and improve the recreational use of the Corridor, whilst also maintaining ecological integrity through continued adaptive management.

Improvement of habitat condition and ecosystem service provision of the Kuils River Corridor can be achieved in various ways – 1) by creating picnic spots, walk-ways, jungle gyms, exercise equipment and enhancing educational experiences (increasing education, recreational and cultural services; refer to Figure 5.3 - 2AU5 to 3AU2 for the current ecosystem services supplied), 2) by removing alien vegetation and reintroducing indigenous vegetation, 3) by improving the sinuosity and the flow of the water in the system, and 4) by formally recognising the park as a protected or managed area. The more the park is used by the community, the more likely they are to advocate for the improvement or enhancement of the Corridor and other systems. Similar measures have been implemented worldwide (Renn et al., 2008; Svendsen, 2013; Schanze et al., 2004). Moreover, the Corridor would be better equipped for improvements in the control of potential floods, the maintenance of biodiversity goals and the reduction in natural hazard threats to communities (Tomsett and Leyland, 2019). Similar suggestions such as these have been recommended for other parts of the Kuils River (e.g. see Ninham Shand, 1994; Shand et al., 1999), but these have yet to be implemented.

Furthermore, restorative action should pay attention to the drivers that result in the degradation of the system (Russel, 2009; Halajova, Halaj, Macura and Škrinár, 2019). In the case of the Kuils River, this centres around urbanisation resulting in increased stormwater and effluent discharge into the Kuils River and, therefore, poor water quality (Chapter 4). Considering the three ecosystem assessment approaches used in this study, the following section discusses the limitations of the research. Thereafter, the potential restorative measures that can be taken to lessen the anthropogenic impacts on the Kuils River are discussed.

## 6.1 Limitations and Future Research

The results of this study showed the linkages between urban waterways and people. From a land-use mapping perspective, the lack of high-resolution images in the older aerial imagery made it difficult to detect differences between patches of native vegetation and alien vegetation. Regardless, the general trend still showed a change in vegetation structure from a mostly native species composition to one containing more alien vegetation, as is the case in many other riparian systems worldwide. Additionally, from an ecosystem service perspective, the WET-EcoServices tool lacked the incorporation of wetland size which could affect the prioritisation of wetlands for rehabilitation, as the ecosystem service provision estimates are not as accurate. It is therefore suggested that the WET-EcoServices tool incorporates wetland size in the algorithm to more accurately describe the ecosystem services supplied by the wetlands. Alternatively, future studies could conduct an economic valuation of the ecosystem services supplied by wetlands along the Kuils River and compare the results to that of the WET-EcoServices tool to assess the accuracy of the tool. Future studies should also consider reassessing wetland conditions and ecosystem service provision after rehabilitation measures are implemented as part of an adaptive management process and to generate best-practice evidence.

This study thus highlighted the importance of managing and protecting urban rivers at a local scale to contribute to achieving some of the SDGs at a national level, which in turn will contribute to the achievement of the goals of planetary boundaries. Rehabilitating sections of the Kuils River has the potential to not only affect the provision of water and ecosystem services in an already water-scarce country but can also contribute to much-needed job creation, improved health and safety, biodiversity and goals of sustainable cities. Further research could include assessing the most appropriate restorative action given the social and governmental setting of the Cape and running pilot projects to assess the suggested restoration activities. Findings should thus be combined with the remainder of the interdisciplinary study (which includes the governance and social context of thr Kuils River) to develop a feasible governance framework for the restoration of the Kuils River.

## 6.2 Driver-Specific Recommendations

When planning management and restorative action along the Kuils River, urban runoff, water quality improvement and alien vegetation removal require particular attention. It should be noted that the recommendations listed below not only reflect on the findings of this study, but also include actions that can be incorporated to improve the conditions of the functioning of the Kuils River and its wetlands.

### *6.2.1 Urban Runoff and Water Quality Improvement*

There are various ways in which water quality impacts can be addressed. Firstly, floodplain-like areas and wetlands can be created to trap nutrients in sediment, by reintroducing native riparian vegetation on the river banks to filter the water systems, and by ensuring that waterways are kept clear of litter (Luger, 1998; Russel, 2009). Secondly, stormwater flows can be constructed to mimic the natural hydrology of the system, for example, by creating regenerative stormwater conveyance systems using riffles and cobbles (Ravit et al., 2017). This also has the potential to increase the already low dissolved oxygen levels of the Kuils River system. Lastly, stormwater and residential runoff can be diverted directly into gardens to relieve the river from excessive water input (Luger, 1998), or neighbouring communities can be incentivised (and financially assisted, if required) to install rainwater tanks. Such initiatives would not only address water quality impacts and improve the seasonality of the river, but it can also assist the City with managing water supply and demand (Liang, Matteo, Maier and Thyer, 2019). These suggestions however only address surface water and stormwater runoff and might not be sufficient in addressing the impact of wastewater discharge. Such restorative measures are likely to be more successful in the upper reaches of the Kuils River, above the Bellville WWTF discharge point. Such actions could still provide downstream benefits such as on water quality and the impact of floods in the region of Khayelitsha.

### *6.2.2 Alien Vegetation Management*

Regarding alien vegetation, some of the abundant invasive tree species found in the Kuils River are Acacia spp., Prosopis spp. and poplars. Other plants that have become problematic are *Typha capensis* (bulrush) and *Pennisetum cladestinum* (Kikuyu grass). Managing alien vegetation in urban areas can be a very precarious task due to conflicting opinions in the management thereof (e.g. see Richardson and van Wilgen, 2004; van Wilgen and Richardson, 2012; Potgieter et al., 2017, 2019, 2020; Potgieter, 2019; Qukula, 2021), often requiring community and other stakeholder involvement. As a result, authors mention the need for careful consideration of the restorative goals regarding alien invasive species clearing (e.g. see Richardson et al., 2007). Despite the presence of alien trees, most of the alien vegetation in the Kuils River is comprised of non-native weeds and grasses. Urban ecologists have suggested that such species should be incorporated into restorative action management plans (Hobbs et al., 2006; Richardson et al., 2007; Francis, 2014). For example, active rehabilitation by planting and sowing seeds of native trees and shrubs could help tip the competitive balance away from grass and herbaceous weed dominance (Prins, Holmes and Richardson, 2004; Pretorius et al., 2008), but such actions can be costly (Holmes et al., 2020). If restoration goals include alien vegetation control, the priority should be to remove vegetation that causes the most damage or is more likely to lead to further ecological threshold shifts (Richardson et al., 2007). The upper and middle reaches of the Kuils River should thus be prioritised to limit further spread of alien trees in the lower reaches. The fell and remove treatment would be most suited for clearing acacias (the most common invader) in the Kuils River area with continuous seedling follow-up treatment (Holmes, Esler, Richardson and Witkowsk, 2008; Rebelo et al., 2011), to allow native trees and shrubs to recolonise, with the active reintroduction of native species were necessary (Martens et al., 2021).

Shand et al. (1994) recommended the complete removal and replacement of *Typha capensis* from wetlands along the Kuils. However, this should be carefully considered because of the conflict between the provision of ecosystem services (biodiversity, nutrient trapping and job creation services) and the health impacts (Voigt, 2007; Mathenjwa, 2017; Potgieter, 2019). For example, *T. capensis* has various social and ecological benefits (including job creation and sediment and nutrient trapping services), but its windblown seeds are known to be a health hazard to communities living near it. There is thus conflict regarding its management. If and where native plants are reintroduced, these plants should preferably be resilient to alien plant re-invasion and flooding and should contribute to the achievement of the restoration goals (Holmes et al., 2008; Ravit et al., 2017). Holmes (2007) lists possible common native species that can be planted to restore ecosystem function in fynbos areas. These include *Juncus* species, native *Pennisetum* and restios, among others. Target sites can then for example be landscaped into ‘urban parks’ that house a variety of native vegetation. Biennial active monitoring of the cleared sites should be incorporated into the management plans to prevent further regrowth of cleared alien vegetation (Ractliffe and Day, 2001; Holmes, 2007).

These recommendations could have positive implications for water quality management and indirect effects on the structure and functioning of the wetlands along the Kuils River, and the river as a whole. Many restorative projects can be used as guidelines for the management of the Kuils River. Potential lessons include the need for very clear goals and objectives when planning restorative activities, the need to consider the dynamics of the river reaches, the recognition that restoration takes time and ecological, economic and social aspects should be taken into account (Crimmens and Larson, 2006; Vall-Casas et al., 2019).

## 6.3 General Considerations in Urban River Management

Research suggests that there are some aspects that should be considered when conducting urban river management. Firstly, Hobbs et al. (2006) and Francis (2012) suggests that the novelty of urban systems should be embraced by incorporating the urbanised nature and occurrence of non-threatening non-native species into management plans rather than opting for full restoration. Secondly, an appropriate reference site should be chosen to guide management and restorative decisions. In the case of the Kuils River, the Liesbeek River in Cape Town provides a good example. With a similar history to the Kuils River, extensive research (including different management and restoration projects) has already been conducted on the Liesbeek (e.g. Communitree, no date; Carden and Winter, 2015; Source to Sea, 2016; Edwards, 2018; Muntjewerff et al., 2019; Kotzé, 2020), which can be applied to the Kuils River. Lastly, developing a Sustainable Kuils River Management Plan, as suggested by Henning et al. (2007), could help to solve the problem of addressing the root causes of biological impacts on urban rivers (Groffman et al., 2006; Beechie et al., 2009; Russel, 2009). Once a thorough Sustainable Urban Kuils River Management Plan has been created and implemented, only then can plans be broadened to include the greater catchment and its tributaries through the development of integrated river management plans, catchment management plans and associated goals for restorative measures.

Furthermore, while beyond the scope of the current study, the literature also suggests the inclusion of the society in restorative action. As governmental funding and resources for restoration activities in developing countries like South Africa are limited, there is a strong need to involve communities in restoration activities along the Kuils River (Wantzen et al., 2019). Without community support and involvement in ecological rehabilitation, restorative action might not succeed (Gross and Hoffmann-Riem, 2005; Richardson et al., 2007). Involving multiple stakeholders can thus result in a more holistic objective for rehabilitation along the Kuils River (Tunstall, Penning-Rowsell, Tapsell and Eden, 2000; Hall, 2005; Wohl, Lane and Wilcox, 2015; Metzger et al., 2017; Ravit et al., 2017; Gann et al., 2019), while also ensuring the success of the rehabilitation measures implemented and constibuting to the improvement of the economy through job creation. Public involvement in urban rehabilitation projects can also assist by identifying sites that might need restorative action. This can include the use of citizen science, through the incorporation of social media and community programs (e.g. Dumakude and Graham, 2017; Khumalo, 2019). Available photographic technology, such as iNaturalist, could be used to record species occurrence in open spaces over consecutive years to assess changes in the park and implement management measures. Making the rehabilitated sites available for human recreational use could then contribute to the need for continuous monitoring and the upkeep of these sites and will also address some of the social problems that the communities along the Kuils River are facing (see Mathenjwa, 2017; Myeza and Mdunyelwa, 2020).

Therefore, for restoration to be successful in urban settings, clear goals and objectives are needed that are based on ecosystem services, potential human use and people's needs (Chan, Satterfield and Goldstein, 2012; Shackleton and Blair, 2013; Greeff, 2014; Chou, 2016; Pascual et al., 2017; Douglas, 2018; Shackleton et al., 2018). In the case of the Kuils River, while the demands for ecosystem services in communities in the upper reaches of the Kuils River are focussed around supporting and regulating services, in the lower reaches where municipal service delivery is limited and floods occur regularly, demands for ecosystem services include provisioning and cultural services (Greeff, 2014). The management and restoration options for wetlands in the different reaches would thus be dependent on the community needs.

This thesis assessed the condition of the Kuils River and the drivers causing the change in its condition, over the last fifty years, to inform prospects for future restorative action. While each assessment technique (land-use change, water quality and ecosystem services) provided insight into the state of the Kuils River, they also provided a more conceptual understanding of the drivers that have resulted in the degradation of urban rivers in general, while simultaneously provided a justification for the rehabilitation (rather than restoration) of the Kuils River, and urban rivers globally. Not rehabilitating urban rivers can have serious economic, social and health implications and can impede sustainability efforts, both locally and at an international scale. Restorative efforts should therefore incorporate the drivers that result in the degradation of urban water systems to promote successful urban river rehabilitation.

