AN INVESTIGATION INTO THE HEALTH STATUS OF THE WETLANDS DELINEATING THE LOWER UMNGENI RIVER FLOODPLAIN, KWAZULU-NATAL, SOUTH AFRICA

By

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Submitted in fulfillment of the academic requirements
for the degree of

MASTER OF SCIENCE
in the
School of Agricultural, Earth and Environmental Sciences
University of KwaZulu-Natal

November 2014

PREFACE

All of the work presented henceforth is ultimately based on the experimental work conducted in the School of Agricultural, Earth and Environmental Sciences, University of KwaZulu-Natal, Durban (South Africa) – from February 2011 to November 2014 under the supervision of Dr. Srinivasan Pillay.

This dissertation comprises the original intellectual product of the author, Samantha Naidoo, and has not been submitted in substance for any other degree or award at this or any other learning institution, nor is it being submitted concurrently for any other degree or award. Information sources and the work of others are duly acknowledged as such.

ACKNOWLEDGEMENTS

My sincere gratitude goes out to the following people for their assistance and contribution towards the successful completion of this study:

1. God, for never failing to provide me with guidance, strength and the courage to persevere.
2. My beloved parents, for your unwavering love and support, not only during the compilation of this thesis, but throughout my life. A special thanks to my dad, who accompanied me on my site visits and provided assistance during my many hours in the laboratory.
3. Dr. Srinivasan Pillay, for your assistance, guidance and mentorship throughout my undergraduate and postgraduate years in UKZN. Your willingness to provide support when I needed it most will not be forgotten.
4. My dearest Patrick, for your continuous and unconditional love and support.
5. My friend Rohaida, for your assistance, advice and moral support. I am truly grateful for all that you've done for me.
6. The National Research Foundation (NRF) for funding this research.
7. Mr. Ajay Bissessur, for all your efforts and assistance in analysing the data for this study.
8. Mr. Eddie Powys, for your assistance in the laboratory.
9. Mr. Andrew Thurlow, for your help with creating the maps utilised in this study.
10. Mr. George Biermann and Mrs. Karin Burns, for your assistance with the illustrations and formatting of this document respectively.
11. To anybody I've failed to recognize who has helped in the compilation of this thesis. It is most appreciated.

DECLARATION - PLAGIARISM

I, Samantha Naidoo, declare that:

1. The research reported in this thesis, except where otherwise indicated, is my original research.
2. This thesis has not been submitted for any degree or examination at any other university.
3. This thesis does not contain other persons' data, pictures, graphs or other information unless specifically acknowledged as being sourced from other persons'.
4. This thesis does not contain other persons' writing unless specifically acknowledged as being sourced from other researchers. Where other written sources have been quoted, then:
    a) Their words have been re-written but the general information attributed to them has been referenced
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5. This thesis does not contain text, graphics or tables copied and pasted from the internet, unless specifically acknowledged, and the source being detailed in the thesis and in the Reference sections.

Signed:

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Samantha Naidoo

As the candidate's supervisor I have/have not approved this dissertation/thesis for submission.

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DECLARATION - PUBLICATIONS

The chapters of this thesis were synthesized as individual research papers which were submitted for publication in the journals listed hereunder:

Publication 1

An assessment of the anthropogenic contamination of sediments constituting the floodplain wetlands of the Lower uMngeni River, KwaZulu-Natal, South Africa.

This article has been submitted to the *Environmental Earth Sciences Journal* and is under review.

Publication 2

A comparative study of the water quality status of the Lower uMngeni River and adjacent floodplain wetlands, KwaZulu-Natal, South Africa.

This article has not yet been submitted to a relevant academic journal for publication.

Signed:

Samantha Naidoo

ABSTRACT

An amalgamation of a number of factors, *inter alia,* increased anthropogenic activity, reclaiming wetlands for other uses, poor planning of projected developments, and exponential population growth, have spurred a decline in the health status of wetland environments and interconnected river systems. This is particularly true of wetlands located within developed areas and is also characteristic of the floodplain wetlands of the Lower uMngeni River located in the eThekwini Municipality, KwaZulu-Natal, South Africa. The Lower uMngeni sub-catchment is heavily utilized and characterised by a multitude of human activities and land uses including urban and residential development, subsistence agriculture, recreational activity and, light and heavy industrial operations.

Given this intensive development and the recognized value of wetlands, this study sought to determine the health status of the Lower uMngeni River and adjacent wetland systems by assessing contamination levels of the wetland soil, the wetland interstitial water and river water.

Water and sediment samples were collected along transects established at strategically selected locations, each of which are situated at the intersection of different land uses or within close proximity of the major human activities within the locality. Such an approach allowed for inferences to be made regarding the potential anthropogenic sources of pollution. This was achieved by performing laboratory assessments, calculating geochemical indices and conducting statistical analyses for a range of physico-chemical parameters. The investigation also took into account the effect of seasonality on detected indicator concentrations in water samples, and determined compliance of measured concentrations with the prescribed DWAF limits.

The findings indicate that wetland sediment displays signs of anthropogenic contamination, especially within the uppermost soil lamina, indicating the recency of pollution events. The dominance of fine-grained sediment in the wetlands indicated increased vulnerability to heavy metal and nutrient pollution. A strong positive relationship exists between wetland interstitial water and river water at most sample sites, indicative of the easy transfer of contaminants between these environments. Furthermore, elemental concentrations detected in

interstitial water and/or river water varied according to seasonal rainfall patterns and were directly influenced by adjacent land uses. The majority of these concentrations were also non-compliant with the stipulated DWAF limits; necessitating that management, rehabilitation and monitoring measures be identified and implemented, where and if relevant, as a matter of urgency.

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LIST OF ABBREVIATIONS

Al	Aluminium
ANOVA	Analysis of Variance
As	Arsenic
Ba	Barium
Ca	Calcium
$CaCO_3$	Calcium carbonate
CF	Contamination Factor
Co	Cobalt
Cr	Chromium
CSIR	Council for Scientific and Industrial Research
Cu	Copper
DEA	Department of Environmental Affairs
DEAT	Department of Environmental Affairs and Tourism
DO	Dissolved Oxygen
DUCT	Duzi-uMngeni Conservation Trust
DWA	Department of Water Affairs
DWAF	Department of Water Affairs and Forestry
DWS	Department of Water and Sanitation
EF	Enrichment Factor
ESMP	Environmental Services Management Plan
Fe	Iron
GPS	Global Positioning System
ICP-OES	Inductively Coupled Plasma – Optical Emission Spectroscopy
IDP	Integrated Development Planning
HGM	Hydrogeomorphic
HKS	Hill Kaplan Scott Incorporated Consulting Engineers
IRC	International Water and Sanitation Centre
KZN	KwaZulu-Natal
LUMS	Land Use Management System
MC	Moisture Content
MCMP	Mgeni Catchment Management Plan
Mg	Magnesium
Mn	Manganese
NEMA	National Environmental Management Act (No. 107 of 1998)
NEMBA	National Environmental Management: Biodiversity Act (Act 10 of 2004)

NEMICMA	National Environmental Management: Integrated Coastal Management Act (Act 24 of 2008)
NFEPA	National Freshwater Ecosystem Priority Areas
NH_4^+	Ammonium ion
Ni	Nickel
NPAES	National Protected Areas Expansion Strategy
NWA	National Water Act (No. 36 of 1998)
NWCS	National Wetland Classification System
NWI	National Wetland Inventory
NWRS	National Water Resource Strategy
OMC	Organic Matter Content
P	Phosphorous
Pb	Lead
pH	Potential of Hydrogen
PLI	Pollution Load Index
ppm	Parts per million
PVC	Polyvinyl Chloride
RCS	Ramsar Convention Secretariat
r	Pearson Correlation Coefficient
S	Sulphur
SANBI	South African National Biodiversity Institute
SDF	Spatial Development Framework
SO_4^{2-}	Sulphate ions
TDS	Total Dissolved Solids
UKZN	University of KwaZulu-Natal
UNDP	United Nations Development Programme
USEPA	United States Environmental Protection Agency
V	Vanadium
WHO	World Health Organization
WRC	Water Research Commission
Zn	Zinc

LIST OF UNITS

cc	Cubic Centimetre
cm	Centimetre
g	Gram
km	Kilometre
km^2	Square kilometre
L	Litre
m	Metre
m^2	Square metres
mg/kg	Milligrams per Kilogram
mg/L	Milligrams per litre
mL	Millilitre
%	Percentage
µm	Micrometres
°C	Degrees centigrade
<	Less than
>	Greater than

CHAPTER 1: INTRODUCTION

1.1 Preamble

A number of major landscapes in the world have taken predominance as subjects of interest within various disciplines of academia (Williams, 1990). Amongst these, wetlands have only recently captured the attention of scholars (Bassi *et al.*, 2014). The general lack of interest in building a substantive knowledge-base on wetland environments, through increased investigation and research efforts, stemmed from the misconception that dominated scientific communities, and societies at large, regarding the importance and functioning of such systems (Reddy and Gale, 1994). In the past, wetlands were perceived as insignificant patches of marsh or swampland, which were known to prompt timely outbreaks of scourge and disease, bringing adversity and strife to nearby communities (Horwitz *et al.*, 2012).

Apart from past perceptions, however, the recency of their recognition as systems of immense value can also be attributed to the inconspicuous nature of their defining characteristics (Williams, 1990). Unlike climatically-based landscapes; such as mountainous regions that cover large stretches of land; wetlands are comparably less majestic, being characterised by considerably smaller surface areas (Williams, 1990). This often served as justification for the possible environmental significance of such landscapes being overlooked by researchers (Williams, 1990). Over the past few years, however, studies have provided vast amounts of information on the role of wetlands as facilitators of important environmental processes, drivers of socio-economic uplift, and hubs of biotic activity (Mitsch & Gosselink, 2011). This in turn afforded the motive required for conducting more detailed research concerning the health of such landscapes, and the monitoring and management thereof.

Scientific literature provides numerous definitions of wetlands. One of the most widely used is *"… land which is transitional between terrestrial and aquatic ecosystems 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"* (Day, 2009: 846). They can also more simply be defined as *"lands with soils that are periodically flooded"* (Williams, 1990:1). These landscapes are ubiquitous, appearing in almost every climatic zone in the world (Williams, 1990). They occur in areas that range widely in altitude, and vary in degree of wetness depending on climatic, hydrologic and topographic influences (Begg, 1986). As a result, there are different types of wetlands. Although not all types of wetlands offer all ecological and economic benefits that may be derived from such systems, most wetland types provide several, depending largely on their unique characteristics and site-specific conditions (USEPA, 1996).

Wetlands function in a number of ways to improve the health status of water-dominated environments on a global scale, and contribute significantly to building the economy of nations (Mitsch & Gosselink, 2000). This is linked to their inherent ability to serve as water purification systems; their utilitarian values; and their usefulness in crop, pasture, timber and raw material production; waste assimilation; and erosion and pest control (Begg, 1986). Furthermore, these landscapes take on important ecological and cultural roles; having immense recreational, aesthetic and educational value; within their geographical boundaries both nationally and internationally (Begg, 1986).

It is thus apparent that wetlands are features of remarkable value. However, unrestricted human activity has resulted in encroachment upon these systems and unsustainable utilization and damage of wetland resources. Consequential is the accelerated pace of wetland degradation, which in the absence of appropriate management measures, leads to their widespread destruction (Kusler & Opheim, 1996).

1.2 Problem Contextualization and Motivation for the Study

Wetlands are being obliterated at an alarming rate on a global and national scale, becoming a cause for much concern (Gibbs, 2000). Research has shown that since 1990, more than half of the world's wetlands may have disappeared (Barbier, 1993). Such destruction is the outcome of external forces that act on these systems, the most powerful of which are anthropogenic activities (Wuver and Attuquayefio, 2006). Inclusive are dredging, draining, infilling, water abstraction and inappropriate agricultural practices (Gibbs, 2000). Wetlands have also been used as "wastelands", into which a variety of urban and industrial wastes are disposed by both point and non-point sources of pollution (Barbier, 1993; Hendricks, 2004).This is often to such extents that it results in the exceedance of their respective assimilative capacities, causing deterioration and ultimately ecosystem demise (Barbier, 1993; Hendricks, 2004).

According to Harvey (2007), the elimination of wetlands must be prevented, as the resulting loss to society and the environment is much too significant. The downstream consequences of wetland destruction are substantial and include increased severity of flood and drought events; increased siltation; reduced productivity and recreational value of impoundments, estuaries and lagoons; a marked deterioration in water quality; and the disruption of wildlife (Begg, 1986). Apart from these, wetlands serve as habitats for a multitude of biota adapted to survive only in conditions that typify wetland environments (Cwikiel, 1996). Therefore, wetland degradation also endangers the lives of wetland-dependent and -reliant species, thereby increasing their probability of extinction (Cwikiel, 1996).

In South Africa, like in many other parts of the world, development decisions often prove unfavourable to the health and continued existence of wetlands. Studies have revealed that wetlands previously occupied between 10% and 15% of every catchment in KwaZulu-Natal (Begg, 1986). However, they have now been virtually eliminated or reduced to a few scattered remnants over the past 50 years (Begg, 1986). In addition, the structure and functioning of the remaining wetlands are being constantly altered by a number of impetuous human activities that continue to increase in number and intensity (Begg, 1986).

Of these remaining wetlands, coastal floodplain wetlands are at the greatest risk of degradation and destruction (Cleugh *et al.*, 2011). Floodplain wetlands occur along the lower reaches of most riverine systems, and perform functions that yield both environmental and socio-economic benefits (Nagle, 1999). Considering this, it is precisely these areas that are sought after for development, particularly where major towns and cities are located in close proximity to river mouths and estuaries (Shine & de Clemm, 1999). The cumulative impact of human disturbance within the vicinity of wetland systems, through land modification and the continued release of harmful and toxic wastes, is a decline in wetland health, which concomitantly incites a loss of habitat and ecosystem integrity (Millennium Ecosystem Assessment, 2005).

Amongst the existing floodplain wetlands in KwaZulu-Natal, those delineating the uMngeni River, within the Lower uMngeni sub-catchment region, form the focus of this particular study.

The uMngeni fluvial system is recognized as the lifeblood of the development core of KwaZulu-Natal as it traverses the major developed areas of the province (Brijlal, 2005). The system is recognised by the plethora of human activities taking place within its watershed, including agricultural, industrial, urban and both formal and informal residential activities (WRC, 2002). As a result, it has been impacted on progressively over the years (Department of Water Affairs and Forestry (DWAF) and Umgeni Water, 1996). Numerous scientific investigations aimed at determining the impact of these activities on the health of the river system have been undertaken (WRC, 2002). However, there are limited records to date of investigations conducted for the purpose of ascertaining the health of the floodplain wetlands situated along the boundaries of the uMngeni River channel.

Specific activities that contribute to the degradation and destruction of wetlands in the Lower uMngeni sub-catchment include dam construction, agricultural practices, industrialization, urbanisation and sand mining (WRC, 2002). Such occurrences endanger the lives of wetland biota; cause ecosystem disequilibrium; and may result in the eventual collapse of these important systems should these conditions persist in the long-term (Brijlal, 2005). Therefore, in order to prevent these wetlands from becoming void of any future benefits (both ecosystem and human), careful and effective management is necessary (DWAF and Umgeni Water, 1996). Taking the above into account, it is essential that these wetlands be studied.

1.3 Aim

The aim of this study is to determine the physico-chemical characteristics of sediment and interstitial water comprising the floodplain wetlands of the Lower uMngeni River, and thus provide reference data and an assessment of natural and anthropogenic impacts on the health of these major wetlands.

1.4 Objectives

The objectives of the study are:

1. To determine the physico-chemical properties of water in the Lower uMngeni River, wetland interstitial water and wetland sediment by measuring:

a) Chemical parameters, which include, inter alia -
   i. Nutrients;
   ii. Heavy metals; and
   iii. Other elements.
b) Physical parameters, which include, inter alia -
   i. Organic matter content (OMC);
   ii. Calcium carbonate ($CaCO_3$);
   iii. Textural character; and
   iv. Moisture content.

2. To identify patterns, trends or relationships based on detected levels of physico-chemical parameters relating to the level of sediment contamination, river water quality and wetland interstitial water quality.

3. To relate nutrient and heavy metal concentrations detected in wetland sediment, samples, wetland interstitial water sample and river water samples to background or accepted values which enables the assessment of contamination and the identification of potential pollution sources.

4. To assess the enrichment and pollution status of selected wetlands along the Lower uMngeni River using geochemical indices.

5. To provide physico-chemical baseline data, based on which rehabilitation, management and monitoring actions will be recommended to supplement the existing Catchment Management Plan developed for the uMngeni catchment.

1.5 Assumptions and Limitations

1. It is assumed that the information presented in all literature used as reference sources, and spatial data such as shapefiles and geodatabases, are accurate and reliable.

2. The facts and figures presented and utilised in this study are limited to information that was freely available and accessible at the time of preparation.

1.6 Chapter Sequence and Summation

Chapter One of this dissertation introduces wetlands, states their most widely used definitions, and explains how past perceptions of these systems have changed over time. It also provides context on the problem of wetland degradation and loss; the causes; and the environmental and socio-economic repercussions thereof.

Chapter Two provides an overview of literature on the topic of study, highlighting and expanding on components that have particular relevance to the investigation being conducted. It describes wetland ecosystems, more specifically the physico-chemical environment of wetlands, with particular attention being afforded to the properties of soil and wetland interstitial water. Thereafter, wetland health and integrity, their indicators, and factors that pose a threat to their health and functioning are discussed. The values and roles of wetlands, the issue of wetland loss in South Africa, and the legal framework that enforces the need to protect, conserve, manage and rehabilitate wetlands are also detailed.

A description of the site in terms of its defining attributes, geographic location, and regional and catchment characteristics are presented in Chapter Three. The focal area for this study is defined in this Chapter, and the various land cover features and land uses that potentially impact on water quality, the level of sediment contamination, and therefore wetland and river health, are demarcated in relation to the floodplain wetlands of interest and the Lower uMngeni River. Existing management plans and institutional arrangements for the Lower uMngeni sub-catchment are also described.

The materials and methods employed to collect and analyse the water and sediment samples for this investigation are elucidated in Chapter Four. Also highlighted in this chapter are: the method utilised to select the wetlands for investigation, the location of selected sampling sites, the sampling protocol followed, procedures involved in the laboratory analysis of physical and chemical indicators of water and sediment samples, the types of statistical analyses undertaken for measured values, and the indices utilised to assess the level of pollution in the systems of concern.

Chapter Five is the results and discussion section of this dissertation, and is presented in the form of academic papers. The intent is to submit these papers to scientific journals for publication. This chapter focuses on the possible impacts of measured physical and chemical parameters in wetland interstitial water samples, river water samples and wetland sediment samples on the health of the Lower uMngeni River floodplain wetlands. Results are discussed in relation to key anthropogenic activities that potentially contribute to a decline in water and sediment quality. The pollution status of these floodplain wetlands are determined by conducting a geochemical assessment on wetland sediments, and comparing detected concentrations of water quality parameters to the acceptable limits prescribed by DWAF (1996a, 1996b). Statistical methods of analysis, specifically ANOVA and the correlation function, are utilised to show the variance of elemental concentrations at each sediment sampling site and ascertain whether a relationship exists between wetland interstitial water and river water respectively.

Chapter Six identifies viable rehabilitation, management and monitoring measures that should be formalised and executed with the aim of restoring and protecting the floodplain wetlands of the Lower uMngeni River.

A management and monitoring programme for the recommended rehabilitation initiative is proposed. Chapter Seven outlines the salient findings and conclusions of the investigation in line with the study aim and objectives.

CHAPTER 2: LITERATURE REVIEW

2.1 Introduction

This chapter provides a detailed description of wetland ecosystems, their health and integrity, their distinguishing characteristics, and an explanation of their importance which stems from the many resources and services they provide. In addition, wetland classification, factors that threaten the health and continued existence of wetland systems, and the resulting national, and global crisis of wetland loss are further addressed. An outline of the existing legislation, policies and guidelines used to inform and promote the application of management and rehabilitation strategies within a national context are also provided.

2.2 Wetland Ecosystems

Wetland ecosystems are shown to be three-component ecosystems, comprising of both biotic and abiotic elements (Cronk, 2001:371; Mitsch *et al.*, 2009). Three important components underpinning the very functioning of wetland ecosystems; namely hydrology, the physiochemical environment and biota; are interrelated and interact intimately with each other (Mitsch *et al.*, 2009). This is such that the hydrology of the landscape influences the physiochemical environment, which in turn, together with hydrology determines the biotic constituents of wetlands (Mitsch *et al.*, 2009).

Apart from this, factors external to the immediate ecosystem, such as climate (including solar energy, temperature patterns and precipitation); and geomorphology of the landscape, jointly determine the existence of wetlands given their strong influence over the location and duration of water presence in an area (Mitsch *et al.*, 2009, van der Valk, 2012). This renders climate and landscape geomorphology the principal determinants of wetland formation and existence (van der Valk, 2012).

Figure 2.1 provides an illustration of the network of interactions between these major influencing factors.

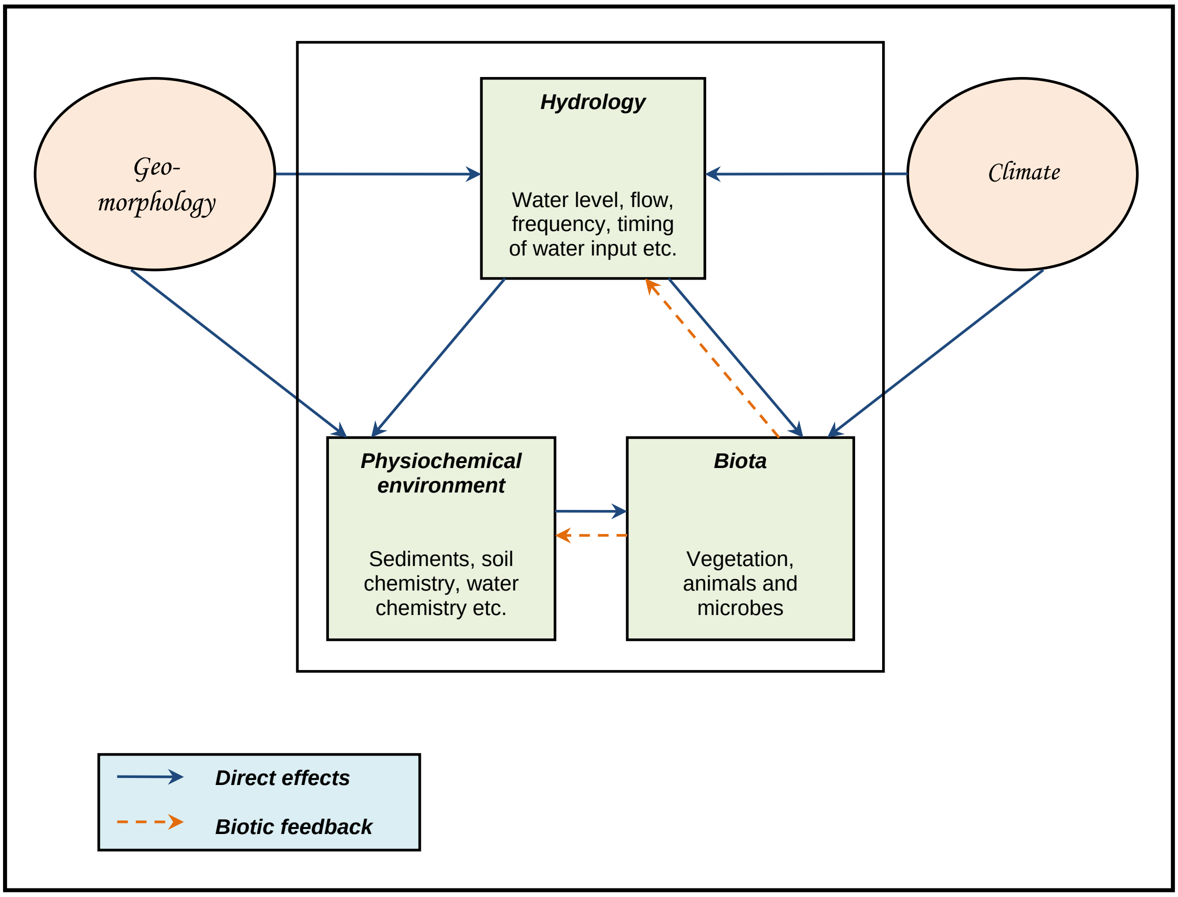


Figure 2.1: Conceptual model of wetland ecosystems, showing the three-component basis of wetlands, the direct and indirect effects resulting from the interconnectedness of the components, and the principal determinants of wetlands (adapted from Mitsch & Gosselink, 2007, 2000; NRC, 1995 cited in Mitsch *et al.*, 2009)

This study aims to inform a more in-depth investigation required to ascertain the health of wetlands along the Lower uMngeni riverine system by establishing the status of their physiochemical environment. Considering the interrelatedness of wetland ecosystem components, the hydrological environment of these wetlands will be considered to a certain extent, where relevant in explaining the observable variances in the physiochemical environment. This will be accomplished by determining the characteristics and concentrations of certain key physical and chemical parameters within constituent sediment and interstitial water, and relating these measurements to rainfall data and water level in wetlands and the adjacent river system.

The following section provides a description of wetland physiochemical environments, and how the associated indicators and attributes can be used to ascertain the susceptibility of wetland systems to contamination.

2.2.1 The Physiochemical Environment of Wetlands

2.2.1.1 Wetland Interstitial Water

The major inflows and outflows of water in wetland systems include surface water inflows and outflows, precipitation, groundwater inflows, groundwater to or from rivers, tidal inflow and outflows, groundwater flows from uplands and evapotranspiration (van der Valk, 2012). In order for a wetland to occur in a specific area, the sum of the input components must exceed the sum of the output components (Schwirzer, 2006). Rivers interact with floodplain wetlands in three ways: 1) through direct surface runoff, 2) through sub-surface water flow, and 3) by losing water to wetlands by seepage through the river walls (Junk, 1989).

Relative to this, it must be mentioned that water flows and surface and sub-surface water levels in most wetlands are dynamic, being subject to the effects of seasonal change, and thus climate and geomorphology (Schwirzer, 2006). Floodplain wetlands in particular, which delineate river channels, usually receive water from rivers systems during dry seasons and drought events, when the water table drops below that of prevailing water levels in the adjacent river (Nyarko, 2007). Nyarko (2007) showed that a correlation exists between surface and wetland interstitial water quality. This is such that changes in the wetland environment, resulting from the influence of external forces, such climatic and seasonal variations, are accompanied by changes in the river system and vice versa. Noteworthy in these cases are the consequential effects on the physical and chemical properties of river water, and wetland interstitial water (Kingsford, 2000).

In line with the above, an objective of this study is to determine the water quality of wetland interstitial water constituting wetlands along the Lower uMngeni River, and that of the uMngeni River itself, in relation to anthropogenic activities that occur in its catchment and hydrological patterns incumbent upon seasonal change.

Considering the intricate link between floodplain wetlands and river systems, the possible contribution of surface water and groundwater flows from the watershed on the water quality of these wetlands and the river system are of substantial importance in this investigation.

2.2.1.2 Water Quality

‘Water quality’ is the term used to describe the physical, chemical, biological and aesthetic characteristics of water that determine its suitability for a multitude of different uses and for the protection of aquatic and semi-aquatic environments (DWAF, 1996a). Water quality is a critical factor that helps determine the health and integrity of wetland ecosystems (Reddy and Gale, 1994). Water quality assessment has also been considered and employed as an important water resource management tool over recent years (Antonopoulos *et al.*, 2001 in Sukdeo, 2010).

The use of water quality in determining ecosystem health has become protocol for several Catchment Management Areas (CMAs) in South Africa, to the extent that water quality testing and monitoring form vital components of Catchment Management Plans (Dickens *et al.*, 2003).

Poor water quality, resulting from pollution or contamination, not only qualifies as unsuitable for a multitude of human uses, but also proves unfit for the support of ecosystem functioning (DWAF, 1996a; DWAF, 1996b).

2.2.1.3 Water Pollution

Water pollution in South African wetlands has resulted in their eutrophication, their identification as notorious breeding grounds for human pathogens and disease, and their inability to support their biotic constituents; collectively resulting in ecosystem degradation and collapse (Gleick *et al.*, 2002; Carpenter *et al.*, 1998; Mitsch & Gosselink, 2011). The following consequences of uninhibited anthropogenic input into wetland ecosystems are of particular relevance to this study:

2.2.1.3.1 Nutrient pollution

Nutrients are essential for the growth of algae and vegetation. However, excessive amounts of nutrients, over-stimulate plant and algal growth, and decrease the amount of available oxygen for other dependent biota (Carpenter *et al.*, 2008). Eutrophication is most often the result of increased nutrient input, principally nitrogen and phosphorous, that are originally lacking in water-based environments (Carpenter *et al.*, 2008). According to Sukdeo (2010), eutrophication results in water quality decline and creates conditions that are unsuitable for the support of biota. Although natural sources of nitrogen and phosphorous may result in eutrophication of aquatic and semi-aquatic systems; the anthropogenic deposition of large quantities of these nutrients acts to exacerbate existing unfavourable conditions (Camargo & Alonso, 2006). Drainage from agricultural land and certain recreational activities which require the application of fertilisers and pest control products are amongst the potential human sources of nutrients.

2.2.1.3.2 Heavy and trace metal pollution

Metals can easily accumulate in wetland environments and can persist for extended periods of time (Greaney, 2005). They are able to accumulate in successive levels of the biological food chain and are known to have lethal effects on all inhabitants of an ecosystem (Canadell *et al.*, 2007). Heavy and trace metals exist in the natural environment in very low concentrations and excessive concentrations in wetland environments usually result from the deposition of waste generated by human activities (Coetzee, 1995; Zhang *et al.*, 2012). Examples include cadmium, copper, selenium, lead, mercury, and chromium; which are derived from industrial effluent (Hillel, 1998).

The persistence of high heavy and trace metal levels in wetland systems for prolonged periods of time will result in adversities materialising within the biotic component of wetland ecosystems; concomitant to the exposure of wetland plants and animals to physiochemical conditions that are unsuitable to their health and well-being (Greaney, 2005). All wetland-dependent and -reliant species, including humans, eventually experience the negative effects of utilising 'polluted' wetland resources (Coetzee, 1995; Zhang *et al.*, 2012). These include:

2.2.1.3.3 Water-related diseases

The issue of water-related diseases continues to be a subject of great concern in African and other developing countries alike, where pathogen-infested water bodies have staggeringly increased the number of diagnosed infection cases and mortality rates within communities (United Nations Environment Programme, 2006). As a consequence, significant time and money losses become the plight of affected nations (Cairncross & Valdmanis, 2006). Water contaminated by human or animal wastes containing pathogenic bacteria and viruses bring about diseases when consumed, or when made direct contact with (Gleick, 2002). Examples of such diseases include diarrhoea, cholera, typhoid and bacillary dysentery (Gleick, 2002). They are also incited by parasites living in intermediate organisms that inhabit contaminated water, or parasites with an insect vector which breed in water (Appleton *et al.*, 1995; Gleick, 2002). For instance, mosquitoes are the notorious carriers of malaria (Appleton *et al.*, 1995; Gleick, 2002). The prevalence of these diseases within national bounds is the culmination of a number of service delivery shortfalls, including a lack of reticulated sanitation, access to potable water and the release of insufficiently treated or untreated sewage from sewage treatment plants. (Appleton *et al.*, 1995; Macaskill, 2010).

2.2.1.3.4 Degradation of wetland ecosystems

When hydrological conditions in wetland environments change even slightly, the constituent biota respond with substantial changes in species composition, richness and ecosystem productivity (Mitsch & Gosselink, 2011). Considering that the physical and chemical requirements of wetland species differ, alterations to the physical and chemical characteristics of the system will affect each to varying extents (Wrona *et al.*, 2006). Hence, spikes or falls in concentration levels of the physical and chemical constituents of water may prove unfavourable to several existing species, resulting in their eventual disappearance and replacement by new species adapted to flourish in prevailing conditions (Mitsch & Gosselink, 2011). This affects species richness if the number of new species is actually fewer than the number of species lost, and as a result, the system experiences a decrease in its integrity as a habitat (Davies and Day, 1998 in Sukdeo). Furthermore, a decline in indigenous species coupled with uncontrolled human disturbance may act to increase the presence of exotic alien and troublesome biota within the environment (Ciruna *et al.*, 2004). A reduction in species diversity and an increase in diseases and parasitism represent other key impacts of a decline in water quality within water-based ecosystems (Malan and Day, 2002 in Sukdeo, 2010).

2.2.1.4 Water Quality Management

2.2.1.4.1 Water Quality Guidelines

Focus on the dire need to monitor the quality of water comprising aquatic and semi-aquatic ecosystems in South Africa, resulting from growing concerns regarding imminent water stress in an already water scarce country, and the costs associated with ecosystem collapse, has highlighted the importance of measuring water quality variables to identify trends, detect problems and apply suitable limits (Sukdeo, 2010). The management of water quality in South Africa is informed by the water quality guidelines developed by the Department of Water and Sanitation (DWS), referred to as DWAF before the year 2009, and as the Department of Water Affairs (DWA) from the year 2009 to 2014. These guidelines aim to enforce standards on water quality variables according to the activities for which the resource will be used (DWAF, 1996a). These guidelines address the management of water resources used for domestic, recreational, industrial and agricultural purposes, as well as the protection of marine environments, and aquatic- and semi-aquatic ecosystems (DWAF, 1996a). Of specific relevance to this study are the guidelines that have been developed to manage the quality of water comprising aquatic ecosystems (DWAF, 1996a) and water used for domestic purposes (DWAF, 1996b).

2.2.1.5 Wetland Sediment

Wetland sediment, often referred to as hydric soils, can be defined as soil that has been saturated, flooded or ponded for a prolonged period of time resulting in the dominance of anaerobic conditions (Tiner, 1999; Maltby, 2009). Wetland soils play a crucial role in the functioning of wetlands (Environmental Law Institute, 2002). They have a major influence on hydraulic conductivity, the availability of nutrients, groundwater, seed germination, the rooting and growth of wetland flora, and provide habitats for a number of wetland fauna (Environmental Law Institute, 2002).

2.2.1.5.1 Types of wetland sediment

There are two major classes of wetland soils, namely organic soils and mineral soils (Sprecher, 2001). The distinction is based on their source material and, in practice, by their difference in organic carbon levels (Sprecher, 2001).

Organic soils form from plant debris and are prevalent in wetland environments where the pace of decomposition is slower due to the high moisture content (Aber *et al.*, 2012). They are black in colour, porous, light in weight, and commonly referred to as “peats” or “mucks” (Sprecher, 2001). For soils to be categorised as organic they have to meet certain criteria, which include a minimum proportion of constituent organic carbon equivalent to 10%, and 200 mm of organic material within the upper 800 mm of soil, or that of any thickness extending from solid surface to rock or gravel (Mitsch & Gosselink, 2000 cited in Schwirzer, 2006).

Mineral soils are formed by the weathering of rock or material transported by wind, water, landslide or ice (Weaver, 1976). As a consequence, their constituents include differing amounts of sand, silt and clay (Weaver, 1976). Unlike organic soils, mineral soils are documented to consist of less than 10% organic carbon (Boettinger *et al.*, 2010). They comprise the majority of soils worldwide and can be found in wetlands and a multitude of other environments (Sprecher, 2001). Mineral soils in the field can be identified by its grittiness, stickiness or otherwise texture that resists compression (Sprecher, 2001).

2.2.1.5.2 Delivery to wetlands

According to Maltby and Barker (2009), sediments in wetlands derived from external sources, otherwise referred to as 'allochthonous sediment', are predominantly mineral in nature. Sediments derived from internal sources on the other hand, or 'autochthonous sediment', are organic sediments of biogenic or chemical origin, and are formed by the decomposition of plant and animal matter through *in situ* processes (Maltby and Barker, 2009).

Allochthonous sediment in wetland environments includes fluvial sediment, which originates in catchments and rivers, and is reflective of the catchment parent material and physiographic character (Abed, 2009). Various modes of transportation are utilised in facilitating the movement of river - and catchment - derived sediment, namely wash load, suspended load and bed load (Sukdeo, 2010). Apart from sediment within the wetlands themselves, sediments are in constant movement in and out of wetland systems *via* both fluvial and marine sources (Abed, 2009; Sukdeo, 2010).

With regards to this particular study, fluvial sediment from the adjacent river and catchment area are of extreme importance in assessing the contribution of catchment land use to the level of contaminants within the Lower uMngeni wetlands, and the resulting effects on wetland ecosystem health.

2.2.1.5.3 Hydric soil chemistry and sediment-water interactions

The chemistry of hydric soils are based on oxidation-reduction reactions that affect the properties and functions of hydric soils, and therefore often support the identification of hydric soils (Vepraskas & Faulkner, 2001). As sediments are moved from one environment to another, changes in their chemical composition occur to establish a state of equilibrium under conditions prevalent in the receiving environment (Sukdeo, 2010). Chemical elements within sediments experience a set of redox reactions in their new environment that differs from those reflecting conditions in their former location (Sukdeo, 2010). It is thus evident that oxidation-reduction reactions strongly influence the chemical properties and composition of sediments (Chapin (III) *et al.*, 2011).

The major chemical processes occurring in hydric soils including denitrification, and the production of mottled soil colours, hydrogen sulphide and methane gases, are created by reducing reactions, mainly those utilising electron acceptors other than oxygen (Chapin (III) *et al.*, 2011). These reactions are known to have an effect on soil colour and water quality (Vepraskas & Faulkner, 2001). There are four specific conditions required for soils to reach an anaerobic state and support the abovementioned reducing reactions (Vepraskas & Faulkner, 2001). These are (Meek *et al.*, 1986; Bouma, 1983 cited in Vepraskas & Faulkner, 2001): (1) the soil must be saturated or inundated to exclude the presence of oxygen; (2) organic tissue that can be oxidised or decomposed must be present in soil; (3) microbe communities to decompose organic matter must exist in soil; and (4) water should ideally be stagnant, or otherwise slow-moving, to prevent oxygen from being carried into the soil (Meek *et al.*, 1986; Bouma, 1983 cited in Vepraskas & Faulkner, 2001).