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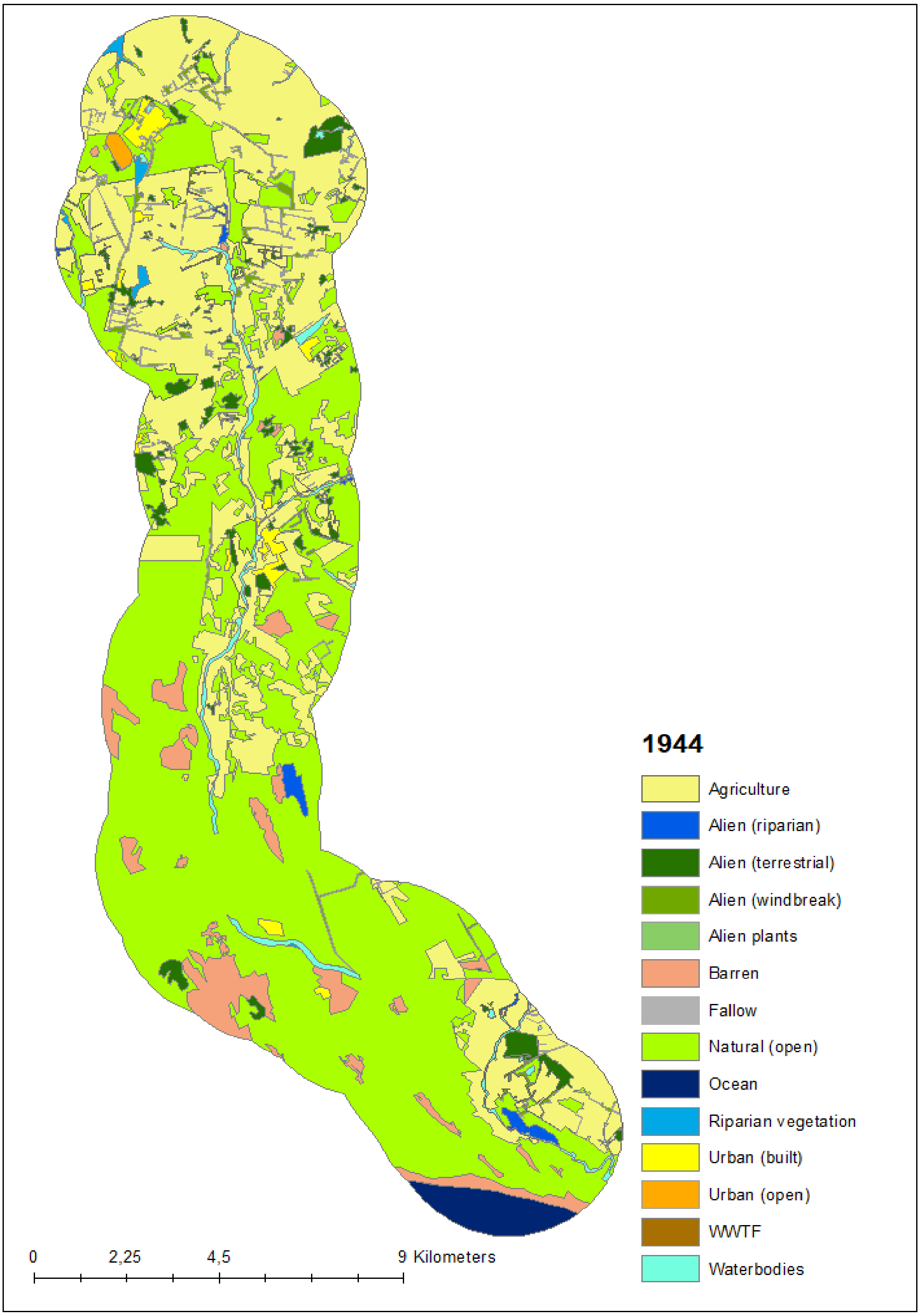
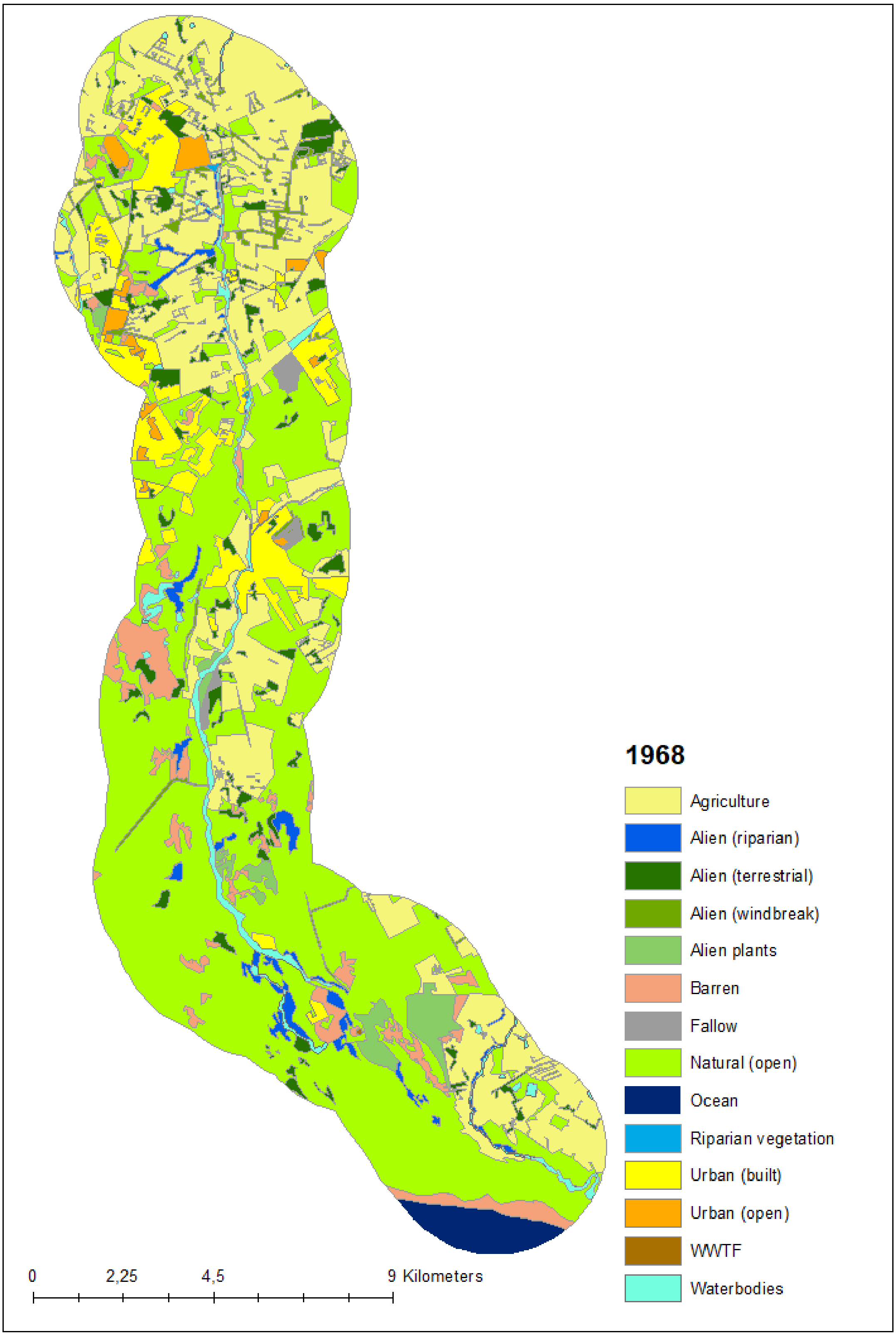
# Addenda

## Addendum A: LULC Maps

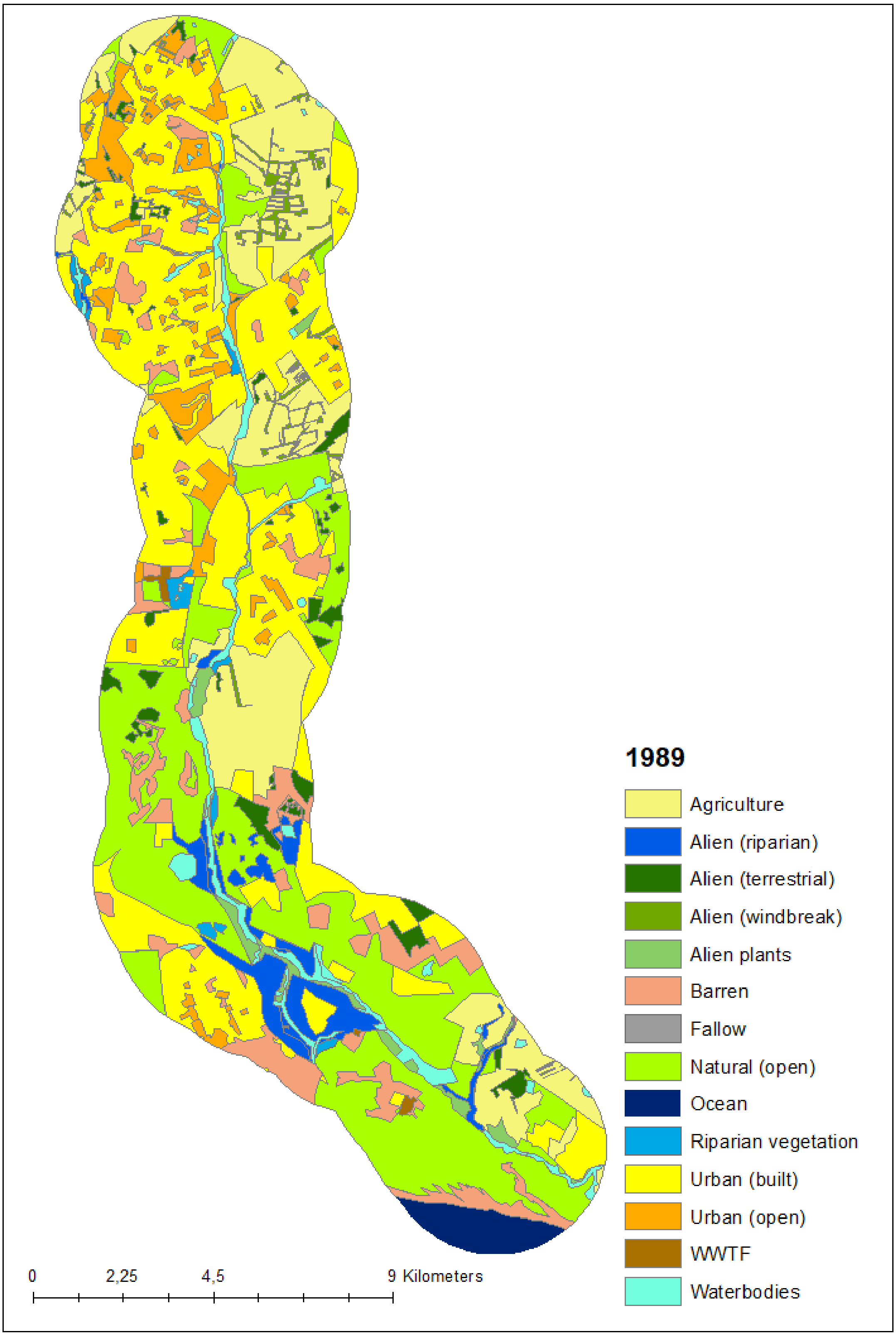
### *1944*



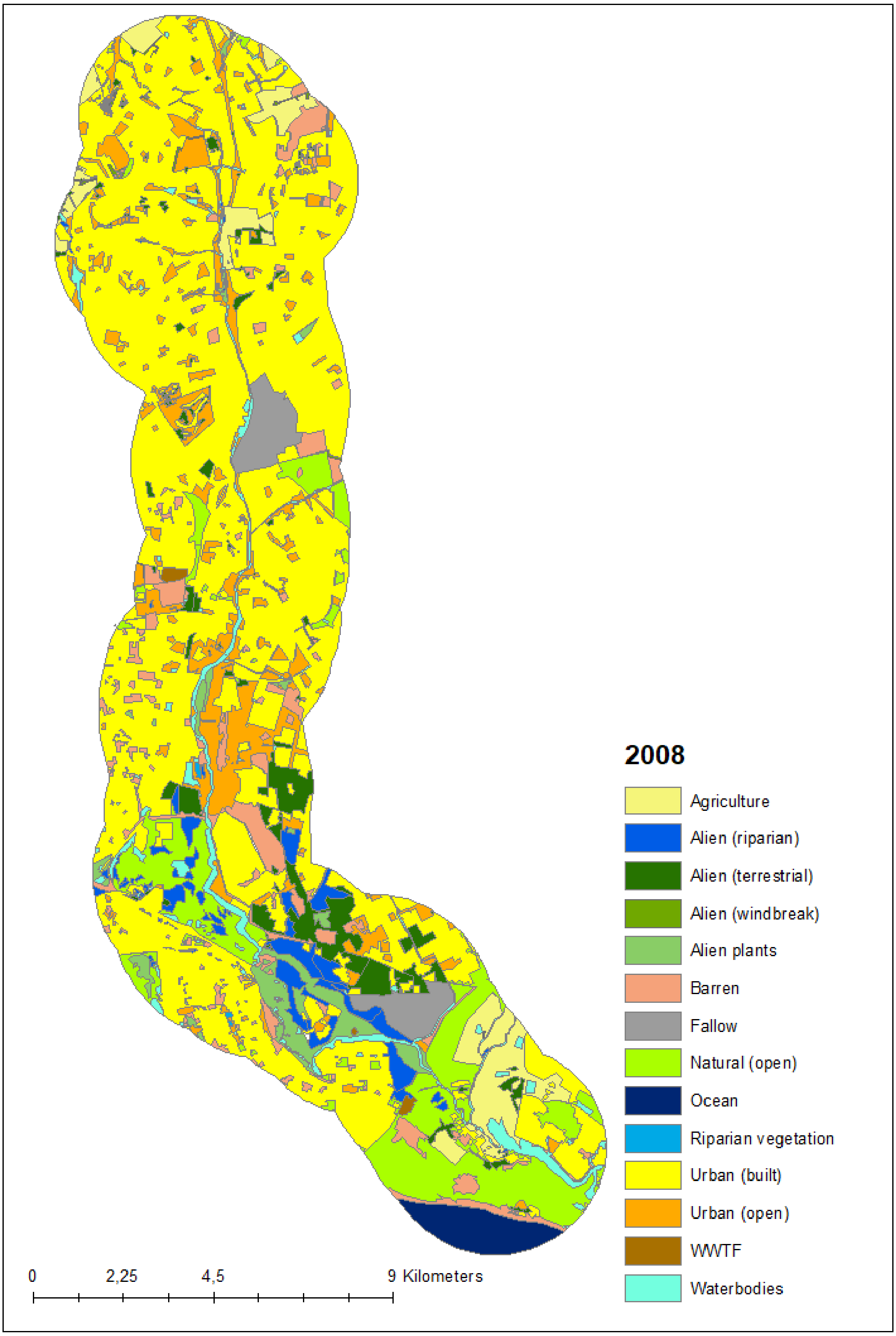
***1968***


1968
Agriculture
Alien (riparian)
Alien (terrestrial)
Alien (windbreak)
Alien plants
Barren
Fallow
Natural (open)
Ocean
Riparian vegetation
Urban (built)
Urban (open)
WWTF
Waterbodies
0
2,25
4,5
9 Kilometers

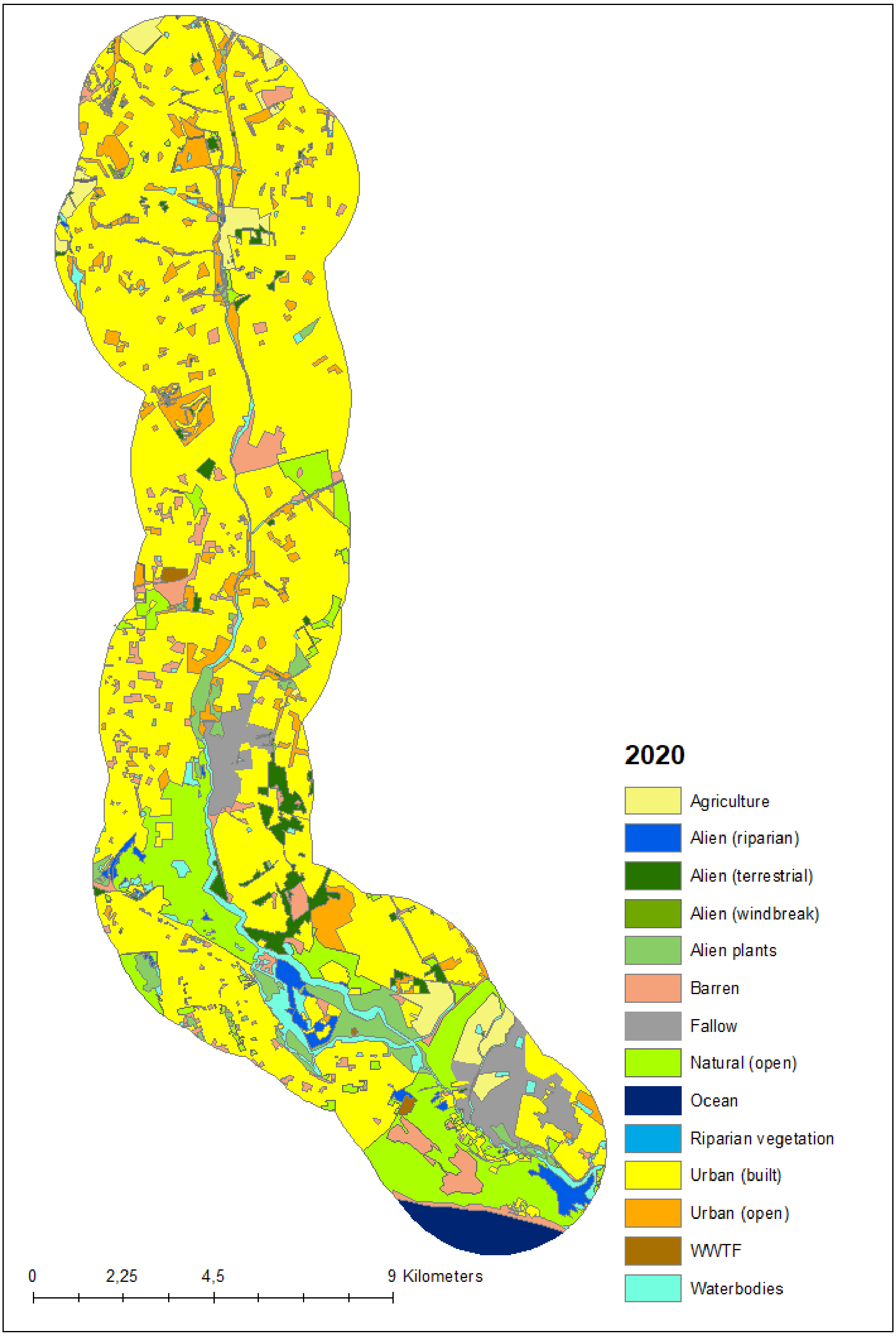
***1989***


1989
Agriculture
Alien (riparian)
Alien (terrestrial)
Alien (windbreak)
Alien plants
Barren
Fallow
Natural (open)
Ocean
Riparian vegetation
Urban (built)
Urban (open)
WWTF
Waterbodies
0
2,25
4,5
9 Kilometers