As stated by Burich (2007), the physical properties of individual sediment grains have a major influence on adsorption, desorption, ion exchange, transport and deposition. The grain size of the sediment, more specifically, is most responsible for metal interactions as it directly affects the surface area, cation exchange capacity and surface charge of a sediment particle (Horowitz, 1985). In line with this, studies have shown that sediment-water interactions that promote adsorption, desorption and ion-exchange, have a significant influence on sediment chemistry and composition (Sukdeo, 2010). Furthermore, other sediment properties that affect soil chemistry, and therefore sediment quality, include organic matter content and calcium carbonate content. This is explained further in the sub-section below.

2.2.1.5.4 Sediment-contaminant relationship

As stated by Barceló and Petrovic (2007), the contamination of aquatic and semi-aquatic sediment is largely influenced by sediment texture, particularly sediment grain size. Numerous investigations have revealed that the content of heavy metals and organic contaminants in soil are principally governed by grain size (Poleto & Charlesworth, 2010).

The various sediment grain sizes occurring in an aquatic and semi-aquatic system exhibit neither the same elemental concentrations, nor equal propensities to retain metal contaminants (Miller & Orbock Miller, 2007). A contributing factor is the differential surface areas of the various particle types, which is such that surface area increases with decreasing grain size (Poleto & Charlesworth, 2010). The larger surface area of finer sediments provides a larger platform for metals to accumulate (Miller & Orbock Miller, 2007). Therefore, the finer the sediment grain, the higher will be the level of contamination (Barceló and Petrovic, 2007). Furthermore, studies have shown that both organic matter content and moisture content are negatively correlated to sediment grain size such that a decrease in sediment grain size is accompanied by an increase in organic matter and moisture content (Frei, 2008; Um *et al.*, 2009). Therefore, by determining the average sediment grain size in phi units within each sediment lamina comprising a sediment core, comparisons between laminae regarding their physical and chemical characteristics can be made (Abed, 2009).

Considering that the adsorption of contaminants, however, is highly dependent on the physiochemical factors of the system, the fixation of pollutants to sediment particles are not permanent as their remobilisation into the water column may result from the various physical, chemical and biological processes occurring in the system (Barceló and Petrovic, 2007). The properties of sediment, however, have a large influence in this regard. For instance, the calcium carbonate content of sediment can be used to determine the system's susceptibility to contamination, resulting from changes in prevailing conditions in the system. According to Frei (2008), calcium carbonate is the predominant factor affecting the "buffer capacity" of sediments which acts to neutralise hydrogen and hydroxyl ions resulting from the entry of anthropogenic contaminants into the system, or the deposition or re-suspension of anoxic sediment. In doing so, it prevents the dominance of acidic conditions in a system and therefore the release of metals from sediment. (Frei, 2008) Hence, ecosystems containing sediment characterised by low amounts of calcium carbonate face a higher risk of contamination.

The bio-availability of contaminants, including heavy and trace metals, resulting from prevalent conditions in an ecosystem, poses a threat to the well-being of inhabitant organisms (Landajo *et al.*, 2004 In Sukdeo, 2010; Chatterjee *et al.*, 2006 in Sukdeo, 2010; Jordao *et al.*, 2007 in Sukdeo 2010). Sediments are thus renowned as the most important sinks, accumulators, carriers and possible future source of metals and other pollutants in water-based environments (Chatterjee *et al.*, 2006).

According to Lewis and McConchie (1994), sediments are useful indicators of pollution levels in water-based environments, and thus assist in identifying appropriate methods of pollution control. Sediments also provide information regarding the inter- and intra-component transfer of pollutants (between sediments, water and biota) within an aquatic and semi-aquatic ecosystem and help predict future levels of pollution induced by various anthropogenic activities (Lewis and McConchie, 1994). The analysis of wetland sediment may also assist in identifying the potential sources of pollution by informing inferences regarding the contribution of natural sources verses anthropogenic sources of contaminants (Chatterjee *et al.*, 2006)

2.2.1.5.5 Sediment Quality

Determination of the concentration of contaminants within sediments comprising a particular environment, and comparing them to limits or criteria specified in sediment quality guideline documents, can be used to establish sediment quality in an ecosystem (Lee and Jones-Lee, 1993). However, in South Africa, a set of guidelines governing sediment quality, as well as specific background concentration values for the uMngeni catchment, have either not been developed or are unavailable (Orr, 2008 in Sukdeo, 2010). Hence, for purposes of this study, background values or 'Clarke" values have been utilised.

Clarke values serve as background values for the mean elemental composition of the Earth's crust which are determined based on 'geochemical backgrounds' (Martinez *et al.*, 2007 in Sukdeo, 2010). Geochemical backgrounds refer to the normal abundance of elements in barren Earth material, and are primarily used to differentiate between normal element concentrations and anomalies which represent deviations from normal concentrations (Yousif, 2007). Clarke or background values for a specific region or geological type are used to determine: (1) the presence and concentration of pollutants in sediments; (2) if they are derived from natural or anthropogenic sources; and (3) if they occur within acceptable limits or cause pollution (Chenhall *et al.*, 2004 in Sukdeo, 2010; Martinez *et al.*, 2007 in Sukdeo, 2010).

Background values are important tools in the assessment of sediment quality, and thus enrichment detection or sediment pollution, in aquatic and semi-aquatic systems (Harikumar and Jisha, 2010). Common methods employed to determine the level of pollution in such systems require that background values be factored into the relevant index equations

(Harikumar and Jisha, 2010). These indices include contamination factors (CF), enrichment factors (EF) and the pollution load index (PLI) (Chatterjee *et al.*, 2006; Harikumar and Jisha, 2010). All the aforementioned indices will be utilised for purposes of this study.

In order to effectively manage wetlands such that their physiochemical environments create conditions that are suitable for the support of wetland organisms, it is essential that wetlands be classified (Uys, 2004). Classification provides information that is particularly relevant to the wetland systems of interest and thereby assists in identifying appropriate management actions for implementation at ground-level (Uys, 2004).

2.3 Wetland Classification

In order to give momentum to priority actions that promote the national endeavour to protect wetland systems, it was critical that a wetland classification system encompassing the diversity of wetland types in South Africa be developed (Uys, 2004). Classification systems have been developed internationally (Cowardin *et al*., 1979; Brinson, 1993), within the African continent and nationally (Dini and Cowan, 2000; Kotze *et al*., 2001; Kotze *et al.*, 2005; Ewart-Smith *et. al*, 2006) to guide the categorisation of wetlands (Schwirzer, 2006; Botes, 2009).

With the development of a National Wetland Inventory (NWI) by the South African National Biodiversity Institute (SANBI), the need to apply a suitable wetland classification system to this newly invented NWI stood at the fore of environmental objectives on a national scale (SANBI, 2009). Recognising the need for such a classification system, SANBI in association with other key organisations and renowned wetland specialists, produced a National Wetland Classification System (NWCS) in 2009 (SANBI, 2009). This classification system utilises the hydrogeomorphic approach of classification, and as such distinguishes primary wetland units based on hydrological and geomorphological characteristics (Uys, 2004). Such classification criteria directly contrasts the more traditional classification system developed by the United States Fish and Wildlife Service (Cowardin *et al*., 1979), which identifies and differentiates wetlands on a very broad level based on structural features; such as size, depth, vegetation cover and the presence of surface water (Schwirzer, 2006). Given this, experts presume that applying the Cowardin-based approach for the management and conservation of wetlands in South Africa will lack in overall effectiveness, as the significance attached to hydrology and geomorphology as fundamental factors in wetland systems are largely under-estimated (Botes, 2009).

Ewart-Smith *et al.* (2006) recognised that apart from methods, like the Cowardin-based approach, that were internationally developed and adapted for implementation in South Africa, a number of classification systems also existed within national bounds (Botes, 2009). These existing classification systems, however, were structurally-based, or developed for specific regions in South Africa, and thus did not consider the broad range of wetland types occurring throughout the country (Schwirzer, 2006). Ewart-Smith *et al.* (2006), subsequent to Kotze *et al.* (2005), created a prototype classification system founded on principles of the hydrogeomorphic approach at higher levels of classification, which attempts to incorporate structural features at finer levels (Botes, 2009). Realisation of the necessity to further develop, rigorously test and refine this system spurred the collaborative initiative of creating the updated NWCS in 2009 (SANBI, 2009). Characteristics of the different hydrogeomorphic (HGM) types included in the National Wetland Classification System are illustrated in Table 2.1 below. Since then, the NWCS was applied to wetlands in the National Wetland Inventory's map through the National Freshwater Ecosystem Priority Areas (NFEPA) project which involved automation of the classification system (SANBI, 2009). The resultant map contains all wetlands in South Africa mapped to date and their respective classifications.

Based on criteria prescribed by the NWCS (2009), the wetlands of particular interest situated along the Lower uMngeni River can be classified as 'floodplain wetlands'. According to the NWCS, floodplain wetlands are those that are mostly flat or otherwise gently sloping wetland areas adjacent to, and formed by, a Lowland or Upland Floodplain River; and which are subject to periodic inundation by overtopping of the channel bank (Uys, 2004). These wetlands receive water and sediment inputs mainly *via* overtopping of a major channel, as well as some overland or subsurface flow from adjacent valley side-slopes (Macfarlane *et al.*, 2008). Water movement in these wetlands are generally horizontal and bi-directional in the form of diffuse surface flow and interflow (SANBI, 2009). Infiltration and evaporation within floodplain wetlands can also be significant (Uys, 2004; Macfarlane *et al.*, 2008). Floodplain wetlands within the Lower uMngeni sub-catchment occur within a "Valley Floor" landscape setting (Macfarlane *et al.*, 2008). They also fall under the broader landscape category of 'Coastal Plains' located in the 'Natal Coastal Bioregion', which altogether constitute what is referred to as an 'Inland System' by the NWCS (SANBI, 2009).

According to DWAF (2004), wetland classification provides a basis for guiding the appropriate wetland assessment activities (where assessment is taken to be the preliminary identification of the health status of wetlands) and threats to their current conditions.

Table 2.1….\

Table 2.1: Characteristics of the different Hydrogeomorphic (HGM) Types included in the proposed National Wetland Classification System (Macfarlane *et al*., 2008)

Primary (Level 4A) HGM Type*	Secondary (Level 4B) HGM Units (Longitudinal Zonation / Landform)	Landscape setting/s	Dominant hydrological characteristics			Dominant hydrodynamics
			Inputs	Throughputs	Outputs	
CHANNEL	Mountain Headwater Stream	Slope	Overland flow from catchment runoff, concentrated surface flow from upstream channels and tributaries, diffuse surface flow from an unchannelled upstream drainage line (i.e. an unchannelled valley-bottom wetland), seepage from adjacent hillslope or valleyhead seeps, and/or groundwater (e.g. via in-channel springs)	Concentrated surface flow	Concentrated surface flow, generally, but can be diffuse surface flow (e.g. where a channelled valley-bottom wetland becomes an unchannelled valley-bottom wetland because of a change in gradient or geological control)	Horizontal: unidirectional
	Mountain Stream	Slope / Valley floor				
	Transitional River	Slope / Valley floor				
	Upper Foothill River	Valley floor				
	Lower Foothill River	Valley floor				
	Lowland River	Valley floor / Plain				
	Rejuvenated Bedrock Fall (gorge)	Slope / Valley floor				
	Rejuvenated Foothill River	Slope / Valley floor				
	Upland Floodplain River	Valley floor / Plain (specifically a plateau)				
CHANNELLED VALLEY-BOTTOM WETLAND	Valley-bottom flat	Valley floor	Overland flow from adjacent valley-side slopes, lateral seepage (interflow) from adjacent hillslope seeps, channel overspill during flooding	Diffuse surface flow, temporary containment and storage of water in depressional areas, possible short-lived concentrated flows during flooding events	Diffuse surface flow and interflow into adjacent channel, infiltration and evaporation (particularly from depressional areas)	Horizontal: bidirectional; limited vertical: bidirectional (mostly in depressions)
	Valley-bottom depression	Valley floor				
UNCHANNELLED VALLEY-BOTTOM WETLAND	Valley-bottom flat	Valley floor / Plain	Concentrated or diffuse surface flow from upstream channels and tributaries; overland flow from adjacent valley-side slopes (if present); lateral seepage from adjacent hillslope seeps (if present); groundwater	Diffuse surface flow, interflow, temporary containment and storage of water in depressional areas, possible short-lived concentrated flows during high-flow events	Diffuse or concentrated surface flow, infiltration and evaporation (particularly from depressional areas)	Horizontal: unidirectional; limited vertical: bidirectional (mostly in depressions)
	Valley-bottom depression	Valley floor / Plain				
FLOODPLAIN WETLAND	Floodplain flat	Valley floor / Plain	Channel overspill during flooding (predominantly), but there could also be	Diffuse surface flow, interflow, temporary containment and	Diffuse surface flow and interflow into adjacent channel,	Horizontal: bidirectional; limited vertical: bidirectional
	Floodplain depression	Valley floor / Plain				

Primary (Level 4A) HGM Type*	Secondary (Level 4B) HGM Units (Longitudinal Zonation / Landform)	Landscape setting/s	Dominant hydrological characteristics			Dominant hydrodynamics
			Inputs	Throughputs	Outputs	
			some overland flow from adjacent valley-side slopes (if present) and lateral seepage from adjacent hillslope seeps (if present)	storage of water in depressional areas, possible short-lived concentrated flows during flooding events	infiltration and evaporation (particularly from depressional areas)	(mostly in depressions)
DEPRESSION (EXORHEIC, with channelled inflow)	n/a	Slope / Valley floor / Plain / Bench	Precipitation, concentrated and (possibly) diffuse surface flow, interflow, groundwater	Containment and storage of water, slow through-flow	Concentrated surface flow	Horizontal: unidirectional; vertical: bidirectional
DEPRESSION (EXORHEIC, without channelled inflow)	n/a	Slope / Valley floor / Plain / Bench	Precipitation, diffuse surface flow, interflow, groundwater	Containment and storage of water, slow through-flow	Concentrated surface flow	Horizontal: unidirectional; vertical: bidirectional
DEPRESSION (ENDORHEIC, with channelled inflow)	n/a	Slope / Valley floor / Plain / Bench	Precipitation, concentrated and (possibly) diffuse surface flow, interflow, groundwater	Containment and storage of water	Evaporation, infiltration	Vertical: bidirectional
DEPRESSION (ENDORHEIC, without channelled inflow)	n/a	Slope / Valley floor / Plain / Bench	Precipitation, diffuse surface flow, interflow, groundwater	Containment and storage of water	Evaporation, infiltration	Vertical: bidirectional
FLAT	n/a	Plain / Bench	Precipitation, groundwater	Containment of water, some diffuse surface flow and/or interflow	Evaporation, infiltration	Vertical: bidirectional; limited horizontal: multidirectional
HILLSLOPE SEEP (with channelled outflow)	n/a	Slope	Groundwater, precipitation (perched)	Diffuse surface flow, interflow	Concentrated surface flow	Horizontal: unidirectional
HILLSLOPE SEEP (without channelled outflow)	n/a	Slope	Groundwater, precipitation (perched)	Diffuse surface flow, interflow	Diffuse surface flow, interflow, evaporation, infiltration	Horizontal: unidirectional
VALLEYHEAD SEEP	n/a	Valley floor	Groundwater, diffuse surface flow, precipitation	Diffuse surface flow, interflow	Concentrated surface flow	Horizontal: unidirectional

* For completeness, in this list a distinction is also made between *depressions* and *hillslope seeps* with different ***drainage (outflow and inflow)*** characteristics, as recorded at Levels 4C and 4D of the proposed NWCS (the drainage criteria are not applicable to other HGM Types).

2.4 Wetland Health and Integrity

The ratification of environmental legislation and policy pertaining to the protection and conservation of South African wetlands initially prompted a preoccupation with ensuring that the areal extent of wetlands are minimally diminished, if not completely undisturbed (Cronk, 2001). The quantified impacts on wetland areas documented in monitoring reports were therefore based solely on the acreage of wetland loss (Cronk, 2001). However, later recognition of the accelerated pace at which existing wetlands are being degraded highlighted the importance of maintaining the health and functionality of wetland systems (Horwitz, 2012). This impelled an increased focus on assessing the quality or condition of wetlands rather than simply monitoring changes in their size and shape (Cronk, 2001; Horwitz, 2012).

According to the Macfarlane *et al.* (2008), wetland health can be defined as “a measure of the deviation of wetland structure and function from the wetland’s natural reference condition”. The reference condition of a wetland refers to the wetland’s state of health prior to adulteration by any form of human disturbance or activity that may act to transform it such that the system no longer functions at its optimum (Uys, 2004). Considering this, a healthy wetland ecosystem is one fully capable of carrying out its ecological and economic functions as a result of the interaction between its physical, chemical and biological parameters and the status of each (Mitsch & Gosselink, 2000 cited in Wray & Bayley, 2006). A healthy wetland is thus well-equipped to support biological communities and is characterised by physical and chemical attributes similar to those of natural habitats within the same region (Mitsch & Gosselink, 2000 cited in Wray & Bayley, 2006).

Wetland environments, however, are disturbed by a multitude of human activities that cause ecosystem disequilibrium and declining wetland health (Horwitz, 2012), as discussed in the section that follows.

2.5 Factors that Pose a Threat to Wetland Health and Integrity

Although scientific studies indicate that natural factors, such as sea level rise and climate change, are likely to impact on wetland ecosystem functioning; anthropogenic factors undoubtedly have the most significant influence on the health of such environments (Dickens *et al.*, 2003).

Anthropogenic factors are known to place growing pressure on wetland ecosystems, resulting in their deterioration and eventual destruction (Horwitz, 2012). The most influential anthropogenic activities and their resulting socio-economic and environmental impacts are illustrated in Figure 2.2 below. A lack of awareness and education concerning the benefits and services provided by wetland systems promotes indifferent human attitudes and behaviour towards wetland resources, and therefore the continuation of unsustainable practices (Dickens *et al.*, 2003). In the present age of industrialisation, short term economic benefits are prioritised and the associated external costs incurred by the environment and human well-being are largely overlooked (Uys, 2004; Dickens *et al.*, 2003). This compounded by a lack of local institutional capacity; the poor enforcement of legislation; and the lack of development and implementation initiatives aligned with existing policies, have cumulatively attached a 'dire' status to the need for effective wetland conservation and protection in South Africa (Dickens *et al.*, 2003).

Apart from the above, burgeoning increases in human population coupled with urbanisation, industrialisation and the continuation of poor agricultural practices have placed increased pressure on wetland ecosystems, compromising their health and longevity (Sahu & Choudhury, 2005). This is supported by Kotze *et al.* (1995), who state that the most dominant direct cause of wetland destruction is the loss of wetland area through draining or filling for purposes of human settlement, agriculture and silviculture. The principal causes of wetland degradation include erosional degradation, dam and road construction, afforestation, mining, water abstraction and the disposal of solid and toxic wastes (Kotze *et al.*, 1995). The most common indirect causes of wetland loss, on the other hand, are the anthropogenic input of nutrients and other contaminants into wetland ecosystems, and invasion by alien species (Zedler, 2004). Of all the above-mentioned factors, the off-site causes of wetland destruction are most influential due to the large spatial extent of catchment areas in relation to identified wetlands systems (Kotze *et al.*, 1995; Dickens *et al.*, 2003).

Disturbances may be measured directly by determining the response of wetland systems to changes brought about by such anthropogenic influences (Mitsch & Gosselink, 2000 cited in Wray & Bayley, 2006). Regarding the numerous pathways of wetland disturbance, it is neither feasible nor convenient to measure all potential disturbances to the environment and their possible impacts on ecosystem functioning (Mitsch & Gosselink, 2000 cited in Wray &

Bayley, 2006). Alternatively, freshwater ecology and wetland specialists utilise key parameters as indicators in assessing wetland health (Uys, 2004).

Figure 2.2….\

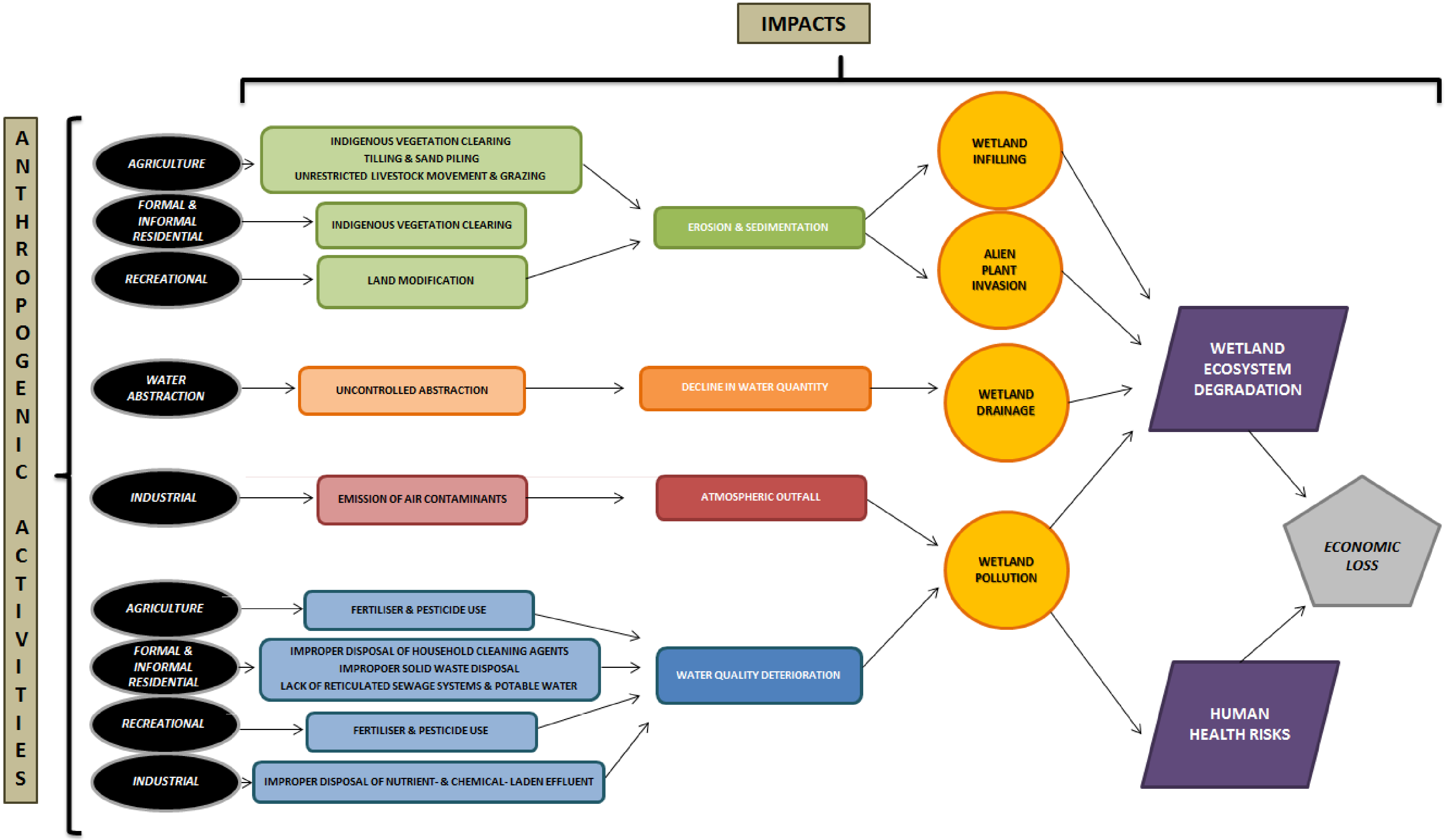


Figure 2.2: Anthropogenic activities and the resulting environmental and socio-economic impacts

2.6 Assessment of Wetland Health and Integrity

Methods have been established both nationally and internationally which may be used to determine the health of wetlands (Uys, 2004). For instance, the United States Environmental Protection Agency (USEPA) developed a system that assists in ascertaining wetland health by undertaking bio-assessments (Uys, 2004). This system is based on the assumption that biota inhabiting wetland ecosystems have occupied these environments for thousands of years and are thus resilient to changes induced by external factors. Hence, it is expected that changes to biotic communities resulting from varying intensities of human disturbance can be predicted (Botes, 2009). However, the most significant disadvantage of utilising this system is that it was created specifically for the North American context, and therefore its application in other parts of the world may produce findings that are questionable (Botes, 2009).

Another approach to wetland health assessment developed by Brinson and Rheinhardt (1996) is the hydrogeomorphic approach, which utilises reference wetland functions to define standards and goals for wetland creation and restoration purposes (USEPA, 2002). Although this approach is widely applied, its application within South Africa may be limited by the lack of data on wetland reference sites (Botes, 2009). However, this may change in the near future, with the many detailed studies being undertaken on the origin and evolution of South African wetlands in recent years (Botes, 2009).

Although more than one system has been developed within national bounds for the assessment of wetland health, the WET-Health tool is by far the most comprehensive and widely accepted (Botes, 2009). The success of this tool is linked to its focus on the underlying hydrological, geomorphological and biological processes that support wetland environments and their constituent species, rather than biotic or indicator communities in isolation (Macfarlane *et al.*, 2008). The tool thus allows for the assessment of these three components in separate modules and measures the deviation of current conditions from the pristine state (Macfarlane *et al.*, 2008). A scoring system is used to grade a number of attributes related to impacts and indicators of human activity relevant to each of the three components, which are thereafter drawn together in a highly structured way to produce final health scores for each (Macfarlane *et al.*, 2008). In addition, this tool further enables the estimation of the likely trajectory of change in wetland condition in the short-term (Botes, 2009).

This study may therefore serve to provide information on the current status of the physio-chemical environment of wetlands within the Lower uMngeni sub-catchment, through the measurement of specific physical and chemical indicators. The findings can then be used in future studies to inform further investigations undertaken through utilisation of the WET-Health tool.

2.7 Indicators of Wetland Health and Integrity

Indicators ideally display a cause-and-effect response and provide a general description of wetland health (Mitsch & Gosselink, 2000 cited in Wray & Bayley, 2006). The use of indicators to evaluate the ecological well-being of wetlands and the extent of wetland degradation prove to be both time- and cost-effective as they provide a diagnosis on wetland health without requiring the measurement of numerous processes and parameters (Mitsch & Gosselink, 2000 cited in Wray & Bayley, 2006).

To assess the disturbance or overall health of wetlands; physical, chemical or biological indicators may be used (Sims *et al.*, 2013). For purposes of this study, specific water and sediment quality parameters will be used to make deductions relating to the health of selected wetlands in the Lower uMngeni sub-catchment.

According to Wray and Bayley (2006), water and sediment chemistry parameters are useful in indicators of wetland health. Chemical parameters of particular interest in this study are listed and described below. Where relevant, the potential impacts of too high or too low a concentration of these chemicals on wetland biota and humans are explained.

2.7.1 Nutrients and other Chemical Characteristics

2.7.1.1 Ammonium Ions

Ammonia is a common pollutant that contributes significantly to the eutrophication of wetland ecosystems (DWAF, 1996a). Common anthropogenic sources of ammonia include fish-farm effluent, sewage discharge, industrial discharge, manufacture and use of explosives and the atmospheric deposition of ammonia (DWAF, 1996a). The increased concentration of ammonia in wetland systems adversely impacts wetland organisms by causing respiratory disorders, reducing hatching success, reducing growth rate and morphological development, and causing pathological changes in tissue (DWAF, 1996a). High levels of ammonia in water-based environments may also result from the presence of organic wastes in waters, which are known to have adverse effects on human health, especially that of children, if ingested (Sukdeo, 2010).

2.7.1.2 pH

pH is basically a measure of the hydrogen ion conductivity of a water sample (DWAF, 1996a). As the concentration of hydrogen ions in a water sample increases, pH decreases and the sample becomes more acidic (DWAF, 1996a). A decrease in hydrogen ions in a sample however results in an increase in pH, and a more alkaline or basic sample (DWAF, 1996a).

Anthropogenic activities that may act to alter pH concentrations in wetlands include industrial, mining, and pollution emitting activities that generally cause 'acid rain' (DWAF, 1996a). Effects of changes in pH on aquatic organisms include alterations of the ionic and osmotic balances of individual organisms, slow growth and reduced fecundity (DWAF, 1996a). Changes in the availability of toxic substances such as ammonia and aluminium are amongst other indirect impacts of pH alterations (DWAF, 1996a).

2.7.1.3 Phosphorous (P)

Elevated concentrations of phosphorous in wetland environments may result from point-source discharges such as domestic and industrial effluents, and non-point discharges including atmospheric precipitation, urban runoff and drainage from agricultural land (DWAF, 1996a). The most poignant effect of elevated phosphorous levels in a wetland system is the resultant increased productivity of the system (DWAF, 1996a). Reduced species diversity, algal proliferation and an increased presence of nuisance and toxic species are direct indications of high phosphorous levels in wetland environments (DWAF, 1996a).

2.7.1.4 Sulphate Ions

Sulphate is a common constituent of water (DWAF, 1996b). The anthropogenic sources of sulphates in wetland environments include acid mine drainage, industrial wastes and 'acid rain' events which result from sulphur dioxide emissions (DWAF, 1996b). The consumption of low concentrations of sulphate ions do not pose health risks to people, and is therefore not considered a toxic ion (DWAF, 1996b). High concentrations of sulphate ingested by humans, however, may cause diarrhoea (DWAF, 1996b). In excess, sulphates form sulphuric acid, which reduces pH and has adverse effects on humans and biota (Brijlal, 2005).

2.7.2 Heavy Metals, Trace Metals and Other Parameters

2.7.2.1 Aluminium (Al)

Aluminium, that was once considered a ‘non critical’ element, is now becoming a growing concern due to the detrimental effects of elevated levels in the environment (DWAF, 1996a). Water treatment plants, aluminium smelters, paper industries, and leather and textile industries, all use aluminium in either their processes or products, and are therefore amongst the major sources of aluminium that reach wetlands and aquatic and semi-aquatic ecosystems (DWAF, 1996a). Excessive concentration of bio-available aluminium proves toxic to most aquatic-dependent and –reliant species (DWAF, 1996a). For instance, species of fish experience ionic and osmotic imbalances, respiratory and calcium metabolism problems, and neuromuscular function if exposed to high levels of aluminium (DWAF, 1996a).

2.7.2.2 Arsenic (As)

Arsenic is present in minute amounts in the environment due to its poor solubility, and occurs in both an organic and inorganic form, the latter of which is most toxic (Sukdeo, 2010). Naturally occurring levels of arsenic in the environment are produced by volcanic activity and the weathering of rocks (Boyd, 2000). However, the main sources of elevated levels of arsenic in ecosystems are usually derived from anthropogenic activity, mainly industrial processes (DWAF, 1996a). Humans are relatively more sensitive to arsenic than aquatic organisms such that the ingestion of water containing high levels of arsenic may result in hyperkeratosis, hyperpigmentation as well as cancer of the skin and other internal organs (Sukdeo, 2010).

2.7.2.3 Barium (Ba)

As stated by Lenntech (2009), barium is the 14th most abundant element in the earth’s crust and is capable of reacting with numerous non-metallic elements thereby forming toxic substances. Aquatic organisms inhabiting environments that have a high concentration of water soluble barium are likely to experience a decline in health (Sukdeo, 2010). Exposure to barium also results in a number of adverse human health effects including increases in blood pressure, stomach irritation and swelling of internal organs (Sukdeo, 2010).

2.7.2.4 Calcium (Ca)

As stated by DWAF (1996b), calcium is an essential element for all living organisms and comprises an important part of the bony skeleton in mammals. There is no reliable information supported by scientific studies that indicate that calcium may have negative impacts on human health (DWAF, 1996b). However, it is evident that calcium affects the use and efficiency of household and industrial appliances (DWAF, 1996b).

2.7.2.5 Chromium (Cr)

Given its relative scarcity, the probability of occurrence of chromium in aquatic and semi-aquatic ecosystems are low (DWAF, 1996a). Chromium occurs in various forms, the most common of which is chromite, characterised by a trivalent state of chromium (DWAF, 1996a). Chromium salts are largely utilised by leather industries, manufacturers of paints, dyes, explosives, ceramics and photography (DWAF, 199a). Elevated levels of chromium in aquatic and semi-aquatic ecosystems result in reduced growth in fish, and have toxic effects on a range of other organisms, including invertebrates, daphiniids and green algae (DWAF, 1996a).

2.7.2.6 Cobalt (Co)

Cobalt is naturally present in mineral water and surface water in low concentrations, and is beneficial to humans in small amounts as it forms part of vitamin B12 (Ackermann and Sommer, 1988; Sukdeo, 2010). However, given its propensity to accumulate in the environment, high levels of cobalt in water-based ecosystems is a likely result of waste deposition by human activities, such a mining. Human exposure to significant levels of cobalt may result in vomiting and nausea, vision impairments, heart problems and thyroid damage (Lenntech, 2009).

2.7.2.7 Copper (Cu)

As stated by DWAF (1996a), copper is one of the most widely used metals. Metallic copper occurs in four oxidation states, and although generally insoluble in water, copper salts such as cupric or cuprous ions are highly soluble (DWAF, 1996a). The major anthropogenic sources of copper include corrosion of brass and copper pipes by acidic waters, sewage treatment plant effluents, fungicides and pesticides, liquid effluent, atmospheric fallouts from mining, smelting and refining industries, and coal-burning (DWAF, 1996a). The adverse effects of increased copper levels in aquatic and semi-aquatic environments include brain damage in mammals, reduced nitrogen-fixation by blue algae, and negative impacts on species composition and richness (DWAF, 1996a).

2.7.2.8 Dissolved Oxygen (DO)

It is vital that an adequate amount of dissolved oxygen in aquatic and semi-aquatic ecosystems is maintained, considering that aerobic aquatic organisms depend on it for respiration, and thus their very functioning and survival (DWAF, 1996a). According to DWAF (1996a), oxygen is moderately soluble in water. A decrease in oxygen levels in aquatic and semi-aquatic systems can be attributed to the re-suspension of anoxic sediments, the release of anoxic bottom water, the presence of large quantities of organic matter and high turbidity levels (DWAF, 1996a). Adverse changes in feeding and reproduction patterns, reduced growth, and physiological stress are examples of the negative effects brought about by reduced dissolved oxygen levels in water-based ecosystems (DWAF, 1996a).

2.7.2.9 Iron (Fe)

Apart from their natural deposition in wetland environments, iron is released by human activities such as the burning of coke and coal, acid mine drainage, mineral processing, sewage, landfill leachates, corrosion of iron and steel, and various industries that use iron in their processes (DWAF, 1996a). Documentation of the acute and chronic effects of increased iron concentrations on wetland organisms is limited (DWAF, 1996a). However, considering that associated impacts may prove deleterious to biota, a tentative guideline has been provided by DWAF regarding its acceptable concentrations in water-based environments (DWAF, 1996a).

2.7.2.10 Lead (Pb)

Lead exists in several oxidation states (DWAF, 1996a). The major sources of lead in wetland environments are derived from anthropogenic sources and include precipitation of lead emissions; industrial and municipal wastewater discharges; mining, milling and smelting of lead metals; and the combustion of fossil fuels (DWAF, 1996a). Lead, being a common toxic trace metal, has several negative impacts on wetland organisms (DWAF, 1996a). For instance, it interferes with haemoglobin synthesis, inhibits some enzymes involved in energy metabolism, and affects membrane permeability by displacing calcium at functional sites resulting in bone deformities in certain fish (DWAF, 1996a). It also has adverse impacts on humans, more especially children and unborn babies (DWAF, 1996b). In children, the negative impacts include behavioural changes and neurological impairment, whilst in adults the effects are less pronounced and take the form of anaemia and lead colic (DWAF, 1996b).

2.7.2.11 Magnesium (Mg)

Magnesium is an important macronutrient required by plants and animals in wetland systems (Chapman, 2002 in Sukdeo, 2010). Human activities resulting in the increase of magnesium concentrations in wetland environments include fertiliser and chemical industries (DWAF, 1996a). Low levels of magnesium in wetland environments result in skeletal deformities and reduced reproductive capabilities in vertebrates, whilst excessive levels may result in disturbances to metabolism and the central nervous system (DWAF, 1996a).

2.7.2.12 Manganese (Mn)

As stated by DWAF (1996a), Manganese occurs in a number of ores (DWAF, 1996a). In wetland systems manganese is usually found in the form of salts or minerals, commonly in association with iron compounds (DWAF, 1996a). Industrial discharges and acid mine drainage are the main anthropogenic sources of manganese found in wetland ecosystems (DWAF, 1996a). High levels of manganese are toxic and may lead to disturbances of the central nervous system in wetland fauna (DAWF, 1996a).

2.7.2.13 Nickel (Ni)

Nickel is present at naturally high concentrations in the environment, which can be further elevated by the input of anthropogenic wastes from mining, refineries, Nickel-Cadmium battery manufacturing and fossil fuel industries (David, 2006 in Sukdeo, 2010). Increased levels of nickel in wetland environments diminish algae growth, and result in higher mortality rates of primary producers, primary consumers and secondary consumers (David, 2006 in Sukdeo, 2010). Studies have also shown that prolonged exposure to high levels of nickel may result in carcinoma of various vital organs (Hardy *et al.*, 2008 in Sukdeo, 2010).

2.7.2.14 Total Dissolved Solids (TDS)

Total Dissolved Solids (TDS) is simply a measurement of total inorganic salts, organic compounds and other dissolved solids in water (DWAF, 1996a). The presence of total dissolved solids or salts in excess of the ‘normal’ quantities in wetland ecosystems acts to deteriorate water quality (DWAF, 1996a). Changes in the concentration of TDS affects aquatic organisms to varying degrees depending on their individual abilities to maintain the balance of water and dissolved ions in their cells and tissues (DWAF, 1996a). Individual ions constituting TDS also have physiological effects on aquatic organisms (DWAF, 1996a).

2.7.2.15 Vanadium (V)

Vanadium is a soft, white, ductile metal which is resistant to corrosion and occurs in several oxidation states (DWAF, 1996b). Vanadium salts are highly soluble and readily taken up by living organisms. In some animals, vanadium can cause several neurological effects (Lenntech, 2009). Symptoms of vanadium toxicity are conjunctivitis, rhinitis, a sore throat and a persistent cough (DWAF, 1996b; Lenntech, 2009). Exposure to high levels may result in chronic bronchitis, paralysis and adverse effects on the liver and kidneys (DWAF, 1996b; Lenntech, 2009).