***2008***


2008
Agriculture
Alien (riparian)
Alien (terrestrial)
Alien (windbreak)
Alien plants
Barren
Fallow
Natural (open)
Ocean
Riparian vegetation
Urban (built)
Urban (open)
WWTF
Waterbodies
0
2,25
4,5
9 Kilometers

***2020***


2020
Agriculture
Alien (riparian)
Alien (terrestrial)
Alien (windbreak)
Alien plants
Barren
Fallow
Natural (open)
Ocean
Riparian vegetation
Urban (built)
Urban (open)
WWTF
Waterbodies
0
2,25
4,5
9 Kilometers

# Addendum B: Water Quality Data

## *Addendum B1: Water Quality Data Sources*

***Table S4.1*** List of sources used in water quality assessments of the Kuils River.

| State, Tertiary and Private Institutions | Source |
|---|---|
| Department of Water and Sanitation (National) | |
| City of Cape Town | |
| Ninham Shand | |
| University of Western Cape | Feng, X.Y. 2005. A Survey of Inorganic Chemical Pollution in the Bottelary River, Cape Town. MSc Thesis, University of the Western Cape. Bellville, South Africa.<br>Mwangi, F.N. 2014. Land Use Practices and Their Impact on the Water Quality of the Upper Kuils River (Western Cape Province, South Africa). MSc Thesis, University of the Western Cape. Bellville, South Africa.<br>Ngwenya, F. 2006. Water Quality Trends in the Eerste River, Western Cape, 1990-2005. MSc Thesis, University of the Western Cape. Bellville, South Africa. |
| University of Cape Town | Govender, K. 2004. The Effect of Development on Seasonal Wetlands on the Cape Flats, Western Cape, South Arica. MSc Thesis, University of Cape Town. Cape Town, South Africa. |
| Stellenbosch University | Fourie, S. 2005. An Assessment of Water Quality and Endocrine Disruption Activities in the Eerste / Kuils River Catchment System, Western Cape, South Africa. MSc Thesis, Stellenbosch University. Stellenbosch, South Africa. |

## *Addendum B2: Results of Analysis for the Additional Variables Assessed*

***Table S4.2*** Summary statistics of the temporal changes in water quality of the Kuils River with statistical analyses (Mann-Kendall tests) of trends. S.D. – standard deviation, Temp – temperature, DO – dissolved oxygen, POS – percentage oxygen demand, $NH_4$ – ammonium ion, TP – total phosphorous, TDS – total dissolved solids, TSS – total suspended solids, F. coli – faecal coliforms.

| Variable | Unit | Summary Statistics of Variables | | | Kendall's tau | p value[1] | National Water Quality Guidelines (NWQG) | | Health Risk Rating Ranges[2] |
|---|---|---|---|---|---|---|---|---|---|
| | | Min | Max | Mean (S.D.) | | | Domestic Use | Aquatic Life | |
| Temp | °C | 11.6 | 21.6 | 18.1 (1.9) | 0.1 | 0.365 | - | * | - |
| DO | mg/l O | 3.47 | 11.5 | 6.7 (2.1) | -0.55 | <0.01 | - | - | < 4 = Unacceptable |
| POS | % O | 38.93 | 90.2 | 59.9 (13.5) | -0.267 | 0.06 | - | 80-120% | - |
| $NH_4$ | mg/l N | 0.033 | 33 | 5.7 (6.7) | 0.34 | <0.01 | <1 | <0.007 | >0.1 = Unacceptable |
| TP | mg/l P | 1.21 | 3.8 | 2.2 (0.7) | 0.37 | <0.01 | - | <0.005 | >0.125 = Unacceptable |
| TSS | mg/l | 10.75 | 1528 | 75.0 (253.6) | 0.13 | 0.21 | - | <100 | - |
| TDS | mg/l | 496 | 591972.9 | 96291.2 (191771.3) | 0.47 | <0.01 | 450 | * | - |
| F. coli | cfu/ 100 ml | 2420 | 195209.2 | 39398 (64819.1) | -0.391 | <0.01 | <0 | - | >4000 = Unacceptable |

[1] –P-values in *italics* show significant trends, [2] – Health risk rating ranges of Day et al (2020), * According to the National Water Quality Guidelines, the temperature should not vary by more than 10% or 2°C from the background levels. TDS levels should also not vary by more than 15% from the normal cycles of the river at any time.

***Table S4.3*** ANOVA results of the comparison of average water quality before (KR04 and KR08), at (KRB and KRZ) and after (KR06 and KR10) wastewater discharge at the Bellville and Zandvliet Wastewater Treatment Facilities.

| | Bellville WWTF | | | | | Zandvliet WWTF | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| Variables | KR04 | KRB | KR06 | F statistic[1] | p-value | KR08 | KRZ | KR10 | F statistic[1] | p-value |
| Temperature (°C) | 18.00 | - | 20.08 | 17.72[a] | <0.001 | 18.21 | - | 18.13 | 0.03[a] | 0.87 |
| Dissolved Oxygen (mg/l O) | 9.17 | - | 6.42 | 123.31[b] | <0.001 | 6.28 | - | 4.81 | 43.27[b] | <0.001 |
| Percentage Oxygen Saturation (%) | 108.72 | - | 67.94 | 127.67[b] | <0.001 | 70.93 | - | 45.89 | 45.20[a] | <0.001 |
| Ammonium ion ($NH_4$) (mg/l N) | 0.48 | 9.57 | 5.82 | 93.48[b] | <0.001 | 1.82 | 11.95 | 6.69 | 37.02[b] | <0.001 |
| Total phosphorous (mg/l P) | 0.86 | - | 2.90 | 148.52[b] | <0.001 | 1.89 | - | 2.40 | 21.04[b] | <0.001 |
| Total dissolved solids | Insufficient data | | | | | | | | | |
| Total suspended solids | 32.74 | 113.53 | 53.02 | 8.41[b] | <0.001 | 18.02 | 29.44 | 18.65 | 2.20[b] | 0.11 |
| Faecal coliforms (cfu/ 100ml) | Insufficient data | | | | | | | | | |

[1] – F-values annotated with an (a) were calculated using LS means (current effect) and (b) weighted means; p values in *italics* show significant differences.

## *Addendum B3: Analysis of the Differences between Sections*

***Table S4.4*** Multiple t-tests and significance values for the differences in slopes between sites for each of the five water quality variables included in the body of the chapter. Asterisks (*) indicate significant p-values of <0.05.

| | | pH | | Electrical Conductivity (mS/m) | | Chemical Oxygen Demand (mg/l O) | | Ammonia NH3 (mg/l N) | | Orthophosphates (mg/l P) | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| | | T value | p-value | T value | p-value | T value | p-value | T value | p-value | T value | p-value |
| KR01 vs | KR02 | 2,17 | 0,03* | 0,46 | 0,64 | -1,53 | 0,13 | 0,99 | 0,32 | 0,99 | 0,32 |
| | KR03 | 0,84 | 0,40 | -0,32 | 0,75 | -1,00 | 0,32 | -1,54 | 0,12 | -0,34 | 0,73 |
| | KR04 | -0,61 | 0,54 | -0,12 | 0,90 | -0,49 | 0,63 | -2,61 | 0,01* | -0,47 | 0,64 |
| | KR06 | 0,12 | 0,90 | -0,20 | 0,84 | -0,64 | 0,52 | -4,11 | 0,00* | -0,17 | 0,86 |
| | KR07 | -0,76 | 0,45 | -0,18 | 0,85 | -0,99 | 0,32 | -4,47 | 0,00* | 0,22 | 0,83 |
| | KR08 | 1,95 | 0,05* | 0,03 | 0,97 | -0,29 | 0,77 | -3,63 | 0,00* | 1,31 | 0,19 |
| | KR09 | 3,39 | 0,00* | -0,16 | 0,88 | -0,25 | 0,80 | -3,25 | 0,00* | -1,14 | 0,26 |
| | KR10 | 3,45 | 0,00* | -0,12 | 0,91 | -1,49 | 0,14 | -5,44 | 0,00* | -0,91 | 0,36 |
| | KR11 | 3,62 | 0,00* | -0,36 | 0,72 | -2,01 | 0,05* | -3,33 | 0,00* | -1,18 | 0,24 |
| | KR12 | 2,17 | 0,03* | -5,10 | 0,00* | -1,52 | 0,13 | -4,00 | 0,00* | -0,61 | 0,54 |
| KR02 vs | KR03 | -0,70 | 0,48 | -0,61 | 0,54 | -0,01 | 0,99 | -2,09 | 0,04* | -1,01 | 0,31 |
| | KR04 | -2,69 | 0,01* | -0,58 | 0,56 | 1,20 | 0,23 | -2,99 | 0,00* | -1,27 | 0,20 |
| | KR06 | -2,31 | 0,02* | -0,67 | 0,50 | 1,17 | 0,24 | -4,01 | 0,00* | -1,18 | 0,24 |
| | KR07 | -2,96 | 0,00* | -0,66 | 0,51 | 0,74 | 0,46 | -4,26 | 0,00* | -0,93 | 0,35 |
| | KR08 | 0,47 | 0,64 | -0,26 | 0,80 | 0,76 | 0,45 | -3,95 | 0,00* | 0,49 | 0,62 |
| | KR09 | 0,40 | 0,69 | -0,61 | 0,54 | 1,42 | 0,16 | -3,47 | 0,00* | -1,79 | 0,07 |
| | KR10 | 0,44 | 0,66 | -0,58 | 0,56 | 0,37 | 0,71 | -5,12 | 0,00* | -1,61 | 0,11 |
| | KR11 | 0,76 | 0,45 | -0,76 | 0,45 | -0,20 | 0,84 | -3,56 | 0,00* | -1,81 | 0,07 |
| | KR12 | 0,22 | 0,83 | -4,91 | 0,00* | -0,28 | 0,78 | -4,22 | 0,00* | -1,31 | 0,19 |

| | | pH | | Electrical Conductivity (mS/m) | | Chemical Oxygen Demand (mg/l O) | | Ammonia NH3 (mg/l N) | | Orthophosphates (mg/l O) | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| | | T value | p-value | T value | p-value | T value | p-value | T value | p-value | T value | p-value |
| KR03 vs | KR04 | -1,20 | 0,23 | 0,25 | 0,80 | 0,76 | 0,45 | -0,04 | 0,97 | 0,01 | 0,99 |
| | KR06 | -0,82 | 0,41 | 0,23 | 0,82 | 0,72 | 0,47 | -0,64 | 0,53 | 0,26 | 0,79 |
| | KR07 | -1,28 | 0,20 | 0,24 | 0,81 | 0,48 | 0,63 | -0,84 | 0,40 | 0,47 | 0,64 |
| | KR08 | 0,97 | 0,33 | 0,26 | 0,79 | 0,60 | 0,55 | -1,84 | 0,07 | 1,31 | 0,19 |
| | KR09 | 1,07 | 0,28 | 0,23 | 0,81 | 0,90 | 0,37 | -0,40 | 0,69 | -0,40 | 0,69 |
| | KR10 | 1,11 | 0,27 | 0,26 | 0,80 | 0,24 | 0,81 | -1,73 | 0,09 | -0,27 | 0,79 |
| | KR11 | 1,33 | 0,18 | 0,09 | 0,92 | -0,12 | 0,91 | -0,54 | 0,59 | -0,48 | 0,64 |
| | KR12 | 0,84 | 0,40 | -3,45 | 0,00* | -0,20 | 0,84 | -1,61 | 0,11 | -0,15 | 0,88 |
| KR04 vs | KR06 | 0,86 | 0,39 | -0,06 | 0,95 | -0,13 | 0,90 | -1,14 | 0,25 | 0,38 | 0,70 |
| | KR07 | -0,06 | 0,95 | -0,05 | 0,96 | -0,56 | 0,57 | -1,53 | 0,13 | 0,70 | 0,48 |
| | KR08 | 2,28 | 0,02* | 0,10 | 0,92 | -0,01 | 0,99 | -2,24 | 0,03* | 1,52 | 0,13 |
| | KR09 | 4,16 | 0,00* | -0,04 | 0,97 | 0,27 | 0,79 | -0,62 | 0,54 | -0,57 | 0,57 |
| | KR10 | 4,22 | 0,00* | 0,01 | 0,99 | -1,09 | 0,28 | -2,92 | 0,00 | -0,39 | 0,70 |
| | KR11 | 4,32 | 0,00* | -0,26 | 0,79 | -1,67 | 0,10 | -0,84 | 0,40 | -0,65 | 0,52 |
| | KR12 | 2,62 | 0,01* | -5,17 | 0,00* | -1,25 | 0,21 | -2,19 | 0,03* | -0,20 | 0,84 |
| KR06 vs | KR07 | -1,14 | 0,25 | 0,02 | 0,99 | -0,49 | 0,62 | -0,51 | 0,61 | 0,50 | 0,62 |
| | KR08 | 1,98 | 0,05* | 0,13 | 0,90 | 0,06 | 0,96 | -1,81 | 0,07 | 1,44 | 0,15 |
| | KR09 | 3,97 | 0,00* | 0,02 | 0,98 | 0,42 | 0,67 | 0,44 | 0,66 | -1,14 | 0,26 |
| | KR10 | 4,05 | 0,00* | 0,07 | 0,94 | -1,07 | 0,29 | -2,31 | 0,02* | -0,88 | 0,38 |
| | KR11 | 4,12 | 0,00* | -0,24 | 0,81 | -1,69 | 0,09 | 0,12 | 0,91 | -1,17 | 0,24 |
| | KR12 | 2,26 | 0,02* | -5,45 | 0,00* | -1,21 | 0,23 | -1,63 | 0,10 | -0,54 | 0,59 |

| | | pH | | Electrical Conductivity (mS/m) | | Chemical Oxygen Demand (mg/l O) | | Ammonia NH3 (mg/l N) | | Orthophosphates (mg/l O) | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| | | T value | p-value | T value | p-value | T value | p-value | T value | p-value | T value | p-value |
| KR07 vs | KR08 | 2,39 | 0,02* | 0,12 | 0,90 | 0,30 | 0,76 | -1,63 | 0,10 | 1,26 | 0,21 |
| | KR09 | 4,92 | 0,00* | 0,01 | 0,99 | 0,83 | 0,41 | 0,84 | 0,40 | -1,48 | 0,14 |
| | KR10 | 4,99 | 0,00* | 0,06 | 0,95 | -0,48 | 0,63 | -1,91 | 0,06 | -1,20 | 0,23 |
| | KR11 | 4,97 | 0,00* | -0,25 | 0,80 | -1,11 | 0,27 | 0,49 | 0,62 | -1,47 | 0,14 |
| | KR12 | 2,82 | 0,01* | -5,46 | 0,00* | -0,88 | 0,38 | -1,38 | 0,17 | -0,80 | 0,43 |
| KR08 vs | KR09 | -0,26 | 0,80 | -0,12 | 0,91 | 0,15 | 0,88 | 1,93 | 0,05* | -1,92 | 0,06 |
| | KR10 | -0,23 | 0,82 | -0,10 | 0,92 | -0,58 | 0,56 | 0,74 | 0,46 | -1,79 | 0,08 |
| | KR11 | 0,01 | 0,99 | -0,23 | 0,82 | -0,95 | 0,34 | 1,77 | 0,08 | -1,94 | 0,05 |
| | KR12 | -0,29 | 0,77 | -3,45 | 0,00* | -0,90 | 0,37 | 0,53 | 0,60 | -1,56 | 0,12 |
| KR09 vs | KR10 | 0,06 | 0,95 | 0,04 | 0,97 | -1,39 | 0,17 | -2,35 | 0,02* | 0,17 | 0,87 |
| | KR11 | 0,50 | 0,61 | -0,23 | 0,82 | -1,94 | 0,05* | -0,25 | 0,80 | -0,12 | 0,90 |
| | KR12 | -0,10 | 0,92 | -5,18 | 0,00* | -1,42 | 0,16 | -1,78 | 0,08 | 0,27 | 0,78 |
| KR10 vs | KR11 | 0,45 | 0,65 | -0,27 | 0,79 | -0,70 | 0,49 | 1,97 | 0,05* | -0,28 | 0,78 |
| | KR12 | -0,14 | 0,89 | -5,21 | 0,00* | -0,60 | 0,55 | -0,19 | 0,85 | 0,13 | 0,90 |
| KR11 vs | KR12 | -0,44 | 0,66 | -4,83 | 0,00* | -0,14 | 0,89 | -1,56 | 0,12 | 0,37 | 0,72 |

# Addendum C: Description of the WET-Health and WET-EcoServices Frameworks

## *Addendum C1: Description of the WET-Health Framework*

The abstract was taken from Macfarlane, Ollis and Kotze (2020, p. i-v).

> *"WET-Health Version 2 consists of a series of three tools developed to assess the Present Ecological State (PES) or "ecological health" of wetland ecosystems of different hydrogeomorphic types at three different levels of detail/resolution. These tools build on previous assessment methods, including WET-Health Version 1 and Wetland-IHI, in response to the need that was identified to develop a refined and more robust suite of tools for the assessment of the PES of wetland ecosystems in South Africa. The suite of tools was developed through extensive engagement with key stakeholders to clarify user requirements for different types of wetlands and levels of PES assessment, and the tools have been tested across a variety of wetland hydrogeomorphic types and land-use contexts. Thus, WET-Health Version 2 has achieved the main project aim of integrating the existing Wetland PES assessment tools into a single suite of tools which are in line with user requirements and which address the shortcomings of the previous methods.*
>
> *The conceptual basis and structure of the method*
>
> *WET-Health is designed to assess the PES of a wetland by scoring the perceived deviation from a theoretical reference condition, where the reference condition is defined as the un-impacted condition in which ecosystems show little or no influence of human actions. In thinking about wetland health or PES, it is thus appropriate to consider 'deviation' from the natural or reference condition, with the ecological state of a wetland taken as a measure of the extent to which human impacts have caused the wetland to differ from the natural reference condition.*
>
> *Whilst wetland features vary considerably from one wetland to the next, wetlands are all broadly influenced by their climatic and geological setting and by three core inter-related drivers, namely hydrology, geomorphology and water quality. The biology of the wetland (in which vegetation generally plays a central role) responds to changes in these drivers, and to activities within and around the wetland. The interrelatedness of these four components is illustrated schematically in Figure 1 below and forms the basis of the modular-based approach adopted in WET-Health Version 2.*


***Figure C.1***

*Diagram representing the four key components of Wetland PES considered in WET-Health Version 2.*

*In WET-Health, the natural reference condition of a wetland is inferred from conceptual models relating to the selected hydro-geomorphic (HGM) wetland type, the selected hydro-geological type setting and knowledge of vegetation attributes of similar wetlands in the region. PES is then assessed by evaluating the extent to which anthropogenic activities have altered wetland characteristics across the four inter-related components of wetland health, as follows:*

- ***Hydrology** is defined in this context as the distribution and movement of water through a wetland and its sediments. This module focuses on (i) changes in water inputs that result from human alterations to the catchment which affect water inflow quantity and pattern, and (ii) modifications within the wetland itself that alter the water distribution and retention patterns of the wetland (e.g. artificial drainage channels). These aspects are then integrated into a composite score that reflects the overall change in wetland hydrology.*
- ***Geomorphology** in this context is assessed by assessing changes to (i) geomorphic processes and (ii) the geomorphic structure of the wetland. Geomorphic processes in this context refer to those physical processes that are currently shaping and modifying wetland form and evolution, whilst geomorphic structure refers to the three-dimensional shape of sediment deposits on which wetland habitat is established. Whilst catchment drivers (similar to those assessed in the hydrology module) are integrated as part of the assessment, impacts are ultimately assessed based on an understanding of the degree to which within-wetland geomorphic processes and the associated structure of the wetland have been altered by anthropogenic activities. The module also accounts for differences in geomorphic processes in wetlands characterised by clastic (minerogenic) sedimentation and those characterised by organic sediment accumulation (peat).*
- ***Water quality** is defined as the physico-chemical attributes of the water in a wetland. It is assessed based on considering both potential diffuse runoff from landuses within the wetland and from the areas surrounding the wetland, together with point-source discharges of pollution entering directly into the wetland and/or into streams that flow into that wetland.*
- ***Vegetation** is defined in this context as the structural and compositional state of the vegetation within a wetland. This module evaluates changes in vegetation composition and structure as a consequence of current and historic on-site transformation and/or disturbance. Whilst the assessor needs to have some knowledge of vegetation in a particular region, the method does not require the assessor to be able to identify all wetland plant species. The emphasis is rather on identifying alien and ruderal (weedy) species that indicate disturbance, and assessing their occurrence relative to common naturally occurring indigenous species, including those that are naturally dominant in the wetland.*

*In order to undertake such an assessment, the wetland must be mapped, together with different "external areas of influence" making up the catchment of the wetland. The predefined "external areas of influence" include the area immediately adjacent to a wetland to account for impacts associated with the local upslope catchment (with a 200 m wide GIS buffer around the wetland recommended to represent this area), and the broader remaining catchment area that includes all portions of the catchment beyond the 200 m GIS buffer areas. In the case of more detailed assessments (at Level 1B and Level 2, as explained below), the areas along the edges of the inflowing channels of a wetland up to the catchment boundary are also delineated and considered separately to account for the increased impact of landuses close to influent streams relative to those occurring further away (again, a 200 m wide GIS buffer adjacent to inflowing streams is recommended to represent this area). This approach is used to account for variation* *in the natural functioning of different HGM types by applying different relative weightings to activities in each of the external areas of influence.*

*A standardised approach to scoring impacts has been adopted, which involves quantifying the extent and intensity of impacts to determine an overall "magnitude of impact" score. The extent of impact is measured as the proportion of a wetland and/or area of influence in the catchment that is affected by an activity, expressed as a percentage. The intensity of impact is estimated by evaluating the degree of alteration that results from a given activity, with scores ranging from 0 (no impact) to 10 (critical impact). The assessment accounts for a broad variety of stressors, including catchment alterations (e.g. extent of tree plantations in the catchment) and impacts in the wetland itself (e.g. the excavation of artificial drainage furrows). These impacts are assessed either directly by rating response indicators (e.g. species composition of the vegetation relative to its natural composition) or by assessing stressor indicators (e.g. extent of landcover in the catchment or the density, depth and orientation of artificial drainage furrows), to estimate impacts on wetland condition. Given the lack of baseline reference-wetland studies in South Africa, WET-Health focuses mainly on stressor indicators, but uses some response indicators, particularly when assessing impacts to the composition of wetland vegetation.*

*Indicators are aggregated in structured algorithms to derive an overall impact score for each component of wetland health. The algorithms are not simulation models but are designed to generate an index that reflects the extent of departure from natural reference conditions. The indicators are combined in a way that represents the authors' understanding of their relative importance at the time of developing this method. The rationale behind the selection of each indicator is provided, together with the rationale for combining the scores for the different areas of influence and indicators into a final impact score. Although the relationship of the aggregated scores and indicators of the method to the underlying processes have not been validated in a quantitative sense, the assumptions behind the method are provided and are supported by the international literature. Therefore WET-Health is open to progressive refinement where specific assumptions are found to be inadequate or incorrect.*

*The impact scores that generated are translated into PES scores, which reflect the similarity to the natural reference condition. The PES scores are then used to place a wetland into one of several Ecological Categories that are consistently applied across all freshwater assessments in South Africa. These categories (and associated ranges of PES scores) are as follows:*

- *Ecological Category A: Natural (90-100%)*
- *Ecological Category B: Largely natural with few modifications (80-89%)*
- *Ecological Category C: Moderately modified (60-79%)*
- *Ecological Category D: Largely modified (40-59%)*
- *Ecological Category E: Seriously modified (20-39%)*
- *Ecological Category F: Critically modified (0-19%)*

*Whilst scores for each component of wetland PES are calculated separately, individual scores from each component may also be combined into an overall PES score by weighting the component scores according to a pre-defined formula, as follows:*

*[Overall (Combined) PES Score] = ([(Hydrology score) x 3] + [(Geomorphology score) x 2] + [(Water Quality score) x 2] + [(Vegetation score) x 2)] ÷ 9).*

*This formula is refined by doubling the weighting for hydrology in situations where the wetland's hydrology has been seriously or critically modified (Ecological Category E/F). This reflects the overriding importance of hydrology in maintaining wetland processes.*

*Levels of assessment*

*Three different levels of assessment have been developed to account for a broad range of user requirements, ranging from regional assessments involving thousands of wetlands through to detailed site-based assessments used to identify specific stressors and impacts on a single wetland for management and rehabilitation planning. In each instance, the assessment is based initially on a landcover assessment that seeks to provide an initial indication of wetland condition based on a generic understanding of the impacts of different landuses on catchment and wetland processes and characteristics. The assessment is refined for more detailed assessments by integrating finer-scale mapping, and a combination of additional desktop and site-based indicators to refine and improve the accuracy of the assessments. The following three levels of assessment are catered for in the method:*

- ***Level 1A (desktop-based, low resolution)**, is an entirely desktop-based assessment and uses only preexisting landcover data (i.e. no interpretation of aerial imagery by an assessor is required) and for which default impact intensity scores have been allocated for each component of wetland PES. In many cases, particularly when applied at a national level, it is not possible to delineate the upslope catchment of each of the individual wetlands. Instead, the landcover types in a GIS buffer around a wetland and within a "pseudo-catchment" selected to represent the true catchment (such as a sub-quaternary catchment) is used as a coarse proxy of the impacts on the wetland arising from its upslope catchment. Impacts arising from the wetland and catchment are then integrated through structured algorithms to provide a coarse indication of wetland health.*
- ***Level 1B (desktop-based, high resolution)**, is also largely desktop-based using pre-existing landcover data but makes a few finer distinctions than Level 1A in terms of landcover types and usually requires interpretation of the best available aerial imagery in order to do so. This also allows the pre-defined landcover types to be mapped more accurately. Furthermore, the upslope catchment of each wetland can be individually delineated at this level, and landcover in this area is used as a proxy of the impacts on a wetland arising from its upslope catchment. As for Level 1A, impacts arising from within individual wetlands are inferred from landcover types occurring within the delineated wetlands.*
- ***Level 2 (rapid field-based assessment)**, starts with landcover mapping, but is refined by assessing a range of catchment and wetland-related indicators that are known to affect wetland health. Impacts arising from the upslope catchment of a wetland are inferred from landcover mapping but are refined based on additional information (e.g. for plantations, the user must indicate whether the trees making up the plantations are eucalypts or pines and/or wattle). Landcover types occurring within the wetland are used as the starting point for assessing human impacts arising from within the wetland. However, this initial assessment is refined considerably by sub-dividing the wetland into relatively homogenous "disturbance units" and answering a suite of site-based wetland questions which provide a more direct assessment of change (e.g. the density, depth and orientation of artificial drainage channels, and the texture of the soil in the wetland).*

*Accounting for differences and evaluating the anticipated trajectory of change*

*WET-Health Version 2 caters for differences in the inherent vulnerability of wetlands to anthropogenic impacts. This is accounted for in the method by introducing a range of modifiers that account for aspects such as the wetland's climatic setting, linkage to regional aquifers and susceptibility to erosion. Differences between HGM types are also specifically accounted for by varying the weightings of different indicators to account for differences in sensitivity to* *anthropogenic impacts and the relative importance of different areas of influence in driving natural wetland processes.*

*It is recognised that the method may not adequately cater for every situation, and expert review and refinement of impact scores is encouraged based on additional information and expert interpretation. This is accommodated in the Level 1B and Level 2 assessments by allowing the assessor to review and moderate scores with appropriate justification.*

*Whilst the main emphasis of the assessment is on assessing PES, a more thorough understanding of wetland health requires not only a diagnosis of current condition but also requires an evaluation of the trajectory of change. This is particularly relevant to detailed assessments that are undertaken to inform management actions and decision making. The likely trajectory of change is therefore also integrated into the Level 1B and Level 2 assessments. This involves simply rating the following categories: large improvement (↑↑), slight improvement (↑), remains the same (→), slight decline (↓) and large decline (↓↓) for each component of wetland health. The overall health of a wetland is then presented for each module by jointly presenting the PES category and the Trajectory of Change, e.g. C↓.*

*Key changes made to WET-Health Version 2.0*

*WET-Health Version 2.0 has a similar structure to the original version but has been comprehensively revised, incorporating elements of the Wetland IHI method and previous work undertaken to infer impacts on wetlands based on landcover types in the wetland and its catchment. Some specific changes made to the method, in going from Version 1 to Version 2.0, include:*