2.7.2.16 Zinc (Zn)

Zinc is a metallic element which is an essential micronutrient for all living organisms (DWAF, 1996a). Both soluble and insoluble zinc salts occur in industrial wastes from metal galvanising, dye processing, paint and cosmetics, pharmaceuticals, and fertiliser and insecticide industries (DWAF, 1996a). The effects of organism exposure to high levels of zinc include oedema and liver necrosis in fish fry; and reduced levels of shell growth, oxygen uptake and larval development in invertebrates (DWAF, 1996a). Elevated concentrations in consumed water also cause gastrointestinal disturbances in humans (DWAF, 1996b).

Considering the potential adverse effects on the health and well-being of wetland biota and dependent humans, resulting from increased nutrient and elemental concentrations in wetlands, it is essential that these ecosystems be protected. Furthermore, the multitude of functions performed by naturally occurring wetlands, as discussed in the section that follows, augments the need to prioritise the health and continued existence of these important systems.

2.8 The Importance of Wetlands: Their Values and Functions

The immense importance attached to wetland systems is corollary to the recent acknowledgement of their many present and possible future, functions and values which prove beneficial to society in a number of ways (Scodari, 1997). As postulated by Howe *et al.* (1991), wetland benefits refer to *"those functions, products, attributes, and services provided by the ecosystem that have values to humans in terms of worth, merit, quality or importance"* (cited in Collins, 2005:41). These benefits can be derived directly through consumption of wetland resources or outputs; or indirectly through wetland uses resulting from ecosystem services that these landscapes provide (Georgiou & Turner, 2012). Benefits obtained from wetland environments are specified in Table 2.2 below and expounded on in the sub-sections that follow.

Table 2.2: Wetland functions included in WET-EcoServices (adapted from Kotze *et al.*, 2005)

<table>
<tr><td rowspan="15">ECOSYSTEM SERVICES SUPPLIED BY WETLANDS</td><td rowspan="9">Indirect Benefits</td><td rowspan="8">Hydro-geochemical benefits</td><td colspan="2">Flood attenuation</td></tr>
<tr><td colspan="2">Streamflow regulation</td></tr>
<tr><td rowspan="5">Water quality enhancement benefits</td><td>Sediment trapping</td></tr>
<tr><td>Phosphate assimilation</td></tr>
<tr><td>Nitrate assimilation</td></tr>
<tr><td>Toxicant assimilation</td></tr>
<tr><td>Erosion control</td></tr>
<tr><td colspan="2">Carbon storage</td></tr>
<tr><td colspan="3">Biodiversity maintenance</td></tr>
<tr><td rowspan="6">Direct Benefits</td><td colspan="3">Provision of water for human use</td></tr>
<tr><td colspan="3">Provision of harvestable resources</td></tr>
<tr><td colspan="3">Provision of cultivated foods</td></tr>
<tr><td colspan="3">Cultural significance</td></tr>
<tr><td colspan="3">Tourism and recreation</td></tr>
<tr><td colspan="3">Education and research</td></tr>
</table>

2.8.1 Direct Benefits

Wetlands often serve as sources of water for domestic, agricultural and industrial use, which are accessed either by direct abstraction or *via* shallow wells (Dickens *et al.*, 2003). Apart from this, water from wetlands that move into underlying aquifers may serve as a source of water supply or move deeper into the groundwater system where it can provide a longer-term source of water for more distant communities (Dickens *et al.*, 2003). It may also serve as natural water resources that may be important for abstraction or the support of downstream ecosystems (Dickens *et al.*, 2003).

There are numerous plant and animal products that are harvested from wetlands throughout the world. These are harvested by people to be used as food, building materials, fuel, medicine, animal fodder and for craft-making (Day, 2009). They also produce harvestable products that travel beyond the system, such as migratory fish and birds, which benefit people at off-site locations or perform a variety of other functions in those ecosystems (Scodari, 1997). The socio-cultural value of wetlands is intrinsically linked to the benefits derived from the utilisation of its resources (Maltby & Barker, 2009). Communities that accommodate land peripheral to wetland systems, rural settlements in particular, depend largely on these environments; for subsistence, for water in areas where potable water is unavailable, and as a source of income (Maltby & Barker, 2009). Wetlands also provide a place for conducting cultural and religious ceremonies, and increase the beauty of landscapes from which many obtain spiritual upliftment (Dickens *et al.*, 2003). Wetlands may also be valued as sites of historical significance and constitute an important component of the nation’s cultural heritage (Dickens *et al.*, 2003).

Wetlands in South Africa are renowned worldwide for adding beauty to landscapes and their inhabiting communities of rare flora and fauna (Alexander *et al.*, 2000). It is for this reason that areas in close proximity to wetlands, such as the Greater Saint Lucia Wetland Park and the Isimangaliso Wetland Park, have become popular tourist destinations, contributing significantly to the growth of the South African economy (Dickens *et al.*, 2003). Wetlands are also widely used for recreational purposes, such as angling, fishing and canoeing (Day, 2009). Many wetlands are used as sites for scientific research, including monitoring, experimentation, reference, and the determination of long-term environmental trends (Day, 2009). They can also be used as a powerful educational tool, as they contain evidence of past and present processes, which may lead to a better understanding of human occupation, wetland species, communities and habitats (Dickens *et al.*, 2003).

2.8.2 Indirect Benefits

As stated by the North Atlantic Treaty Organization (2006), wetlands are considered to be both “the kidneys of the landscape”, due to their vital role in hydrological and chemical cycles, as well as “biological supermarkets”, given the array of biota and extensive food chains they support.

Wetlands act as sediment traps, and are therefore more widely considered as areas of sediment deposition as oppose to sediment sources (Mullins, 2012). The slowing down of water as it flows through riparian vegetation comprising wetland environments results in approximately 80-90% of sediment in water being removed (Schwirzer, 2006). This is commonly referred to as ‘sedimentation’ (Schwirzer, 2006). Wetlands are known to provide food and refuge to a plethora of plant and animal species; including birds, micro-organisms, invertebrates, fish, amphibians, reptiles and mammals (Aber *et al.*, 2012). Wetland ecosystems serve as conservancies in their own right, housing a number of rare and endemic species (Brijlal, 2005). Hence, the disturbance of wetland environments endangers constituent biota, compromises habitat integrity, and may cause the irreplaceable loss of important ecosystems if not properly managed (Schwirzer, 2006).

Wetlands are most renowned for their vital role in the hydrological cycle (Begg, 1986; Bullock & Acreman, 2003). Wetlands act to dampen the intensity of devastation accompanying flood events by serving as storage areas for high volumes of water and sediment (Renwick and Eden, 1999). In this way they reduce and control the amount of water reaching downstream portions of the river system at the time of the flood and prevent substantial infrastructural damage (Dickens *et al.*, 2003). Consequential to the role of wetlands in the storage of runoff water generated by snowmelt or rainfall, is their function of sustaining stream and river flow and reducing peak flow during small flood events (USEPA, 1995). Wetlands also play a key role in maintaining soil water balance as a result of their complex relationship with groundwater (USEPA, 1995). Their association with groundwater is such that during weather conditions that favour a rise in the water table, aquifers beneath the surface supply wetlands with water; whilst in the case of a low water table and hydraulic gradient reversal, wetlands assume the role of supplier and aquifers the role of receptor (Dickens *et al.*, 2003). The processes described above are referred to as ‘groundwater discharge’ and ‘groundwater recharge’ respectively (Dickens *et al.*, 2003).

Amongst the numerous functions performed by wetland systems, their role in water purification is most distinguished. Wetlands act as natural filters as they improve the quality of water from inland areas through a number of different processes (Begg, 1986; Collins, 2005). For instance, the uptake of minerals by resident plant communities; the accumulation of organic matter; the prevalent aerobic, anaerobic and decomposition processes; and the reduction of water flow velocity in wetlands, all result in the removal of minerals and chemicals from wetland waters (Dickens *et al.*, 2003). This effectively reduces any potential hazards to the environment (Dickens *et al.*, 2003). The role of wetlands as natural water purification systems has resulted in great support being awarded to the idea of using constructed wetlands as water treatment facilities (Kivaisi, 2001). The concept has already been realised in other parts of the world as an alternative to conventional wastewater treatment works; and according to research, the potential for its successful application in developing countries is considerably high (Kivaisi, 2001).

2.9 Constructed Wetlands as Wastewater Treatment Systems

Documentation of natural wetlands being used as wastewater treatment systems in different parts of the world dates as far back as 1912 (Kadlec and Knight, 1996 in Zhang *et al*., 2010). The use of constructed wetlands, however, as a means of water purification is one of the most recently proven efficient technologies, based on extensive research initiated in the 1960s and 1970s (Kivaisi, 2001; Vymazal, 2009). According to Vymazal (2009), constructed wetlands can now be used for the treatment of a wide variety of pollution, including domestic wastewaters, agricultural and industrial wastewaters, various runoff waters, as well as landfill leachate. Furthermore, the use of constructed wetlands for water treatment is cost-effective, easily operated and maintained, allows for water recycling and re-use, and requires minimal fossil fuel and no chemicals for operation (Vymazal, 2009). Therefore, a strong potential exists for its application in developing countries where the effects of water scarcity and the uninhibited depletion of natural water reserves are augmented by a lack of wastewater treatment facilities (Kivaisi, 2001). These systems are also designed to function at their optimum according to specifications of its surrounding environment, provide a habitat for numerous plant and animal species, and form open spaces of immense aesthetic value (Vymazal, 2009).

However, it must be noted that although constructed wetlands are ecologically engineered systems that are analogous to natural wetlands, they cannot completely replace the functions and services performed by natural wetlands (Mitsch & Gosselink, 2011). This holds especially true with regards to water purification, as their efficacy in providing this service is stunted by a lack of diversified plant communities, mature microbial populations and stable rhizosphere environments (Mitsch & Gosselink, 2011). Other limitations of utilising constructed wetlands as water treatment systems include that they require large tracts of land, a longer retention period to achieve effluent water quality of an acceptable level, and display unstable performance in the start-up period. Constructed wetlands also contain biota that show a greater sensitivity to chemicals, and water flow surges in these systems may act to reduce the effectiveness of water purification (Vymazal, 2009).

Types of constructed wetlands can be broadly categorised as surface flow, sub-surface flow and hybrid wetlands (Vymazal, 2009). The combination of various types of constructed wetlands, to produce what is referred to as ‘hybrid constructed wetlands’, can be used to achieve higher water treatment effectiveness if the combination is dependent on target pollutants in each case (Vymazal, 2008). On the other hand, given that the pros and cons of utilising natural and constructed wetlands differ, Zhang *et al*. (2010), proposed the use of an integrated wetland system, composed of natural and constructed wetlands, with the view that complementary advantages can achieve greater treatment capacity than any individual system by itself.

It is thus apparent that natural wetlands are features of remarkable value, which cannot truly be replaced by man-made replicates (Dickens *et al.*, 2003). Noteworthy however, is that their prevalence in South Africa is limited in comparison to that of other parts of the world (Begg, 1986).

2.10 Distribution of Wetlands in South Africa

In general, wetlands are most prevalent in regions characterised by high mean annual rainfall such that the quantity of water received through precipitation exceeds the quantity lost through evapotranspiration and surface runoff (Botes, 2009). Literature on national wetland environments indicates that the number of wetlands in South Africa is low, and their extent limited, due to the climatic and physiographic regimes dominating the landscape (Begg, 1986). The low annual rainfall and predominantly steep topography characterising the inland margin and coastal belt zones of the country have rendered these expanses of land unsuitable for wetland formation (Begg, 1986).

Hence wetland occurrence is most commonly noted within the interior plateau zone of South Africa due to its distinctly flat topography, despite its reception of less than 500 mm of mean annual rainfall (Begg, 1986; Barnes *et al.*, 2001). However, riverine wetlands occupy areas along river banks and drainage lines within the inland margin zone, and more limitedly, within the coastal belt of the country (Begg, 1986, Barnes *et al.*, 2001). The floodplain wetlands along the Lower uMngeni River system are amongst the few wetland areas occurring within the coastal belt of the country. It is thus important that these wetlands be studied so that effective management strategies can be put in place to ensure that growing anthropogenic pressures in this region do not result in the degradation and eventual disappearance of these ecosystems.

Given the undoubted importance of smaller riverine wetland systems, like those characterising the prominent uMngeni system, it is crucial that the required steps be taken to ensure their conservation and protection (Kotze *et al.*, 1995). Figure 2.3 depicts the distribution of wetlands in South Africa as classified by the NWCS and NFEPA project, and the location of wetland sites protected under the Ramsar Convention. Observation of the figure will reveal that the wetlands identified as important ecosystems by the Ramsar Convention are those that are typically larger in areal extent, and which serve as popular tourist destinations (Kotze *et al.*, 1995). However, the numerous smaller wetlands that exist in many parts of the country, especially those along important river systems, have generally been overlooked (Kotze *et al.*, 1995; Ramsar Convention Secretariat (RCS), 2007). It is thus essential that greater importance and conservation value be attached to the many smaller wetlands in South Africa to support sustainable utilisation and longevity of these systems (Kotze *et al.*, 1995).

However, the accelerated pace of wetland degradation, concomitant to the action of a multitude of factors on these systems, has resulted in their widespread destruction (Kusler & Opheim, 1996).

Figure 2.3….\

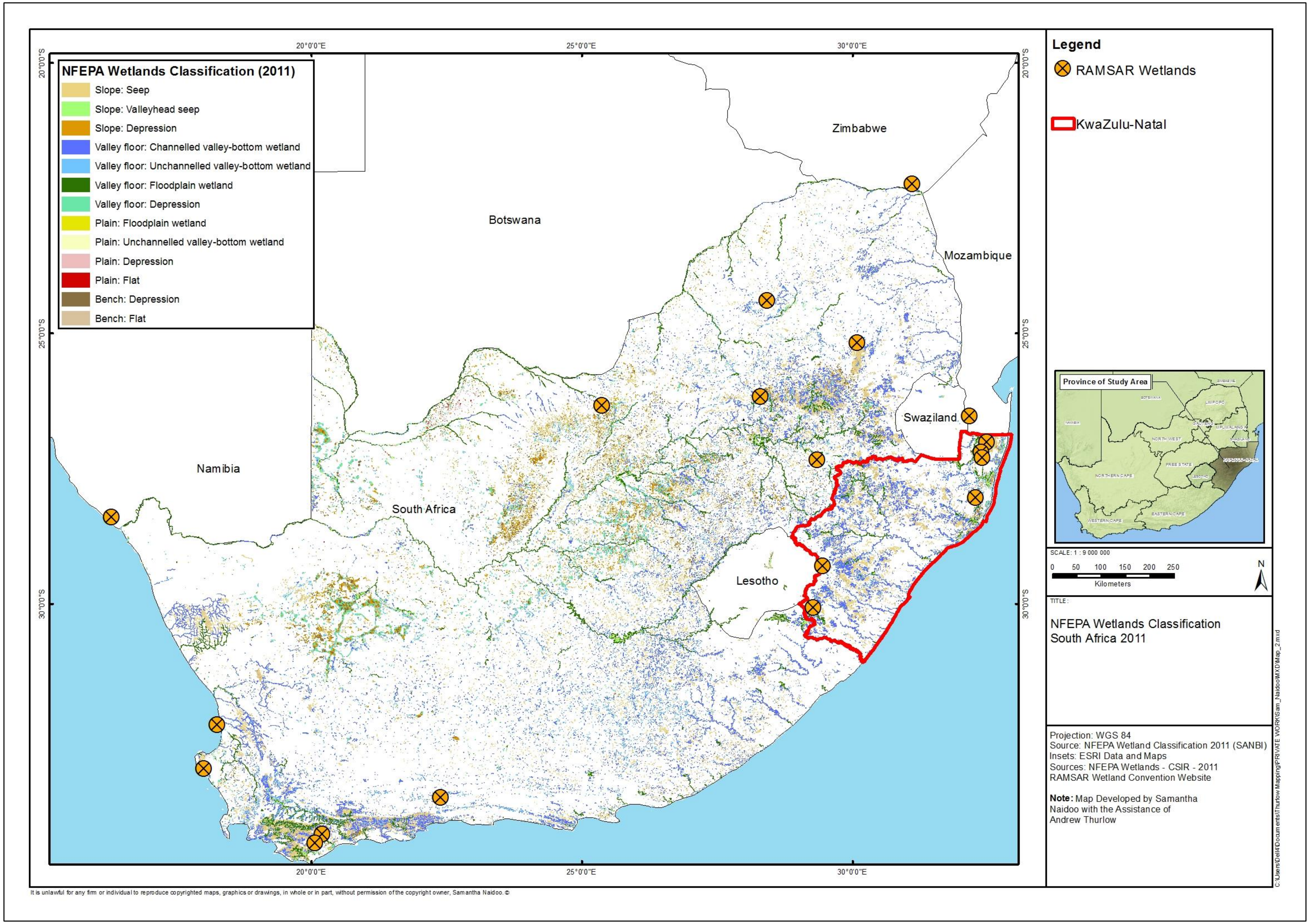


Figure 2.3: The distribution of wetlands identified as ‘Wetlands of International Importance’ by the Ramsar Convention and other wetlands in South Africa

2.11 Wetland Loss

According to Fraser and Keddy (2005), the spatial extent of wetland landscapes has reduced by almost 50% within a global context in the last century. Studies within several major catchments in South Africa have revealed that between 35% and 50% of national wetlands have already been lost or severely degraded; according well with the global estimate (Dini, 2004).

Wetland loss can be attributed to both direct and indirect anthropogenic activities (McInnes, 2010). Many would argue that although the pace of wetland destruction has rapidly accelerated over the past decade, the restoration and construction of man-made wetlands should mitigate, if not negate, the loss of these existing ecosystems (Fraser and Keddy, 2005). However, although such an ideology sounds reasonably logical at a conceptual level, its practicality and success thereof has been subject to much speculation (Fraser and Keddy, 2005). In light of this, research and statistics have shown that attempts by humans to construct fully functional wetland ecosystems have often proved futile, being informed by an insufficient understanding of factors that are imperative to ascertaining plant and animal community structure in wetlands (Fraser and Keddy, 2005). It is for this reason that the global success rate of constructed wetlands in fulfilling the role of such a highly dynamic and functional ecosystem is extremely low (Álvarez and Bécares, 2008).

It is evident that South Africa is collectively not conducive to the formation of wetlands, therefore magnifying the seriousness of issues relating to wetland loss on a national scale and the urgent need to implement better management, conservation and rehabilitation programmes (Kotze *et al.*, 1995). It can therefore be postulated that the significance of wetland destruction and loss in South Africa is comparatively greater than that of other countries, given the sporadic location and lack of prominence of wetlands within national bounds (Begg, 1986; Kotze *et al.*, 1995).

Cognisance must be taken of the adverse consequences associated with wetland degradation and loss in a semi-arid and developing country like South Africa (Turner, 1991). The imminent results of continued and/or accelerated wetland destruction include lower agricultural productivity, poorer water quality, less reliable water supplies, increased incidence and severity of downstream flooding, threatened wildlife resources, and an inevitable increase in incidence of species extinction (Millennium Ecosystem Assessment, 2005).

These outcomes act to deteriorate the facets upon which well-being and optimum functionality of ecosystems (including mankind) depend on (Kotze *et al.*, 1995). The loss of wetlands causes ecosystem disintegration and thereby biodiversity loss and an increase in the unattended needs of dependent poor and rural communities (Kotze *et al.*, 1995). It also has a crippling effect on the economy by creating monetary losses equivalent to billions which are squandered by the development and implementation of rehabilitation programmes and the construction of man-made wetlands (Kotze *et al.*, 1995).

Thus, legislation, conventions and policies, aimed specifically at promoting the protection, conservation, management and sustainable use of wetlands in South Africa should be strongly enforced by all tiers of government (Dickens *et al.*, 2003).

2.12 Wetland Legislation, Policy, Programmes and Conventions

The existing suite of legislation in South Africa provides great opportunities for the protection and conservation of wetlands within national bounds, and clearly indicates the obligations and rights of all interested and affected parties, including government officials, land owners and civil society (Kotze, 2000). The provisions of these pieces of legislation can either be used in a proactive way to ensure that wetland deterioration is minimised, if not negated, when faced with the threat of development, or in a reactive manner by enforcing clean-up or rehabilitation actions with the aim of curtailing existing adverse impacts on these systems (Dickens *et al.*, 2003). The former of the two approaches is preferable from both an environmental and economic perspective, and relevant steps should be taken to make mandatory the application of this particular approach (Dickens *et al.*, 2003). The relevant legislation that promotes the conservation and sustainable use of wetlands in South Africa are outlined below.

The constitution is the supreme law of South Africa and no law, whether national, provincial or local, may conflict with it (Teixeira-Leite & Macfarlane, 2012). The constitution calls for the prevention of pollution, the promotion of protection and the sustainable development and use of natural resources (Dickens *et al.*, 2003). The Bill of Rights in the constitution also makes explicit the right of all citizens to an environment that safeguards their health and well-being (Dickens *et al.*, 2003).

The National Water Act is the overarching law that provides for the protection, use, development, conservation, management and control of water resources, based fundamentally on the guiding principles of equity and sustainability (Teixeira-Leite & Macfarlane, 2012). It considers the needs of present and future generations, and addresses the competing uses of water resources by attempting to balance the need for water resource protection with the need for economic development (Dickens *et al.*, 2003). It also considers wetland systems as an integral water resource by defining them as a type of 'watercourse' that requires due consideration and management (Kotze, 2000). The Act also requires that the Minister establish Catchment Management Areas, and delegate to catchment level certain powers and duties relating to water resource management to be carried out within the respective jurisdictions (Dickens *et al.*, 2003).

The objective of the National Environmental Management: Biodiversity Act (NEMBA) (Act 10 of 2004) is to provide for the conservation of biological diversity, regulate the sustainable use of biological resources and to ensure a fair and equitable sharing of the benefits arising from the use of genetic resources (Council for Scientific and Industrial Research (CSIR), 2014a). The National Department of Environmental Affairs (DEA), previously known as the Department of Environmental Affairs and Tourism (DEAT), is the lead agent for the NEMBA and the custodian for the nation's biological diversity, and is therefore mandated to ensure that the constitutional rights of the nation's citizens are respected, protected, promoted and fulfilled (CSIR, 2014a).

The Conservation of Agricultural Resources Act stipulates that the Minister of Land and Agricultural Affairs may prescribe land use measures which all land users are obligated by law to comply with (Kotze, 2000). This Act requires that control be exercised on land use activities, such as burning and weed control, and pollution and erosion prevention (Dickens *et al.*, 2003). It also includes measures related to the utilisation and protection of wetlands (Teixeira-Leite & Macfarlane, 2012).

The Environmental Conservation Act (ECA) (1989) requires that the effects of all proposed developments be identified, and their significance determined, prior to decision-making by the DEA (Kotze, 2000). The Act further ensures public involvement in the decision-making process through the EIA regulations, which requires that all persons involved in activities that cause environmental damage be responsible for its rehabilitation, and confers the power to the Minister of Environmental Affairs to either accept or prohibit proposed developments subsequent to evaluation of the associated impacts (Dickens *et al.*, 2003).

The National Environmental Management Act (NEMA) (Act 107 of 1998), was promulgated to support the constitutional principle of sustainable development and is concerned primarily with the effective management of natural resources, including those associated with catchment management (Dickens *et al.*, 2003; Teixeira-Leite & Macfarlane, 2012). It aims to integrate social, environmental and economic factors into decision-making, integrate legislative input from all spheres of government and integrate sustainable environmental management into all development activities (Dickens *et al.*, 2003). Noteworthy is that the ECA (1989) and associated EIA regulations were repealed in 2006 and promulgated under the National Environmental Management Act (NEMA) (2006). Subsequently, the NEMA and EIA regulations were further amended in 2010 and replaced the regulations and listing notices promulgated in 2006.

The promulgation and enactment of the National Environmental Management: Integrated Coastal Management Act (NEMICMA) (Act 24 of 2008) on 1 December 2009 promoted a paradigm shift with regard to coastal management in South Africa (Celliers *et al.*, 2009). The Act defines the coastal zone and has significant legal implications for land-based coastal activities (e.g. coastal development, access to the coastal zone and the release of effluent) and marine-based coastal activities (e.g. dumping at sea) (Celliers *et al.*, 2009). The aim of the Act is to ensure that all developments, and the use of coastal resources, are socially and economically justifiable and ecologically sustainable (Celliers *et al.*, 2009). Therefore, it establishes new and innovative regulatory instruments to provide for the co-ordinated and integrated management of the coastal zone by all spheres of government (Celliers *et al.*, 2009).

The Convention on Wetlands and the South African Wetlands Conservation Programme has also been developed for the purpose of facilitating wetland conservation in South Africa (Kotze, 2000). The former is an intergovernmental treaty which is aimed at promoting the conservation and wise use of wetlands through local, regional and national actions as well as international co-operation (RCS, 2007). The latter, the South African Wetlands Conservation Programme, is also based on the fundamentals of wetland conservation and sustainability (Kotze, 2000). This programme is governed by the Sub-directorate Ecosystems of the DEA and has been implemented to ensure that South Africa meets its obligations in terms of the Convention on Wetlands of International Importance (RCS, 2007).

With regards to international Agreements and Conventions, South Africa is signatory to the Ramsar Convention on Wetlands since 1975, and comprises one of 136 countries who have pledged to uphold the values and fulfil the objectives as stipulated by the Convention (Dickens *et al.*, 2003). The Convention recognises the importance of the wise use of wetlands, given their immense value in maintaining ecological processes, providing refuge to a plethora of species, and affording a number of benefits to local communities (Dickens *et al.*, 2003). Parties of the convention aim to ensure the conservation of wetlands and all their constituents by combining far-sighted national policies with co-ordinated international action (Dickens *et al.*, 2003).

Apart from being legal instruments, the Conventions, Acts and Programmes discussed above provide an overarching framework for managing water resources and other natural resources in South Africa (Kotze, 2000; Dickens *et al.*, 2003). Although plans relating to conventions, such as the Ramsar Convention, are set at national level, implementation needs to take place at regional and local levels where resources are available and where people interact with resources (Dickens *et al.*, 2003). It is therefore imperative that Conventions and Acts be translated and integrated into catchment specific strategies, and wetland rehabilitation and management measures to be implemented on ground-level (Dickens *et al.*, 2003).

2.13 Wetland Management and Rehabilitation

The disconcertion incited by wetland loss has prompted the application of more effective and efficient conservation, management and rehabilitation strategies. Despite this effort, wetlands in South Africa, particularly those outside protected areas, are subject to high levels of degradation (Dickens *et al.*, 2003).

It is important to note at the very outset that the formulation of such strategies requires the availability of vital information pertaining to wetland functioning, inter- and intra-relationships within and between these and other systems respectively, and most importantly, a valid compilation of baseline information on wetland environments (Kotze *et al.*, 1994). Although much progress has been made over the last few years in developing more comprehensive databases of South African wetlands, much of the abovementioned requirements, to a certain extent, remain unfulfilled (Kotze *et al.*, 1994). This in effect limits the effectiveness of all attempts made to remedy the impacts of wetland loss (Kotze *et al.*, 1994).

These limitations are often viewed as, and referred to, as weak points in management systems employed to properly manage wetland environments (Dickens *et al.*, 2003). Included is the fact that past perceptions of wetlands as disease- and filth-infested landscape features resulted in no prior attempts being made to record their presence and distribution within the country (Kotze *et al.*, 1994). Hence, South Africa possesses an inadequate database of wetland loss documentation (Kotze *et al.*, 1995).

Of equal importance to the existence of data and technical tools, is the need for supporting legal and institutional mechanisms (Kotze *et al.*, 1995). These include the need for institutions with the appropriate technical expertise to manage wetlands; the development of national policies to promote wetland conservation and wise use; co-ordinating mechanisms to allow stakeholders to meet and exchange information; and awareness amongst planners, decision-makers and the general public of the value of wetland ecosystems (Dickens *et al.*, 2003).

Development of the National Wetland Inventory in 2009 which provides information on wetland location, size and type will assist in guiding the assessment of the impact of wetland loss on biotic diversity, conservation status and urgency of protection (SANBI, 2009). This paves the way for further progress in the management and sustainable use of wetlands in South Africa (SANBI, 2009). However, in terms of conserving the biotic dependents and larger-scale ecosystem services and processes of wetland systems, it is essential that concerted efforts be made to measure within- and between- system diversity and understand the mechanism regulating diversity (Kotze *et al.*, 1995; Dickinson, 2007).

In addition, the importance of considering the inter-relationship between catchment areas and their inherent freshwater features, such as rivers, streams and wetlands, should provide impetus for a shift towards aligning land use management and water resources management strategies, so that their joint implementation can ensure the maintenance of healthy, functional wetlands and other ecosystems alike (Dickens *et al.*, 2003).

Catchment Management Areas should also bring water resource management perspectives to the Integrated Development Planning (IDP), Spatial Development Framework (SDF) and Land Use Management System (LUMS) planning processes, undertaken by local municipalities (SANBI, 2009). Given that co-operative governance is provided for in the Municipal Systems Act, it is imperative that plans and strategies of municipalities and other organs of state be aligned and complement each other (Dickens *et al.*, 2003; SANBI, 2009).

According to the Wetland Research Commission (2009), wetland rehabilitation makes reference to actions performed with the view of reversing or halting the declining health of a wetland system, thereby returning it to a healthy state and allowing some, if not all, wetland services originally lost to be recovered. This is usually done by trying to re-instate the driving forces of wetlands, namely, hydrological, ecological and geomorphological forces, through implementing a number of measures (Jaganath, 2009). South African experience has revealed that it is possible to recover the health and values of the many degraded wetlands in the country through rehabilitation (Macaskill, 2010).

The major challenges of wetland rehabilitation can be addressed by consulting openly and comprehensively with landowners, land users, and other key stakeholders such as municipalities and provincial departments who will either benefit from, or are mandated to facilitate, rehabilitation initiatives (SANBI, 2009). Also, both causes and symptoms of wetland degradation must be accounted for in rehabilitation programmes, and clear measurable objectives must be developed with a multidisciplinary team including experts in ecological functioning and the design of rehabilitation structures (SANBI, 2009).

It must be noted that the Water Research Commission has developed a series of wetland management and rehabilitation tools, referred to as the ‘WET-Management Series’, which offer a sound scientific basis for the planning, implementation and evaluation of wetland management and rehabilitation. These guidelines can be used to steer management applications in South Africa and internationally (SANBI, 2009).

2.14 Conclusion

It can be concluded from the above that wetland ecosystems are important 'assets' to society in both the environmental and socio-economic senses (WRC, 2003). The increased incidence of wetland degradation and loss in South Africa, as a result of human activities, has become a cause for great concern (Kotze *et al.*, 1995). This has effectively increased research initiatives regarding wetlands, their location, distribution, classification and functioning (Kotze *et al.*, 1995). It also expedited the process of compiling a comprehensive national wetland inventory and classification system, which can be used to guide wetland management and rehabilitation strategies (SANBI, 2009). The protection and conservation of wetlands can be achieved by capacitating and re-enforcing the role of local institutions; strengthening enforcement of legislation, policies and conventions relating to wetlands by all levels of government; and applying management and rehabilitation strategies informed by sound scientific research (Kotze *et al.*, 1995).

CHAPTER 3: REGIONAL SETTING

3.1 Introduction

The prevailing environmental and socio-economic conditions and processes that govern the health and functioning of the uMngeni River and connected wetland features are described in this chapter. Characteristics including dominant land uses and climatic, hydrological, geological, ecological and socio-economic attributes of the uMngeni catchment, and particularly the Lower uMngeni sub-catchment, are discussed. An overview of the history of wetlands within the uMngeni catchment is provided as well as a description of their current status, classification and occurrence. An account of existing management plans and institutional arrangements for the Lower uMngeni sub-catchment is also provided.

3.2 Site Description

3.2.1 The uMngeni Catchment and Fluvial System

The uMngeni catchment is situated within the province of KwaZulu-Natal, covering a mean area of approximately 4 740 km^2; approximately 4 079 km^2 of which is situated upstream of Inanda Dam (Brijlal, 2005). The uMngeni River has an approximate length of 255 km and represents one of the largest river systems in KwaZulu-Natal (Brijlal, 2005; Cooper, 1993). The source of the uMngeni River is the uMngeni vlei, which has been declared a nature reserve and a Ramsar site in light of its significant ecosystem service and biodiversity value (Johnson *et al.*, 1998). The river then flows through a number of major urban, industrial and agricultural areas before reaching the coast. At the coast, waters are then transported *via* a highly modified river channel through the areas of Pinetown and Springfield Flats in Durban, prior to its entry into the Indian Ocean through the river mouth (van der Zel, 1975). The topography of the catchment varies as altitude decreases along the course of the river, reaching the lowest point at the river mouth (van der Zel, 1975).

The uMngeni River is closely linked to a number of other river systems that either branch off into tributaries of the uMngeni River or converge with the uMngeni River at specific locations along its course. Inclusive are the uMsunduzi, uMlazi and Mqeku Rivers. The lower section of uMngeni catchment in particular is drained by a number of rivers, including the uMbilo, uMhlatuzana, uMhlangane, Molweni and Palmiet Rivers, which are likely to carry pollutants from their respective catchments (WRC, 2002; Arcus Gibb, 2005). Furthermore, a total of five dams have been constructed along the uMngeni River, namely, Midmar Dam, Henley Dam, Albert Falls Dam, Nagle Dam and Inanda Dam (Brijlal, 2005).

As part of river management initiatives, the uMngeni catchment was divided into sub-catchments, otherwise referred to as resource units, for reporting purposes (WRC, 2002). The entire catchment area falls within the jurisdictional boundaries of the following municipalities in KwaZulu-Natal: the eThekwini Metropolitan Municipality, the Ndwedwe Local Municipality within the iLembe District, and the Mkhambathini, uMshwati, uMngeni, Richmond and uMsunduzi Local Municipalities within the uMgungundlovu District. Figure 3.1 depicts the location of the uMngeni catchment in relation to municipal, provincial and national boundaries; and the extent of the study area falling within the Lower uMngeni and Durban sub-catchment.

The area of interest for this particular study falls within the Lower uMngeni and Durban sub-catchment, otherwise referred to as the “Lower uMngeni sub-catchment”. This is a 5 km stretch of the river channel, from approximately 3 km upstream of the N2 Bridge to the Athlone Bridge. The rationale for limiting the study to this area stems from the findings of Jewitt & Kotze (2000) and WRC (2002), which infer that the greatest historical loss of wetlands were recorded in the lower reaches of the uMngeni catchment. Furthermore, the lower region of the catchment is characterised by increasing development pressures which result in the expansion of existing urban, industrial and commercial areas; thereby increasing the susceptibility of environmental features to damage and destruction.

The physical and biological attributes of the uMngeni catchment are described in the sections that follow.

Figure 3.1….\

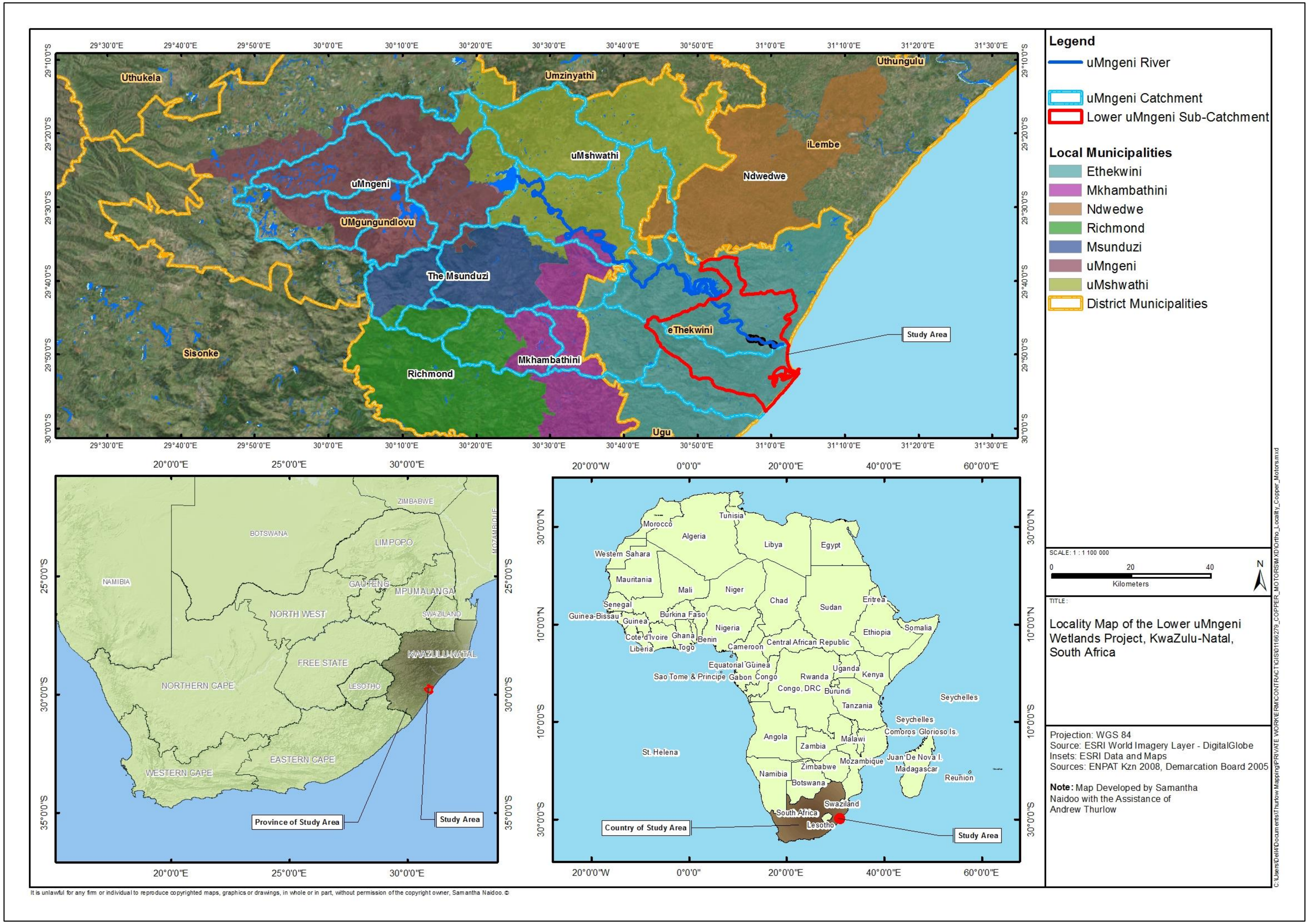


Figure 3.1: Location of the study area within the Lower uMngeni sub-catchment in relation to national, provincial and local geographic bounds

3.2.1.1 Climatology and Hydrology

A warm subtropical climatic regime dominates the uMngeni catchment (Govender, 2013). In the summer months, the weather is generally hot and humid with temperatures ranging between 20°C and 30°C (Summerton and Schulze, 2008). In winter, on the other hand, temperatures range between 13°C and 25°C, and dry conditions prevail (Summerton and Schulze, 2008; Palmer *et al.*, 2011). Annual potential evaporation fluctuates according to these climate and temperature patterns, and was recorded to be approximately 1400 mm in the coastal area within the Lower uMngeni sub-catchment (DWAF & Umgeni Water, 1996).

Mean Annual Precipitation (MAP) is approximately 900 mm in the lower eastern parts of the uMngeni catchment (Summerton and Schulze, 2008). Falling within a summer rainfall region, the catchment receives approximately 80% of its annual rainfall between the months of October and March, due largely to convectional storms (DWAF & Umgeni Water, 1996; Summerton and Schulze, 2008). Rainfall is further enhanced in the headwaters by orographic activity; whilst large weather fronts, heavy winds and cyclones contribute to heavy rainfall along the coast (DWAF & Umgeni Water, 1996).

Prevailing wind directions along the KwaZulu-Natal coastal belt are predominantly from the north east and south west, and occur with frequencies in excess of 255 days per year (Miller, 2001). South westerly winds are generally stronger and may be accompanied by rain (Golder Associates Africa (Pty) Ltd., 2012). Mean monthly variations in wind speed are such that it reaches recorded lows in the months of May and June, and highs in the transitional period at the end of winter in the months of September and October (Golder Associates Africa (Pty) Ltd., 2012). Maximum wind speeds occur in the early afternoon, at around 14h00, and minimum wind speeds after dawn between 06h00 and 08h00 (Golder Associates Africa (Pty) Ltd., 2012).

The climate and weather patterns that typify the Lower uMngeni sub-catchment, and more particularly the Durban area, are illustrated in Figures 3.2, 3.3, 3.4 and 3.5 below. These figures depict the average monthly variations in temperature, rainfall, wind speed and wind direction in Durban respectively.

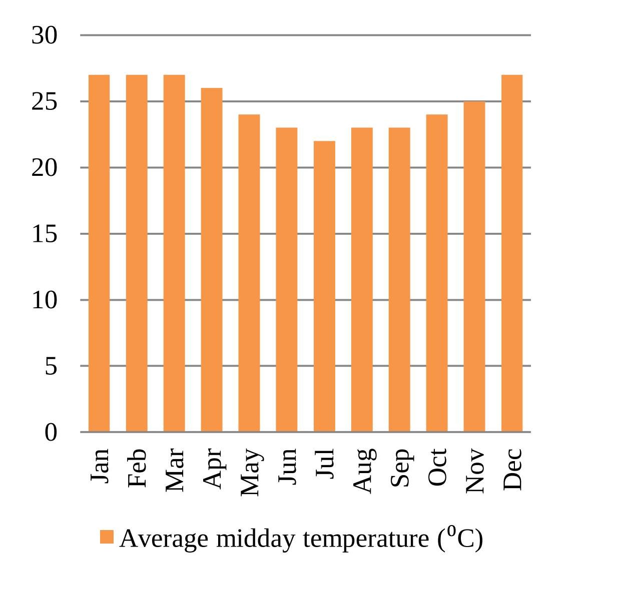


Figure 3.2: Mean monthly midday temperature in Durban (Windfinder, 2014)

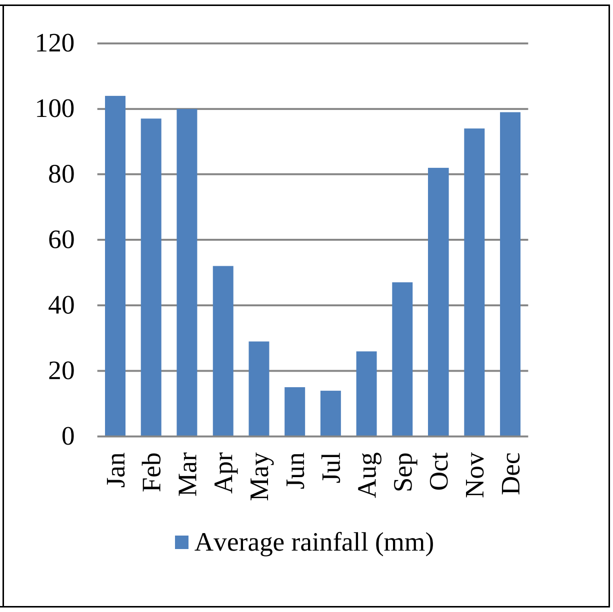


Figure 3.3: Mean monthly rainfall in Durban (Windfinder, 2014)

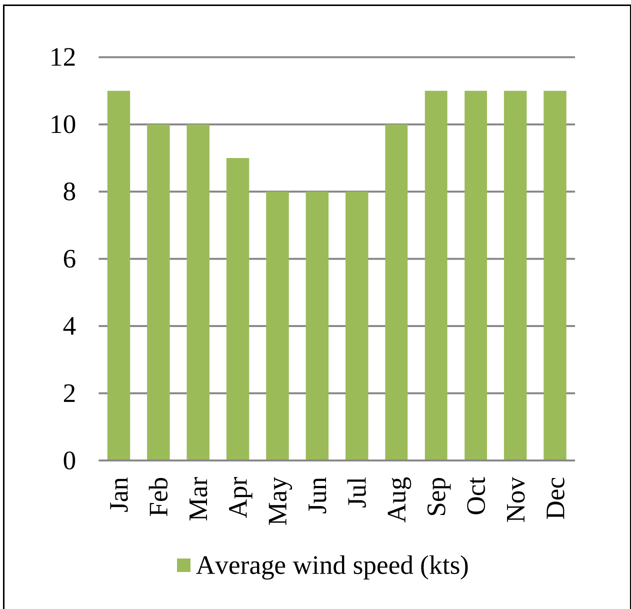


Figure 3.4: Mean monthly wind speed in Durban (SAExplorer, 2014)

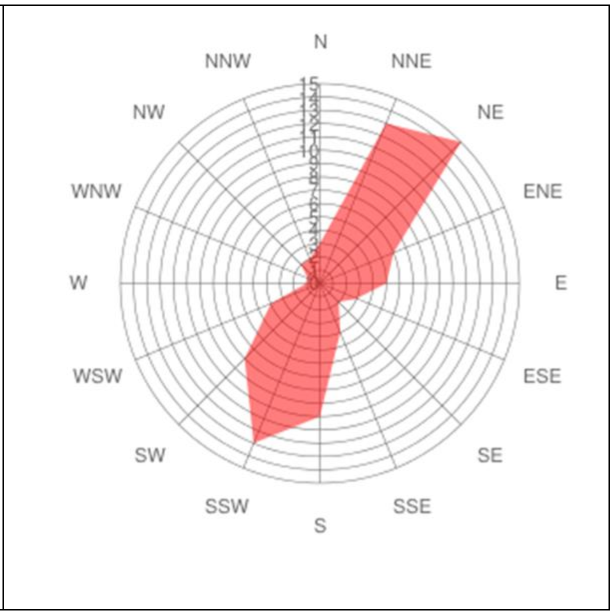


Figure 3.5: Dominant annual wind direction (SAExplorer, 2014)

Water flow in the uMngeni River is intrinsically linked to the rainfall pattern throughout the catchment, with approximately two thirds of natural river flow occurring between November and March annually (DWAF & Umgeni Water, 1996). However, river flow is significantly more variable than rainfall, having been documented to deviate to much greater degrees away from the recorded average (DWAF & Umgeni Water, 1996). Low flows can be attributed to drought periods and may occur during the drier seasons of the year (DWAF & Umgeni Water, 1996). Releases from impoundments may, however, act to mitigate against the adverse effects of low flows and considerable seasonal variations (DWAF & Umgeni Water, 1996).

The contribution of rainfall to river water levels at the coast is such that approximately 20% to 30% of precipitation reaches the river (DWAF & Umgeni Water, 1996). Land surface runoff comprises a significant component of the total river flow in the Lower uMngeni Sub-catchment, owing to the higher rainfall and more impervious surfaces in the highly developed towns of Durban and Pietermaritzburg (DWAF & Umgeni Water, 1996).

3.2.1.2 Geology and Soils

Figure 3.6 depicts the geological characteristics of the Lower uMngeni sub-catchment. It can be noted that the sub-catchment is dominantly underlain by Natal Group sandstone, whilst Pietermaritzburg and Vryheid shales, Vryheid sandstone and Natal granites occur in the northern-most parts of the sub-catchment. It is also characterised by alluvium of the Quaternary age, consisting predominantly of silts, sands and clays (Arcus Gibb, 2005). The alluvium layer, which is approximately 30 m thick, is underlain by Permian age dark grey shales and sandstone interspersed with boulder shale of the Karoo sequence and Dwyka formation (Arcus Gibb, 2005). Furthermore, on the southern edge of the alluvium, a southwest - northeast trending fault was identified (Arcus Gibb, 2005). Closer to the river mouth, Harbour beds and arenite of the Berea formation occurs.

As illustrated Figure 3.7, a significant portion of the soils occurring here are Glenrosa or Mispah forms; one or more of undifferentiated vertic, melanic and red structured diagnostic soils; miscellaneous land classes with undifferentiated deep deposits; red-yellow apedal freely drained soils; and grey regic sand and other soils. According to DWAF and Umgeni Water (1996), the red and grey sands at the coast within the Lower uMngeni sub-catchment are also characterised by soil depths, soil permeability and soil fertility that are considered ‘low’. These variables are higher than those relevant to soils within the Valley of a Thousand Hills, and are also less erodible (DWAF & Umgeni Water, 1996). However, erosion within this part of the catchment remains a major problem.

The underlying geology and predominant soil types in the Lower uMngeni sub-catchment provide an indication of the susceptibility of soils to erosive forces, thereby influencing sediment yield in this section of the uMngeni River.

3.2.1.3 Sediment Yield

It has been documented that the construction of dams on the uMngeni River may have reduced the sediment yield from the reported figures of 1.6 x 106 tonnes (Rooseboom, 1975) and 1.0 x 106 tonnes (Pillay, 1980) that reaches the Lower uMngeni River and uMngeni Estuary per year (Brijlal, 2005). As mentioned in section 3.2.1.2 above, the Lower uMngeni River consists mainly of sand and is thus highly erodible, especially during flood events, therefore prompting changes in the river bed (HKS, 1989 cited in Arcus Gibb, 2005). In light of the afore-mentioned, and given the high degree of river bank modification resulting from increased rural dwellings, development, urbanisation and industrialisation, it is likely that severe erosion may have increased downstream sediment yield in current times (Brijlal, 2005).

Furthermore, taking into consideration the prevailing north easterly - south westerly wind direction in the Durban area, in relation to the north west - south east positioning of the river channel, aeolian forces are likely to impel significant erosion and sedimentation along both the northern and southern banks of the channel resulting in increased sediment yield and siltation of the river. In addition, the consecutive occurrence of high wind speed months and high rainfall months per annum results in a six-month period between September and March annually, in which erosion (resulting from the action of wind and water), and thus sediment yield will be most significant (Newman *et al*., 2008). The occurrence of erodible and shallow silts, sand and clays, with low permeability further facilitates erosion processes along the coast (DWAF & Umgeni Water, 1996).

To this point, it can be stated that changes in topography, climatology and geology collectively create conditions within the Lower uMngeni sub-catchment that determine its suitability for the location of different anthropogenic land uses. With regards to sediment yield, land use activities also increase the vulnerability of cleared and modified areas to erosive forces, which in turn increases the sediment yield of rivers (Owens & Collins, 2006).

Land use activities taking place within the uMngeni catchment, and more specifically within the Lower uMngeni sub-catchment, are described in the section that follows.

Figure 3.6….\

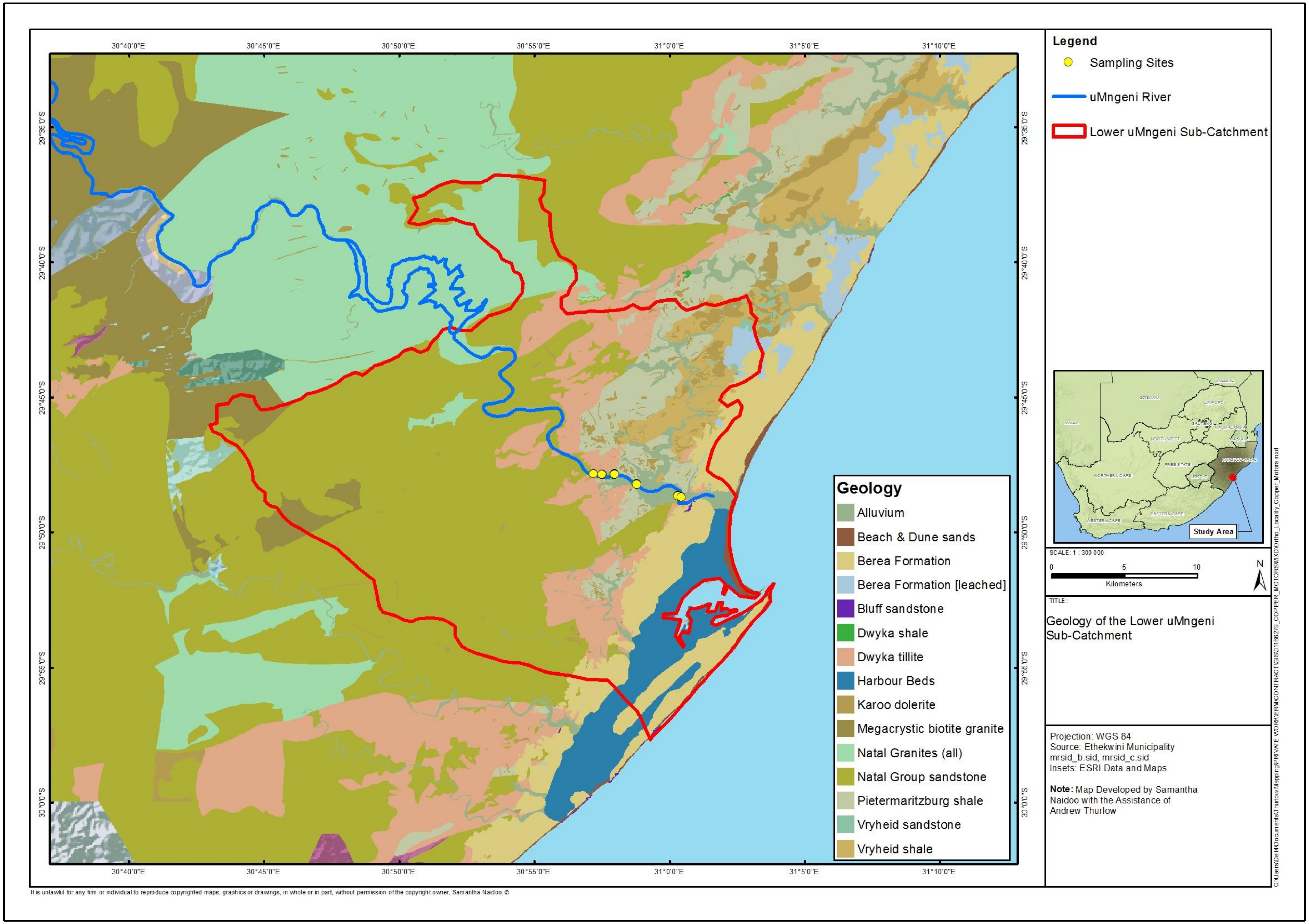


Figure 3.6: Geology of the Lower uMngeni sub-catchment

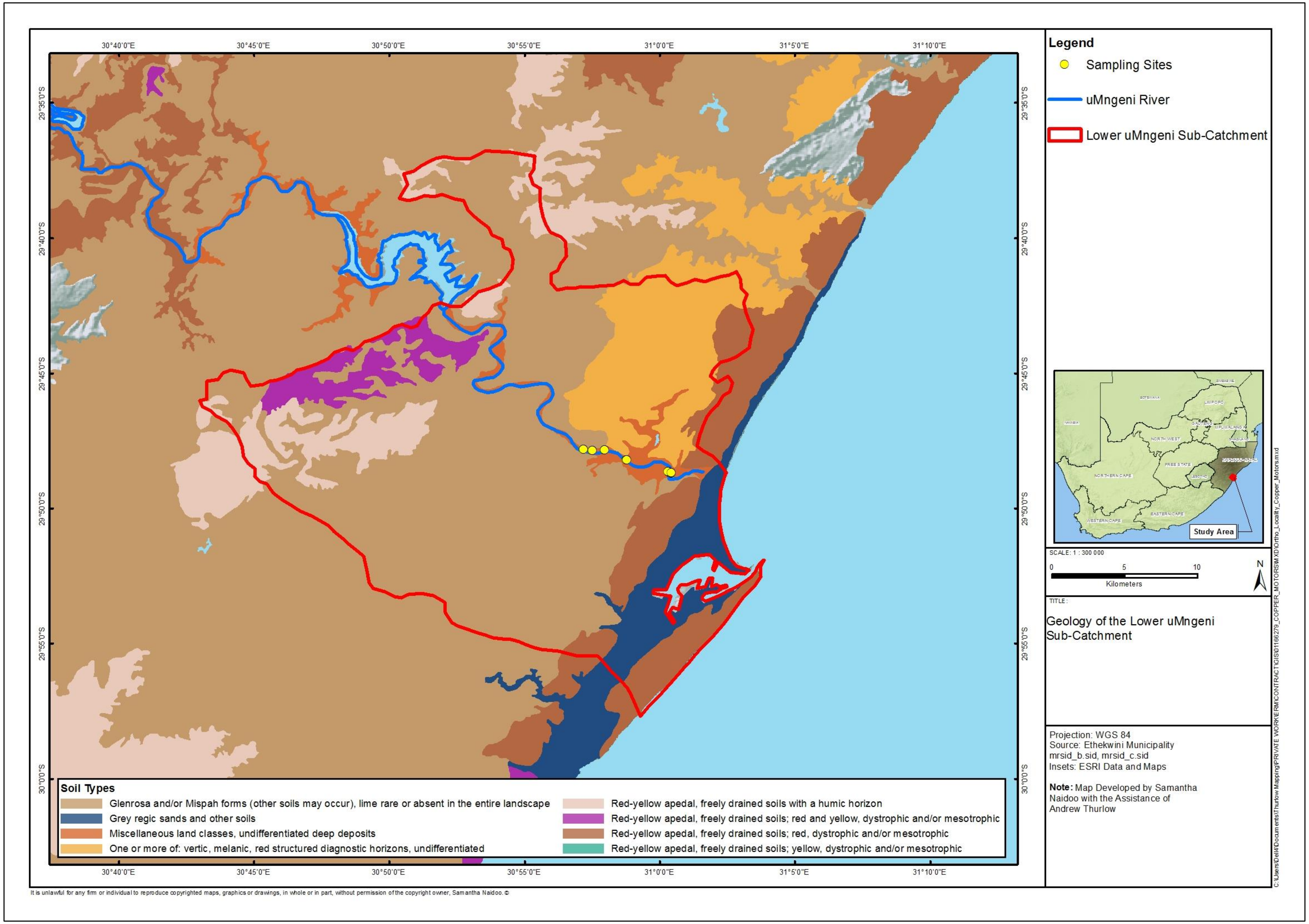


Figure 3.7: Soils of the Lower uMngeni sub-catchment

3.2.1.4 Land Uses

Land use activities within the uMngeni catchment have the greatest influence on river water level, water use and water quality (DWAF & Umgeni Water, 1996; WRC, 2002). The approximate land cover percentage of natural, degraded bush/shrub land, agricultural and urban environments within the uMngeni catchment are 52%, 3%, 37% and 8% respectively (Arcus Gibb, 2005).

The city of Durban, within the Lower uMngeni sub-catchment, is predominantly urban in nature, consisting mainly of residential, industrial, and commercial development (DWAF & Umgeni Water, 1996; Arcus Gibb, 2005). This region was subject to significant modification to accommodate development, including canalisation of the uMngeni, uMbilo and uMhlatuzana Rivers, which resulted in the clearance of natural habitat in its lower reaches (WRC, 2002). Polluted run-off from urban areas and unserviced informal settlements lead to water contamination in the river and wetland systems (Brijlal, 2005). Sand mining activities are also undertaken in this region of the catchment although the construction of the Inanda Dam may have diminished the volumes of sand in downstream areas (Brijlal, 2005). The adverse impacts on ecosystem health therefore include bacterial contamination, soil erosion, flooding and the entry of contaminants into the system sourced from urban and industrial activities (DWAF & Umgeni Water, 1996). The consequential decline in water quality poses a number of human health risks resulting from the utilisation of polluted water and infection by water-borne diseases (DWAF & Umgeni Water, 1996).

Considering the fact that the uMngeni catchment area represents regions encompassing ecologically important ecosystems, unmanaged nutrient and contaminant input into the river and hydrologically linked systems may result in the loss of a number of floral and faunal species (WRC, 2002).

3.2.1.5 Ecological Sensitivity and Biotic Characteristics

Figure 3.8 below illustrates the study location in relation to formally protected spaces, the National Protected Areas Expansion Strategy (NPAES) areas, and the ranks of threatened ecosystems within its immediate surrounds. According to DEA & South African National Biodiversity Institute (SANBI) (2009), the South African protected areas network currently falls short of sustaining the biological and ecological processes of ecosystems within the country (DEA & SANBI, 2009). With the aim of improving the current situation, the goal of the NPAES is to achieve protected area expansion, promote ecological sustainability and increase the resilience of national ecosystems to climate change (DEA & SANBI, 2009). The categorisation of areas as "threatened ecosystems", based on their susceptibility to ecosystem and species extinction, was presented as the "National list of ecosystems that are threatened and in need of protection", approved and gazetted as part of the National Environmental Management: Biodiversity Act (DEA, 2011).

It can be observed by examination of Figure 3.8 below that terrestrial ecosystems situated approximately 10 km and 30 km upland of the river channel along the southern and northern banks respectively, are classified as "endangered". This implies recognition of the considerable degradation that these ecosystems are subject to, and acknowledgement of the fact that these areas should be awarded formal protection precedence.

With regards to vegetation, the Lower uMngeni sub-catchment, and more specifically the Durban Metropolitan, falls within the Indian Ocean Coastal Belt biome. Examination of Figure 3.9 below illustrates that the dominant vegetation types within this area include the KwaZulu-Natal Coastal Belt Grassland, KwaZulu-Natal Coastal Belt Thornveld, KwaZulu-Natal Sandstone Sourveld and Eastern Valley Bushveld. It must be noted that vegetation characterising freshwater wetlands also occur within the Lower uMngeni sub-catchment, and typically along the uMngeni River channel. Noteworthy is the fact that all the above-mentioned vegetation types fall within the "endangered" zone in this particular geographic location. It is therefore absolutely necessary that continued destruction of vegetation within the riparian zone and areas further inland be prevented in order to avert species extinction and ecosystem collapse.

A number of plankton, invertebrate, fish, amphibian, crustacean, reptile and bird species either inhabit or rely on the resources and features within the riverine environment and its immediate surrounds (Brijlal, 2005). Although river systems, together with other linked hydrological features, are able to purify their constituent waters, large numbers of macro-invertebrates, bacteria and algae are known to absorb nutrients and pollutants from waters, therefore assisting to a large degree in these purification processes (WRC, 2002). Some bacteria also convert ammonia to less noxious forms, and provide food for organisms that occupy higher trophic levels of the food chain (WRC, 2002).

Of concern however is the increasing number of alien terrestrial and aquatic species within the uMngeni River and its environs (DWAF & Umgeni Water, 1996). Alien fish species, found to occur in a number of the uMngeni sub-catchments, include carp, bass and the redbreast tilapia (WRC, 2002). Also evident is the ecosystem disruption caused by aquatic weed species such as water hyacinth, water lettuce, Kariba weed and parrots feather. These have also been recorded within several of the uMngeni sub-catchments (DWAF & Umgeni Water, 1996; WRC, 2002).

Furthermore, increased human disturbance, and uncontrolled development within the catchment, has fast resulted in a number of dependent and reliant species becoming vulnerable to extinction (Brijlal, 2005).

Apart from the danger posed by anthropogenic activities to floral and faunal species that inhabit or depend on the riverine and wetland environments, the resulting deterioration in water quality and decline in water quantity also result in social and economic loss.

Figure 3.8….\

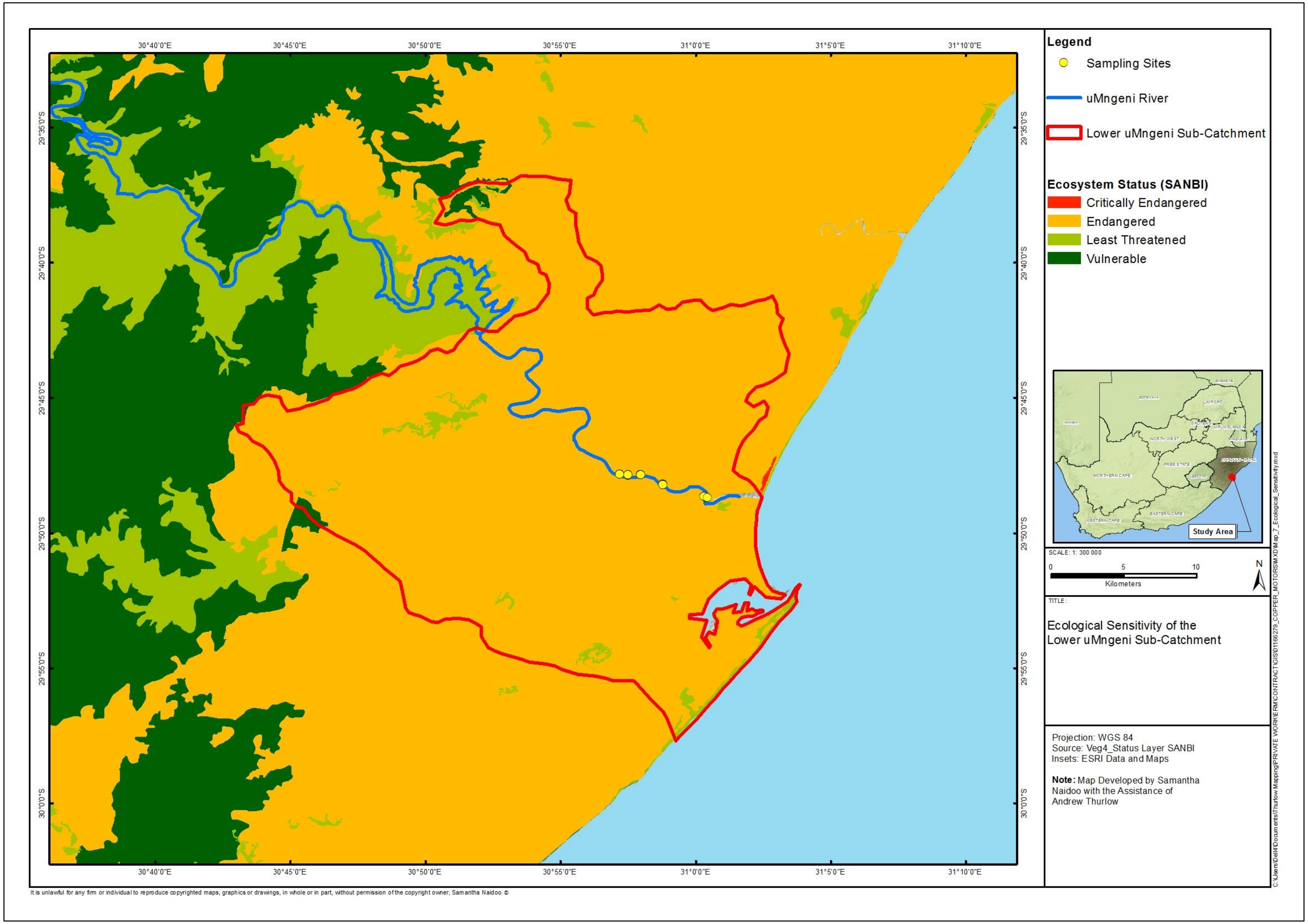


Figure 3.8: Ecological Sensitivity of the Lower uMngeni sub-catchment

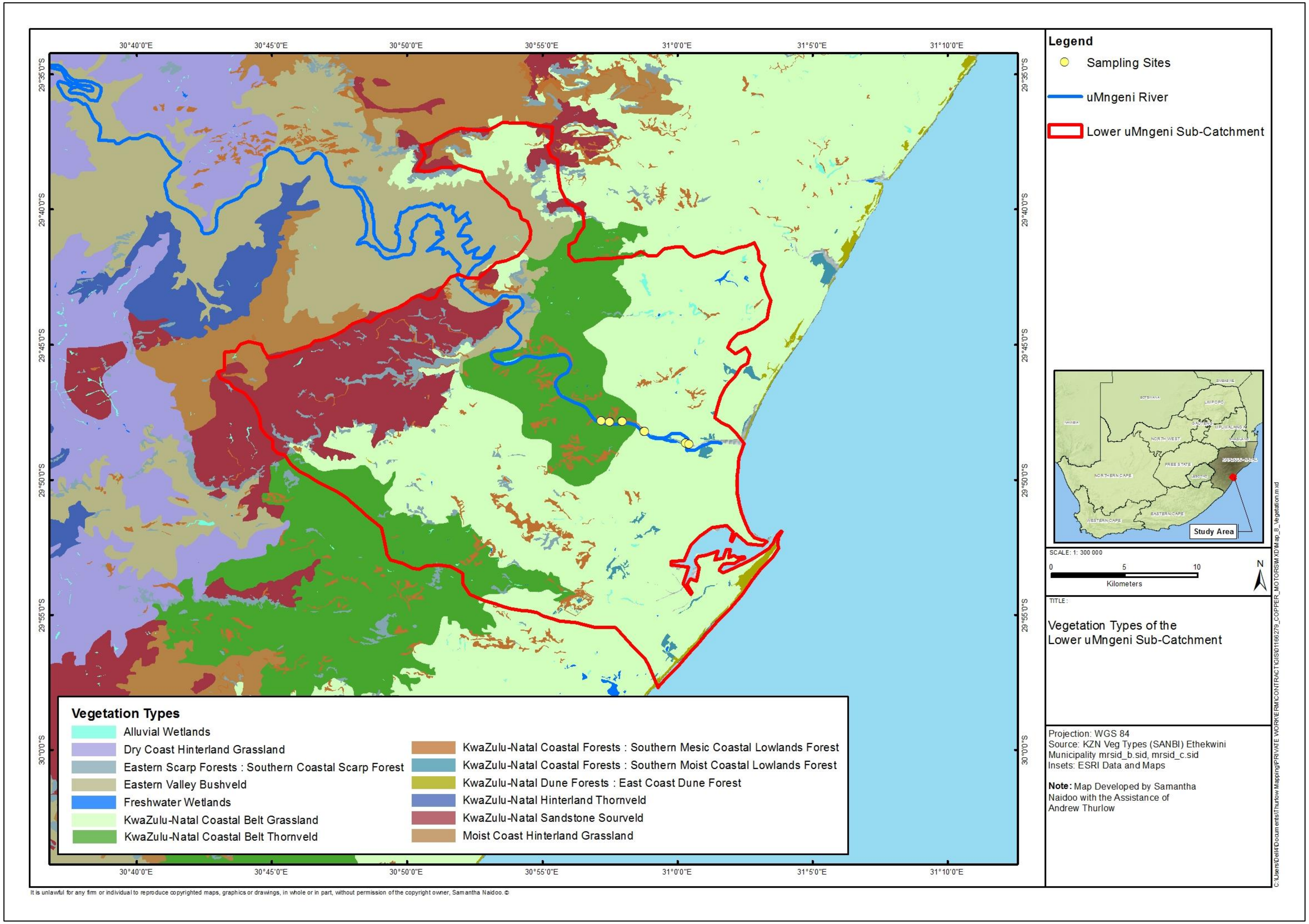


Figure 3.9: Vegetation of the Lower uMngeni sub-catchment

3.2.1.6 Socio-Economic Profile

Decreased water quantity and quality have significant environmental and financial cost implications due to the loss of ecosystem services. Increased monetary costs may also result from the potential need to purchase water from elsewhere and the additional water treatment and purification processes required (Engel *et al.*, 2011). According to Brijlal (2005), both the Greater Durban and Pietermaritzburg areas in KwaZulu-Natal depend significantly on the uMngeni River as a primary source of water supply for their communities. The river supports approximately 40% of the population within the Pietermaritzburg-Durban corridor alone (DWAF & Umgeni Water, 1996).

On the other hand, the multiple land uses within the catchment have significant, social and economic value, providing employment to a large proportion of local residents (WRC, 2002). This in turn provides motive for the migration of people from other regions and provinces into the area, resulting in population increases within the catchment (WRC, 2002). Considerable urban and rural development has markedly changed catchment demographics over recent years y, and will continue to do so in future (DWAF & Umgeni Water, 1996).

Noteworthy is the fact that potable water and ablution facilities are inaccessible to approximately one fifth of the catchment population (DWAF & Umgeni Water, 1996). These residents usually occupy informal areas within the catchment and depend largely on commercial and rural subsistence farming to sustain their livelihoods. Furthermore, regular refuse collection services are unavailable to approximately one third of the catchment population, posing significant water quality and human health risks. This is further compounded by the continued influx of people and population growth within the catchment (WRC, 2002; Brijlal, 2005). On the other hand, urban areas form hubs of formal residential, commercial and industrial activity which are currently the highest water users in the catchment (WRC, 2002). It is projected that the greatest future water demands will be related to the upgrade and formalisation of informal settlements and the development of further formal residential areas to accommodate increasing populations (DWAF & Umgeni Water, 1996).

In keeping with the general trend of income distribution on a national scale, income distribution in the uMngeni catchment is extremely skewed, with residents of informal areas receiving less than R 1250 per month whilst the average monthly income of urban dwellers is in excess of R 3000 (DWAF & Umgeni Water, 1996). Overall, approximately 30% of the uMngeni catchment's population receives a monthly income that is lower than the subsistence level of R 750 per month (DWAF & Umgeni Water, 1996). These community members usually reside in informal dwellings where access to basic services is limited (DWAF & Umgeni Water, 1996).

It is therefore not surprising that residents of the informal settlements, situated within the catchment, depend largely on the resources provided by the uMngeni River and surrounding features for sustenance. For instance, subsistence fishing contributes considerably to the well-being of poorer communities within the uMngeni catchment (WRC, 2002). The members of some rural households also collect plant species that flourish within the riparian zone to be used as raw materials in their craft-making. They derive monetary benefits by selling these crafts to other residents and tourists, which in turn supplements their income and contributes towards their financial stability (WRC, 2002).

With regards to economic production within the catchment, approximately 55% is service-based, 30% is derived from manufacturing and construction, 3% from agricultural practices and an estimated 12% from informal sector activities (DWAF & Umgeni Water, 1996). The gross geographic product value, including the value produced by non-agricultural activities occurring beyond the catchment boundaries but supplied with water from within the catchment, represents approximately 70% of the total gross geographic product of KwaZulu-Natal (DWAF & Umgeni Water, 1996).

Furthermore, the uMngeni River and surrounding areas provide several socio-economic benefits in the form of opportunities for tourism and both passive and active recreation (WRC, 2002). Inclusive are fishing, angling, swimming, bird watching, and the renowned "Dusi Canoe Marathon" from Pietermaritzburg to Durban, which is known to attract approximately 2000 paddlers annually (WRC, 2002; Brijlal, 2005). Figure 3.10 below captures participants of the Dusi Canoe Marathon that took place in March 2011. Furthermore, the uMngeni Bird Park, Beachwood Mangroves and the Midmar/uMngeni Valley Nature Reserve have considerable educational and research value (WRC, 2002; Brijlal, 2005).

Figure 3.10: Participants of the Dusi Canoe Marathon canoeing: along the lower reaches of the uMngeni River, in close proximity to the Recreational Field on Siripat Road within the Reservoir Hills area (March, 2011)

3.2.2 Wetlands of the uMngeni Catchment

As part of the existing "Mgeni Catchment Management Plan" (MCMP), a study was conducted with inputs from a number of organisations and stakeholders to inform conservation strategies for wetlands within the uMngeni catchment (Jewitt & Kotze, 2000). According to the study, a total of 169 wetlands occur within the catchment and cover 6227 ha (5.7%) of the catchment area (Jewitt & Kotze, 2000). Almost all of the identified wetlands are riparian wetlands and are associated with the river network (WRC, 2002). All of these wetlands were found to be palustrine, and emergent, with natural tree and shrub wetlands being largely absent (Jewitt & Kotze, 2000).

The findings of the investigation indicated that anthropogenic activities within the catchment resulted in 66% of the wetland area being lost (Jewitt & Kotze, 2000; WRC, 2002). Historical loss across the catchment showed an increase from upper to lower altitudinal zones such that 47% (intermediate level loss), 67% (high level loss) and 73% (high level loss) of the original wetland area was lost in the upper altitudinal zone, mid-altitudinal zone, and lower altitudinal zone respectively (Jewitt & Kotze, 2000; WRC, 2002). Furthermore, it was found that alien plant infestation within wetlands in the catchment decreased with increasing altitude, therefore posing the greatest problem within the lower altitudinal region (Jewitt & Kotze, 2000). The lack of more recent information necessitates that present day investigations be undertaken to ascertain the percentage of wetland loss and *status quo* conditions of wetlands within the various altitudinal zones in relation to results obtained from the studies conducted in 2000 and 2002.

Figure 3.11 portrays the wetlands situated within the Lower uMngeni sub-catchment. These wetlands are ecologically important systems that provide refuge to numerous plant and animal species (Brijlal, 2005). However, the large number of remaining wetlands within the uMngeni catchment continues to be threatened and degraded by human activities (DWAF and Umgeni Water, 1996). It was also established that the greatest loss of wetlands were a result of drainage and cultivation, due to the generally fertile soils and favourable locations for irrigation, followed by permanent flooding by dams (Jewitt & Kotze, 2000). Other significant contributors to degradation of the uMngeni floodplain wetlands include artificial drainage, urbanisation, alien plant invasion, overgrazing and frequent burning of riparian plants (WRC, 2002).