- *A stronger emphasis on landcover information and the mapping of landuses, particularly for desktop assessments;*
- *The explicit provision of two desktop-based levels of assessment (i.e. Levels 1A and 1B), with the spreadsheet for Level 1A purposefully designed to cater for the assessment of multiple wetlands (up to 2 000) at a time for broad-scale applications;*
- *More focussed integration of differences in wetland type, and an expansion of the method to specifically cater for wetland types not previously accommodated;*
- *A comprehensive revision of the geomorphology module, which now clearly differentiates between impacts to geomorphic structure and impacts to geomorphic processes;*
- *The addition of a water quality module, which was not included in the previous version;*
- *An overhaul of the spreadsheets, with the inclusion of additional drop-down boxes for the selection of options and many more automated calculations, in an attempt to make the spreadsheets more userfriendly;*
- *Restructuring of the spreadsheet for a Level 2 assessment to facilitate more rapid data entry and avoid duplication, by grouping the indicators that need to be considered into a list of "catchment questions" and a list of "wetland-related questions", instead of grouping the indicators by module;*
- *Integration of new research and information to refine scoring guidelines.*

*Conclusions and recommendations*

*The development of the elements listed above all represent important findings of the project and should contribute significantly to the generation of new knowledge relating to the ecological condition of wetlands in South Africa. This knowledge represents both the knowledge distilled into the tools themselves, as well as how the tools are anticipated to better equip practitioners* *and scientists in the future to add more effectively to the pool of knowledge relating to the PES of South Africa's wetlands.*

*Whilst the development of the refined suite of tools making up WET-Health Version 2.0 is thought to represent a significant advancement in the practice of wetland assessment in South Africa, it is important to emphasise that some assumptions of the revised method remain largely untested and should be validated and refined through further research and testing. As such, it is recommended that research on factors affecting the various components of wetland condition should be actively encouraged and be used to provide recommendations for further refinements to the method. Application of the method in other countries should also be actively encouraged, since the underlying principles and associated scoring system are likely to be valid, particularly in other temperate regions.*

*It is strongly recommended that a User Manual should be produced for WET-Health Version 2, to accompany and support the Technical Guide that has been produced in the current report. In addition, training material should be developed and a series of training courses should be rolled out across the country for WET-Health Version 2, once the User Manual has been produced. This will greatly facilitate the proper application of the tools, and should lead to more robust wetland PES assessments in South Africa with better consistency between assessors."*

## *Addendum C2: Parameters Used to Assess the Components of Wetland PES*

***Table S5.5*** A detailed overview of the parameters used to measure wetland health (adapted from Dada et al., 2007 and Macfarlane, Ollis and Kotze, 2020).

| Component of Wetland PES | Definition | Parameters used to assess the component (scores from 0-4) |
|---|---|---|
| Hydrology | The distribution and movement of water through a wetland and its sediments | • Changes in water inputs that affect water inflow quantity and pattern (as a result of human alterations to the catchment); and<br>• Modifications within the wetland that alter the water distribution and retention patterns of the wetland (e.g. artificial drainage channels). |
| Geomorphology | Geomorphic processes are the physical processes that are currently shaping and modifying wetland form and evolution<br>The geomorphic structure is the three-dimensional shape of sediment deposits on which wetland habitat is established | Changes in geomorphic processes and geomorphic structure of the wetland. This includes:<br>• Catchment drivers;<br>• Within wetland geomorphic processes and the structure of the wetland;<br>• Sedimentation. |
| Water Quality | The physico-chemical attributes of water in a wetland | • Diffuse runoff from land-uses within the wetland and surrounds; and<br>• Point-source discharges of pollution entering the system upstream or in the wetland itself |
| Vegetation | The structural and compositional state of the vegetation in a wetland | • Evaluates changes to vegetation due to on-site disturbance and/or transformation – mainly based on the identification of alien and weedy species in comparison with native vegetation |

## *Addendum C3: Example of a WET-Health Score Sheet*

| Wetland Attributes | |
|---|---|
| The information in this sheet must be captured before continuing with any other aspects of the assessment. Not capturing all the information required will lead to errors in the spreadsheet calculations, which will prevent a final outcome being obtained. | |
| Wetland Name | |
| Assessment Unit Name / No. | |
| Assessor | |
| Date of Assessment | |
| HGM Type (Basic) | |
| | |
| HGM Type (Refined) | |
| | |
| Conceptual model | |
| Wetland size (Ha) | |
| Upslope catchment size (Ha) | |
| Quaternary Catchment[1] | |
| MAR (Mm3) | |
| MAR per unit area (m3/Ha) | |
| MAP (mm) | |
| PET (mm) | |
| MAP:PET ratio | |
| Vulnerability Factor | |
| Hydrogeological Type Setting[2] | |

| | |
|---|---|
| Connectivity of wetland to a regional aquifer | |
| Change in groundwater levels in the regional aquifer | |
| Water quality of regional aquifer | |
| Channel characteristics (if present) | |
| Natural wetness regimes | |
| Broad vegetation attributes | |
| Number of dams in the catchment | |
| Average surface area of dams (m2) | |
| Perimeter of wetland (m) | |
| Perimeter-to-area ratio (m/ha) | |
| Down-slope length of wetland (m) | |
| Elevation change over length (m) | |
| Longitudinal Slope (%) | |
| Propensity to erode (Category)[3] | |
| Propensity to erode (Score) | |
| Dominant sediment accumulation process | |

| Catchment Land Cover Assessment | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| This sheet must be populated by estimating the extent of each landcover category in each area of influence of the wetland's catchment | | | | | | | | | | |
| Level 1B & Level 2 Landcover Categories | Proportional extent within each "area of influence" within the topographically defined wetland catchment. | | | | | | | | | |
| | Wetland Buffer (200m wide, excl. the wetland itself) | | Catchment area outside of wetland buffer | | | | | | Total catchment extent | |
| | | | Inflowing stream buffers | | Broader catchment (excl. stream buffers and wetland buffer) | | Total upstream catchment (Buffers around inflowing streams & broader catchment) | | | |
| | Area (Ha) | Area (%) | Area (Ha) | Area (%) | Area (Ha) | Area (%) | Area (Ha) | Area (%) | Area (Ha) | Area (%) |
| Open Water – Natural | | | | | | | | | | |
| Water supply dam | | | | | | | | | | |
| Aquaculture dams/ponds | | | | | | | | | | |
| Natural / Minimally impacted | | | | | | | | | | |
| Semi-natural | | | | | | | | | | |
| Moderately degraded land | | | | | | | | | | |
| Orchards and vineyards | | | | | | | | | | |
| Sugar cane | | | | | | | | | | |
| Commercial annual crops (irrigated) | | | | | | | | | | |
| Commercial annual crops (non-irrigated) | | | | | | | | | | |
| Subsistence crops | | | | | | | | | | |
| Tree plantations | | | | | | | | | | |
| Dense infestations of invasive alien plants | | | | | | | | | | |
| Quarrying (sand, stone, diamonds) | | | | | | | | | | |
| Coal mining | | | | | | | | | | |
| Ore mining | | | | | | | | | | |
| Eroded areas (& heavily degraded lands) | | | | | | | | | | |
| Urban Industrial/Commercial | | | | | | | | | | |
| Urban Informal | | | | | | | | | | |

| | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| Urban Residential – high density | | | | | | | | | | |
| Urban Residential – low density | | | | | | | | | | |
| Urban Open Space | | | | | | | | | | |
| Livestock feedlots (cattle and pigs) | | | | | | | | | | |
| Chicken farms | | | | | | | | | | |
| Planted pastures (irrigated) | | | | | | | | | | |
| Total Area | | | | | | | | | | |

| Wetland Land Cover Assessment | | |
|---|---|---|
| This sheet must be populated by estimating the extent of each landcover category in the wetland being assessed | | |
| Level 1B & Level 2 Landcover Categories | Wetland | |
| | Extent | |
| | Area (Ha) | Area (%) |
| Open Water – Natural | | |
| Deep flooding from impoundments | | |
| Shallow flooding from impoundments | | |
| Aquaculture dams/ponds | | |
| Natural / Minimally impacted | | |
| Semi-natural (undrained) | | |
| Semi-natural (drained) | | |
| Moderately degraded land | | |
| Orchards and vineyards | | |
| Sugar cane | | |
| Commercial annual crops (irrigated) | | |

| | | |
|---|---|---|
| Commercial annual crops (non-irrigated) | | |
| Subsistence crops | | |
| Tree plantations | | |
| Dense infestations of invasive alien plants | | |
| Quarrying (sand, stone, diamonds) | | |
| Coal mining | | |
| Ore mining | | |
| Eroded areas (& heavily degraded lands) | | |
| Urban Industrial/Commercial | | |
| Urban Informal | | |
| Urban Residential – high density | | |
| Urban Residential – low density | | |
| Urban Open Space | | |
| Livestock feedlots (cattle and pigs) | | |
| Chicken farms | | |
| Planted pastures (irrigated) | | |
| Infilling (incl. infrastructure) | | |
| Sediment deposits | | |
| Artificially wetter areas (e.g. seepage below dams) | | |
| | | |

| Additional Catchment-Related Questions | | | | | | | |
|---|---|---|---|---|---|---|---|
| For a Level 2 assessment, answers only need to be provided for indicators relating to catchment impacts that are highlighted in green | | | | | | | |
| INDICATOR | Table Reference | | | Assess? | ANSWER (select from drop-down options) | NUMERICAL RESULT | ADDITIONAL NOTES |
| | Hydro | Geo | WQ | | | | |
| Catchment attributes | | | | | | | |
| Average slope of the catchment[4] | | | | | | | |
| Inherent runoff potential of soils in the catchment[5] | | | | | | | |
| Wetland buffer zone characteristics (within 200m of the wetland boundary, upslope of the wetland) | | | | | | | |
| Average slope of the buffer | | | | | | | |
| Soil permeability | | | | | | | |
| Location of largely untransformed, vegetated land (natural and near-natural areas) within the buffer, upslope of the wetland | | | | | | | |
| Structural characteristics of the dominant vegetation in the buffer | | | | | | | |
| Concentration of flows | | | | | | 4 | |
| Transfers / other abstractions (Excluding irrigation in the catchment) | | | | | | | |
| Degree to which water inputs are reduced | | | | | | | |
| Flow diversion structures in the catchment | | | | | | | |
| Degree to which water inputs are intercepted and diverted | | | | | | | |
| Degree to which floodwaters are intercepted and diverted | | | | | | | |

| | | | | | | | |
|---|---|---|---|---|---|---|---|
| Dams in the catchment | | | | | | | |
| Extent: Proportion of upstream catchment occupied by dams (%) | | | | | | | |
| Interception by dams of the streams entering the wetland | | | | | | | |
| Collective volumes of dams in the wetland's catchment in relation to mean annual runoff (MAR) | | | | | | | |
| Level of abstraction from the dams | | | | | | | |
| Specific allowance for natural floods within the operating rules of the dam | | | | | | | |
| Relative importance of bedload relative to suspended load | | | | | | | |
| Invasive alien woody plants | | | | | | | |
| Plant type in upstream catchment | | | | | | | |
| Plant type in buffer | | | | | | | |
| Plantations | | | | | | | |
| Tree type in upstream catchment | | | | | | | |
| Tree type in buffer | | | | | | | |
| Erosion in the catchment | | | | | | | |
| Desktop review of catchment impacts | | | | | | | |
| Presence, size and distribution of gullies or active erosion within the catchment (including stream channels) | | | | | | | |
| Presence / extent of dirt roads in the catchment | | | | | | | |

| | | | | | | | |
|---|---|---|---|---|---|---|---|
| Breaching of upstream dams in the catchment or wetland causing an increase in sediment supply | | | | | | | |
| Transfers / point source discharges | | | | | | | |
| Increase in water input volumes from transfers / point-source discharges. | | | | | | | |
| Change in seasonality of inputs from point-source discharges. | | | | | | | |
| Point-source discharges into inflowing streams within the catchment of the wetland | | | | | | | |
| Number of known MAJOR point-source discharges into inflowing streams of the wetland[9] | | | | | | | |
| Point-source discharge #1 | | | | | | | |
| Type of effluent | | | | N | | | |
| Discharge volume | | | | N | | | # Override auto-populated answer for discharge volume, if necessary, using drop-down options |
| Distance upstream from wetland | | | | N | | | |
| Point-source discharge #2 | | | | | | | |
| Type of effluent | | | | N | | | |
| Discharge volume | | | | N | | | # Override auto-populated answer for discharge volume, if necessary, using drop-down options |
| Distance upstream from wetland | | | | N | | | |
| Point-source discharge #3 | | | | | | | |
| Type of effluent | | | | N | | | |
| Discharge volume | | | | N | | | # Override auto-populated answer for discharge volume, if necessary, using drop-down options |

| | | | | | | | |
|---|---|---|---|---|---|---|---|
| Distance upstream from wetland | | | | N | | | |
| Point-source discharge #4 | | | | | | | |
| Type of effluent | | | | N | | | |
| Discharge volume | | | 9,9 | N | | | # Override auto-populated answer for discharge volume, if necessary, using drop-down options |
| Distance upstream from wetland | | | | N | | | |
| Point-source discharge #5 | | | | | | | |
| Type of effluent | | | | N | | | |
| Discharge volume | | | | N | | | # Override auto-populated answer for discharge volume, if necessary, using drop-down options |
| Distance upstream from wetland | | | | N | | | |

[4] Refer to national map of broad-scale relief categories (provided in KMZ and GIS shapefile formats)

[5] Refer to national map of "hydrological soil groups" based on Soil Conservation Services method for Southern Africa (SCS-SA)

[6] Refer to national Sediment Yield Map

[7] Refer to national Erosion Hazard Map

[8] Refer to national Gully Erosion Map

[9] Refer to spatial database of major WWTW discharge points in South Africa from DWS (provided in KMZ format)

| | | | |
|---|---|---|---|
| Classification and Assessment of Disturbance Units | | | |
| Capture details of each Disturbance Unit here, including the extent of each Disturbance Unit and the vegetation condition of each unit relative to reference conditions. Further information on each Disturbance Unit is then captured in the "Wetland Questions" tab. | | | |
| | Number of Disturbance Units: | | |

| Disturbance Unit (DU) number | Land-cover category (Level 2) | Description | Default extent | Revised (actual) Extent | Proportional extent (%) | Typical Vegetation Impact Intensity Scores (based on landcover) | Vegetation Impact Intensity Score (0 - 10) |
|---|---|---|---|---|---|---|---|
| DU#1 | | | | | | | |
| DU#2 | | | | | | | |
| DU#3 | | | | | | | |
| DU#4 | | | | | | | |
| DU#5 | | | | | | | |
| DU#6 | | | | | | | |
| DU#7 | | | | | | | |
| DU#8 | | | | | | | |
| DU#9 | | | | | | | |
| DU#10 | | | | | | | |

| | |
|---|---|
| | 100% |
| Adjust areas entered above to ensure that the sum equals the total wetland extent | |
| Wetland extent (Ha) | |

Wetland Questions

Impacts affecting each Disturbance Unit are assessed based on a range of indicators for each impact type. Where relevant to a Disturbance Unit, indicators must be assessed by selecting from the drop-down list of attributes provided.