The uninhibited degradation and loss of the uMngeni wetlands is a cause for much concern in light of the important ecosystem services that they render, including water purification, erosion control and water flow regulation. Furthermore, these wetlands provide resources which a number of people rely on as a source of income (WRC, 2002; Schuyt, 2005).

According to the NWI and NFEPA project, the wetlands falling within the Lower uMngeni sub-catchment, in the vicinity of Durban, are categorised as floodplain, channelled valley-bottom, or unchanneled valley-bottom wetlands; all of which occur within a “Valley Floor” type of landscape (Macfarlane *et al.*, 2008). The wetlands of particular interest in this study have been identified as floodplain wetlands (as discussed in Chapter 2). This portion of the river channel has been highly disturbed and modified by the multitude of human activities that characterise its peripheral catchment area.

As illustrated in Figure 3.12 below, urbanisation and industrialisation form the dominant land use activities, covering approximately 53% of the specific area of interest for this study, hereinafter referred to as the “study area”. Grasslands and marine water are also poignant features, constituting approximately 20% and 6% of the study area respectively. Commercial sugarcane plantations, rural dwellings, national roads, the golf course and grassland and bush clumps occupy much smaller surface areas, whilst wetlands amongst other features are shown to be least prominent within this zone. It can thus be postulated that the wetlands of interest in this study not only receive return flow from agricultural, informal residential and recreational land uses, but are also likely to receive input from dense urban and industrial activities, that cover more than 50 times the area covered by the wetlands themselves. Reference can be made to Appendix A which presents the findings of the GIS pixel count.

Figure 3.11….\

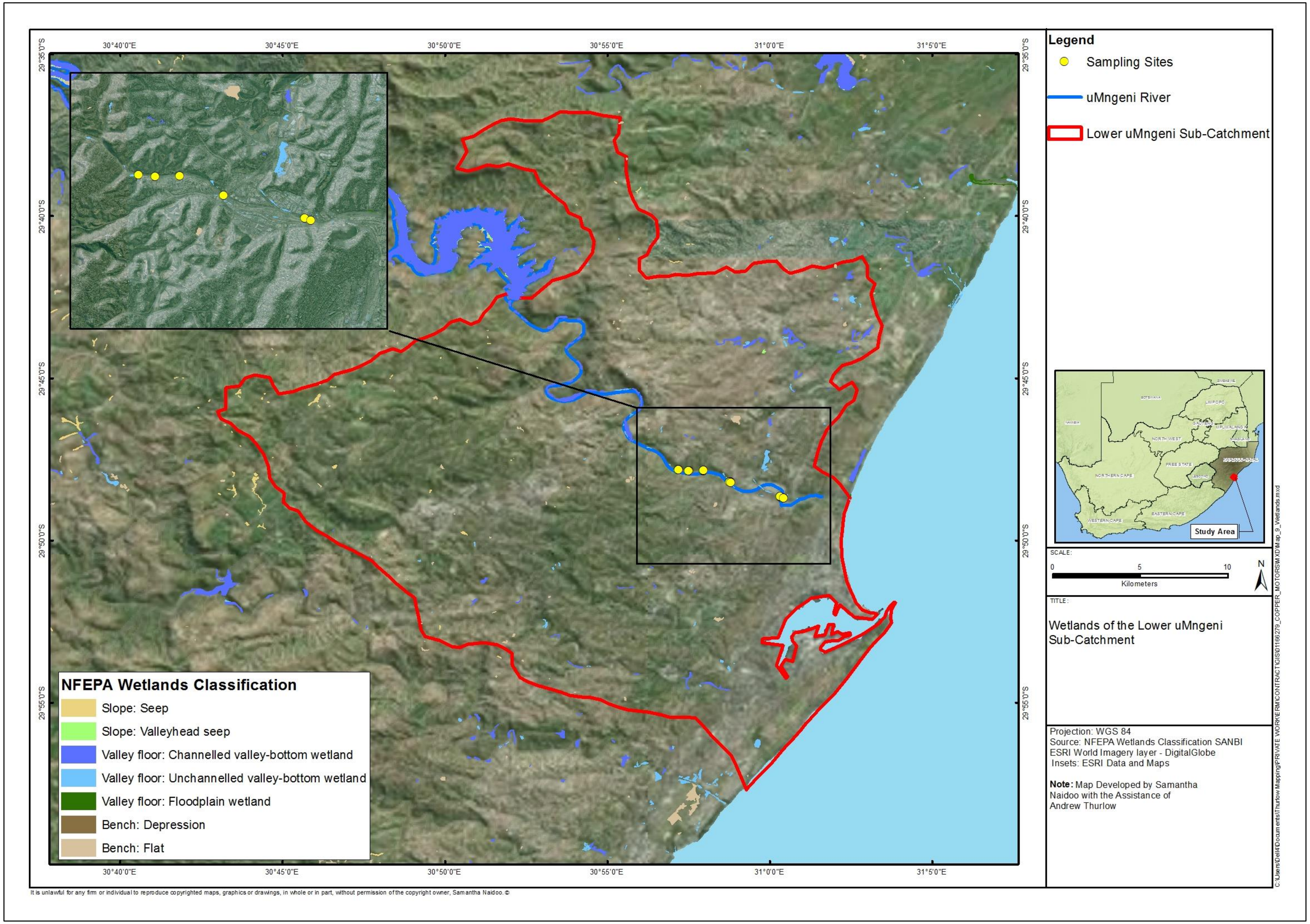


Figure 3.11: Wetlands of the Lower uMngeni sub-catchment

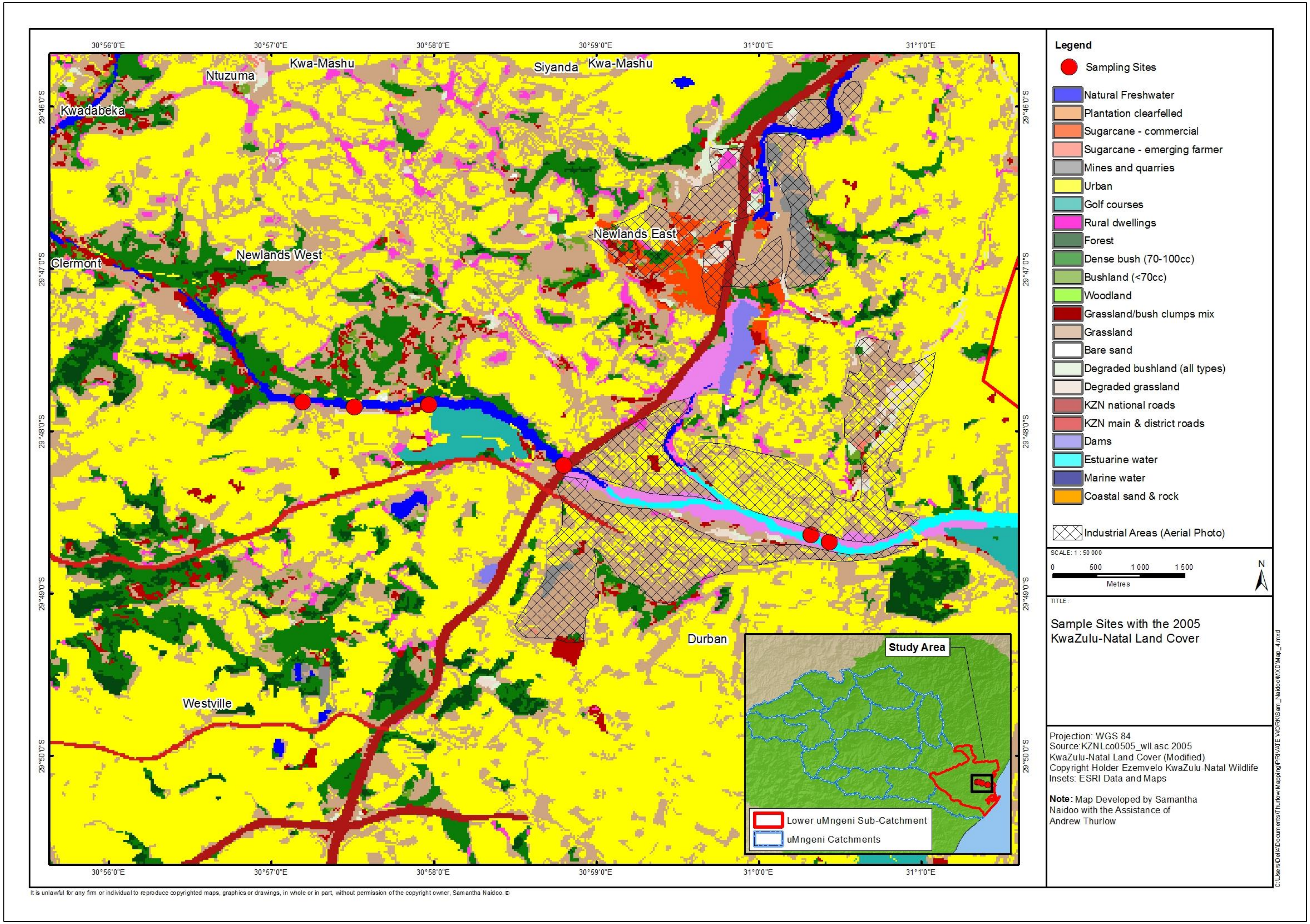


Figure 3.12: Land uses within the immediate vicinity of the wetlands and sampling sites relevant to this study

3.3 Existing Management Plans & Institutional Arrangements

In 1993, Umgeni Water and the Department of Water and Sanitation jointly initiated the MCMP (Karar *et al.*, 1999). The uMngeni Catchment was the first in South Africa in which integrated catchment management strategies were applied (Karar *et al.*, 1999).

More than ten international projects and countries participated in an initiative to evaluate the water resource management practices implemented in the catchment against eight key principles for "effective water resource management" identified by the International Water and Sanitation Centre (IRC) and the United Nations Development Programme (UNDP) (Karar *et al.*, 1999). These eight principles were: (1) water source and catchment protection are essential; (2) water must be adequately allocated; (3) water must be used efficiently; (4) management must be delegated to the lowest appropriate level; (5) all stakeholders need to be involved in decision making; (6) there should be gender equity in water resources management; (7) skills and capacity should be built; and (8) water should be treated as an economic and a social good (Karar *et al.*, 1999). The study identified strengths and weaknesses in the implementation of each principle (Karar *et al.*, 1999). Ideally, weaknesses identified should be used to amend existing management actions or inform additional management actions for inclusion in the MCMP.

Considering that the findings of the MCMP (DWAF & Umgeni Water, 1996) highlighted issues of soil loss and water quality and quantity decline, a number of management strategies to curtail these negative impacts were also proposed in the Plan. However, identification of the responsible parties, estimation of potential costs, and ascertainment of physical and social sustainability of the strategy as well as time frames for their implementation is the responsibility of the respective catchment management forums (DWAF & Umgeni Water, 1996). Management strategies for the entire catchment, as specified in the MCMP, which are relevant and applicable to the study area include:

- Optimise the yield from the Mgeni system through on-going analysis, taking consideration of water quality issues;
- Provide a framework within which to manage groundwater recharge, utilisation, discharge and contamination throughout the Mgeni Catchment;

- Remove alien invasive vegetation from river courses and rehabilitate the riparian zone (DWAF & Umgeni Water, 1996).

The MCMP also specifies management strategies that are particularly pertinent to the Lower uMngeni sub-catchment (previously referred to as the Durban Management Unit) and therefore the study area, which include the following:

- Monitor and assess lead and chromium concentrations in Durban and the estuary;
- Rehabilitate rivers and aquatic vegetation through Durban to provide a healthy resource base, flood attenuation and pollution assimilation; and
- Remove alien riparian vegetation and gross pollutants (DWAF & Umgeni Water, 1996).

To be noted is that documents detailing amendments of and additions to existing strategies, and progress regarding implementation of these strategies from the year 1996 to 2014 are unavailable. Amendments that may have been made during this period include the specification of responsible parties, approximate cost, sustainability and time frames for each proposed strategy.

Furthermore, a water quality management plan for the uMngeni catchment, namely the Mgeni Catchment Water Quality Management Plan (MCWQMP), as well as a GIS-based Catchment Management Information System (CMIS), were initiated by DWS and Umgeni Water to assist in the management of water quality within the catchment (Jewitt *et al.*, n.d.; Randall & Dutlow, 1996). However, the MCWQMP document itself and GIS data constituting the CMIS, as well as any possible updates to the respective documents and data, are either inaccessible or unavailable within the public domain.

The eThekwini Metropolitan Municipality Environmental Services Management Plan (ESMP) (2001) highlights management actions that have been tailored to improve prevalent conditions within particular sections of the uMngeni River. Noteworthy is that management actions and tools for floodplains and wetlands within the area of study for this project have not been included in the ESMP. Furthermore, only high-level management actions have been proposed for other sections of the uMngeni River and are not accompanied by details on the assessment criteria, management methodology, responsible parties, frequency, timeframe and output for each recommended management action.

With regards to institutional arrangements, the Duzi-uMngeni Conservation Trust (DUCT), which is a non-profit organisation dedicated to championing the health of the uMngeni and uMsunduzi Rivers, is represented in a number of public forums (DUCT, 2014). Amongst these public forums is the Lower uMngeni Catchment Management Forum, which is responsible for discussing and addressing all water resource issues experienced in the Lower uMngeni sub-catchment (DUCT, 2014). This forum, also referred to as DUCT's Durban Working Group, constitutes representatives of the eThekwini Metropolitan Municipality, DWS and a number of non-profit organisations (NGOs) (DUCT, 2014). Members of the forum have been meeting bi-monthly since 2006 to discuss water resource issues, propose corrective actions and provide updates on the progress of implemented management actions (DUCT, 2014). However, minutes of all these meetings have not been made available to the public. Furthermore, there is no indication of whether the status of water resource issues is being tracked, and whether proposed management measures are being documented in formal plans for approval prior to execution.

Given the above, it can be stated that a comprehensive management plan for the Lower uMngeni sub-catchment has not yet been developed, or is otherwise unavailable or inaccessible to the public. Minutes of all the Lower uMngeni Catchment Management Forum meetings, which should ideally inform amendments to the management plan for the Lower uMngeni sub-catchment, have either not been released into the public domain or are inaccessible. Similarly, the MCWQMP and data informing the CMIS, as well as all associated amendments to these documents and data are unavailable to the public. Furthermore, there is no available evidence indicating that a rehabilitation plan, specifying appropriate management and monitoring measures, for the wetlands delineating the Lower uMngeni River has been developed.

3.4 Conclusion

The characteristics of the uMngeni catchment and fluvial system are described in this chapter including their general geographic location, spatial extent, ecological attributes and dominant climatic conditions. Also detailed in this chapter are the hydrological, geological and soil characteristics of the catchment and more specifically the Lower uMngeni sub-catchment. The history of wetlands within the catchment, their current status, occurrence and classification are highlighted. The exact location of the study area within the Lower uMngeni sub-catchment is delineated and the dominant land uses within these boundaries are identified. Apart from the environmental attributes, the socio-economic profile of the uMngeni catchment is also provided, and existing management plans and institutional arrangements discussed.

CHAPTER 4: RESEARCH METHODOLOGY

4.1 Introduction

In this chapter, the methods of data collection and analysis employed for the study are specified. The location of the selected sampling sites, and the timing and frequency of sampling are indicated. Laboratory procedures conducted, including chemical and granulometric analyses processes, are elucidated. This is followed by an explanation of the statistical and pollution assessment methods of data analysis which were subsequently undertaken.

4.2 Site selection

A total of 6 sampling sites were selected for sample collection. These points were selected based on accessibility and strategically positioned within each of the major riverine wetlands occurring along the periphery of uMngeni River, extending from approximately 3.4 km upstream of the river mouth at Blue Lagoon north-westwards towards Reservoir Hills. Furthermore, the location of these wetlands within 2 distinctly differing zones of the Lower uMngeni sub-catchment, namely the residential and industrial zones, will inform inferences regarding the possible contribution of land uses within these zones to the health of the riverine and wetland systems. The geographical location co-ordinates of each site were recorded using the Magellan Meridian Marine Global Positioning System (GPS) (Refer to Appendix B for GPS co-ordinates of the sampling sites).

At each site, piezometers were installed in each of the major wetlands. Water samples were collected from the river at points either adjacent or proximal to each piezometer, and sediment cores were extracted a short distance up-bank from the installed piezometers.

Figure 4.1 illustrates the geographic location of the river water, wetland interstitial water and wetland sediment sampling points along the Lower uMngeni River, thereby marking the extent of the study area.

Figure 4.1: Sampling sites selected along the Lower uMngeni River

4.2.1 River water and wetland interstitial water sampling sites

The six river sampling points selected are labelled R1, R2, R3.... R6, where 'R' denotes 'River' and the number that follows makes reference to one of the six selected sites. Likewise, the wetland interstitial water sampling points, where piezometers were installed, are referred to as W1, W2, W3... W6, where 'W' indicates 'Wetland'. Sites R1 to R4 and sites W1 to W4 are situated along or in close proximity to the southern banks of the uMngeni River, whilst R5, R6, W5 and W6 occur along or close to the northern banks of the river.

Sites R1 and W1 are furthest upstream, located approximately 3 km away from the N2 bridge travelling towards Reservoir Hills. The site can be easily accessed *via* Siripat Road, which if followed to the very end leads to the property of a manufacturing business. The portion of the property closest to the river bank is used as a storage area for materials utilised by the company. R1 is located in the river adjacent to this storage area and is slightly disturbed due to the creation of a man-made pathway to allow easy access to the river. Hence, rubble and rocks, although not present in overwhelmingly large quantities, are found along the steep river bank. The wetland however, is minimally disturbed, with characteristic tall reeds and wetland vegetation in this area being lush, predominantly intact and undisturbed.

Sites R2, R3, W2 and W3 can also be accessed *via* Siripat Road, and occur opposite the recreational field and the Papwa Sewgolum golf course respectively. Site R2 is situated approximately 0.5 km east of R1, and R3 is located approximately 0.7 km eastward of R2.
Informal settlements and subsistence agricultural activities are located a short distance away from R2, and encroach slightly upon the wetlands represented by site W2. The wetlands in this area are highly disturbed as a result of the proximal farming practices and the movement of people to and from the river. The vegetation however remains tall and dense, with signs of clearing only present along footpaths.

Figure 4.2 below depicts the selected sediment and water sampling sites in relation to the surrounding land uses and land cover features.

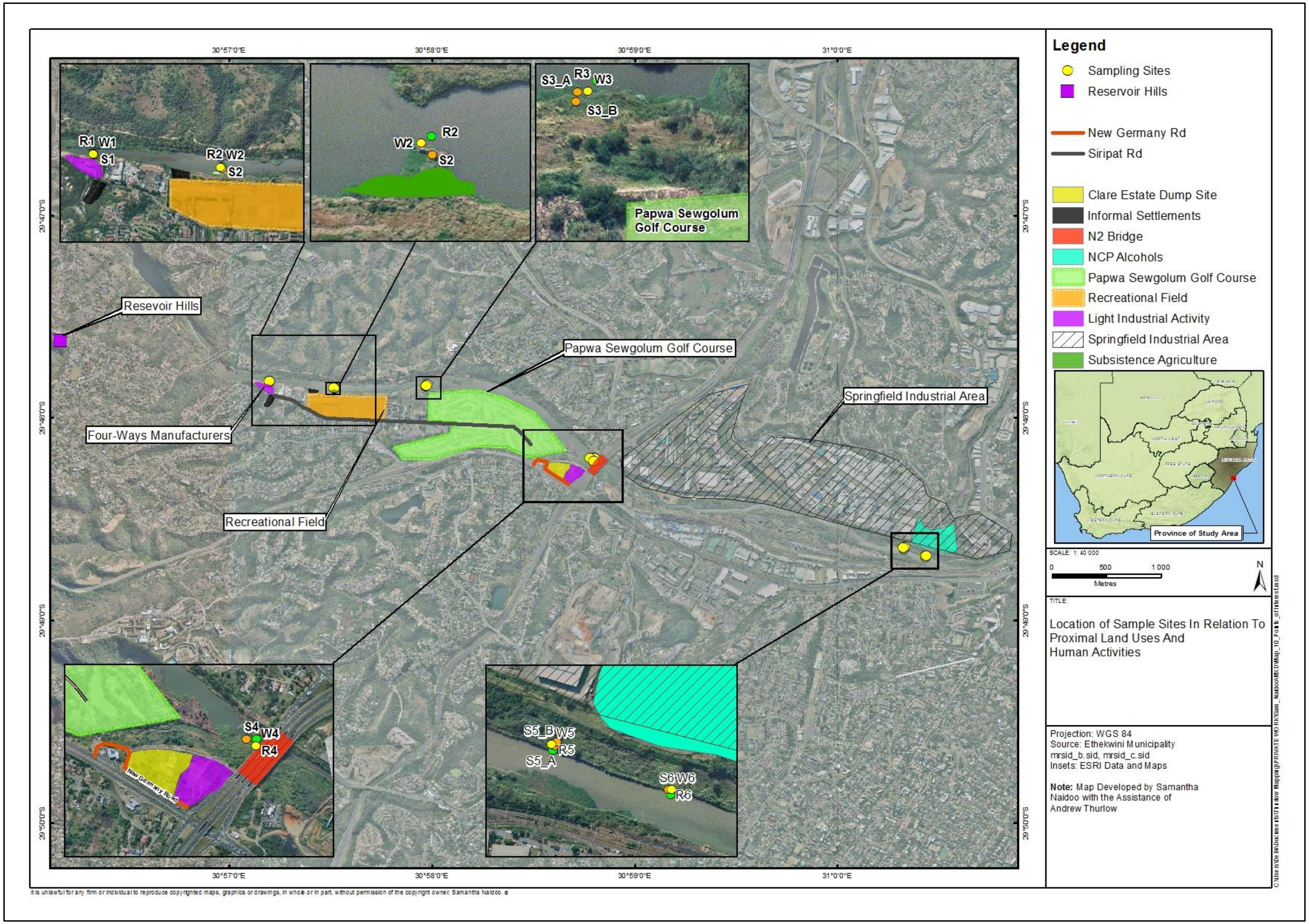


Figure 4.2: Sampling sites in relation to land uses and land cover features in the study area

Figure 4.3(a): Informal settlements within the immediate vicinity of sample sites S2, R2 and W2, opposite the recreational field on Siripat Road **(b)** Proximity of some low-lying settlements to the river channel, represented by site R2, and thus the wetland area represented by sites S2 and W2.

Figures 4.3(a) and (b) portray the proximity of some informal dwellings to the river channel and thus existing wetlands represented by sample sites R2 and W2 respectively.

Sampling points R3 and W3 are located opposite the Papwa Sewgolum golf course. Apart from the man-made footpaths and signs of vegetation burning in the upper reaches of the river bank, no indication of substantial anthropogenic disturbance are present at this site. However, very sparsely located and single-standing informal dwellings on the northern banks of the uMngeni, within the vicinity of sites R3 and W4, are possible sources of contamination.

Sites R4 and W4 are located under the N2 bridge , within the vicinity of the Clare Estate dump site and light industrial activity, approximately 1.5 km seaward of sites R3 and W3.

These sites can be easily accessed *via* New Germany Road, which becomes a gravel road when travelling in an easterly direction towards the bridge. The wetland is characterised by dense riparian vegetation, with no signs of clearing or disturbance. A short distance upstream of R4, however, dumping of large quantities of domestic waste in the system has been noted, as depicted in Figure 4.4(a). Figure 4.4(b) shows children swimming in the river near site R4, despite the obvious visual signs of solid waste pollution. This highlights the importance of maintaining acceptable water quality at all reaches of the river system.

Furthermore, the floodplain in this section of the study area has been transformed into untarred roads for the movement of heavy vehicles to and from the light industrial activity situated in close proximity to the N2 Bridge. The result is significant erosion, and slumping due to soil compaction.

Figure 4.4(a): Improper disposal of solid waste near the N2 bridge, in close proximity to sampling sites S4, R4 and W4 **(b)** Children swimming in a portion of the river near the N2 bridge, near sampling sites S4, R4 and W4.

Sites R5, R6, W5 and W6 are situated opposite the Springfield Flats industrial area, near Natal Chemical Producers (NCP), also referred to as NCP Alcohols, in Sea Cow Lake. Sites R5 and W5 are about 2.7 km downstream of R4 and W4, and sites R6 and W6 are approximately 0.3 km downstream of R5 and W5. As shown in Figures 4.5 (a) and (b) below, the vegetation in this area is largely undisturbed, with dense, tall riparian flora characterising the wetlands and river banks. Disturbance of vegetated areas at these sites are limited to clearance for footpaths and the periodic trimming and/or removal of tall plant species. It is possible that residents of near-by informal dwellings, including those sparsely located within the vicinity of these sites, are responsible for the clearing of riparian vegetation on and along footpaths for ease of access to the river. It is also likely that the River Care Team, created by DUCT, have cleared footpaths for easy access to sections of the riparian habitat that require invasive alien weed control, trash collection and removal, and inspection for the reporting of illegal dumping and pollution (Ward, 2013).

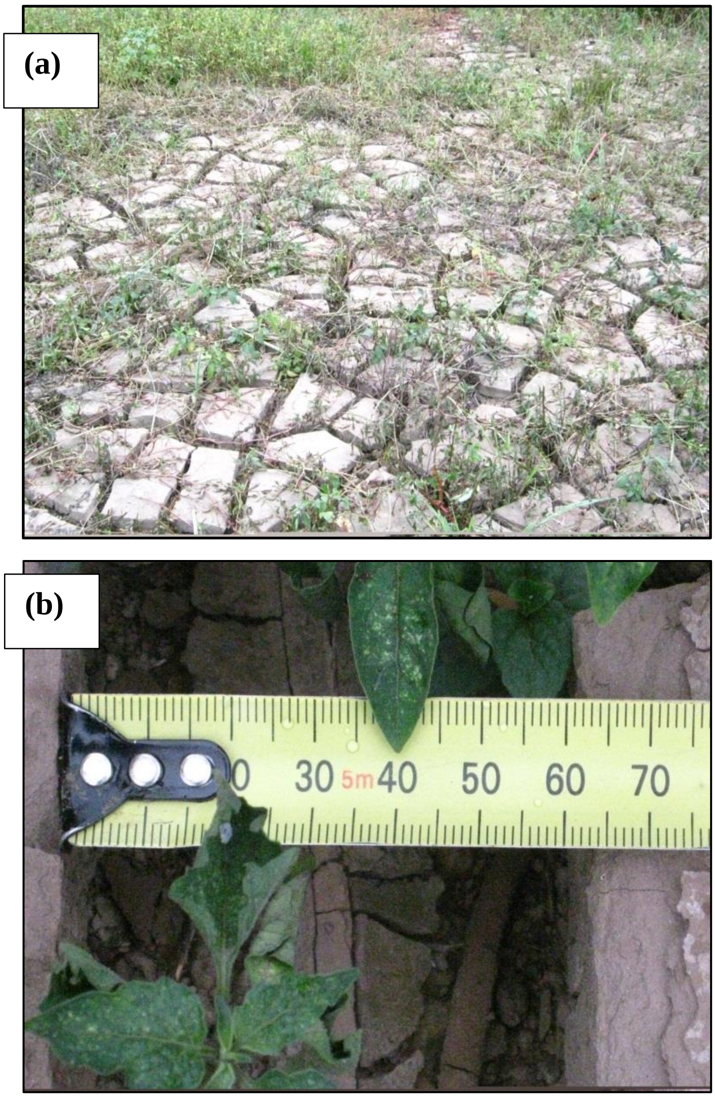
Figure 4.5(a): Dense riparian vegetation and a cleared footpath in close proximity to sampling points S6, R6 and W6, opposite NCP Alcohols in the Springfield Industrial Area **(b)** Dense riparian vegetation within the vicinity of sampling points S6, R6 and W6 to the south of the Springfield Industrial Area.

Inspection of the sampling sites and their surrounds during site visits revealed that a number of alien plant species, including *Ricinus communis L.*, commonly known as the castor oil bush, infringe the uMngeni River banks and wetland areas. Figure 4.6 depicts one such a specimen. Although they were noted to occur within the surrounds of all sampling sites, they were particularly well-established and dense within the vicinity of sites S5, R5, W5, S6, R6 and W6.

Figure 4.6: Castor oil bush, *Ricinus communis L*. noted within the vicinity of sampling sites S5, R5, W5, S6, R6 and W6, near the Springfield Industrial Area

4.2.2 Soil sampling sites

Soil sampling sites were situated between 2-10 m up-bank of the corresponding river and wetland interstitial water sampling points. Location of the soil sampling points was largely dependent on the width of the wetland in each case, and the ease of soil auger and corer penetration along the delineated transects. In cases where the soil along the established transects was impenetrable, soil cores were extracted from areas containing permeable soil in closest proximity to transects. For instance, Figures 4.7(a) and (b) illustrate the hard and impermeable nature of the soil in close proximity to sample site S4. These figures depict fine grained alluvial sediment, containing significant clay content with high swell-shrink properties. Upon drying out, considerable shrinkage has taken place, forming a large network of cracks. Although the floodplain at this site is approximately 87 m wide, the dense clays, and thin soils with underlying rock strata and rubble deposits created unsuitable conditions for the extraction of an additional sediment core further up-bank, in parallel to sample site S4. The sediment sampling sites are labelled S1, S2, S3A, S3B, S4, S5A, S5B and S6.



Figures 4.7(a) and (b): Illustrate dry, hardened and impenetrable top soil layers situated land-ward of the sampling sites S4, R4 and W4 near the N2 bridge. Cracks of between 6 - 6.5 cm wide characterised the first lamina of soil in this area.

In cases where the wetland of interest was relatively wide, two soil cores along the respective transects were collected instead of one. Two soil cores were collected at site S3 near the Papwa Sewgolum golf course and at site S5 opposite the Springfield Flats industrial area. The soil cores were thus labelled S3AS3B S5A and S5B. Sample sites S3A and S5A are located to the south of S3B and S5B respectively. In the case of sample site S6, although the wetland is approximately 35 m wide, the hard, dry, impermeable upper layers of gravel further up-bank made the collection of an additional sample in this area impossible. As a result, only one sediment core was extracted at this site.

4.3 Sampling protocol

The sampling period extended from March 2012 to January 2013. Piezometers were installed in March 2012, and river water and wetland interstitial water samples were collected on a monthly basis during the above-mentioned period. These samples were contained in sterilised 1L polyethylene bottles and kept at cool temperatures to prevent changes in water quality parameters resulting from changes in water temperature. The samples were thereafter transported to the laboratory in the School of Chemistry at the University of KwaZulu-Natal (Westville Campus) where all chemical analyses were conducted.

Soil coring was undertaken in February 2012. Open-ended PVC pipes (2 m in length and 4 cm in diameter) were driven into the wetland soil using a 5 kg metal weight. Once driven to a depth of approximately 1.5 m, depending on depth to the water table, the open end of the pipe was sealed with a custom-made cap and taped over with buff tape. The core was then carefully extracted by pulling the PVC pipe out of the ground. At times this required the assistance of two field workers; one to pull the PVC pipe out of the ground and the other to quickly seal the rear end of the PVC pipe to prevent soil loss. Once the PVC pipes were extracted, they were sealed securely at both ends and labelled with Redfern 70 mm x 37 mm adhesive labels for ease of reference. Details, including the date, time, GPS co-ordinates, and sampling site number were recorded on the labels.

Relevant observations, including the prevalent conditions in the immediate environs of each soil sampling site, were also recorded in a notebook. Once all soil cores were collected and labelled, the assistance of two field workers were required to carry them to the vehicle. The sediment samples were then transported to the Soils Laboratory in the School of Environmental Science at the University of KwaZulu-Natal (Westville Campus).

4.4 Physical Analysis

Once transported to the University of KwaZulu-Natal, the soil cores were sliced longitudinally in half at the Engineering Workshop before being taken to the Soils Laboratory. Thereafter, the soil laminae comprising each sediment core were carefully logged based on differences in morphological characteristics between each visible lamina. The attributes and thickness of each lamina were recorded. Thereafter, each lamina was carefully removed with a spatula. Two samples were taken from each identified lamina, one for determining the physical properties of constituent sediments including moisture content, organic matter content and calcium carbonate content, and the other for chemical analyses. These were placed in separate plastic bags and tightly sealed prior to analyses.

4.4.1 Moisture Content

The following procedure was followed for each of the sediment sub-samples:-
An empty sterilised 20 mL polyethylene scintillation vial was weighed and the weight was recorded. Thereafter, a 1.0 cm^3 core sub-sample was placed on the scintillation vial. The mass of the wet sub-sample including the weight of the scintillation vial was determined and recorded. Thereafter, the mass of the sediment sub-sample was calculated by subtracting the weight of the scintillation vial from the corresponding weight of the vial containing the wet sub-sample. The scintillation vial was labelled with a Redfern 70 mm x 37 mm adhesive label for ease of identification and distinction between other sub-samples.

Following completion of the above process for each sediment sub-sample, all sub-samples were then dried overnight at 110°C in a low-temperature oven (Rosenmeier, 2005). After overnight drying, samples were kept in a low-temperature oven at 60°C prior to weighing, with the aim of preventing moisture absorption from the ambient atmosphere (Rosenmeier, 2005).

Subsequent to heating, weights of each vial containing the dry sediment sub-sample were recorded. Moisture loss was therefore calculated as the difference between the wet sample weight and dry sample weight subsequent to drying at 60°C (Reeb and Milota, 1999):

$$\textit{Water loss (g)} = \textit{(Sample Wet Weight)} - \textit{(Sample Dry Weight)} \qquad (1)$$

4.4.2 Organic Matter Content (OMC)

Loss on ignition is one of the most commonly utilised procedures to ascertain the organic matter content of sediment core sub-samples (Heiri *et al*., 2001). The total organic matter content of sub-samples can be determined, subsequent to heating at 60°C, by measuring the weight before and after further ignition at a temperature of 550°C (Rose Rosenmeier, 2005).

Sediment samples contained in the scintillation vials from the procedure undertaken to determine moisture content were transferred to a requisite number of clean ceramic crucibles. These crucibles were pre-dried in the low-temperature furnace at 60°C for a period of 4 hours (Rosenmeier, 2005). Each dried crucible was weighed both prior and subsequent to the transfer of sub-samples, and the respective weights were recorded. The crucibles were then labelled with a permanent marker for ease of distinction between the sub-samples. The sub-samples were thereafter placed in a muffle furnace and ignited at a temperature of no less than 550°C for 2 hours to facilitate the oxidisation of organic matter to carbon dioxide and ash (Maher *et al*., 2012).

Subsequent to ignition, a period of approximately 2 hours was required for cooling of the sub-samples to a temperature that is suitable for safe handling. Thereafter, the crucibles were carefully removed from the furnace. Weights of the dried crucibles containing the post-550°C sediment were recorded. Organic matter content was therefore calculated as the mass difference between the sediment dried at 60°C (determined in the procedure used to determine moisture content) and the ash produced following ignition at 550°C (Veres, 2012). Hence, the following equation was used to establish the percentage of total organic matter content in sediment sub-samples (Rosenmeier, 2005):

$$\%\ OM = \frac{\text{(Weight of post 550°C Ash)}}{\text{(Weight post 60°C Dry Sample)}} \times 100 \qquad (2)$$

4.4.3 Calcium carbonate content

The loss on ignition method of analysis was also employed to determine the calcium carbonate content of sediment sub-samples. Following the ignition of sub-samples at 550°C to establish organic matter content, sub-samples were heated in the muffle furnace for a further 2 hours at 1000°C to determine their respective calcium carbonate contents (Heiri *et al*., 2001; Wang *et al*., 2012). Following firing at 1000°C, the ignited sub-samples were transferred to the low-temperature drying oven, which was set to 60°C, for a 2 hour cooling period (Maher, 1998). Here again, the weights of the dried crucibles containing the post-1000°C dried sediment were recorded (Rosenmeier, 2005). The calcium carbonate content of sub-samples were therefore determined by calculating the difference in weights between sub-samples heated at 550°C and thereafter at 1000°C (Maher, 1998). The following mathematical expression was used (Rosenmeier, 2005):

$$\%CaCO_3 = \frac{(\text{Weight of Post } 550°C \text{ Ash} - \text{Weight of Post } 1000°C \text{ Ash})}{(\text{Weight Post } 60°C \text{ Dry Sample})} \times 2.274 \times 100 \qquad (3)$$

4.4.4 Granulometric analysis

Textural analysis of the sediment sub-samples was carried out *via* the dry sieving method (Di Stefano *et al*., 2010) using a Retsch® sieve shaker. Sediment sub-samples were dried in the low-temperature oven at 110°C for 48 hours (Rosenmeier, 2005). They were subsequently disaggregated, using a pestle and mortar, and thereafter split into two portions. The first portion of the split samples was sent to the School of Chemistry at the University of KwaZulu-Natal (Westville Campus) for chemical analyses. The second portion, on the other hand, was used to carry out the granulometric analysis. In order to perform the granulometric analysis, samples were placed on the uppermost member of a column of metal sieves, and passed through sieves of aperture sizes 2 mm, 1 mm, 0.5 mm, 0.25mm, 0.125 mm and 0.053 mm from top to bottom respectively. These sieves were manually shaken for 8-10 minutes. To be noted is that the period of shaking was largely dependent on sample size. After sieving, the quantity of sediment retained on each sieve was weighed, recorded, and used to plot particle size distribution graphs.

4.5 Chemical analysis

A series of laboratory procedures were performed to ascertain the concentration of specific parameters in river water samples, wetland interstitial water samples and the portion of each sediment sample set aside for chemical analyses. . Water samples were tested to measure pH, and the level of Total Dissolved Solids, Dissolved Oxygen, Ammonium ions, Aluminium, Vanadium, Barium, Chromium, Nickel, Cobalt, Iron, Zinc, Lead, Copper and Manganese. Sediment samples were also analysed to determine the presence and concentration of various elements, namely Zinc, Copper, Lead, Nickel, Chromium, Iron, Aluminium, Arsenic, Vanadium, Phosphorous, Calcium, Magnesium, Manganese and Sulphate ions.

4.5.1 In Situ Recordings

pH, TDS, DO and ammonium ion levels in water samples were measured on-site using the YSI 6920 Multi-parameter Sonde and the 650 MDS Multi-parameter Display System.