| INDICATOR | Table Reference | | | Assess? | ANSWER FOR EACH DISTURBANCE UNIT IDENTIFIED WITHIN THE WETLAND (select from drop-down options, where relevant) | | | | | | | | | | ADDITIONAL NOTES |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | Hydro | Geo | WQ | | DU#1 | DU#2 | DU#3 | DU#4 | DU#5 | DU#6 | DU#7 | DU#8 | DU#9 | DU#10 | |
| Stream channel modification | | | | | | | | | | | | | | | |
| Assessing changes in channel form | | | | Relevant? | N | N | N | N | N | | | | | | |
| Length of channel under Present Conditions - metres | | | | N | | | | | | | | | | | |
| Length of channel under "Reference" Conditions - metres | | | | N | | | | | | | | | | | |
| Reduction in length of stream | 7.1<br>0 | 8.14<br>8.30 | | | | | | | | | | | | | |
| Channel width under "Reference Conditions" - metres | | | | N | | | | | | | | | | | |
| Channel width under Present Conditions - metres | | | | N | | | | | | | | | | | |

| | | | | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Wetland width - metres | | | | N | | | | | | | | | | | |
| Increase in channel width (current vs reference) relative to the wetland width | | | | | | | | | | | | | | | |
| Channel depth under "Reference Conditions" - metres | | | | N | | | | | | | | | | | |
| Channel depth under Present Conditions - metres | | | | N | | | | | | | | | | | |
| Average depth of sediment deposits - metres | | | | N | | | | | | | | | | | |
| Increase in channel depth (current vs reference) in relation to the depth of sediment deposits | | | | | | | | | | | | | | | |
| Change in Cross-Sectional Area (%) | | | | | | | | | | | | | | | |

| | | | | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Category for change in cross-sectional area of the channel | | | | | | | | | | | | | | | |
| Channel Roughness under Present Conditions | | | | N | | | | | | | | | | | |
| Channel Roughness under "Reference" Conditions | | | | N | | | | | | | | | | | |
| Changes in the roughness of the stream channel. | | | | | | | | | | | | | | | |
| Evidence of within wetland anthropogenic factors causing changes in channel form? | | | | N | | | | | | | | | | | |
| Anthropogenic deposition, infilling / excavation - ***Direct impacts*** | | | | | | | | | | | | | | | |
| Excavation & Removal of Sediment (e.g. by mining) | | | | Relevant? | N | N | N | N | N | | | | | | |
| Average depth of excavation and removal | 7.1 5 | 8.20 8.28 | | N | | | | | | | | | | | |

| | | | | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Depositional Features (linked with increased sediment inputs) | | | | Relevant? | N | N | N | N | N | | | | | | |
| Average depth of recent sedimentary deposits | 7.15 | 8.33 | | N | | | | | | | | | | | |
| Direct evidence of human-induced erosion / activities in the catchment linked to identified sediment deposits? | 7.14 | 8.33 | | N | | | | | | | | | | | |
| Direct evidence of within wetland anthropogenic factors contributing to sediment deposition? | | 8.29 | | N | | | | | | | | | | | |
| Infilling (e.g. roads, dams, platforms) | | | | Relevant? | N | Y | N | N | N | | | | | | |
| Average depth of infill | 7.15 | 8.20 | | Y | | | | | | | | | | | |
| Anthropogenic deposition, infilling /excavation - *Indirect impacts* | | | | | | | | | | | | | | | |
| Anthropogenic deposition / infilling including roads & berms (Upstream effects) | | | | Relevant? | N | Y | N | N | N | | | | | | |

| | | | | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Degree to which flows are impounded | 7.16 | 8.21 | | Y | | | | | | | | | | | |
| Dams that extend laterally across the wetland (Upstream effects) | | | | Relevant? | N | N | N | N | N | | | | | | |
| Volume of dam/s in relation to MAR (in the wetland) | 7.16 | 8.21 | | N | | | | | | | | | | | |
| Relative importance of bedload relative to suspended load | | 8.21 | | N | | | | | | | | | | | |
| Representation of different hydrological zones prior to impoundment | 7.16 | | | N | | | | | | | | | | | |
| Anthropogenic deposition / infilling including roads & berms (Downstream effects) | | | | Relevant? | N | N | Y | Y | Y | | | | | | |
| Degree to which flows are intercepted and deflected away from the disturbance unit | 7.16 | 8.21 | | Y | | | | | | | | | | | |
| Dams upstream of the disturbance unit (Downstream effects) | | | | Relevant? | N | N | N | N | N | | | | | | |

| | | | | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Degree to which dam/weir interrupts low flows to downstream or lateral areas | 7.1 6 | | | N | | | | | | | | | | | |
| Level of abstraction from the dam/s | 7.6 7.1 6 | | | N | | | | | | | | | | | |
| Interception by the dam of the streams entering the downstream wetland area | 7.1 6 | 8.21 | | N | | | | | | | | | | | |
| Volume of upstream dam/s in relation to MAR (in the wetland) | 7.1 6 | 8.21 | | N | | | | | | | | | | | |
| Importance of bedload relative to suspended load | | 8.21 | | N | | | | | | | | | | | |
| Age of the dam | | 8.34 | | N | | | | | | | | | | | |
| Drains & Erosion Gullies | | | | Relevant? | Y | N | Y | N | N | | | | | | |
| Type of features present (drain | 7.1 4 | 8.19 | | Y | | | | | | | | | | | |

| | | | | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| or erosion gully)? | | | | | | | | | | | | | | | |
| Location of drains and gullies in relation to flows delivered into the disturbance unit from the upstream catchment | 7.14 | 8.19 | | Y | | | | | | | | | | | |
| Location of drains and gullies in relation to diffuse flows delivered from lateral inputs into the disturbance unit | 7.14 | | | Y | | | | | | | | | | | |
| Average depth of the drains/gullies | 7.14 | 8.19<br>8.29 | | Y | | | | | | | | | | | |
| Texture of mineral soil, if present | 7.14 | | | Y | | | | | | | | | | | |
| Degree of humification of organic soil, if present | 7.14 | | | Y | | | | | | | | | | | |
| Average gully width in relation to wetland width | | 8.29 | | Y | | | | | | | | | | | |

| | | | | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| (do not rate for drains) | | | | | | | | | | | | | | | |
| Density of drains /gullies (metres per hectare of wetland) | 7.1 4 | 8.19 | | Y | | | | | | | | | | | |
| Obstructions in the drains/ gullies | 7.1 4 | 8.19 | | Y | | | | | | | | | | | |
| Evidence of within wetland anthropogenic factors causing erosion (do not rate for drains) | | 8.19 8.29 | | Y | | | | | | | | | | | |
| Changes in Surface Roughness of the wetland | | | | Relevant? | Y | Y | Y | Y | Y | | | | | | |
| Surface Roughness (Present Conditions) | 7.1 7 | 8.22 | | Y | | | | | | | | | | | |
| Surface Roughness (Reference Conditions) | | | | Y | | | | | | | | | | | |
| Changes in the surface roughness | | | | | | | | | | | | | | | |
| Peat Fires | | | | Relevant? | N | N | N | N | N | | | | | | |

| | | | | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Depth of peat fires | | 8.31 | | N | | | | | | | | | | | |
| Direct water abstraction | | | | Relevant? | N | N | N | N | N | | | | | | |
| Direct water abstractions | 7.18 | | | N | | | | | | | | | | | |
| Increased transpiration from plantations, alien woody plants or sugarcane | | | | Relevant? | N | N | N | N | N | | | | | | |
| Dominated by alien woody plants: Type of woody plants? | 7.18 | | | N | | | | | | | | | | | |
| Dominated by plantations: Species present? | | | | N | | | | | | | | | | | |
| Dominated by sugarcane: Growth rate? | | | | N | | | | | | | | | | | |
| Reference state of native vegetation? | | | | N | | | | | | | | | | | |
| Point source discharges directly into wetland | | | | | | | | | | | | | | | |
| Number of point-source discharges entering directly into the wetland | | | 9.8 | Y | | | | | | | | | | | |
| Point-source discharge #1 | | | | | | | | | | | | | | | |

| | | | | | | |
|---|---|---|---|---|---|---|
| Type of effluent | | | | Y | | |
| Discharge volume | | | 9.9 | Y | # Override auto-populated answer for discharge volume, if necessary, using drop-down options | |
| Proportion of wetland affected (%) | | | | Y | | |
| Point-source discharge #2 | | | | | | |
| Type of effluent | | | | N | | |
| Discharge volume | | | 9.9 | N | # Override auto-populated answer for discharge volume, if necessary, using drop-down options | |
| Proportion of wetland affected (%) | | | | N | | |
| Point-source discharge #3 | | | | | | |

| | | | | | | |
|---|---|---|---|---|---|---|
| Type of effluent | | | | N | | |
| Discharge volume | | | 9.9 | N | # Override auto-populated answer for discharge volume, if necessary, using drop-down options | |
| Proportion of wetland affected (%) | | | | N | | |
| Presence of inflowing channels for depressions | | | | | | |
| Inflowing channel/s into an endorheic or exorheic depression | | | 9.10 | N | | |
| Availability of data on wetland water quality | | | | | | |
| Are water quality data available for the wetland? | | | | Y | | |
| Presence or absence of vegetation in natural reference state | | | | | | |
| Would the wetland have had less than 5% coverage of vegetation in its natural reference | | | | Y | | |

| | | | | | | |
|---|---|---|---|---|---|---|
| state (i.e. the wetland is naturally non-vegetated)? | | | | | | |

Refined Dam Calculations (Optional)

This tab may be used to refine the preliminary estimate of dam volumes used in the assessment. This is likely to be particularly relevant to wetlands with relatively small catchments, where the volumes of dams can be assessed. If not populated, the default calculations will be used in the assessment.

DAM VOLUMES (REFINED ESTIMATES)

| DAMS | Dam Shape Factor (F) | Wall Lemgth in m (L) | Throwback in m (O) | Max Depth in m (D) | Volume (Qm3 = FLOD) |
|---|---|---|---|---|---|
| DAM 1 | | | | | |
| DAM 2 | | | | | |
| DAM 3 | | | | | |
| DAM 4 | | | | | |
| DAM 5 | | | | | |
| | | | | | |
| | | | | | |
| | | | | | |
| Collective Dam Volume (refined estimate): | | | | | 0 |

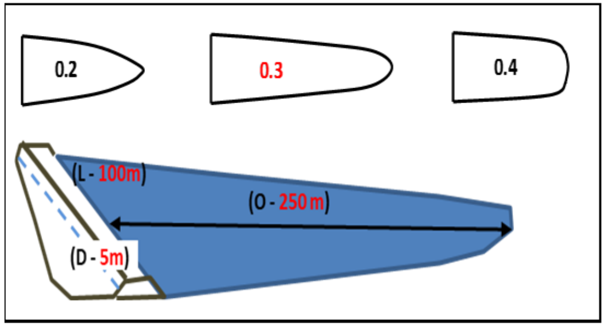
DAM VOLUME IN RELATION TO MAR

| MAR per unit area (m3/Ha) | Focal catchment Area (Ha) | MAR in m3 | Collective dam volume (m3), predicted | Collective dam volume (m3), refined | Collective volume wetland's catchmen mean annual runoff |
|---|---|---|---|---|---|
| | | | 0 | | 0% |

Rationale for refining predicted collective dam volume (if applicable):

# Overide refined collective dam volume (as pred if it is considered to be inaccurate [requires worksheet to be "unp

Dam Shape Factor (F)



* This score is calculated initially based on a desktop estimate as reflected in the "Defalt Dam Ratings" Tab, formula of Maaren and Moolman (1985) to estimate dam volume based on the number & collective area catchment. This calculation can be refined, however, if detailed information on individual dams is c spreadsheet.

## *Addendum C4: Description of the WET-EcoServices Framework*

The abstract was taken from Kotze, Macfarlane and Edwards (2020, p. ii-iv).

An overview of WET-EcoServices Version 2

*"The overall purpose of WET-EcoServices is to assist decision makers, government officials, planners, consultants and educators in undertaking rapid assessments of individual wetlands and riparian areas in order to reveal the importance of the ecosystem services which the individual wetland/riparian areas provide.*

*The specific uses for which the results of a WET-EcoServices assessment might be applied include:*

- *Prioritise resource allocation for management and rehabilitation across a set of wetland/riparian areas.*
- *Assess ecosystem service outcomes and associated trade-offs of wetland rehabilitation projects by applying the technique to "with rehabilitation" and "without rehabilitation" situations.*
- *Flag important ecosystem services as part of an Environmental Impact Assessment or to help direct further investigations, including monetary valuation of key ecosystem services.*
- *Educate and raise awareness about the importance of wetland and riparian areas.*
- *Flag important ecosystem services to be considered when managing individual wetland/riparian areas.*
- *Inform offset planning, in particular offset calculations related to water resource management.*
- *Assist in Ecological Reserve Determinations, specifically where the importance of the wetland/riparian area needs to be accounted for when deciding on ecological flows.*
- *Assist in sustainability-of-use assessments.*
- *Determine the relative importance of individual wetland/riparian areas in a catchment context.*

*Users of WET-EcoServices should have good general experience and training, preferably with a minimum of a diploma or degree in the biophysical sciences. Further, they should have attended at least a basic introductory course on wetland functioning and values and should have had at least eight weeks experience in field assessment of wetlands. In addition, input is required of someone with specific local knowledge of the geographical area to which WET-EcoServices is to be applied.*

*WET-EcoServices Version 2 includes 16 different ecosystem services, which were selected for their specific relevance to the South African situation.*

- *Flood attenuation*
- *Streamflow regulation*
- *Sediment trapping*
- *Phosphate assimilation*
- *Nitrate assimilation*
- *Toxicant assimilation*
- *Erosion control*
- *Carbon storage*
- *Biodiversity maintenance*
- *Provision of water for human use*
- *Provision of harvestable resources*
- *Food for livestock*

- *Provision of cultivated foods*
- *Cultural and spiritual experience*
- *Tourism and recreation*
- *Education and research*

*WET-EcoServices Version 2 is specifically designed for rapid field assessment, generally taking no more than two people a half day in the field and requiring no more than a half day of office preparation and data analysis. WET-EcoServices provides a set of indicators (e.g. slope of the wetland) rated on a five-point scale of 0 to 4 that reflect the supply/capability of a wetland for each of the 16 different ecosystem services listed above. An Excel™ based spreadsheet tool has been developed to conduct the assessment.*

*For each ecosystem service, indicator scores are combined automatically in an algorithm given in the spreadsheet that has been designed to reflect the relative importance and interactions of the attributes represented by the indicators to arrive at an overall supply score. In addition, the demand for the ecosystem service is assessed based on the wetland's catchment context (e.g. toxicant sources upstream), the number of beneficiaries and their level of dependency, which are also all rated on a five-point scale. Again, an algorithm automatically combines the indicator scores relevant to demand to generate a demand score.*

*Applying WET-EcoServices Version 2 encompasses seven primary steps:*

***1. Define the objectives and scope of the assessment**, based on, amongst others, the following key questions: How will the importance scores be used? What specific decisions are to be informed? and What are the specific information needs for these decisions?*