4.5.2 Ion Chromatography

The concentrations of sulphate ions in water and sediment samples were measured using Ion Chromatography. Ion Chromatography is an analytical technique used for the separation and measurement of ionic species in solution (Acikara, 2013). Water and sediment samples were prepared for introduction into the chromatograph by undergoing a series of pre-analysis steps.

Approximately 10 mL of each water sample was extracted using a sterile syringe, which was rinsed three times with sample water. The extracted water samples were then filtered through 0.45 µm filters to remove sediment and other particulate matter, and thereby limit the potential for microbial alteration of samples prior to analysis. The filtrate collection vial was rinsed three times with sample filtrate before being used to contain the rest of the filtrate. Samples were then stored in a refrigerator, set to a low temperature, until being removed for processing. In the case of sediment samples, representative liquid samples were extracted through the addition of acid, which acts to remove ions from the sample surface. The liquid extracted from each sample was similarly filtered through 0.45 µm filters, refrigerated at a low temperature, and removed for analysis. Figure 4.8 below illustrates the analysis process followed for each sediment and water sample prepared.

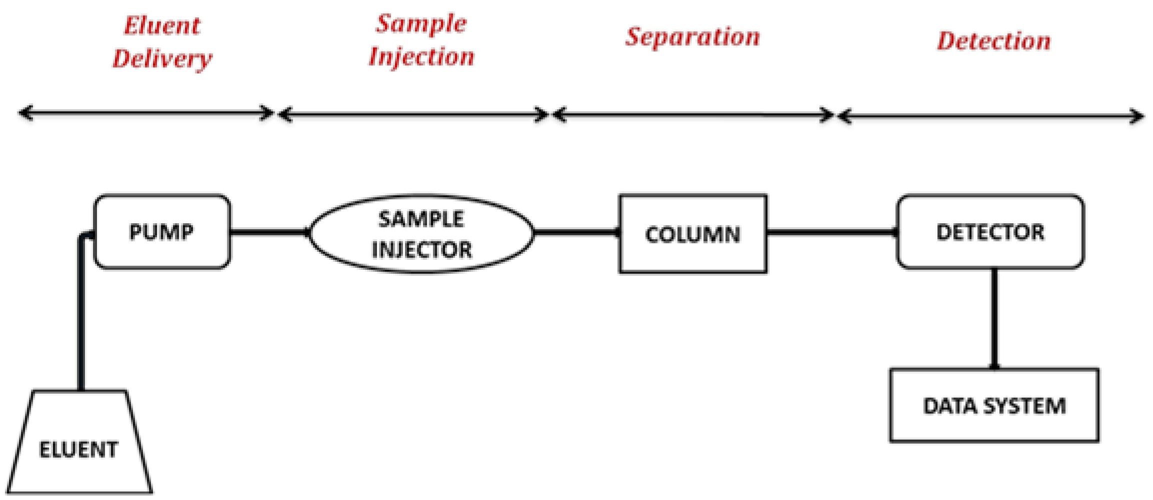


Figure 4.8: Ion Chromatography Components and Analysis Process

As displayed in Figure 4.8 above, a high pressure pump was first used to force a liquid carbonate/bicarbonate eluent through a solid stationary phase (an ion-exchanger in the form of a resin-packed column), and thereafter to a flow-through detector (Haddad & Jackson, 1990). The sample to be separated was injected into the flow-path prior to its entrance into the ion-exchanger (Haddad & Jackson, 1990). Ions in the sample were separated due to their differing affinities to the ion-exchanger resin. The conductivity of the eluent was restrained whilst passing through the ion-exchanger with the separated ions (Ahuja, 2009). This enhanced conductivity detection and measurement of separated ions by transforming them into highly conductive forms (Ahuja, 2009). They separated ions were identified using respective retention times, quantified by comparing calibration standards and their peak areas, and thus differentiated from each other.

The principal advantages of ion chromatography include its selectivity, speed of analysis, sensitivity of detection and its facilitation of simultaneous analysis (Khopkar, 1998).

4.5.3 Inductively-coupled plasma optic emission spectrometry

The Inductively-coupled plasma optic emission spectrometry (ICP-OES) method of analysis was utilised to ascertain the concentrations of barium, calcium, cobalt, aluminium, chromium, copper, iron, lead, magnesium, manganese, nickel, phosphorous, vanadium and zinc in water and sediment samples. ICP-OES is a common spectrophotometric technique that allows for the simultaneous determination of various element concentrations in water solutions using multiple wavelengths and calibration ranges (Boss & Fredeen, 1997).

At the outset, all necessary steps were followed to prepare the water and sediment samples for analysis. Water samples were filtered through 0.45 µm filters and thereafter acidified with nitric acid at a ratio of 1.5 mL of concentrated nitric acid to 1L of sample. Soil samples were prepared for analysis by measuring 0.5 g of each oven-dried sample and adding these to 15 mL of aqua regia (mixture comprising one part nitric acid and three parts hydrochloric acid). This mixture was then boiled for 30 minutes, cooled, filtered under gravity, and subsequently introduced to the ICP-OES machine.

The water and sediment samples were each introduced into the instrument by using an autosampler and peristaltic pump. A peristaltic pump comprises of three sampling streams which facilitates mixture of the sample stream with standards of ionisation suppressants (Ahuja, 2009). The mixed sample thereafter moved into a nebulizer where it was mixed with a plasma gas (argon) to form an aerosol. It subsequently moved into a spray chamber and a plasma stream sequentially (Pakade, 2000; Ahuja, 2009). The aerosol vaporised, evaporated and dissociated into atoms comprising of electrons that were 'excited' to higher energy levels at specific wavelengths, and emitted energy on their return to ground state (Ahuja, 2009). Noteworthy is that a given element emits energy at specific wavelengths based on its chemical character. The emitted energy or spectra at a specific wavelength was measured by the spectrometer using a charge-coupled device (Ahuja, 2009). Calibration curves were then attained for each element from the measured intensities of the metal concentrations. These curves were in turn used to determine the metal ion concentrations of the sample through substitution into the representative curve equation (Ahuja, 2009; Sukdeo, 2010).

The use of ICP-OES is widely favoured due to its multi-element determination ability; low detection limits; few inter-element effects; large analytical range; and its precise, accurate and fast method (Sukdeo, 2010).

4.6 Data analysis

4.6.1 Statistical analysis

Statistics allows researchers to interpret information or data in a scientifically meaningful manner (Naidoo, 2005 in Sukdeo, 2010). The statistical programme SPPS (version 15.0 for Windows) and Excel 2010 were utilised to statistically analyse parameter levels in water and sediment samples derived from laboratory analyses. Analysis of Variance (ANOVA) was used to assess the variance of elemental concentrations detected in sediment cores and the geochemical indices calculated. The correlation function was used to determine whether a correlation exists between concentrations of particular parameters in river and wetland interstitial water samples, and the seasonal variation thereof. Furthermore, the mean parameter levels detected in river surface water samples and wetland interstitial water samples for each season were compared to the acceptable limits, specified in the South African Water Quality Guideline Series developed by DWAF (1996a; 1996b).

4.6.2 Pollution assessments

Amongst the various indices of sediment pollution, the contamination factor (CF), the enrichment factor (EF) and the pollution load index (PLI) will be utilised to establish the level of sediment pollution within the study area.

a) Contamination factor (CF)

The contamination factor ratio is obtained by dividing the concentration of each metal of interest by its respective baseline or background value to establish the level of contamination by each element in sediments (Mmolawa *et al.*, 2011). The following equation can be used to determine CF:

$$CF = \frac{\text{(Concentration of element in the sediment)}}{\text{(Background value of element)}} \qquad (4)$$

The level of elemental contamination is based on their respective intensities, ranging from 0 to 6; where a value of zero indicates no contamination, and a value of 6 denotes very strong contamination (0= none, 1 = none to moderate, 2 = moderate, 3 = moderate to strong, 4 = strong, 5 = strong to very strong, 6 = very strong) (Bhuiyan *et al.*, 2010). Obtaining a value of 6 also indicates that the metal concentration is a hundred times higher than the expected concentration in the crust (Bhuiyan *et al.*, 2010).

b) Enrichment factor (EF)

According to Bhuiyan *et al.* (2010), the enrichment factor of a metal can be determined by dividing its ratio to the normalising element by the same ratio found in the chosen baseline. The enrichment factor can thus be computed by:

$$EF = \frac{(Metal/Fe)_{Sample}}{(Metal/Fe)_{Background}} \qquad (5)$$

Regional background values are used to determine enrichment factors (Mmolawa *et al.*, 2011). Although geochemical background values remain constant, levels of contamination vary with time and place, and are distinctly different among different soil types (Bhuiyan *et al.*, 2010). It should also be noted that the concentration of heavy metals in soil vary over 2-3 orders of magnitude, depending on the parent material (Ji *et a*l., 2008). EF values obtained that are close to unity (a value of 1) suggests that that the metals have a crustal origin, those less than 1 indicates a possible mobilisation or depletion of metals, and those greater than 1 indicates that the metals are of anthropogenic origin (Bhuiyan *et al.*, 2010). In this study, Iron (Fe) was used as the reference element for geochemical normalisation. Fe was selected for various reasons, of which include:

1) Fe is associated with fine solid surfaces;
2) Its geochemistry is similar to that of many trace metals; and
3) Its natural concentration tends to be uniform (Tajam and Kamal, 2013).

c) Pollution load index (PLI)

The extent of pollution at a specific site can be measured by using the Pollution Load index (PLI) (Harikumar and Jisha, 2010). PLI can be determined as the n^{th} root of the product of the n CF as follows (Mmolawa *et al.*, 2011):

$$PLI = (CF_1 \times CF_2 \times CF_3 \times \ldots \times CF_n)^{1/n} \qquad (6)$$

Where:

CF = Contamination Factor

n = number of elements

4.7 Conclusion

Subsequent to sample collection during monthly site visits, a number of analysis procedures were conducted to obtain measurements for particular metals and nutrients in water and sediment samples. Scientifically approved methods of chemical and physical analyses were conducted in the Soils and Chemistry laboratories at the University of KwaZulu-Natal (Westville Campus). Pollution indices were used to determine the level of sediment contamination resulting from anthropogenic inputs into the system. Statistical methods were thereafter utilised to identify relationships between measured values and the implications thereof.

CHAPTER 5: RESULTS AND DISCUSSION

5.1 Introduction

In order to make inferences regarding the health of the wetlands delineating the Lower uMngeni River, laboratory and statistical procedures were conducted on all collected samples, including water samples extracted from installed piezometers, water samples collected from the river surface and wetland sediment cores. The detected values were either entered into pollution index equations or compared to accepted limits to determine the health of the interconnected wetland and river systems. The level of contamination in these systems, based on the findings of analytical assessments, will in turn inform the identification of appropriate rehabilitation, management and monitoring measures to be implemented. .

5.2 Anthropogenic Contamination of Sediments

This section is based on:

An assessment of the anthropogenic contamination of sediments constituting the floodplain wetlands of the Lower uMngeni River, KwaZulu-Natal, South Africa. *In Review*. *Environmental Earth Sciences Journal.*

(Submitted for publication on: 15 October 2013)

5.2.1 Abstract

Despite the immense importance of the functions that wetland ecosystems perform, they are often severely impacted on, as their location frequently coincides with areas receiving the greatest development pressures. Consequently, runoff received by wetland zones may contain a range of contaminants that are detrimental to the water quality, biotic functioning, and biodiversity of these systems. Amongst the vast range of ecosystems that exist today, coastal wetlands are considered one of the most productive. This study focuses on one such a wetland type - the floodplain wetlands of the Lower uMngeni River, located in the vicinity of concentrated development in the city of Durban, KwaZulu-Natal, South Africa.

Geochemical and physico-chemical characterization was performed on soil cores extracted from selected floodplain wetlands to ascertain the contaminant status of comprising sediments. A total of six wetland areas, that lie adjacent to light and heavy industrial, recreational and residential land uses, were selected. Laminae of the sediment cores were analysed using Inductively Coupled Plasma-Optimal Emission Spectroscopy (ICP-OES) for 14 elemental constituents (mainly heavy metals), including: Aluminium (Al), Arsenic (As), Calcium (Ca), Chromium (Cr), Copper (Cu), Iron (Fe), Magnesium (Mg), Manganese (Mn), Nickel (Ni), Phosphorous (P), Lead (Pb), Sulphate ions (S), Vanadium (V) and Zinc (Zn). For each identified lamina, the percentage composition of organic matter, calcium carbonate, particle sizes.

A comprehensive overview of the contamination and enrichment status of each sediment lamina indicated that the highest levels of contamination: (1) occur within the surface laminae, (2) correlated well with organic matter, calcium carbonate and fine sediment content, and (3) decreased with depth. Pb is the most ubiquitous contaminant, occurring dominantly in the surface laminae and in many deeper laminae of sediment cores. Enrichment by Pb and other metals were shown to vary amongst the different laminae. In most cases, findings were indicative of minor to moderate anthropogenic contamination. Given the rapid rate of development in this region, the floodplain wetlands of the Lower uMngeni sub-catchment should be flagged as amongst those requiring conservation priority on a municipal level.

5.2.2 Introduction

Coastal regions contain a multitude of natural resources and therefore epitomise hubs of human activity in both developed and developing countries alike (Cobelo-García & Prego, 2004). Micro-ecosystems, such as wetlands, which were once prevalent and conspicuous features of coastal areas, have been subject to progressive degradation and destruction over time by nations in pursuit of economic growth and development (Vitousek *et al.*, 1997). Previously large functioning wetlands have been reduced to scattered remnants, representing an obvious manifestation of unrestrained human influence in the past, and in many cases, its persistence in the present day (Coleman *et al.*, 2008). Although past perceptions of wetlands have been largely abandoned, and increasing efforts are being made by governments and non-governmental organisations to manage and protect wetland environments, development continues to pose a real and perilous threat to the health of existing wetland ecosystems (Gren *et al.*, 1994).

The mounting pressure on wetland resources and functioning; elicited by increased urbanisation, injudicious agricultural practices and industrialisation, is further compounded by surges in human population and the concentration of anthropogenic activity in close proximity to wetland features (Brijlal, 2005). Densely urbanised localities along the coast act as significant waste- and chemical- producing conglomerates that contribute largely to the enrichment of wetland sediments by contaminants, that typically comprise effluent discharges from industries, and leachate from sources of municipal waste and cultivated land (Cobelo-García & Prego, 2003; 2004; Nabulo *et al.*, 2008).

The correlation between the health and integrity of an environment and the sustainability of its soil ecosystem has been well documented. Sediment pollution is known to decrease soil quality and alter ecosystems in ways that prove detrimental to their biotic dependents (Ayeni *et al.*, 2010). The danger of metals being present in wetlands in quantities exceeding those deposited *via* natural processes is primarily associated with the high potential for their release from sediments into surrounding waters subsequent to changes in prevailing physical and chemical characteristics in the water column (Kotze, 2000). The implications are unattractive from an ecosystem health perspective given the reputed ability of metals to bio-accumulate and bio-magnify in wetland organisms, the consequences of which are fatal to affected biota as well as reliant human populations (Nomaan *et al.*, 2012). The importance of maintaining acceptable soil quality in wetland ecosystems, and in other ecosystems alike, therefore arises from the need to maintain the capacity of sediments to sustain plant and animal productivity, provide refuge for dependent biota, and thereby support ecosystem functioning and human well-being (Karlen *et al.*, 1997).

The effectiveness of sediments as a sensitive indicator, utilised to monitor the level of metal pollution and trace contamination sources, has been largely recognised and accepted by experts and the larger scientific community (Harikumar *et al.*, 2009; Soares *et al.*, 1999). This realised efficacy is based on the ability of sediments to promptly adsorb and accumulate metals carried by the above-flowing water, irrespective of water level at the given time, allowing for a greater degree of accuracy in metal detection than the employment of alternative analysis methods (Soares *et al.*, 1999). The adsorption capacity of wetland sediments, however, is highly variable, being based on the physical and chemical properties of sediment, principally particle size and organic matter content (Sheoran & Sheoran, 2006). Trace metals, for instance, naturally display a higher affiliation for finer-grained sediment, and sediment constituting a higher percentage of organic matter (Khechfe, 1997). Furthermore, sediment cores have been useful in studying the behaviour of metals and broadly reflecting the sediment history, and thus the contamination history of an area (Harikumar *et al.*, 2009).

In order to determine whether the concentration of detected metals in sediments are of natural or anthropogenic origin, and establish their level of acceptability regarding their contribution towards maintaining optimum functionality of these wetlands, it is essential that the results obtained are compared with standard background values for the region or geological type (Martínez-Carballo *et al.*, 2007 and Chenhall *et al.*, 2004 in Sukdeo *et al.*, 2011). 'Geochemical backgrounds' or 'Clarke values' are values that indicate the normal quantity of each metal within a specific geographic location, characterised by a certain geological pattern (Bhuiyan *et al.*, 2010). These values are preferentially used for comparative purposes as standard background values with the aim of identifying anomalies and thus deviations from the 'norm' (Bhuiyan *et al.*, 2010).

This study was aimed at determining and comparing the concentration of specific metal contaminants in the soils of six selected sites, representing the floodplain wetlands of the Lower uMngeni River.

5.2.3 Study Area

The floodplain wetlands along the Lower uMngeni River, located just north of the Durban Central Business District, have become increasingly exposed to human activities and significant development stress over the past decade (Water Research Commission, 2002). The uMngeni River originates in the foothills of the Drakensberg Mountains at an elevation of approximately 2 000 m above sea level, from which it traverses approximately 255 km of land before entering the Indian Ocean to the north of the Durban Harbour (Cooper, 1993; Van der Zel, 1975). The geology of the catchment consists largely of alluvium of the Quarternary age which is underlain by Permian age dark grey shales and sandstone interspersed with boulder shale of the Karoo sequence and Dwyka formation (Arcus Gibb, 2005). The input of heavy metals into the Lower uMngeni wetlands may be attributed to a multitude of catchment land uses, including commercial and small-scale subsistence farming, industrial activity, urban development and, formal and informal settlements (Brijlal, 2005; South Durban Community Environmental Alliance, 2011). Considering the importance of the uMngeni River system in providing water to the greater Durban and Pietermaritzburg metropolitan communities, the value of floodplain wetlands occurring along the channel should not be underestimated, as they may well serve to perform certain distinct functions that facilitate the creation and maintenance of favourable conditions in the adjacent river (Kotze, 2000; Dickens *et al*., 2003), and the estuary further downstream.

5.2.4 Materials and Methods

Sediment cores were collected from each of six identified floodplain wetlands along the uMngeni River as illustrated in Figure 5.1 below.

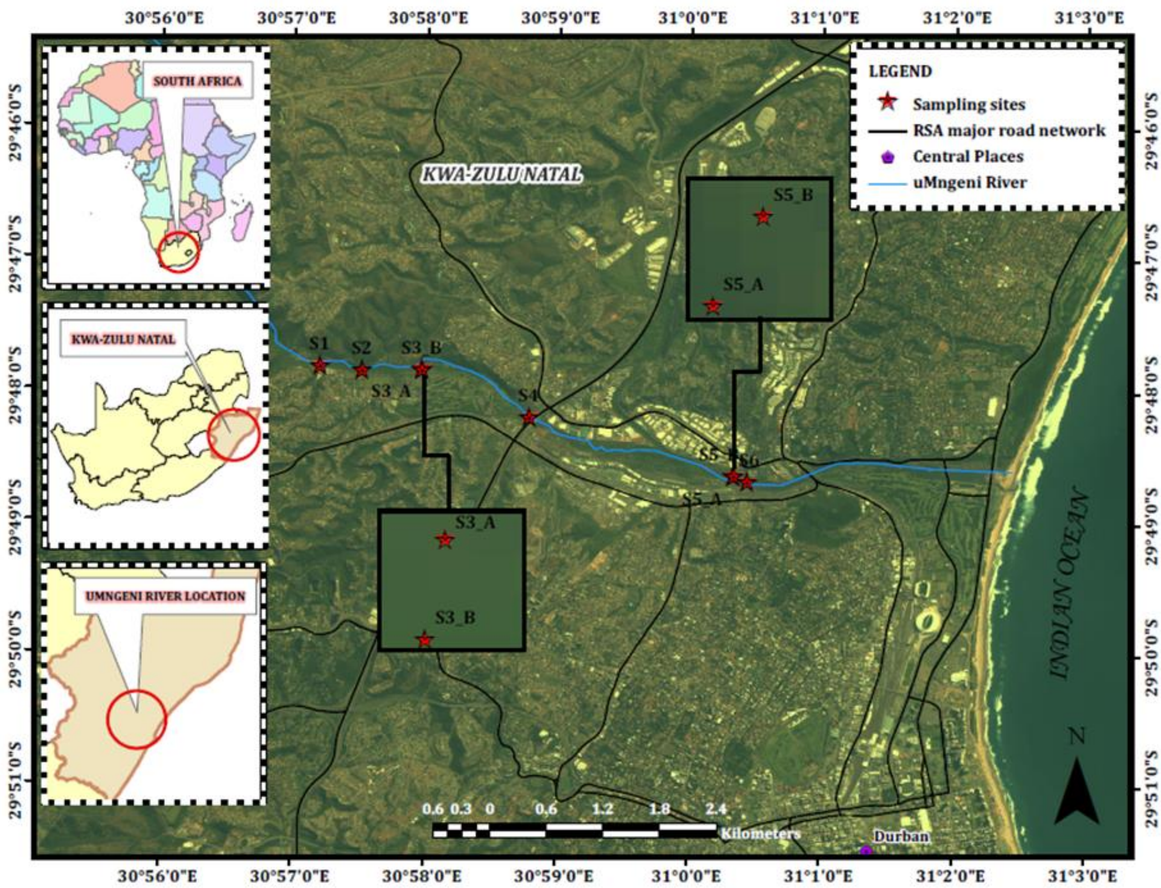


Figure 5.1: Map of the core sampling sites along the floodplain wetlands of the uMngeni River

The sites were selected based on a number of factors including accessibility, the pliability of soil for penetration, and their location within two distinctly differing land use regions, namely residential and industrial. This was done with the intent of enabling comparison between the results attained for each region, and thereby informing deductions related to the potential source of pollutants.

Sediment cores were collected in November 2011 by establishing a transect line across each of the selected wetlands from approximately 30 m up-bank to the river channel. A maximum of two sediment cores were collected at equidistant intervals along each transect. At sites S3 and S5, two cores were collected, whilst only one core was collected at sites S1, S2, S4 and S6 due to a visible narrowing of the wetland areas and the presence of resistant soils at these sites. Core depths ranged from 1 – 1.5 m, and varied according to the thickness of soil horizons and depth to the water table in each case. Land uses adjacent to sampling sites include: S1 – informal housing and light industry; S2 – sports field, informal settlements and subsistence agriculture; S3 – golf course; S4 – light industry; S5 and S6 – broad wetlands with undisturbed vegetation, and a grassed buffer zone of approximately 50 m bordered by industries. Furthermore, a few informal houses are sparsely located on the northern banks of the uMngeni River within the vicinity of sample sites S3, S4, S5 and S6.

The cores were collected using a 40 mm diameter poly-vinyl-compound (PVC) pipe with the assistance of a hand auger. On extraction of the pipe containing the sediment core, the pipe was appropriately labelled indicating site number, date and time. GPS co-ordinates of each site were recorded and both ends of the pipe were securely sealed.

The cores were split longitudinally in half and sediment laminae carefully logged based on compositional characteristics, such as texture, colour, lamination and organic material. Two sub-samples were taken from each layer, one for determining the physical properties of constituent sediments and the other for chemical analysis. Soil moisture content was established by drying; organic matter content was determined by both ignition loss and the Walkley-Black method; and calcium carbonate content was ascertained by ignition loss.

A Retsch® sieve shaker was used to undertake the standard dry sieving method of grain size determination. Elemental concentrations of Al, Ca, Cr, Cu, Fe, Pb, Mg, Mn, Ni, P, S, V and

Zn were detected using ICP-OES, ion chromatography and flame photometry as prescribed by Skoog & West (1982).

5.2.4.1 Statistical analysis

Analysis of Variance (ANOVA) was used to assess the variance of elemental concentrations at the sites and of the geochemical indices used.

5.2.4.2 Assessing contamination using pollution indices

The level of sediment contamination by metals was assessed by computing values measured through laboratory analysis into specific pollution indices. In this study, the contamination factor, enrichment factor and pollution load indices were calculated by utilising the 'Clarke values' for sedimentary rocks and the dominant catchment geology as background concentrations.

Contamination factors are used to determine whether measured concentrations of metals in a sediment sample are of natural or anthropogenic origin, and are calculated as:

$$CF = \frac{[\text{Concentration of element in the sediment}]}{[\text{Background value of element}]} \qquad (1)$$

Contamination factors range in value from 0 to 6, with a value of zero indicating no contamination and a value of six signifying strong contamination (Bhuiyan *et al.*, 2010). More specifically, values less than 1 imply low contamination, values between 1 and 3 imply moderate contamination, and values between 3 and 6 indicate significant contamination (Harikumar and Jisha, 2010).

Enrichment factors are used to ascertain the abundance of a specific metal in a sediment sample in relation to the average concentration in the earth's crust (Harikumar and Jisha, 2010). This calculation can be expressed as:

$$EF = \frac{[\text{Concentration element}]/[\text{Concentration Fe}]}{[\text{Clarke element}]/[\text{Clarke Fe}]} \qquad (2)$$

In the above equation, [Concentration element] refers to the mean concentration of the metal of interest (ppm), [Concentration Fe] is the mean concentration of Fe in the sediment sample (ppm), [Clarke element] is the Clarke value of the metal of interest (ppm) and [Clarke Fe] is the Clarke value of Fe (ppm). Fe was used as the reference element for geochemical normalisation due to its geochemical similarity to many other trace metals, its natural high concentrations and the usual uniformity of its concentration in sediments (Bhuiyan *et al.*, 2010). Values below 0.5 indicate pristine environments or deficient enrichment , between 0.5 and 1.5 suggest metal input from natural sources, values greater than 1.5 indicate possible input from anthropogenic sources, and values greater than 5 imply contamination (Harikumar and Jisha, 2010; Mmolawa *et al.*, 2011).

The pollution load index is calculated to measure the extent of pollution by a metal at a specific site, and can be determined by using the below equation:

$$PLI = (CF1 \times CF2 \times CF3 \times ... \times CFn)\ 1/n \qquad (3)$$

In equation 3 above, CF refers to contamination factor and *n* refers to the number of metals. PLI values are categorised into 3 broad groups, namely those equal to zero (pristine and undisturbed environments), those between zero and one (environments with baseline pollution levels), and those greater than one (polluted environments) (Harikumar and Jisha, 2010).

5.2.5 Results and discussion

5.2.5.1 Physical properties of core sediments

At all six sites, organic matter content (OMC) and calcium carbonate ($CaCO_3$) were highest within the first three laminae of the sediment cores, which then fluctuated with a generally decreasing trend for the subsequent laminae. These percentages are inversely related to the mean sediment grain size (phi) calculated for each lamina such that finer sediment particles contain higher percentages of OMC and $CaCO_3$. Considering the fact that finer sediments retain more water, the percentage of moisture content (MC) was also shown to be high within the first three laminae of the sediment cores, and high within deeper laminae due to increasing proximity to the water table.

The above relationships are illustrated in Figures 5.2(a)–(p). Figures 5.2 (a), (c), (e), (g), (i), (k), (m) and (o) depict the sediment mean grain size variations with core depth for sites S1, S2, S3A, S3B, S4, S5A, S5B and S6 respectively; and Figures 5.2 (b), (d), (f), (h), (j), (l), (n) and (p) show the percentage MC, OMC and $CaCO_3$ variations with core depth at sites S1, S2, S3A, S3B, S4, S5A, S5B and S6 respectively.

The high quantities of fine- to medium-grained material, in the upper laminae of sediment cores are expected in floodplain wetlands that are regularly exposed to overbank flooding. Fine material carried in suspension by flood waters covering the floodplain gradually settle out as a layer as flood waters recede. There is generally a coarsening of grades with depth in such layers, as displayed by several of the laminae comprising sediment cores collected for this study. In general, however, it can be noted that the sediment mean grain size in laminae across all sites ranged from a minimum phi value of 0.56 (coarse sand) to a maximum phi value of 2.29 (fine sand).

The highest OMC value was measured at site 3, located adjacent to the golf course. At this site, riparian vegetation was particularly well established and river morphology allowed for accumulation of organic debris. Further downstream at site 5, characterised by dense natural vegetation, the OMC increases again in the first two laminae. CaCO3 is low with some variability across sites. The implication is that sediments constituting these wetlands have a very low "buffer capacity", therefore increasing the susceptibility of these systems to metal pollution (Frei, 2008). It is therefore likely that high metal concentration in wetland interstitial waters, relative to the concentration detected in river water, may have resulted from a lack of "buffer capacity" that would otherwise counteract the effects of increased acidity due to the deposition or re-suspension of anoxic sediment, or the entry of contaminants derived from anthropogenic activities into these systems. Some increases, however, are noted at sites S2, S4, S5A and S6. At sites S2, S4 and S6, the amount of $CaCO_3$ in sediments were greatest in the surface laminae and decreased with depth.

At site S5A, however, the amount of $CaCO_3$ in sediment is found to be higher in the lower laminae, particularly between a depth of 0.67 and 0.99 m. This may be the result of greater dilution by terrigenous input or a high rate of sedimentation, leading to dissolution of $CaCO_3$ in the upper layers of the core. It may also be possible that higher $CaCO_3$ amounts in the lower laminae may have resulted from the re-precipitation of carbonates in reduced sediment in the lower lamina, or other natural geochemical processes.

In all cases, soil moisture was high in the first two to three laminae of sediment cores, and showed increases with depth, as is expected in environments where the water table is located close to the surface.

Figures 5.2(a) – (p)….\

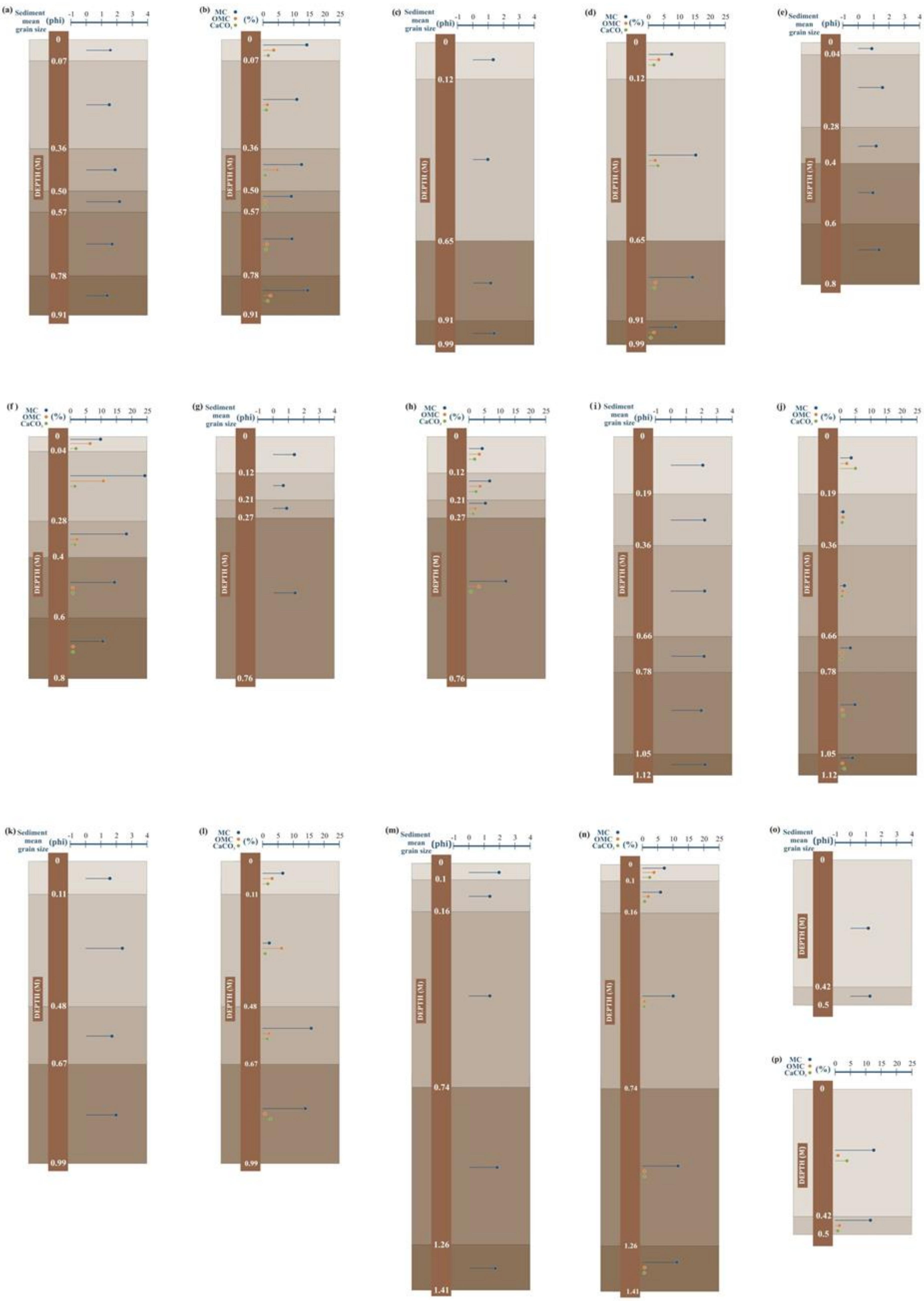


Figure 5.2(a)-(p): Sediment mean grain size (phi) in relation to percentage MC, OMC and $CaCO_3$ contained in core laminae at each site.

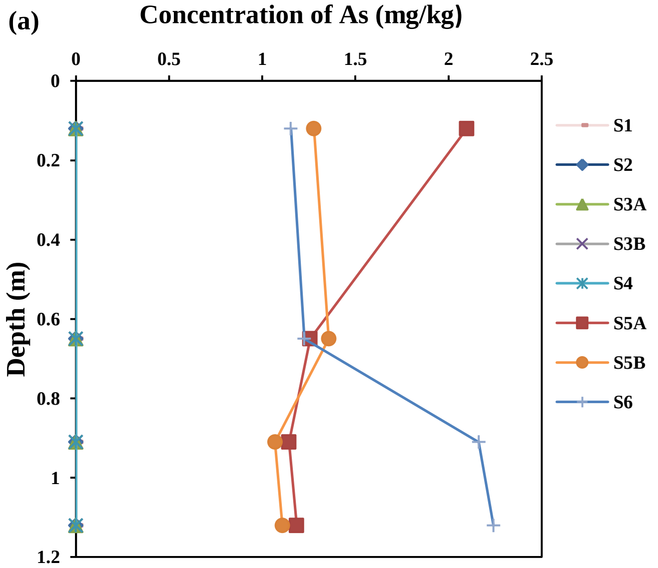
5.2.5.2 Chemical properties of core sediments

Figures 5.3(a)–(l) plot mean metal concentration with depth for each of the six sites. The thickness of each lamina within the various cores was found to differ considerably. This necessitated the standardisation of depths at which comparisons amongst metal levels were made. As such, the mean metal concentration values at particular depths are illustrated, rather than actual detected concentrations in each lamina. Therefore, no values were plotted for a depth of 0-10 cm in each of the graphical representations below.

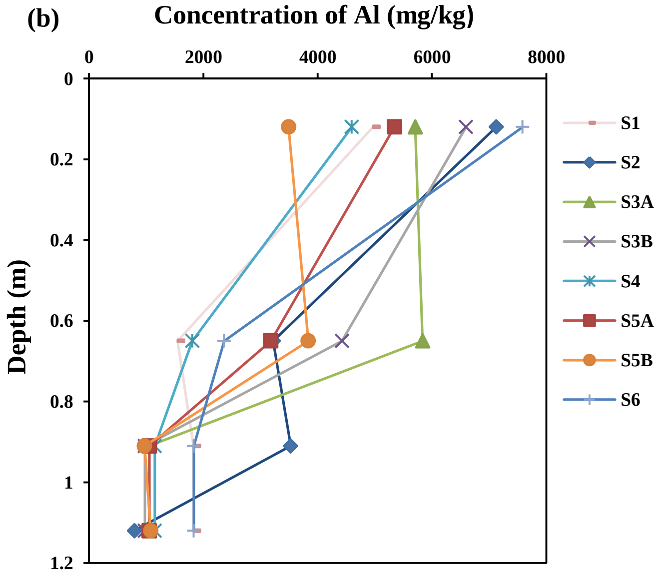
Clearly, a general trend of decreasing metal concentration with increasing core depth is observed, correlating well with the vertical distribution of organic matter and fine sediment. This is in keeping with the high affinity of organic matter and fine sediment for adsorbing metal ions. Exceptions to the trend are displayed in concentration variations of As, Ni and V.

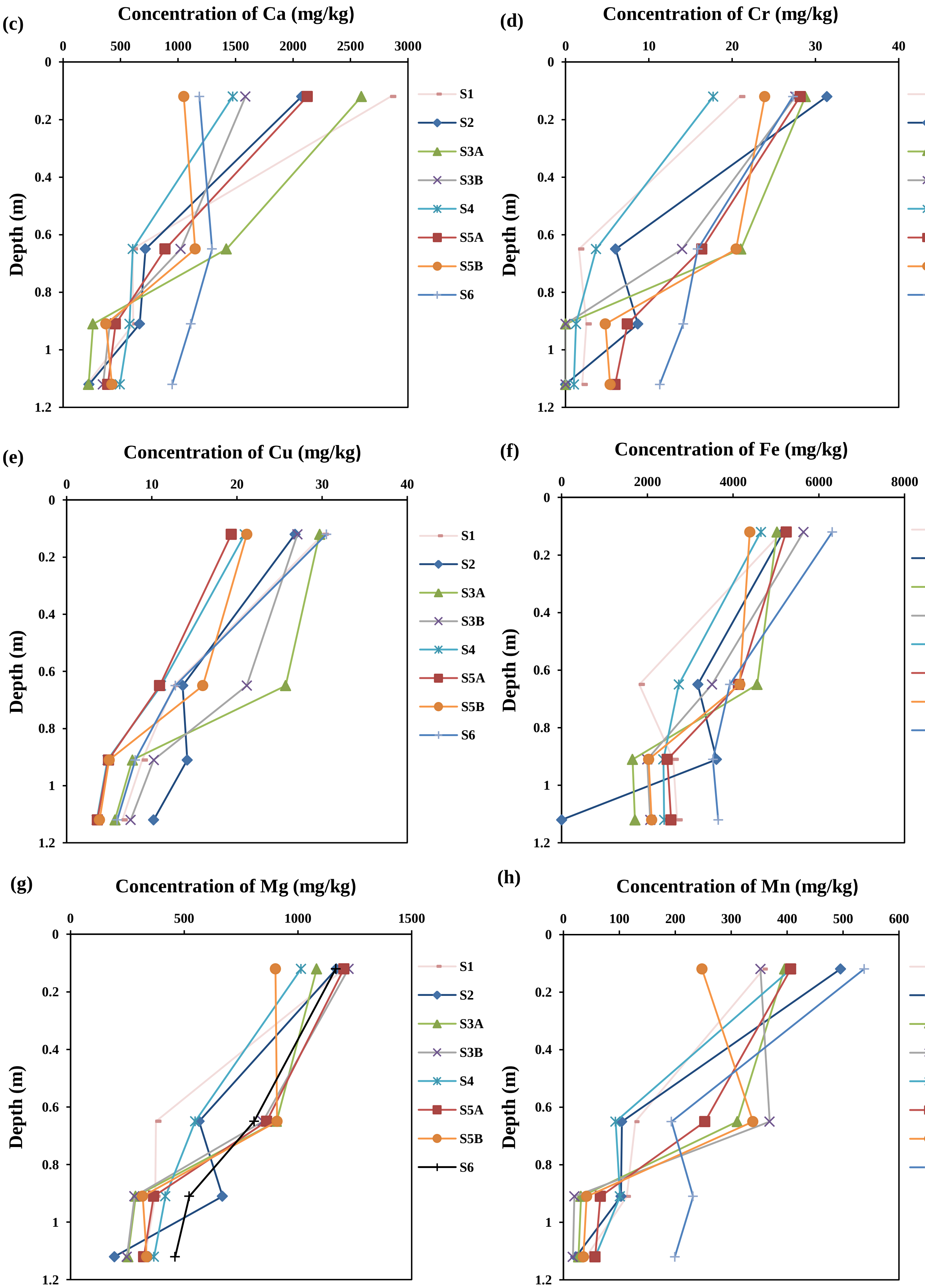
At site S6, As levels in the sediment increase with increasing soil depth. This may be due to the discharge of wastes containing high levels of As by the heavy industrial activity within the vicinity of this site in the past. However, it may also be possible that the high As level is an anomalous, naturally occurring concentration. The high Ni and V levels detected at sites S2 and S3B respectively may have resulted from the drainage of high amounts of fertilisers and pesticides used at the recreational field (near site S2) and the golf course (near site S3B) in the past. Similar to As, it is possible that detected levels of Ni and V have been derived from normal geochemical processes, therefore representing anomalous high natural concentrations.

(a)



(b)




(c)
Concentration of Ca (mg/kg)
0 500 1000 1500 2000 2500 3000
Depth (m)
0 0.2 0.4 0.6 0.8 1 1.2
S1 S2 S3A S3B S4 S5A S5B S6
(d)
Concentration of Cr (mg/kg)
0 10 20 30 40
Depth (m)
0 0.2 0.4 0.6 0.8 1 1.2
S1 S2 S3A S3B S4 S5A S5B S6
(e)
Concentration of Cu (mg/kg)
0 10 20 30 40
Depth (m)
0 0.2 0.4 0.6 0.8 1 1.2
S1 S2 S3A S3B S4 S5A S5B S6
(f)
Concentration of Fe (mg/kg)
0 2000 4000 6000 8000
Depth (m)
0 0.2 0.4 0.6 0.8 1 1.2
S1 S2 S3A S3B S4 S5A S5B S6
(g)
Concentration of Mg (mg/kg)
0 500 1000 1500
Depth (m)
0 0.2 0.4 0.6 0.8 1 1.2
S1 S2 S3A S3B S4 S5A S5B S6
(h)
Concentration of Mn (mg/kg)
0 100 200 300 400 500 600
Depth (m)
0 0.2 0.4 0.6 0.8 1 1.2
S1 S2 S3A S3B S4 S5A S5B S6

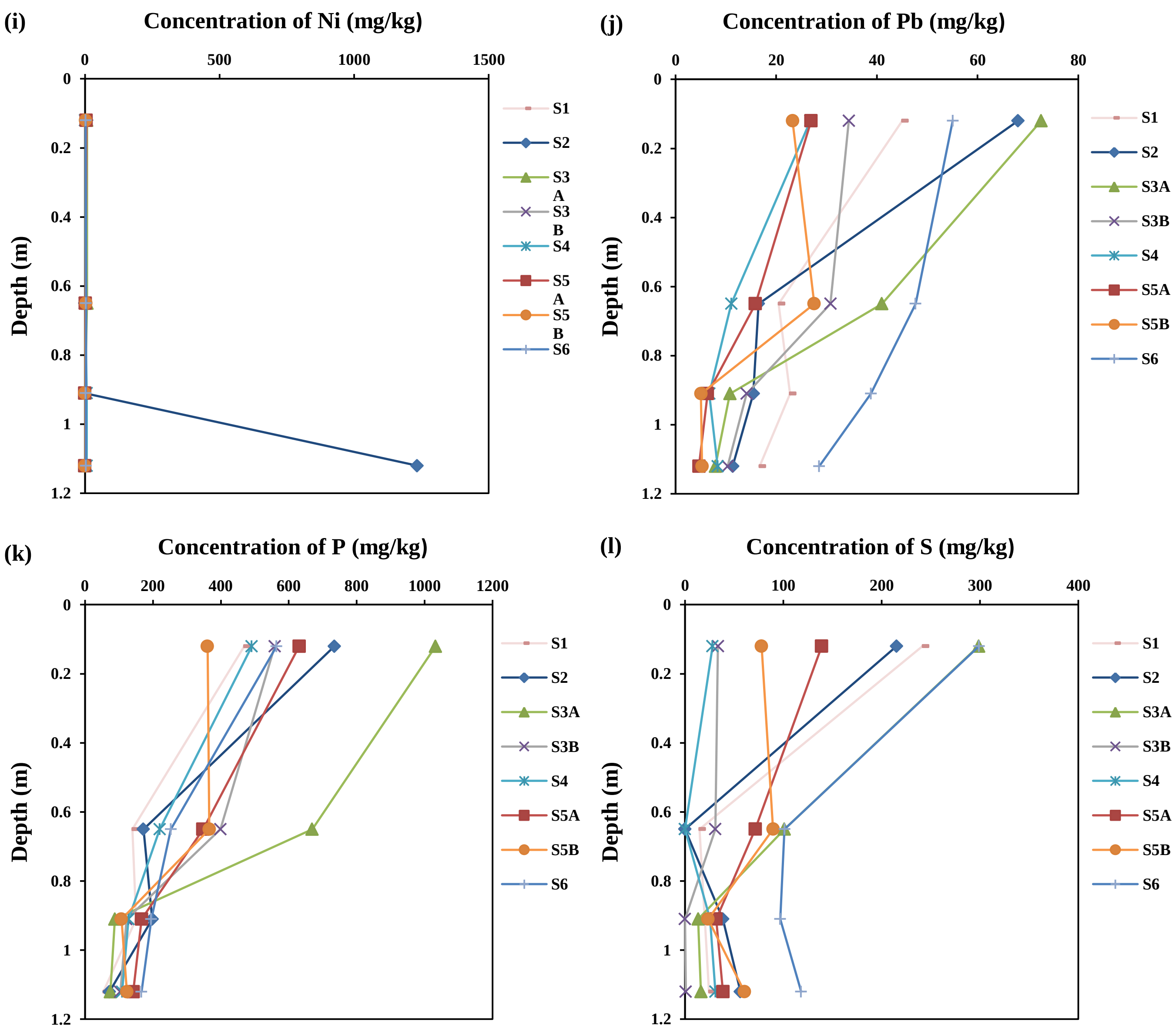


Figure 5.3(a)–(l): Change of element concentrations with depth at each of the study sites

Geochemical indices, calculated using mean metal values detected for the relevant sites, informed inferences regarding the anthropogenic contribution to contaminant enrichment at each site. Table 5.1 below presents the contamination factors (CF) calculated for the measured concentrations of specific elements within sediments at each sampling point. The CF values obtained for most elements were in the range of zero and one, indicating minimal baseline pollution of wetlands. However, contamination factors for Pb and V exceed one at certain sites, implying moderate metal pollution derived from anthropogenic sources. Results of the ANOVA confirm that, although the variances for all elements are low, the highest for this suite of metals were recorded for Pb (0.419) and V (0.253).

Table 5.1: Contamination factor of mean concentration of elements detected within sediments at each sample site

Element	Clarke value	S1	S2	S3A	S3B	S4	S5A	S5B	S6
Al	81300	0.031	0.045	0.042	0.040	0.027	0.033	0.029	0.042
As	5	NE	NE	NE	NE	NE	0.284	0.241	0.339
Ca	36300	0.029	0.025	0.031	0.023	0.022	0.027	0.021	0.031
Cr	200	0.034	0.058	0.062	0.052	0.030	0.072	0.068	0.086
Cu	70	0.208	0.231	0.246	0.236	0.144	0.138	0.164	0.205
Fe	50000	0.061	0.060	0.065	0.066	0.061	0.072	0.063	0.087
Mg	20900	0.027	0.031	0.030	0.031	0.028	0.033	0.029	0.035
Mn	1000	0.160	0.181	0.191	0.189	0.165	0.196	0.166	0.291
Ni	80	0.009	0.020	0.046	0.023	0.045	0.023	0.019	0.042
P	1180	0.172	0.249	0.379	0.251	0.201	0.273	0.203	0.250
Pb	16	1.640	1.740	2.068	1.402	0.827	0.841	0.956	2.658
V	150	0.115	0.126	0.136	1.549	0.121	0.133	0.128	0.121
Zn	132	0.193	0.204	0.426	0.166	0.072	0.329	0.199	0.188

NE: No Enrichment

0 or NE	Pristine and undisturbed environments
0 - 1	Environments with baseline pollution levels
0.75 – 0.99	Border-line polluted environments
> 1	Polluted environments

The contamination factor for Vis highest at site S3B and although its source is unknown, it may be related to the ongoing fertilization and maintenance activities at the golf course. Most conspicuous, however, are the contamination factors for Pb, which are shown to be high across most sites, with sites S1, S2, S3A, S3B S6 being most contaminated. Noteworthy is the fact that Pb input at sites S4, S5Aand S5B in particular; have resulted in border-line to moderate contamination levels.

Tables 5.2 below presents the highest contamination factors calculated for each lamina of sediment cores. These are compared to enrichment factors calculated using mean values displayed in Table 5.3 below.

Table 5.2: Contamination factor of particular elements detected within the laminae of sediment cores at each sample site

Element	Clarke value	L	S1	S2	S3A	S3B	S4	S5A	S5B	S6
Fe	50000	1	1.028	1.035	1.005	1.129	0.930	1.048	0.878	0.785
Pb	16	1	2.813	4.253	4.540	2.151	1.674	1.684	1.454	2.980
Zn	132	1	0.614	0.736	1.137	0.362	0.232	0.910	0.361	0.226
Fe	50000	2	0.510	0.636	1.291	0.872	0.429	0.502	1.353	0.706
Pb	16	2	1.610	1.030	4.609	2.022	0.636	0.401	3.117	2.426
Zn	132	2	0.082	0.040	1.272	0.397	0.000	0.045	0.760	0.171
P	1180	2	0.134	0.146	1.246	0.486	0.155	0.129	0.536	0.165
Pb	16	3	1.153	0.966	2.341	1.827	0.749	1.583	0.327	
Fe	50000	3	0.578	0.723	0.843	0.531	0.665	1.149	1.353	
V	150	3	0.112	0.153	0.164	11.343	0.135	0.184	0.074	
Pb	16	4	1.078	0.710	0.734	0.883	0.420	0.397	0.316	
Pb	16	5	1.700		0.676		0.456		0.344	
Pb	16	6	1.140				0.590			

L: Lamina

0 or NE	Pristine and undisturbed environments
0 - 1	Environments with baseline pollution levels
0.75 – 0.99	Border-line polluted environments
> 1	Polluted environments

From the above table, it can be observed that the contamination factors for Fe and Pb are highest in the 1st, 2nd and 3rd lamina of each sediment core. Contamination factors for Zn, P and V on the other hand are highest in the 1st and 2nd lamina, 2nd lamina, and 3rd lamina respectively.

Fe and Pb contamination is ubiquitous in the surface laminae across all six sites, with contamination factors implying either border-line, moderate or high contamination. Noteworthy are the contamination factors of Pb at sites S2 and S3A, indicating significant metal pollution. Sediment pollution resulting from high levels of Zn is shown to be less problematic across sites, with moderate contamination detected at S3A, and border-line contamination occurring at S5A.

Contamination factors for Fe in the 2nd lamina show border-line contamination at site S3B, and moderate contamination at sites S3A and S5B. Pb contamination is shown to be moderate across all sites except S4 and S5A, and S3B and S5B, which signify low and high contamination respectively.

The 3rd lamina of the sediment cores show low Pb contamination at S5B, border-line contamination at S2 and S4 and moderate contamination across the remaining sites. Fe contamination is moderate at sites S5A and S5B, implies border-line pollution at site S3A, and is low in the remaining sites.

In the 4th, 5th and 6th laminae of sediment cores, Pb is the only metal showing border-line moderate to high contamination levels.

In the 4th lamina, there is moderate Pb contamination at site S1, border-line moderate contamination at site S3B and low contamination in the remaining sites. Pb contamination in the 5th layer is calculated to be moderate at S1, low at sites S3A, S4 and S5B, and none at the remaining sites. Likewise, in the 6th lamina of cores, Pb contamination is moderate only at S1, low at S4 and none in the remaining sites.

Table 5.3: Enrichment factors of the mean concentration of elements detected within sediments at each sample site

Element	Clarke value	S1	S2	S3A	S3B	S4	S5A	S5B	S6
Al	81300	0.158	0.189	0.183	0.175	0.129	0.135	0.130	0.265
As	5	NE	NE	NE	NE	NE	1.170	1.084	2.148
Ca	36300	0.149	0.106	0.135	0.102	0.105	0.109	0.093	0.198
Cr	200	0.171	0.241	0.272	0.228	0.143	0.298	0.307	0.543
Cu	70	1.051	0.967	1.074	1.034	0.695	0.570	0.740	1.295
Fe	50000	NE	0.250	0.283	0.290	0.293	0.296	0.285	0.552
Mg	20900	0.138	0.130	0.132	0.136	0.136	0.136	0.133	0.224
Mn	1000	0.810	0.758	0.836	0.831	0.795	0.805	0.748	1.840
Ni	80	0.046	0.085	0.201	0.102	0.218	0.094	0.085	0.268
P	1180	0.872	1.042	1.657	1.101	0.971	1.124	0.914	1.581
Pb	16	8.294	7.269	9.037	6.149	3.994	3.464	4.303	16.833
V	150	0.582	0.528	0.594	6.794	0.583	0.549	0.575	0.768
Zn	132	0.978	0.851	1.861	0.730	0.347	1.354	0.898	1.190

NE: No Enrichment

0 – 0.49 or NE	Pristine environment or deficient enrichment
0.5 – 1.5	Metal input from natural sources
> 1.5	Possible input from anthropogenic sources
3.50 – 4.99	Border-line anthropogenically polluted environment
> 5	Anthropogenically polluted environment

From Table 5.3 above, it can be seen that the enrichment factors for Pb across all sites imply either possible anthropogenic input or definite contamination. This is consistent with the contamination factor data displayed in Table 5.2, which shows border-line moderate or moderate Pb contamination across the sample sites. Furthermore, the enrichment factors for As, Mn, P, V and Zn are likely to have been derived from anthropogenic sources at certain sites. Enrichment factors for As, Mn and P at site S6 indicate possible anthropogenic pollution, whilst the enrichment factor for Pb at site S6 is more conclusive. The data also shows a possibility of anthropogenic input of P and Zn at site S3A, and definite anthropogenic pollution by V at site S3B. The enrichment factor calculated for the remaining elements measured at each sampling site implied either input from natural sources or pristine conditions. Results of the ANOVA showed high variance for Pb (18.67) as well as V (4.81). The latter is reflective of the anomalous high V level measured at site S3B.

Tables 5.4(a) and (b) below display the enrichment factors determined using actual concentrations of metals detected within each lamina of sediment cores.

Table 5.4(a): Enrichment factor of particular elements detected within lamina 1, 2 and 3 of sediment cores at each sample site

Element	Clarke value	L	S1	S2	S3A	S3B	S4	S5A	S5B	S6
Cu	70	1	4.190	3.701	4.225	3.430	3.214	2.636	3.445	2.323
Pb	16	1	27.371	41.088	45.190	19.057	18.002	16.069	16.565	37.974
Zn	132	1	5.976	7.106	11.320	3.202	2.489	8.685	4.112	2.880
Ni	80	1	0.352	0.787	0.876	0.591	0.141	0.598	0.355	0.641
Cr	200	1	1.018	1.516	1.433	1.222	0.954	1.344	1.363	1.011
As	5	1	NE	NE	NE	NE	NE	4.005	2.911	3.124
V	150	1	1.912	1.468	1.895	2.209	1.994	1.884	1.947	1.306
P	1180	1	3.866	6.015	8.710	4.195	4.477	5.104	3.484	2.732
Mn	1000	1	3.483	4.788	3.938	3.122	4.378	3.878	2.823	2.462
Cu	70	2	3.764	3.063	4.958	4.208	3.568	1.683	2.865	1.631
Pb	16	2	31.547	16.204	35.689	23.191	14.841	7.991	23.035	34.363
Zn	132	2	1.599	9.850	4.555	5.614	2.425	0.900	5.614	2.425
Ni	80	2	NE	NE	1.799	0.611	NE	NE	0.658	0.589
Cr	200	2	0.172	0.473	1.631	1.096	NE	0.646	1.408	1.001
As	5	2	NE	NE	NE	NE	NE	3.928	2.649	6.127
V	150	2	2.153	2.059	2.139	1.836	2.241	1.839	1.815	1.285
P	1180	2	2.627	2.298	9.652	5.577	3.606	2.579	3.964	2.342
Mn	1000	2	2.492	1.644	5.616	3.826	1.713	1.506	4.803	3.285
Cu	70	3	3.177	2.800	3.409	4.476	2.460	1.980	2.232	
Pb	16	3	19.950	13.364	27.778	34.412	11.251	13.779	10.550	
Zn	132	3	1.468	0.542	4.485	3.977	0.212	5.089	0.712	
Ni	80	3	NE	NE	0.563	NE	NE	0.504	NE	

Element	Clarke value	L	S1	S2	S3A	S3B	S4	S5A	S5B	S6
Cr	200	3	0.265	0.601	1.078	0.834	0.550	0.693	0.781	
As	5	3	NE	NE	NE	NE	NE	2.661	5.959	
V	150	3	1.946	2.114	1.951	213.68	2.029	1.598	2.379	
P	1180	3	2.235	2.325	3.581	3.579	3.289	4.002	2.710	
Mn	1000	3	2.522	1.427	1.880	7.607	1.700	3.746	0.905	

0 – 0.49 or NE	Pristine environment or deficient enrichment
0.5 – 1.5	Metal input from natural sources
> 1.5	Possible input from anthropogenic sources
3.50 – 4.99	Border-line anthropogenically polluted environment
> 5	Anthropogenically polluted environment

Table 5.4(b): Enrichment factor of particular elements detected within lamina 4, 5 and 6 of sediment cores at each sample site

Element	Clarke value	L	S1	S2	S3A	S3B	S4	S5A	S5B	S6
Cu	70	4	4.596	5.907	2.904	3.653	1.445	1.420	1.755	
Pb	16	4	30.172	28.75	12.19	22.09	8.838	8.047	7.781	
Zn	132	4	1.050	NE	0.843	NE	0.464	1.214	0.463	
Ni	80	4	NE	NE	0.150	NE	NE	NE	NE	
Cr	200	4	NE	NE	0.237	NE	0.701	0.753	0.589	
As	5	4	NE	NE	NE	NE	5.307	4.635	5.266	
V	150	4	2.388	2.843	1.992	2.368	1.927	1.004	2.154	
P	1180	4	2.573	2.471	2.537	2.593	2.289	2.867	2.245	
Mn	1000	4	3.157	0.908	0.817	0.494	2.133	1.340	1.028	
Cu	70	5	1.081		3.331		1.327		0.896	
Pb	16	5	28.084		20.43		9.352		7.928	
Zn	132	5	1.114		NE		0.369		0.802	
Ni	80	5	NE		NE		NE		NE	
Cr	200	5	0.299		NE		0.702		0.681	
As	5	5	NE		NE		5.011		5.280	
V	150	5	2.057		2.455		2.006		2.239	
P	1180	5	2.338		2.255		2.135		2.716	
Mn	1000	5	2.138		0.954		0.859		0.692	
Cu	70	6	4.316				0.771			
Pb	16	6	26.111				12.57			
Zn	132	6	1.245				0.759			
Ni	80	6	NE				NE			
Cr	200	6	0.159				0.710			
As	5	6	NE				3.393			
V	150	6	2.213				1.775			
P	1180	6	2.589				1.730			
Mn	1000	6	2.227				1.536			

0 – 0.49 or NE	Pristine environment or deficient enrichment
0.5 – 1.5	Metal input from natural sources
> 1.5	Possible input from anthropogenic sources
3.50 – 4.99	Border-line anthropogenically polluted environment
> 5	Anthropogenically polluted environment

In Table 5.4 (a) and (b) above it can be seen that the enrichment factors established for Ni in laminae across all sites imply either pristine conditions or metal input from natural origins An exception is the enrichment factor for Ni calculated for the 2nd laminae of the core extracted from site S3A, which indicates possible input from anthropogenic sources. Similar to Ni, Cr is naturally derived at all sites with an exception of the 1st and 2nd lamina of soil cores collected from sites S2 and S3A respectively. The enrichment factors for Cr at these sites imply that concentrations are potentially anthropogenically derived. All other metals indicate some degree of enrichment across the sampling sites.

Apart from Ni and Cr, it is evident that all metals detected within the 1st layer of sediment cores have possible anthropogenic origins, and of particular concern are those enrichment factors established for Pb across all sites, Zn at sites S1, S2, S3A and S5A, and P at site S2, S3A and S5A, which show definite enrichment. It is also noteworthy that As concentrations were undetected at sites S1-S4 but were found to be possibly or border-line anthropogenically polluted at sites S5A, S5B and S6 where detected.

The enrichment factors calculated for the 2nd lamina of all sediment cores show similar trends to those of the 1st lamina. This is such that Pb contamination occurs across all sites, and all metals indicate possible anthropogenic input or enrichment in each core. The enrichment factors for Ni and Cr show possible input from anthropogenic sources at site S3A, and indicate either pristine conditions or inputs from natural sources across the other sites. Metal contamination within the 2nd lamina results from the input of Zn at sites S2, S3B and S5B; the input of P at sites S3A and S3B, and the addition of As and Mn at sites S6 and S3A respectively.

The enrichment factors calculated for the 3rd and 4th laminae of sediment cores both show Pb contamination across all sites. Unlike in the 3rd lamina however, Zn concentrations in the 4th lamina are of natural origin or are undetected across the sample sites. Furthermore, anthropogenic pollution by As is evident at site S4 in the 4th lamina, and at sites S5B in the 3rd and 4th laminae. . Signs of enrichment to note in the 3rd lamina result from the input of Zn at site S5A, As at site S5B, and V and Mn at site S3B. In addition, Cu enrichment occurs in the 4^{th} lamina at site S2.

A 5^{th} lamina is only present in cores collected from sites S1, S3A, S4 and S5B. The consistent trend of high Pb enrichment across sample sites within preceding sediment layers are displayed in these laminae as well. The enrichment factors calculated for As also indicate anthropogenic pollution at sites S4 and S5B. Ni was undetected within this layer across all sample sites, and the enrichment factors for all other elements at each site indicate either natural origins, the possibility of minor anthropogenic enrichment or pristine conditions.

Amongst the sediment cores of interest in this study, only those collected at sites S1 and S4 contain a 6^{th} lamina. Similar to data obtained for the 5^{th} lamina, Ni concentrations were undetected and Pb values indicate contamination at both sites. Enrichment factors calculated for the remaining metals at each site indicate that the concentrations imply pristine conditions, are naturally derived or possibly sourced from anthropogenic activities.

Results of the ANOVA confirm the above. The highest three variances for each of the lamina are as follows: L1: Pb(145.54) > Zn(9.93) > P(3.46); L2: Pb(100.03) > Zn(8.57) > As(3.09); L3: V(6401.09) > Pb(84.23) > As(5.44); L4: Pb(98.70) > Cu(2.97) > Mn(0.86); L5: Pb(91.40) > Cu(1.27) > Mn(0.44) and L6: Pb(91.68) > Cu(6.28) > P(0.34).

Figure 5.4 below displays the PLI calculated for each sample site. Sites S4 and S6 have a PLI value of 0, implying pristine wetland conditions. The PLI values for the remaining sites, however, range between 0 and 1, indicating baseline pollution levels within these systems. Although pollution loading in the wetlands is generally low, the index does point to concern for sites S1, S2, S3A and S3B.

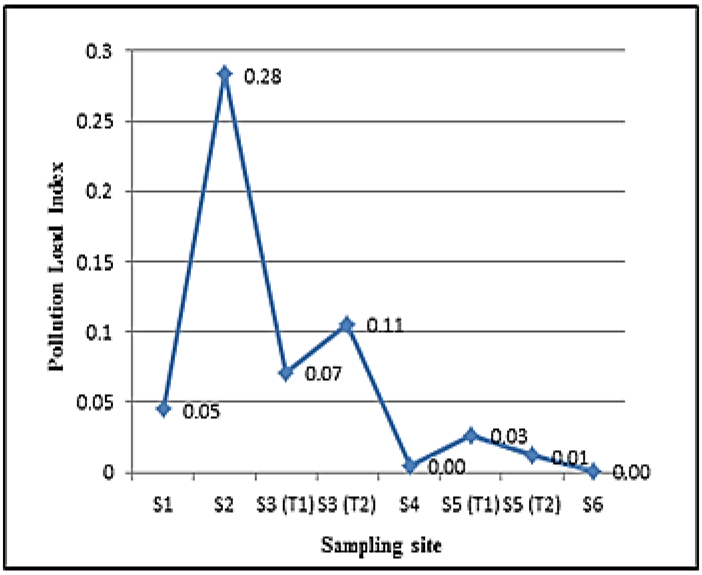


Figure 5.4: Pollution Load Index calculated for each sample site

5.2.6 Conclusion

Overall, the tabulated and graphical data presented in this paper display a general decreasing trend of metal contamination with each subsequent lamina of sediment cores. The highest levels of anthropogenic contamination have been detected within the first 3 layers of sediment cores such that moderate to high contamination levels are most widespread in the 1st lamina and decreases with each of the following laminae. Of particular concern is the high concentration of Pb across the wetlands of interest, particularly within the first three layers of sediment, showing significant contamination levels at sites S2, S3A and S5B in the first two laminae of sediment cores. The prevalence of moderate and high contamination factors for Pb is related to the dominance of fine-grained sediment and high percentages of organic matter within the first 3 laminae of sediment cores. The enrichment values for other elements display no uniform pattern. However, it can be observed that the elemental enrichment of varying laminae and wetland sites has resulted in wetland contamination.

It can therefore be concluded that the anthropogenic input of metals into the Lower uMngeni wetlands had occurred in recent times and persists to the present day. Possible exacerbation of the situation is likely, given the relative fast pace of development in the vicinity of the wetlands and in the hinterland of the catchment. It is therefore of utmost importance that the responsible authorities take the required care in developing and implementing appropriate management and monitoring measures, focused on curtailing current pollution levels and thereby minimising the risk of future degradation.

The references utilised in this paper are provided at the end of this thesis.

5.3 *Water Quality*

This section is based on the following paper which is yet to be submitted to a relevant journal:

A comparative study of the water quality status of the Lower uMngeni River and adjacent floodplain wetlands, KwaZulu-Natal, South Africa.

5.3.1 Abstract

Riverine and wetland systems form the focal points for development and a number of land use activities in South Africa and other countries alike. This together with exponential population growth have resulted in improper waste disposal and the flow of watershed drainage containing high nutrient and metal loads into the Lower uMngeni floodplain wetlands and riverine system, located in KwaZulu-Natal, South Africa. The outcomes include contamination of wetland interstitial water and water constituting the uMngeni River.

wetland interstitial and river water samples collected from the Lower uMngeni wetland and river systems respectively were analysed to detect the level of a number of physico-chemical parameters, including Aluminium (Al), Ammonium ions (NH_4^+), Barium (Ba), Chromium (Cr), Cobalt (Co), Copper (Cu), Dissolved Oxygen (DO), Iron (Fe), Lead (Pb), Manganese (Mn), Nickel (Ni), pH, Total Dissolved Solids (TDS), Vanadium (V) and Zinc (Zn). The derived results were examined and compared, trends were identified, and the potential sources of pollution were determined taking into consideration surrounding land uses and regional rainfall patterns.

Results indicate that variations in detected levels of parameters in both wetland interstitial water and river water samples are closely linked to anthropogenic activities as well as seasonal rainfall variations. The results from the statistical analyses of most measured parameters indicate that a strong positive linear relationship exists between wetland interstitial water and river water samples. Furthermore, a number of water quality indicators were found to be non-compliant with the respective standards for aquatic systems, as stipulated by the Department of Water and Sanitation. It was established that the floodplain wetlands of interest serve as natural water purification systems. It is also highly possible that some detected contaminants in the uMngeni River are sourced from upstream human

activities. Given the dependence of a number of biotic species and community members on the uMngeni River and floodplain wetlands in the Lower uMngeni sub-catchment, water quality

5.3.2 Introduction

Past perceptions of wetland features as wastelands or 'pollution sinks', breeding grounds for a plethora of infectious human diseases, and 'non-functional' expanses of land have drastically changed over the last few decades (Reddy and Gale, 1994). The multitude of beneficial ecological and socio-economic values of wetlands have only recently come to the fore, and have in turn provoked a paradigm shift relating to development initiatives which formerly led to the exploitation and destruction of such systems (Horwitz *et al.*, 2012). The increasingly evident benefits associated with the mere existence and functioning of wetlands had prompted a global peak in scientific efforts, intended to broaden baseline knowledge on wetland dynamics and processes (Mitsch & Gosselink, 2011). Such research was aimed at informing and providing the required impetus for formulating and implementing wetland conservation, management and rehabilitation plans, to be enforced by all tiers of government on a global scale (WRC, 2002).

Discernment elicited by the problem of declining water quality in South African freshwater systems is compounded by the escalating degradation and loss of floodplain wetlands, and in tandem, the role of riparian habitats as water purifiers (Millennium Ecosystem Assessment, 2005; Macaskill, 2010). A number of studies have shown that floodplain wetlands have considerable influence over the quality of water in adjacent rivers (Blackwell *et al.*, 2002). This theory has received such mammoth support that the restoration of natural wetlands and the creation of artificial wetlands have long surpassed the conceptual phase, having materialised both nationally and internationally (Vymazal, 2009). However, the creation of artificial wetlands with the objective of having them serve the functions of naturally occurring wetlands, have in many cases gone in vain (Mitsch & Gosselink, 2011). The underlying factor responsible for the low success rate of such endeavours is their inability to replace the pollutant removal efficacy of natural wetlands, owing to their lack of mature microbial communities, plant communities with high diversities, and stabilized rhizosphere environments (Zhang *et al.*, 2011). It is thus imperative that the remnants of once vast and natural wetland areas are effectively conserved and managed, with the maintenance of acceptable water quality, the protection of biotic habitats and the preservation of irreplaceable ecosystem functions being the prime foci (Collins, 2005).

The floodplain wetlands of interest in this study fall within selected stretches of riparian zone along the banks of the Lower uMngeni River. The uMngeni catchment is amongst the most heavily utilised in KwaZulu-Natal, comprising concentrated residential, agricultural and industrial land-use conglomerates (Department of Water Affairs and Forestry (DWAF) & Umgeni Water, 1996). It is thus likely that nutrient loading has compromised the quality of water in both the existing floodplain wetlands as well as the adjacent river (WRC, 2002). Riverine wetlands are typically dominated by uni-directional horizontal water flow which results in the transportation of drainage from the upland watershed areas, into the wetland environment and adjacent river channel (Smith *et al.*, 1995). However, seasonal dry spells in KwaZulu-Natal causes the water table in wetlands and upland areas to drop below the recorded norm. This incites the movement of water in the opposite direction, such that the higher level of water in the river system acts to recharge groundwater, and thus wetland interstitial water, resulting from the natural propensity of such interconnected systems to restore and maintain a state of equilibrium (Winter, 1999). These seasonal changes in the water table, which result in changes in the direction of groundwater flow, are depicted in Figure 5.5 below.

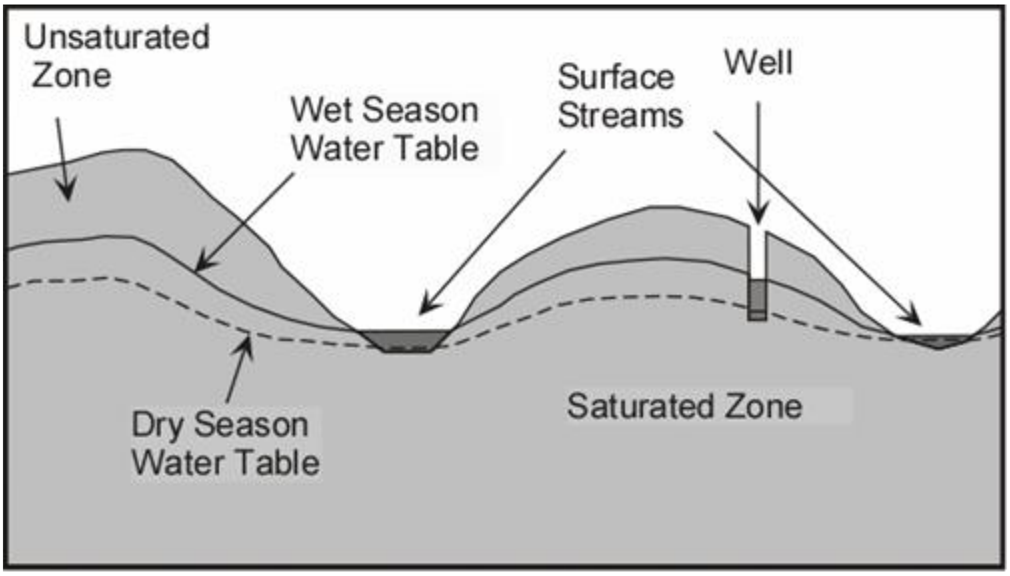


Figure 5.5: Changes in the water table consequential to seasonal variations in rainfall (sourced from Nelson, 2013).

In light of the above, by investigating the changes in water quality of wetland interstitial waters and adjacent river water during the wet and dry periods, in conjunction with annual precipitation data, it is possible to identify the potential sources of pollution and determine the acceptability of constituent element concentrations from an ecosystem health perspective. The objectives of this study are therefore to: (1) establish and compare the level of specific elements in wetland interstitial water and river water samples measured in each season; (2) identify the possible origins of detected elements by examining surrounding land uses and annual precipitation patterns; (3) compare concentrations of physico-chemical parameters in wetland interstitial water samples and river water samples to acceptable DWAF standards.

5.3.3 Site description

The uMngeni River traverses its 4 740 km^2 catchment and is amongst the most important and yet most heavily utilised fluvial systems in the KwaZulu-Natal province, providing refuge to a plethora of biotic species and a source of water supply to surrounding communities (Brijlal, 2005). Anthropogenic influence on the river can be noted from its source in the Drakensburg Mountains to its point of discharge, past the lagoon system within the Durban Metropolitan Area, into the Indian Ocean (DWAF & Umgeni Water, 1996). Human activities situated in close proximity to the river include residential, industrial and agricultural activities (WRC, 2002). Apart from the observable effects of human disturbance on this perennial river system, it is evident that the numerous wetlands delineating the river floodplain have also been subject to the negative impacts of continued, and increasingly intensive, human activities (Jewitt & Kotze, 2000). As propounded by Begg (1986), the size and shape of these wetlands have changed drastically over the last few decades, leaving these micro-environments vulnerable and at a high risk of irreversible damage and destruction.

Uninhibited human influence within the catchment and particularly within close proximity to wetland features may result in encroachment upon wetlands, which if persists indefinitely, will lead to degradation and the ultimate destruction of these systems (WRC, 2002). Hence, it is critical that management measures, informed by policies, are developed and executed with the aim of promoting ecosystem health and human well-being (DWAF & Umgeni Water, 1996).

A revision of the study area, presented in section 3.2 of this thesis, will be included in the version of this paper submitted to the journal.

5.3.4 Methodology

A total of six sampling points were strategically selected in line with the study area's largest floodplain wetlands located within the stretch extending from approximately 3.4 km upstream of the river mouth at Blue Lagoon north-westwards towards the residential area of Reservoir Hills. Piezometers were installed at each of the six selected wetlands within the Lower uMngeni sub-catchment; four along the southern banks of the river and two along the northern banks. The selection was based mainly on anthropogenic activities taking place within the vicinity of the Lower uMngeni sub-catchment and accessibility to the river and identified wetlands.

The six wetland interstitial water sampling points, where piezometers were located, are referred to as W1, W2, W3, W4, W5 and W6. Similarly, the river water sampling points are situated in the water column; either in close proximity to or along the same transects as the installed piezometers, and are labelled R1, R2, R3, R4, R5 and R6 to complement areas represented by the wetland interstitial water sampling points.

Sampling points W1, W2, W3 and R1, R2 and R3 are located within the residential area of Reservoir Hills. The most proximal land uses impacting on sample points W1 and R1 are informal residential and light industry; whilst those impacting on W2 and R2, and W3 and R3 are chiefly recreational facilities, particularly a recreational field and golf course respectively. To be noted however, is that subsistence agriculture and informal residential development characterising sections of the river bank near points W2 and R2 may act to further deteriorate water quality within nearby wetlands and the river. Sample points W4 and R4 are situated under the N2 bridge approximately 3 km south east of sample points W1 and R1, and may be impacted on by the Clare Estate dump site, light industrial activity and the movement of heavy vehicles on untarred roads within the immediate vicinity of the river channel and wetland area. Sample points W5, R5, W6 and R6 are separated from heavy industrial activity taking place in the Springfield Flats industrial area by a 50 m wide vegetated strip.