***2. Identify the Assessment Unit/s and their catchment/s and downstream service area/s**. The Assessment Units within the mapped wetland/riparian area/s are identified by dividing these areas into Hydrogeomorphic (HGM) units - each HGM unit would generally constitute a separate Assessment Unit, but in some cases may be sub-divided or grouped together.*

***3. Prepare before going into the field.** This involves becoming familiar with all of the indicators, and knowing which can be scored based on a desktop assessment and which need to be assessed in the field, and then planning the field visit.*

***4. Assess and score the indicators.** All indicators need to be assessed and scored, and for each indicator, the Rationale underlying the indicator and the Methods which should be used to assess the indicator and decide on the score are provided. It is important that the Rationale provided be understood and that the Methods be followed closely in order to promote consistency of assessment.*

***5. Enter the scores in the spreadsheet.** This is done on the first sheet of the spreadsheet, and the scores are then automatically carried through to the second sheet, where the calculations are carried out automatically.*

***6. Check and where necessary refine the Demand and Supply scores**. A "Demand & Supply" tab is used to integrate the scores for relevant indicators into a Demand and Supply score for each ecosystem service. These scores should be reviewed, and where they are identified to poorly reflect the situation on the ground can be adjusted, provided that well documented justification is given.*

***7. Present and interpret the results**, which are given as a summary table showing the supply, demand and overall importance scores for each ecosystem service and as a spider diagram showing supply and demand. Guidance is provided in terms of interpreting supply relative to demand.*

*A single overall importance score is generated for each ecosystem service by combining the supply and demand scores. Although a future scenario can also be scored, the focus here is the relative importance of the wetland/riparian area in currently providing ecosystem services. This aggregation therefore places somewhat more emphasis on supply than demand, with the supply score acting as the starting score for a "moderate" demand scenario. The importance score is, however, adjusted by up to one class up where demand is "very high" and by up to one class down where demand is "very low". The overall importance score can then be used to derive an importance category for reporting purposes. However, assessors are encouraged not to focus on the overall score (or importance category) alone as this will result in the "loss" of useful information.*

*Finally, guidance is also provided on how the WET-EcoServices technique can be applied in wetland offset calculations. This has particular relevance to assessments concerned with the impact of wetland loss and the consequent loss of wetland functions relevant to water resource management. The approach allows for an offset currency to be developed based on the local catchment context and uses a hectare equivalents approach to assess the gains in response to improvements anticipated with rehabilitation."*

## *Addendum C5: Parameters Used to Assess the Ecosystem Services Provided*

***Table S5.6*** A detailed overview of the indicators used to measure wetland ecosystem services (adapted from Rebelo et al., 2019 and Macfarlane, Ollis and Kotze, 2020) of the WET-EcoServices tool. Wetlands were scored based on their ability to supply the ecosystem services as well as the demand for such services.

| Ecosystem Service | | Definition | Parameters used to assess the ecosystem service provided (scored from 0-4) |
|---|---|---|---|
| Regulating and supporting ecosystem services | Flood attenuation | The spreading and slowing down of floodwaters, reducing the severity of floods downstream and the potential damage that the flooding may cause. | Supply:<br>Size of the upstream topographically defined catchment; Link to the stream network; Slope of the wetland; Current representation of different hydrological zones; Frequency with which storm flows are spread across the wetland; Occurrence of depressions; Soil properties (permeability); Vegetation structure in terms of height and robustness.<br>Demand:<br>Inherent runoff potential of the soils in the wetland's catchment; Rainfall intensity; Contribution of catchment land-uses to increasing runoff intensity and sediment inputs from the natural condition; Extent to which dams and other structures intercept flows from the upstream catchment; Number of people downstream expected to be affected by flooding; Level of risk that flooding poses to downstream users. |
| | Streamflow regulation | The sustaining effect of a wetland on downstream flow during low-flow periods. | Supply:<br>Link to stream network; Wetland occurs on underlying geology with strong surface-groundwater linkages; Aquifer type; Representation of different hydrological zones, Reduction in water availability due to human-caused direct water losses from the wetland; Presence of organic soil; Reduction in evapotranspiration through frosting back of the wetland vegetation.<br>Demand:<br>The number of people downstream of the wetland who are reliant on water for abstraction (run-of-river abstraction and/or from dams) livestock watering and/or recreational use; Dependence of downstream users on the water for abstraction (run-of-river abstraction and/or from dams) livestock watering and/or recreational use. |
| | Sediment trapping | The trapping and retention of sediment are carried by runoff waters. | Supply:<br>Size of the contributing upstream topographically defined catchment; Link to the stream network; Longitudinal slope of the wetland; Flow patterns of low flows within the wetland; Occurrence of depressions in the wetland; Soil properties (permeability); Direct evidence of recent sediment deposition in the wetland; Extent of vegetation cover in the wetland; Vegetation structure in terms of height and robustness.<br>Demand: |

| | | |
|---|---|---|
| | | Contribution of catchment land-uses to increasing sediment inputs from the natural condition; Extent to which dams and other structures intercept flows from the upstream catchment; Number of people downstream of the wetland who are reliant on water for abstraction (run-of-river abstraction and/or from dams) livestock watering and/or recreational use; Dependence of downstream users on the water for abstraction (run-of-river abstraction and/or from dams) livestock watering and/or recreational use; Ecological importance and sensitivity of rivers downstream of the wetland; Presence of wetlands critical to meeting wetland conservation targets downstream of the wetland; Presence and importance of downstream estuaries. |
| Phosphate removal | The removal of phosphates carried by runoff waters, enhancing water quality in the downstream catchment. | Supply:<br><br>Size of the contributing upstream topographically defined catchment; Link to the stream network; Flow patterns of low flows within the wetland; Current representation of different hydrological zones; Occurrence of depressions in the wetland; Soil properties (permeability); Direct evidence of recent sediment deposition in the wetland; Extent of vegetation cover in the wetland; Vegetation structure in terms of height and robustness.<br><br>Demand:<br><br>Extent of phosphate sources in the wetland and associated catchment; Extent to which dams and other structures intercept flows from the upstream catchment; Number of people downstream of the wetland who are reliant on water for abstraction (run-of-river abstraction and/or from dams) livestock watering and/or recreational use; Dependence of downstream users on the water for abstraction (run-of-river abstraction and/or from dams) livestock watering and/or recreational use; Ecological importance and sensitivity of rivers downstream of the wetland; Presence of wetlands critical to meeting wetland conservation targets downstream of the wetland; Presence and importance of downstream estuaries. |
| Nitrate removal | The removal of nitrates carried by runoff waters, enhancing water quality in the catchment. | Supply:<br><br>Size of the contributing upstream topographically defined catchment; Link to the stream network; Flow patterns of low flows within the wetland; Current representation of different hydrological zones; Soil properties (permeability); Extent of vegetation cover in the wetland.<br><br>Demand:<br><br>The extent of nitrate sources in the wetland and associated catchment; Number of people downstream of the wetland who are reliant on water for abstraction (run-of-river abstraction and/or from dams) livestock watering and/or recreational use; Dependence of downstream users on the water for abstraction (run-of-river abstraction and/or from dams) livestock watering and/or recreational use; Ecological importance and sensitivity of rivers downstream of the wetland; Presence of wetlands critical to meeting wetland conservation targets downstream of the wetland; Presence and importance of downstream estuaries. |

| | | |
|---|---|---|
| Toxicant removal | The removal of toxicants carried by runoff waters, enhancing water quality in the downstream catchment. | Supply:<br><br>Size of the contributing upstream topographically defined catchment; Link to the stream network; Flow patterns of low flows within the wetland; Current representation of different hydrological zones; Soil properties (permeability); Direct evidence of recent sediment deposition in the wetland; Extent of vegetation cover in the wetland; Vegetation structure in terms of height and robustness.<br><br>Demand:<br><br>The extent of toxicant sources in the wetland and associated catchment; Number of people downstream of the wetland who are reliant on water for abstraction (run-of-river abstraction and/or from dams) livestock watering and/or recreational use; Dependence of downstream users on the water for abstraction (run-of-river abstraction and/or from dams) livestock watering and/or recreational use; Ecological importance and sensitivity of rivers downstream of the wetland; Presence of wetlands critical to meeting wetland conservation targets downstream of the wetland; Presence and importance of downstream estuaries. |
| Erosion control | The control of erosion at the site through on-site factors that prevent the loss of soil from the hydrogeomorphic unit. | Supply:<br><br>Size of the contributing upstream topographically defined catchment; Longitudinal slope of the wetland; Vulnerability of the wetland to erosion given its slope and size; Frequency with which storm flows are spread across the wetland; Erodibility of the soil in the Assessment Unit based on the inherent erosion potential (K-factor) of catchment soils; Direct evidence of erosion in the wetland; Extent of vegetation cover in the wetland; Vegetation structure in terms of height and robustness<br><br>Demand:<br><br>Average slope of the wetland's catchment; Inherent runoff potential of the soils in the wetland's catchment; Rainfall intensity; Contribution of catchment land-uses to increasing runoff intensity from the natural condition; Extent to which dams and other structures intercept flows from the upstream catchment; Current level of physical disturbance of the soil in the wetland; Number of people downstream of the wetland who are reliant on water for abstraction (run-of-river abstraction and/or from dams) livestock watering and/or recreational use; Dependence of downstream users on water for abstraction (run-of-river abstraction and/or from dams) livestock watering and/or recreational use; Ecological importance and sensitivity of rivers downstream of the wetland; Presence of wetlands critical to meeting wetland conservation targets downstream of the wetland; Presence and importance of downstream estuaries; Number of people reliant on harvestable natural resources (Cumulative across all resources); Dependence of people making use of harvestable natural resources from the wetland; Number of people reliant on the wetland for livestock grazing; Dependence of livestock owners on grazing provided by the wetland; Number of people reliant on the wetland for growing cultivated foods; Dependence of people sourcing cultivated foods provided by the wetland. |

| | | | |
|---|---|---|---|
| | Carbon storage | The trapping of carbon (as soil organic matter). | Supply:<br><br>Current representation of different hydrological zones; Presence of organic soil; Current level of physical disturbance of the soil in the wetland; Vegetation structure in terms of height and robustness.<br><br>Demand:<br><br>The number of people benefiting from carbon storage functions; the Dependence of users on carbon storage provided by the wetland. |
| | Maintenance of biodiversity | The provision of habitat and maintenance of natural processes, contribute to the maintenance of biodiversity. | Supply:<br><br>Ecological condition of wetland / riparian vegetation; Presence of threatened plant and/or animal species; Diversity and heterogeneity of natural habitats; Outstanding biodiversity attributes; Ecological connectivity with other wetlands and aquatic habitats; Width of the intact buffer zone around the wetland.<br><br>Demand:<br><br>Threat status of the wetland or riparian type; Priority in national and regional conservation plans. |
| Provisioning ecosystem services | Provision of water supply for direct human use | The provision of water for direct human use (water extraction directly from a wetland area for domestic, agricultural and other purposes). | Supply:<br><br>Acceptability of water for human consumption; Periodicity of surface and shallow sub-surface water supply available for human use (domestic, agricultural or industrial).<br><br>Demand:<br><br>The number of people making use of water from the wetland/river reach for commercial agriculture or industrial use; Dependence of commercial agricultural/industrial users on abstracted water from the wetland; Number of people using water from the wetland for domestic purposes; Dependence of people using water from the wetland for domestic purposes. |
| | Provision of harvestable natural resources | The wide variety of harvestable resources available in wetlands, which are often imported from a livelihoods perspective. | Supply:<br><br>Availability of sedges, reeds and/or grasses for craft production and/or thatching; Availability of wood for construction/combustion; Availability of medicinal plants; Occurrence of fish and/or game for harvesting; Ecological condition of wetland / riparian vegetation<br><br>Demand:<br><br>The number of people reliant on harvestable natural resources (Cumulative across all resources); Dependence of people making use of harvestable natural resources from the wetland. |

| | | | |
|---|---|---|---|
| | Provision of food for livestock | The contribution of the wetland towards food for livestock. | Supply:<br>Current representation of different hydrological zones; Occurrence of grazable plants.<br>Demand:<br>The number of people reliant on the wetland for livestock grazing; the Dependence of livestock owners on grazing provided by the wetland. |
| | Provision of cultivated foods | The contribution towards food security of subsistence farmers. | Supply:<br>Current representation of different hydrological zones; Frequency with which storm flows are spread across the wetland; Presence of organic soil<br>Demand:<br>The number of people reliant on the wetland for growing cultivated foods; the Dependence of people sourcing cultivated foods provided by the wetland. |
| Cultural ecosystem services | The cultural and spiritual significance | The meanings and values that people attach to the site that arise from their interaction with the site. | Supply:<br>Known local cultural practices in the wetland; Known local taboos or beliefs relating to the wetland.<br>Demand:<br>Is the wetland in a rural communal area; Number of people who use the wetland for cultural and spiritual purposes on an annual basis; Dependence of users on the wetland for cultural and spiritual purposes. |
| | Tourism and recreation | The value of sites for tourism and recreation in terms of abundant wildlife, their scenic beauty and the open water that some wetlands provide for recreation. | Supply:<br>The bird-watching potential of the site; Presence of charismatic species; Scenic beauty of the wetland; Accessibility of the wetland for education/research or tourism/recreation purposes; Security risk associated with accessing the site for education/research or tourism/recreation purposes<br>Demand:<br>The number of people who access the wetland for tourism / recreational purposes on an annual basis; the Dependence of users on the wetland for tourism/recreation |
| | Education and research | The value for education and research, particularly when they are readily accessible. | Supply: |

| | |
|---|---|
| | Accessibility of the wetland for education/research or tourism/recreation purposes; Security risk associated with accessing the site for education/research or tourism/recreation purposes; Suitability of the wetland as a reference and/or demonstration site; Existing data & research.<br><br>Demand:<br><br>The number of people who access the wetland for education/research purposes on an annual basis; the Dependence of users on the wetland as an education/research site. |

## *Addendum C6: Example of a WET- EcoServices Score Sheet*

Sheet for entering the scores, confidence ratings and additional notes for the characteristics assessed

| | | 0 | 1 | 2 | 3 | 4 | HGM unit 1 | | | HGM unit 2 | | | HGM unit 3 |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Date of assessment | | | | | | | | | | | | | |
| Name/s of assessors | | | | | | | | | | | | | |
| Details of owner/authority | O | | | | | | | | | | | | |
| Location (Latitude; Longitude) | O | | | | | | | | | | | | |
| Wetland name | O | | | | | | | | | | | | |
| Hydro-geomorphic setting of wetland | R | F=Floodplain, VC=Valley bottom with channel, V=Valley bottom without channel, HW=Hillslope seepage feeding a water course, H=Hillslope seepage not feeding a watercourse, D=Depression | | | | | | | | | | | |
| Size (hectares) | | | | | | | | | | | | | |
| O=Data should be obtained in the office through desktop investigation prior to the field assessment. R=Data may be available through desktop investigation but is likely to be revised/refined in the field | | 0 | 1 | 2 | 3 | 4 | Score | Conf-idence rating | Additional notes | Score | Confi-dence rating | Add-itional notes | |
| HGM unit'S CATCHMENT | | | | | | | | | | | | | |
| Average slope of the HGM unit's catchment (Box 4.1h) | D | <3% | 3-5% | 6-8% | 9-11% | >11% | | | | | | | |
| Inherent runoff potential of the soils in the HGM unit's catchment (Box 4.1i) | O | Low | Mod low | | Mod high | High | | | | | | | |
| Contribution of catchment land-uses to changing runoff intensity from the natural condition (Box 4.1j) | R | Decrease | Negligible effect | Slight increase | Moderate increase | Marked increase | | | | | | | |

| | | | | | | |
|---|---|---|---|---|---|---|
| Rainfall intensity (Box 4.1k) | O | Low (Zone I) | Moderately low (Zone II) | | Mod. high (Zone III) | High (Zone IV) |
| Extent to which dams are reducing the input of sediment to the HGM unit (Box 4.3c) | R | High | Mod high | Intermediate | Mod low | Low |
| Extent of sediment sources delivering sediment to the HGM unit (box 4.3d) | R | Low | Mod low | Intermediate | Mod high | High |
| Extent of other potential sources of phosphates in the HGM unit's catchment (4.4e) | R | Low | Mod low | Intermediate | Mod high | High |
| Extent of nitrate sources in the HGM unit's catchment (Box 4.5d) | R | Low | Mod low | Intermediate | Mod high | High |
| Extent of toxicant sources in the HGM unit's catchment (Box 4.6b) | R | Low | Mod low | Intermediate | Mod high | High |

| | | | | | | |
|---|---|---|---|---|---|---|
| HGM unit | | | | | | |
| Size of HGM unit relative to the HGM unit's catchment (4.1a) | O | <1% | 1%-2% | 3-5% | 6-10% | >10% |
| Slope of the HGM unit (%) (Box 4.1b) | R | >5% | 2-5% | 1-1.9% | 0.5-0.9% | <0.5% |
| Surface roughness of HGM unit (Box 4.1c) | | Low | Mod. low | | Mod. high | High |
| Depressions (Box 4.1d) | R | None | Present but few or remain permanently filled close to capacity | Intermediate | Moderately abundant | Abundant |

| | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| Frequency with which stormflows are spread across the HGM unit (Box 4.1 e) | | Never | Occasionally but less frequently than every 5 years | | 1 to 5 year frequency | More than once a year | | | |
| Sinuosity of the stream channel (Box 4.1f) | R | Low | Moderately low | Intermediate | Mod. high | High | | | |
| Representation of different hydrological zones (Box 4.1g) | R | Permanent & seasonal zones lacking (i.e. only the temporary zone present) | Seasonal zone present but permanent zone absent | Permanent & seasonal zones both present but collectively <30% | Seasonal & permanent zone both present & collectively 30-60% | Seasonal & permanent zone both present & collectively >60% of total HGM unit area | | | |
| Link to the stream network (Box 4.2a) | R | No link (i.e. hydrologically isolated) | | | | Linked to the stream system | | | |
| Presence of fibrous peat or unconsolidated sediments below a floating marsh (Box 4.2c) | R | Absent | Present but limited in extent/depth | | Moderately abundant | Extensive and relatively deep (>1.5 m) | | | |
| Reduction in evapotranspiration through frosting back of the wetland vegetation (box 4.2d) | R | Low | Moderately low | Intermediate | Moderately high | High | | | |
| HGM unit occurs on underlying geology with strong surface-groundwater linkages (Box 4.2e) | O | No | | Underlying geology quartzite | Underlying geology sandstone | Underlying geology dolomite | | | |
| Direct evidence of sediment deposition in the HGM unit (Box 4.3b) | | Low | Mod low | Intermediate | Mod high | High | | | |
| Flow patterns of low flows within the wetland (Box 4.4b) | R | Strongly channelled | Moderately channelled | Intermediate | Moderately diffuse | Very diffuse | | | |

| | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| Extent of vegetation cover in the HGM unit (Box 4.4c) | R | Low | Mod low | Intermediate | Mod high | High | | | |
| Contribution of sub-surface water inputs relative to surface water inputs (Box 4.5c) | R | Low (<10%) | Moderately low (10-20%) | Intermediate (20-35%) | Moderately high (36-50%) | High (>50%) | | | |
| Application of fertilizers/biocides in the HGM unit (Box 4.4d) | | High | Mod high | Intermediate | Mod low | Low | | | |
| Direct evidence of erosion (Box 4.7a) | R | High | Mod high | Intermediate | Mod low | Low | | | |
| Current level of physical disturbance of the soil in the HGM unit (Box 4.7d) | R | High | Mod high | Intermediate | Mod low | Low | | | |
| Erodibility of the soil in the HGM unit (box 4.7e) | O | Low | Mod low | Intermediate | Mod high | High | | | |
| Abundance of peat (Box 4.8b) | | Absent | Present but limited in extent/depth | Intermediate | Moderately abundant | Extensive and relatively deep (>0.5 m) | | | |
| HGM unit is of a rare type or is of a wetland type or vegetation type subjected to a high level of cumulative loss (Box 4.9a) | O | No | | | | Yes | | | |
| | | | | | | | | | |
| Red Data species or suitable habitat for Red Data species (Box 4.9c) | R | No | | | | Yes | | | |
| Level of significance of other special natural features (Box 4.9d) | R | None | Mod low | Intermediate | Mod high | High | | | |
| Alteration of hydrological regime (Box 4.9g) | R | High | Mod high | Intermediate | Mod low | Low/negligible | | | |
| Complete removal of indigenous vegetation (Box 4.9j) | R | >50% | 25-50% | 5-25% | 1-5% | <1% | | | |
| Invasive and pioneers species encroachment (Box 4.9k) | | >50% | 25-50% | 5-25% | 1-5% | <1% | | | |

| | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| Presence of hazardous/restrictive barriers (Box 4.9l) | R | High | Mod high | Intermediate | Mod low | Low/negligible | | | |
| Current level of use of water for agriculture or industry (Box 4.10c) | | No use | Mod low | Intermediate | Mod high | High | | | |
| Current level of use of water for domestic purposes (Box 4.10c) | | No use | Mod low | Intermediate | Mod high | High | | | |
| Number of dependent households that depend on the direct provision of water from the wetland (Box 4.10d) | | None | 1-2 | 3-4 | 5-6 | >6 | | | |
| Substitutability of the water resource from the HGM unit (Box 4.10e) | | High | Mod high | Intermediate | Mod low | Low | | | |
| Number of different resources used (box 4.11a) | | None | 1 | | 2-3 | >3 | | | |
| Is the wetland in a rural communal area? (Box 4.11b & 4.12b) | O | No | | | | yes | | | |
| Level of poverty in the area (Box 4.11c and 4.12c) | O | Low/ negligible | Mod low | Intermediate | Mod high | High | | | |
| Number of households who depend on the natural resources in the HGM unit (Box 4.11d) | | None | 1 | 2-3 | 4-5 | >6 | | | |
| Substitutability of the natural resources obtained from the wetland (4.11e) | | High | Mod high | Intermediate | Mod low | Low | | | |
| Total number of different crops cultivated in the HGM unit (Box 4.12.a) | | None | 1 | | 2-3 | >3 | | | |
| Number of households who depend on the crops cultivated in the HGM unit (box 4.12d) | | None | 1 | 2-3 | 4-6 | >6 | | | |
| Substitutability of the crops cultivated in the wetland (Box 4.12e) | | High | Mod high | Intermediate | Mod low | Low | | | |
| Registered SAHRA site (Box 4.13a) | O | No | | | | Yes | | | |

| | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| Known local cultural practices in the HGM unit (Box 4.13c) | | None | Historically present but no longer practised | | Present but practised to a limited extent | Present & still actively & widely practised | | | |
| Known local taboos or beliefs relating to the HGM unit (Box 4.13d) | | None | Historically present but no longer so | | Present but held to a limited extent | Present & still actively & widely held | | | |
| Scenic beauty of the HGM unit (Box 4.14a) | | Low/negligible | Mod low | Intermediate | Mod high | High | | | |
| Presence of charismatic species (Box 4.14b) | R | None present | Very seldom seen | Occasionally present | Generally present | Always present | | | |
| Current use for tourism or recreation (Box 4.14c) | | No use | Mod low use | Intermediate use | Mod high use | High | | | |
| 4. Availability of other natural areas providing similar experiences to the HGM unit (Box 4.14d) | R | High | Mod high | Intermediate | Mod low | Low | | | |
| Location within an existing tourism route (box 4.14f) | O | Low/negligible | Mod low | Intermediate | Mod high | High | | | |
| Recreational hunting and fishing and birding opportunities (Box 4.14g) | | None | Mod low | Intermediate | Mod high | High | | | |
| Extent of open water (box 4.14h) | R | None | Present, but very limited | | Extent somewhat limited | Extensive | | | |
| Current use for education/research purposes (Box 4.15a) | R | No use | Mod low | Intermediate | Mod high | High | | | |
| Reference site suitability (Box 4.15b) | | Low | Mod low | Intermediate | Mod high | High | | | |
| Existing data & research (Box 4.15c) | R | None | Mod low | Intermediate detail/ time period | Mod high | Comp-rehensive | | | |

| | | | | | | |
|---|---|---|---|---|---|---|
| | | | | | | data over long period |
| Accessibility (Box 4.15d) | R | Very inaccessible | Moderately inaccessible | Intermediate | Moderately accessible | Very accessible |
| | | | | | | |
| DOWNSTREAM OF HGM UNIT | | | | | | |
| Extent of floodable property (Box 4.1l) | R | Low/ negligible | Moderately low | | Moderately high | High |
| Presence of any important wetlands or aquatic systems downstream (Box 4.2f) | R | None | | Intermediate importance | | High importance |
| | | | | | | |
| LANDSCAPE | | | | | | |
| Extent of buffer around wetland (4.9e) | R | Low | Mod low | Intermediate | Mod high | High |
| Connectivity of wetland in landscape (Box 4.9f) | R | Low | Mod low | Intermediate | Mod high | High |
| Level of cumulative loss of wetlands in overall catchment (box 4.9j) | O | Low | Mod low | Intermediate | Mod high | High |
| | | | | | | |
| | | | | | | |
| THREATS & OPPORTUNITIES | | | | | | |
| Level of threat to existing ecosystem services supplied by the wetland (Table 4.16a) | | Low | Moderately low | Intermediate | Moderately high | High |
| Level of future opportunities for enhancing the supply of ecosystem services (Table 4.16b) | | Low | Moderately low | Intermediate | Moderately high | High |

| | | | | | | |
|---|---|---|---|---|---|---|
| DERIVED CHARACTERISTICS | | | | | | |
| These are characteristics that are derived from other characteristics and therefore do not need to be entered directly | | | | | | |
| Runoff intensity from the HGM unit's catchment | No score | No score | No score | No score | No score | No score |
| Alteration of sediment regime | No score | No score | No score | No score | No score | No score |
| Alteration of nutrient/toxicant regime | No score | No score | No score | No score | No score | No score |

## *Addendum C7: Supply scores for WET-EcoServices Assessments*

***Table S5.7*** Supply scores of the ecosystem services supplied by each assessment unit assessed along the Kuils River. Refer to Figure 5.3 for the graphic representation of this table.

| | | ASSESSMENT UNITS | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| ECOSYSTEM SERVICE | | 1AU1 | 2AU1 | 2AU2 | 2AU3 | 2AU4 | 2AU5 | 2AU6 | 3AU1 | 3AU2 | 4AU1 | 4AU3 | 4AU5 | 4AU6 |
| SUPPORTING AND REGULATING SERVICES | Flood attenuation | 0,0 | 0,0 | 0,0 | 0,0 | 0,0 | 0,0 | 0,5 | 0,0 | 0,1 | 1,1 | 0,6 | 1,6 | 0,9 |
| | Stream flow regulation | 0,8 | 0,5 | 1,2 | 0,8 | 0,4 | 0,8 | 1,2 | 1,2 | 1,2 | 3,7 | 3,7 | 3,0 | 3,7 |
| | Sediment trapping | 0,8 | 1,0 | 2,0 | 1,8 | 2,0 | 2,0 | 2,0 | 1,8 | 2,0 | 1,5 | 0,5 | 1,1 | 1,5 |
| | Erosion control | 0,1 | 0,0 | 1,2 | 1,7 | 0,4 | 1,2 | 1,0 | 1,2 | 1,0 | 1,9 | 1,5 | 1,5 | 1,9 |
| | Phosphate assimilation | 1,6 | 1,8 | 2,9 | 2,7 | 2,9 | 2,9 | 2,9 | 2,7 | 2,9 | 2,0 | 1,4 | 1,7 | 2,0 |
| | Nitrate assimilation | 1,3 | 1,3 | 2,7 | 2,3 | 2,3 | 2,5 | 2,8 | 2,5 | 2,8 | 2,7 | 2,0 | 2,2 | 2,7 |
| | Toxicant assimilation | 0,9 | 0,9 | 2,3 | 2,0 | 2,1 | 2,2 | 2,3 | 2,1 | 2,3 | 2,2 | 1,5 | 1,6 | 2,2 |
| | Carbon storage | 0,6 | 0,4 | 1,5 | 0,9 | 0,4 | 1,3 | 1,5 | 1,5 | 1,5 | 0,8 | 0,6 | 0,3 | 0,8 |
| | Biodiversity maintenance | 0,1 | 0,3 | 1,8 | 3,4 | 1,0 | 1,8 | 1,8 | 1,8 | 1,8 | 4,0 | 4,0 | 4,0 | 4,0 |
| PROVISIONING SERVICES | Water for human use | 0,0 | 0,9 | 1,1 | 1,2 | 1,7 | 1,2 | 1,1 | 1,1 | 1,1 | 1,6 | 1,3 | 1,6 | 1,4 |
| | Harvestable resources | 0,0 | 0,0 | 1,2 | 1,2 | 0,7 | 1,7 | 1,7 | 1,0 | 0,7 | 1,5 | 0,0 | 0,0 | 1,5 |
| | Food for livestock | 1,5 | 2,5 | 0,8 | 1,5 | 2,7 | 1,5 | 0,8 | 0,8 | 0,8 | 1,3 | 0,0 | 0,5 | 1,3 |
| | Cultivated foods | 1,0 | 1,5 | 0,5 | 1,0 | 1,7 | 1,0 | 0,5 | 0,5 | 0,5 | 0,8 | 0,3 | 1,3 | 0,7 |
| CULTURAL SERVICES | Tourism and Recreation | 0,2 | 0,0 | 0,3 | 0,3 | 0,0 | 0,0 | 0,3 | 0,4 | 0,2 | 0,2 | 0,0 | 0,0 | 0,2 |
| | Education and Research | 0,0 | 0,0 | 0,2 | 0,2 | 0,2 | 0,2 | 0,2 | 0,2 | 0,2 | 0,2 | 0,0 | 0,0 | 0,2 |
| | Cultural and Spiritual | 0,0 | 0,0 | 0,7 | 0,5 | 0,0 | 0,0 | 0,5 | 0,5 | 0,5 | 3,5 | 2,5 | 2,5 | 3,5 |