Figure 5.6 below illustrates the location of the selected wetland interstitial water and river water sampling sites within the study area.

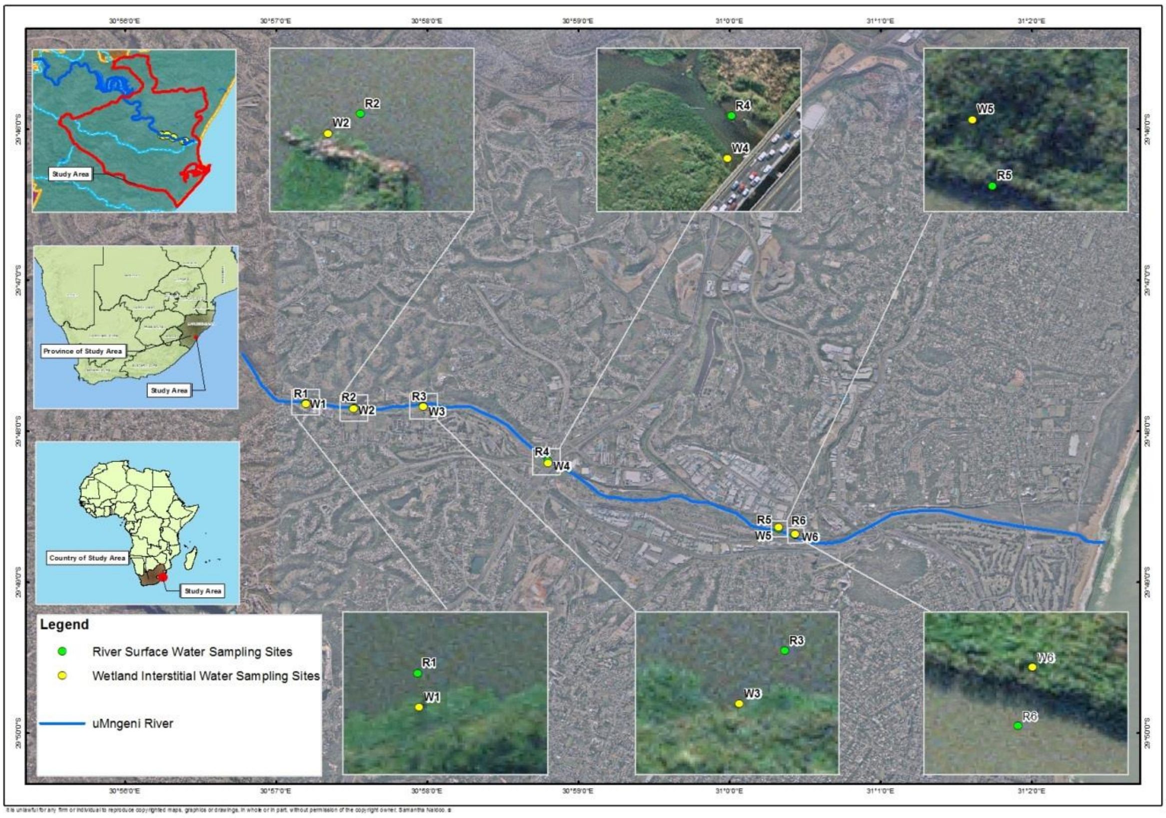


Figure 5.6: Location of wetland interstitial water and river water sampling sites within the study area

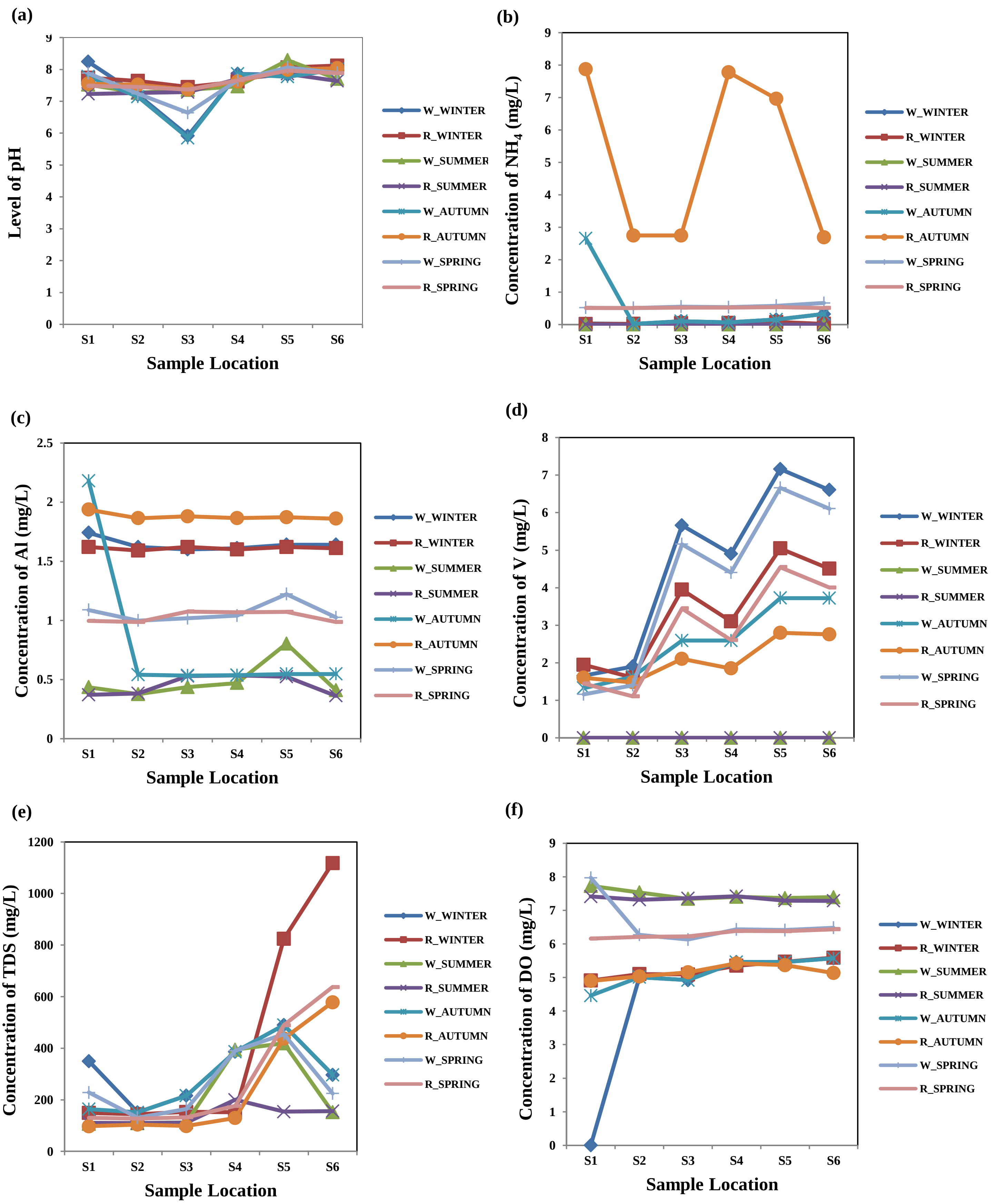
Sampling was conducted monthly from March 2012 to January 2013. This involved the extraction of water samples from installed piezometers and the collection of water samples from adjacent points in the river for comparative purposes. These samples were contained in 1L sterilised polyethylene bottles and kept at cool temperatures prior to, and during, transportation of the samples to the School of Chemistry Laboratory, University of KwaZulu-Natal (Westville Campus). Physical and chemical analyses procedures were undertaken to determine the level of pH, DO, NH_4^+, Al, Ba, Co, V, Cr, Ni, Fe, Zn, Pb, Cu and Mg in water samples using a combination of ion chromatography, flame photometry and ICP-OES. The correlation function was used to statistically analyse the attained results and thereby determine if relationships exist between the concentrations of parameters in wetland interstitial water samples and river water samples. Comparison of the detected concentration of variables with the acceptable DWAF limits provided a basis for appraising the water quality of wetland interstitial water and river water.

5.3.5 Results and Discussion

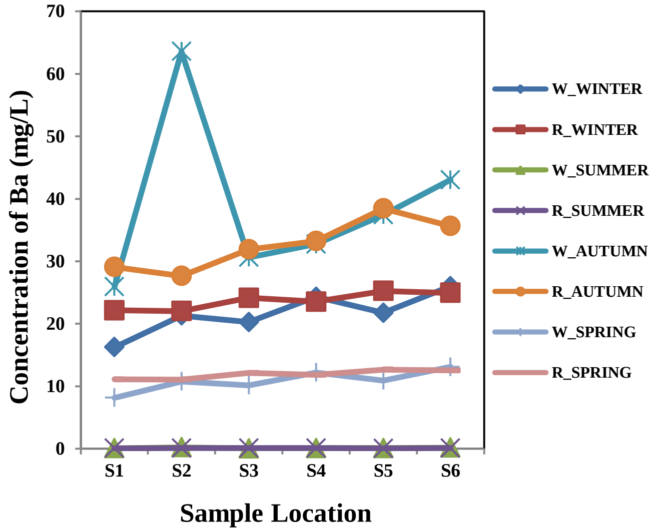
5.3.5.1 Seasonal changes in water quality parameters

Seasonal variations in the water quality parameters measured in wetland interstitial water samples and river water samples are illustrated in Figures 5.7 (a) - (o) below. In these figures, "W" refers to the wetland interstitial water samples and "R" denotes the river water samples. The relevant seasons are specified following these abbreviations in each case. Sample location is denoted by "S1" - "S6", which refer to the general location in which the wetland interstitial water samples and corresponding river water samples were collected.

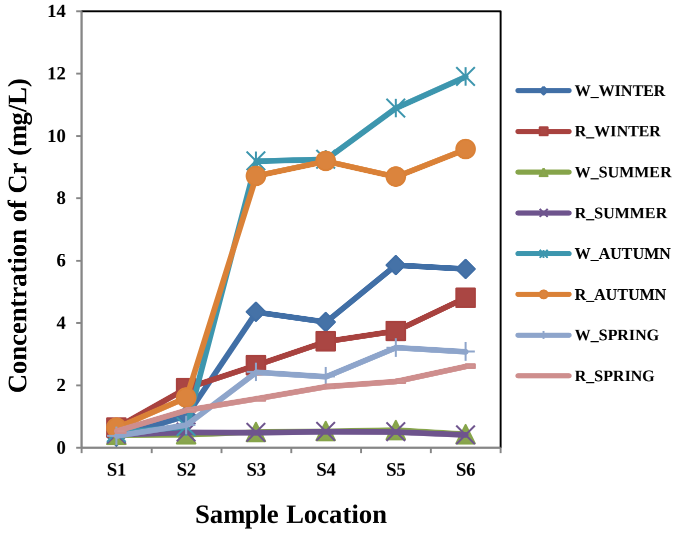
Figure 5.7(a) – (o)….\


(a)
Level of pH
Sample Location
S1 S2 S3 S4 S5 S6
W_WINTER
R_WINTER
W_SUMMER
R_SUMMER
W_AUTUMN
R_AUTUMN
W_SPRING
R_SPRING
(b)
Concentration of NH4 (mg/L)
Sample Location
(c)
Concentration of Al (mg/L)
Sample Location
(d)
Concentration of V (mg/L)
Sample Location
(e)
Concentration of TDS (mg/L)
Sample Location
(f)
Concentration of DO (mg/L)
Sample Location

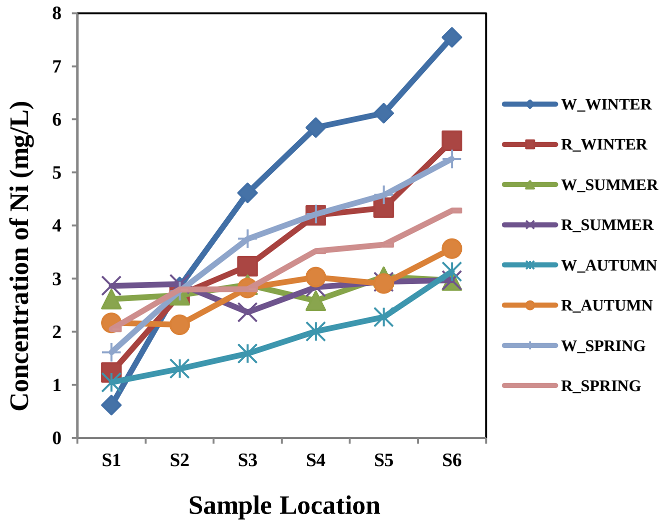
(g)


Concentration of Ba (mg/L)
70
60
50
40
30
20
10
0
S1
S2
S3
S4
S5
S6
Sample Location
W_WINTER
R_WINTER
W_SUMMER
R_SUMMER
W_AUTUMN
R_AUTUMN
W_SPRING
R_SPRING

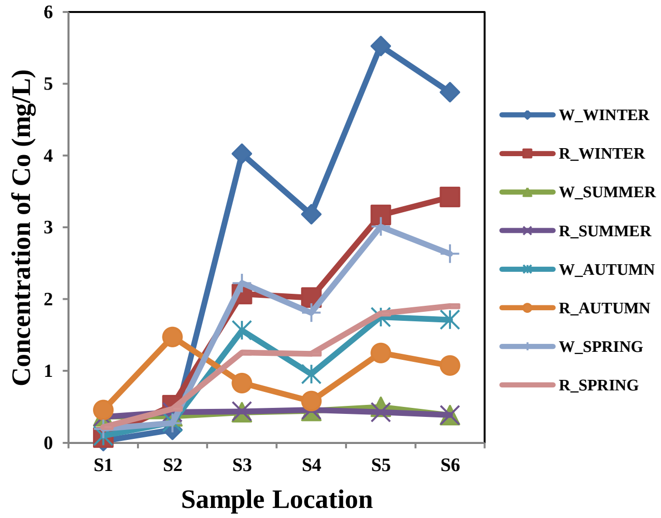
(h)


Concentration of Cr (mg/L)
14
12
10
8
6
4
2
0
S1
S2
S3
S4
S5
S6
Sample Location
W_WINTER
R_WINTER
W_SUMMER
R_SUMMER
W_AUTUMN
R_AUTUMN
W_SPRING
R_SPRING

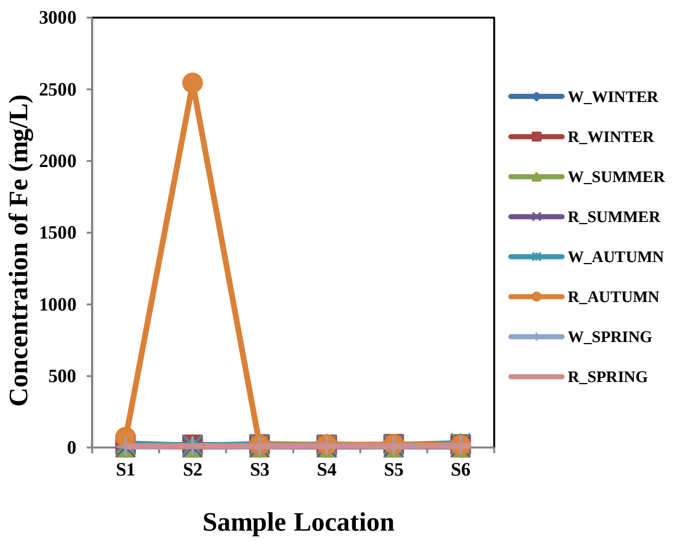
(i)


Concentration of Ni (mg/L)
8
7
6
5
4
3
2
1
0
S1
S2
S3
S4
S5
S6
Sample Location
W_WINTER
R_WINTER
W_SUMMER
R_SUMMER
W_AUTUMN
R_AUTUMN
W_SPRING
R_SPRING

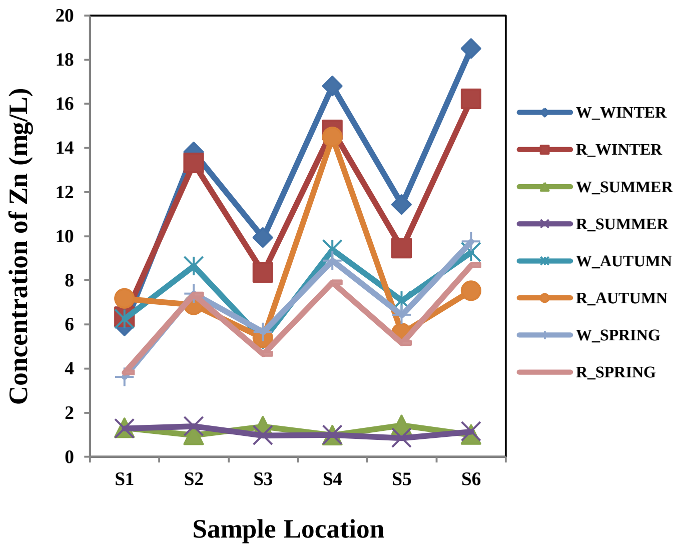
(j)


Concentration of Co (mg/L)
6
5
4
3
2
1
0
S1
S2
S3
S4
S5
S6
Sample Location
W_WINTER
R_WINTER
W_SUMMER
R_SUMMER
W_AUTUMN
R_AUTUMN
W_SPRING
R_SPRING

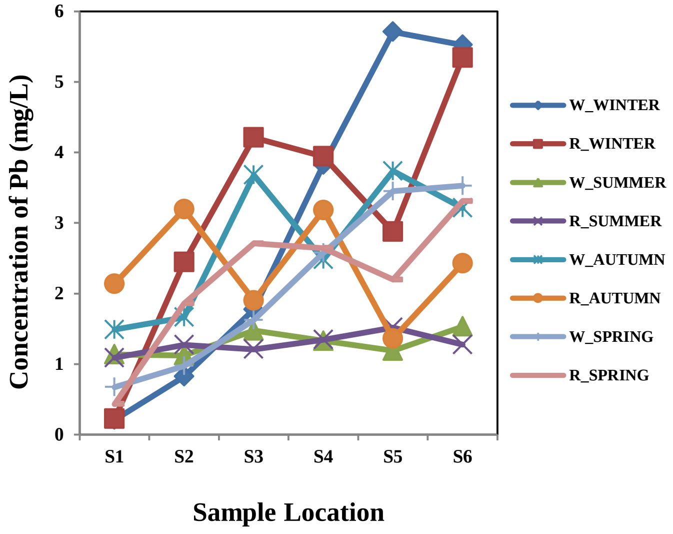
(k)


Concentration of Fe (mg/L)
3000
2500
2000
1500
1000
500
0
S1
S2
S3
S4
S5
S6
Sample Location
W_WINTER
R_WINTER
W_SUMMER
R_SUMMER
W_AUTUMN
R_AUTUMN
W_SPRING
R_SPRING

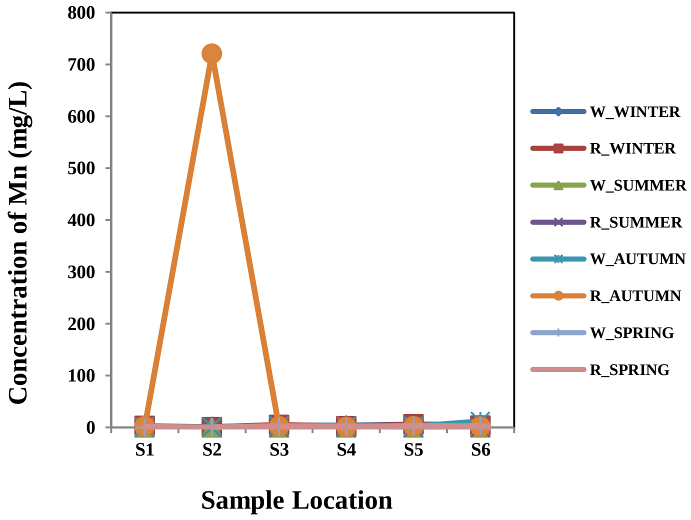
(l)


Concentration of Zn (mg/L)
20
18
16
14
12
10
8
6
4
2
0
S1
S2
S3
S4
S5
S6
Sample Location
W_WINTER
R_WINTER
W_SUMMER
R_SUMMER
W_AUTUMN
R_AUTUMN
W_SPRING
R_SPRING

(m)



(n)



(o)

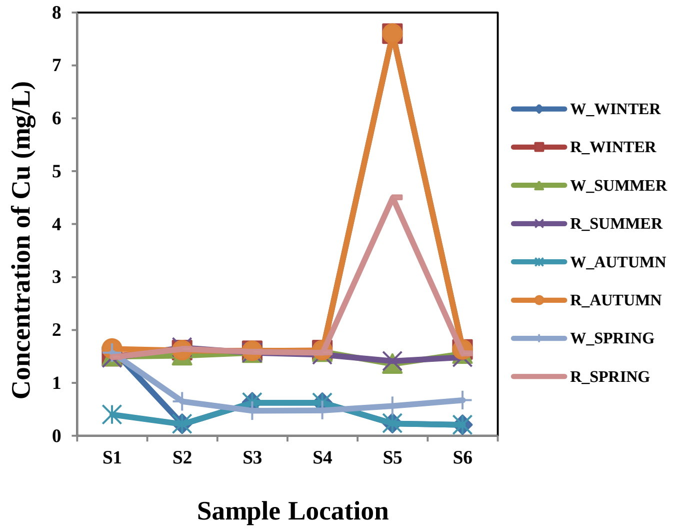


Figure 5.7(a) - (o): Seasonal change in element concentrations at each sampling site

The trends displayed in each of the above figures are explained below. Furthermore, potential anthropogenic sources of, and/or influences on, each of the selected parameters are specified in Table 5.5.

Table 5.5: Potential anthropogenic sources of and/or influences on measured parameters detected at the selected sampling sites

(Lenntech, 2009; Shaffer, 2001; Adriano, 2001; DWAF, 1996a; 1996b; Environment Agency, 2003).

Site	Potential Anthropogenic Sources/Influences	Details	Parameter
S3	Papwa Sewgolum Golf Course	Fertilisers and pesticides	Al; NH_4^+; Ba; Cr; Co; Cu; Fe; Pb; Mn; Ni; pH; V; Zn;
		human and vehicle movement (loose sediment particles – increased suspended solids)	DO
		Organic wastes	DO
		Salt-enriched runoff	TDS
S2	Subsistence agriculture	Fertilisers and pesticides	Ba; Cr; Co; Cu; Fe; Pb; Mn; TDS; Zn
		Human movement (loose sediment particles – increased suspended solids).	DO
		Organic wastes	DO
		Salt-enriched runoff	TDS
S1 and S4	Light industrial activity at Four-Ways Manufacturers and near the N2 bridge	Industrial cleaning agents	NH_4^+; Al; pH
		Waste/by-product discharges	Pb; Zn; Ba; Co; Fe; Mn; Ni; Al; DO; V
		Vehicle movement – fallout of lead dust and runoff of lead emissions	Pb
		Fossil fuel combustion	Pb
		Salt-enriched runoff	TDS
		Human and vehicle movement (loose sediment particles - increased suspended solids)	DO
S5 and S6	Heavy industrial activity in the Springfield Industrial Area	Industrial cleaning agents	Al; NH_4^+; pH
		Industrial wastes/by-products	NH_4^+; Al; V; Cr; Zn; Cu; Co; Fe; Pb; Mn; Ni;
		Atmospheric fallout of barium sulphate	Ba
		Human movement (loose sediment particles – increased suspended solids).	DO
		Organic wastes	DO
		Fossil fuel combustion	Pb
		Salt-enriched runoff	TDS
S1 and S2	Informal residential areas	Domestic cleaning agents	NH_4^+; Fe; Pb; pH
		Coal combustion	Ba
		Improper disposal of electronic wastes	Ba
		Organic matter - raw sewage	pH; DO

Site	Potential Anthropogenic Sources/Influences	Details	Parameter
		Loose sediment particles - human movement	DO
		Salt-enriched runoff	TDS
S2	Recreational field	Fertilisers and pesticides	Ba; Cr; Co; Cu; Fe; Pb; Mn; Zn
		Loose sediment particles - human and vehicle movement	DO
S4	Clare Estate Dump Site	Leachate	Mn; Fe; NH_4^+

*This table only considers water quality parameters that were assessed in this study

5.3.5.1.1 pH

The pH mostly varied between the values of 7 and 8.29 in both wetland interstitial and river water samples over all seasons (Figure 5.7(c)). However in winter, pH of the wetland interstitial water peaked at sample site W1 (8.24), showing alkaline conditions, and troughed at sample site W3 (5.92), implying the prevalence of acidic conditions. In summer, the pH of wetland interstitial water peaked at W5 (8.29) and dipped at sample site W3 in autumn (5.85) and spring (6.64). Acidic conditions at site W3 in summer promotes the desorption of heavy metals by sediment particles, hence decreasing water quality by increasing water toxicity (Hatje *et al.*, 2003). Similarly, alkaline waters, detected at sites W1 and W5 during winter, promote the dissolution of certain ions which also elevate the toxicity of water-based environments (Hatje *et al.*, 2003).

Alkaline conditions are likely a result of natural processes such as increased biological activity, and geological and atmospheric processes at sites W1 and W5. Acidic conditions at site W3 may be related to the influence of organic acids and other polyphenol compounds produced by the dense riparian vegetation at the site which leaches into the system. The high and low pH values recorded within wetlands during the dry seasons may have resulted from the concentration of anthropogenic pollutants from human activities and influences specified in Table 5.5. Overall, pH levels detected over all seasons are acceptable, but should be monitored on a periodic basis.

5.3.5.1.2 Ammonium

Figure 5.7(a) illustrates that NH_4^+ concentrations are generally very high across the sampling sites, ranging between 0.02 mg/L and 7.87 mg/L over the period of study. Detected levels in wetland interstitial waters were found to be higher than that of river water in the seasons of winter and spring. Additionally, in winter and spring, the highest concentrations of NH_4^+ were detected at sample sites W3, W5 and W6, and sample sites W3 and W6 respectively. NH_4^+ was undetected in wetland interstitial water and river water samples collected in the summer months. In autumn, however, the concentrations of NH_4^+ detected in river water samples were considerably higher than those in wetland interstitial water samples, and higher than NH_4^+ concentrations detected over all other seasons as well. Concentrations of NH_4^+ in wetland interstitial water peaked at sample sites R1, R4 and R5 in the autumn months.

With regards to the high detected values in wetlands during winter and spring, possible natural sources of NH_4^+, given the dense and largely undisturbed riparian vegetation at these sites, may include decomposing plant and animal material and animal excretion. However, the location of potential anthropogenic sources of NH_4^+ in close proximity to the afore-mentioned sites (refer to Table 5.5), implies the likelihood of anthropogenic input of NH_4^+ at these sites. The considerably high amounts of NH_4^+ in river water detected during autumn are likely to have been carried from upstream human activities, following an anomalous pollution incident.

5.3.5.1.3 Aluminium

From observation of Figure 5.7(b) it can be noted that the concentrations of Al in collected water samples are high, ranging between 0.36 mg/L and 2.18 mg/L over all seasons. In the winter and spring months, the concentrations of Al in river water samples were found to be lower than that of the corresponding wetland interstitial water samples. Hence, it is unlikely that these high concentrations are a result of the recharge of wetlands by river water. It is possible, however, that drainage and wastes from proximal land uses may have contributed to the high Al levels in these wetland systems (refer to Table 5.5). In line with this, although Al levels in surface waters may have been mobilised from soils by natural weathering and acidification processes, the main sources of Al are anthropogenic (DWAF, 1996a).

Also to be noted is that the concentration of Al peaks in wetland interstitial waters at site W1 in both winter and autumn; at site W5 in winter, summer and spring; and at site W6 in winter; correlating well with potential land use sources of Al near these sites. In autumn, the concentrations of Al at the remaining sites are markedly higher in river water samples (R2 – R6). In this case, it is likely that the high concentrations in river water may have resulted from a pollution event at site S1, or further upstream. Measured levels of Al are lowest in the summer months, and concentrations measured in wetland interstitial water and river water samples are very similar. Dilution of pollutants due to rainfall events and the intricate link between the wetland and river systems are likely causes of the afore-mentioned results.

5.3.5.1.4 Vanadium

With an annual minimum and maximum level of 1.10 mg/L and 7.16 mg/L respectively, it is evident that concentrations of V are extremely high in both the wetland and river systems. This is illustrated in Figure 5.7(d) above. The concentration of V measured for all wetland interstitial water samples was highest in winter and spring. The concentration of V was highest at sites W3 and W5 in winter and spring; and at sites W3, W5 and W6 in autumn. Overall, detected V concentrations in wetland interstitial water samples were higher than those of river water samples over all seasons. Exceptions are the summer months when V was not detected, and at sample site S1 where the concentration detected for river water samples was slightly higher in all seasons.

Given the fact that only small quantities of V occurs in freshwater systems, it is likely that amounts exceeding the acceptable DWAF limit have been sourced from anthropogenic activities. This is supported by the fact that all measured V levels in wetland interstitial water samples exceed those of their corresponding river water samples. Furthermore, wetland interstitial water sites at which peaks in V concentrations were noted are situated in close proximity to potential land use sources (refer to Table 5.5). The higher V levels measured at all river water sampling sites across all seasons (except summer where it is undetected) may be due to the continuous input of chemicals and wastes by upstream anthropogenic activities.

5.3.5.1.5 Total Dissolved Solids (TDS)

As illustrated in Figure 5.7(e), TDS levels range from a low of 97.55 mg/L to a high of 1118 mg/L over the period of study. Levels of measured TDS were found to be higher in wetland interstitial water than in river water over all seasons. The exceptions include sample sites S5 and S6 in winter and spring, and sample site S6 in autumn, where levels in river water are shown to be higher. Peaks in the level of TDS can be observed at sample sites W1, W4 and W5 in winter and spring, and at sites W4 and W5 in summer and winter. It is possible that these high TDS levels are closely related to the location of the afore-mentioned sample sites in close proximity to human activities (refer to Table 5.5). Furthermore, high rainfall in summer may have resulted in the transportation of various salts from the watershed, *via* overland flow, into the wetland systems represented by sample sites W4 and W5, thereby increasing levels of TDS at these sites.

During the dry seasons, the levels of TDS detected in river water samples were higher than those measured in the corresponding wetland interstitial water samples at specific sites. Drainage from upstream anthropogenic activities as well as nearby land uses may have contributed to the high TDS levels at these sites. Also to be noted is that TDS concentrations are highly dependent on physical processes such as rainfall, evaporation and the decomposition of plant material (DWAF, 1996a). Hence, the decomposition of vegetation within the large riparian area immediately up-bank of these sites, as well as high rates of evaporation and low rainfall during the dry seasons, may have also contributed to the high TDS levels measured at these sites.

5.3.5.1.6 Dissolved Oxygen

As illustrated in Figure 5.7(f), DO levels range between 4.5 mg/L and 8 mg/L in both wetland interstitial and river water samples over all seasons. An exception is the DO level in wetland interstitial water samples collected at sample site W1 in winter, which was undetected. The highest levels of DO were measured in the summer and spring months. This could be attributed to rainfall events within these seasons which increase the turbulence of waters, causing entrainment of air from the atmosphere (DWAF, 1996a).

The DO detected in wetland interstitial water and river water samples dropped at sample sites R1, W1, R3 and W3 in winter; R2, W3, R5 and W5 in summer; R1, W1, W3 and R6 in autumn; and W2, and W3 in spring. In the months of winter and autumn, DO levels dropped to below the acceptable minimum limit of 6 mg/L, indicating the prevalence of anoxic conditions. These conditions are a likely result of anthropogenic activity within the study area (refer to Table 5.5). However, a further possibility is that a lack of rainfall during these seasons had reduced the natural diffusion of gaseous oxygen from the atmosphere into water. Additionally, apart from potential anthropogenic influences, the decline in DO at sites R2, W3, R5, and W5 during summer may be a consequence of re-suspended anoxic sediments resulting from rainfall events, or the transportation of suspended solids and organic matter from the vegetated area within the vicinity of these sample sites (DWAF, 199a).

5.3.5.1.7 Barium

As shown in Figure 5.7(g), the levels of detected Ba in both wetland interstitial water and river water were found to be highest in the months of autumn. Concentrations detected over the study period range from a minimum of 0.05 mg/L to a maximum of 63.59 mg/L. This may be due to the concentration of pollutants in wetland and river systems, resulting from a lack of rainfall. In winter and spring, Ba concentrations detected in wetland interstitial water and river water samples showed an inverse relationship, such that concentrations peaked in river water samples at sites R1, R3 and R5 whilst concentrations in wetland interstitial water samples peaked at sites W2, W4 and W6. In summer, Ba concentrations are higher in wetland interstitial water at sites S2, S5 and S6; higher in river water at sites S3 and S4; and are almost the same at wetland interstitial and river water sites at site S1. In spring, the concentrations of Ba in wetland interstitial water and river water are similar across sampling sites except at sites S2 and S6, where concentrations in wetland interstitial water sites are higher. To be noted is that similar peaks in Ba concentrations can be seen at sites S2 and S6 over the summer and spring months.

Proximal land use activities at sites S1, S2, S3, S4, S5 and S6 may be possible contributors to high Ba levels detected in wetland interstitial water and river water samples collected at these sites (refer to Table 5.5), especially during summer. In general, drainage from upstream land uses and activities may have contributed to higher concentrations of Ba detected at certain river water sampling sites within particular seasons (mentioned in the above explanation).

5.3.5.1.8 Chromium

The concentrations of Cr detected in wetland interstitial water samples are only lower than river water samples at sites S1 and S2 over all seasons (Figure 5.7(h)). Concentrations of Cr over all seasons are high. This is such that the lowest detected concentration was 0.35 mg/L and the highest was 11.90 mg/L. The concentration of Cr is also recorded to be highest in the winter and autumn months of the year.

Cr concentrations peak at sites W3 and W5 over all seasons, and are also considerably high at site W6 in the winter, autumn and spring months. The absence of rainfall events in the dry seasons may have resulted in a lack of contaminant dilution in both the river and floodplain wetlands. This may have resulted in the accumulation and concentration of Cr derived from proximal anthropogenic activities (refer to Table 5.5) in the wetland areas in particular.

Higher levels detected in summer, however, may have resulted from the deposition of wastes derived from the catchment *via* surface flow. With regards to river water samples, concentrations of measured Cr at sites R1 and R2 are higher than those detected at sites W1 and W2 over all seasons. It is therefore possible that the higher level of Cr in river water resulted from the discharge of chemicals and wastes by upstream land uses.

5.3.5.1.9 Nickel

As depicted in Figure 5.7(i), the levels of Ni detected across all sites during the period of study ranged from 0.62 mg/L to 7.54 mg/L. The concentration of Ni was found to be highest in samples collected during the winter season. The concentration of Ni was also shown to be higher in river water samples than in the corresponding wetland interstitial water samples at site S1 over all seasons. High levels of Ni at this site throughout the year implies the possibility that upstream activities that either use or produce substances containing Ni continuously discharge operational wastes which eventually enter the river system.

In winter and spring, the level of measured Ni in wetland interstitial water samples were higher than levels detected in river water samples across all sites, except at site S1. Due to the lack of rainfall in these months, a possible contributor to the high concentrations of Ni in these wetland systems are proximal human activities (refer to Table 5.5). It was also noted that in autumn, Ni concentrations were higher in river water samples than in wetland interstitial water samples at all sample sites. Similarly, the lack of rainfall during the months of autumn may have resulted in the concentration of contaminants (including Ni) derived from upstream land uses which are carried by river water to downstream areas. In summer, the levels of Ni in wetland interstitial water samples and river water samples are inversely related. This implies the possibility that geochemical properties in the wetland interstitial soil had hindered the transfer of Ni between the wetland and river systems.

5.3.5.1.10 Cobalt

The concentration of Co detected over all seasons in the selected wetland interstitial water and river water sampling sites ranged in value from 0.02 mg/L to 5.52 mg/L. Furthermore Co levels in wetland interstitial and river water samples were found to be highest in the months of winter, followed by slightly lower levels detected in the months of spring (Figure 5.7(j)). Co concentrations in wetland interstitial water samples were lower than that of the corresponding river water samples at sample sites S1 and S2 for all seasons. This may be attributed to the fact that the land uses within the immediate surrounds of sites S1 and S2, do not use or result in the production of substances containing Co. It could also indicate that the higher Co levels detected in river water at these sites may have resulted from the continuous release of chemicals and wastes from upstream anthropogenic activities into the river system.

In winter, autumn and spring, levels of measured Co were shown to be higher in wetland interstitial water sampling sites than in the corresponding river water sampling sites for all sites (except sites S1 and S2 as explained above). This indicates the possible contribution of anthropogenic activities within the vicinity of these sampling sites to the high levels of Co detected (refer to Table 5.5). In summer, however, Co levels in wetland interstitial water are only higher than those of river water at sample sites S1 and S5, suggesting the input of Co by human activities within the vicinity of these sample sites (refer to Table 5.5). The higher Co concentrations measured in river water at sites S2, S3, S4 and S6 imply possible input from upstream land uses.

5.3.5.1.11 Iron

As depicted in Figure 5.7(k), measured concentrations of Fe across all sites and seasons range from a minimum of 0.53 mg/L to a maximum of 2443.83 mg/L. Detected levels in wetland interstitial water and river water samples are inversely related in winter and spring. This implies the possibility that the transfer of Fe between the wetland and river systems had been hindered by geochemical properties of wetland soils. Concentrations peak in wetland interstitial water at sample sites W2, W4 and W6 in the winter and spring months.

In summer, Fe levels are higher in wetland interstitial water than in river water at sites S2, S5 and S6, whilst concentrations in river water are higher in the remaining sites. Furthermore, in summer, concentrations of Fe peak at sites W2 and W5 in wetland interstitial water, and at sites R2 and R4 in river water. Rainfall events in the summer months may have resulted in the entrance of drainage from proximal land uses into the wetland environment and connected river system (refer to Table 5.5). In autumn however, Fe was found to be higher in river water than in wetland interstitial water at all sites except sites S3 and S6. Furthermore, a marked peak in Fe levels was detected at site R2 in autumn. The strikingly high level of Fe detected at site R2 during autumn may have been deposited by upstream land uses or anthropogenic activities (refer to Table 5.5) that take place in close proximity to the river channel.

5.3.5.1.12 Zinc

The concentrations of detected Z were considerably high across all sites, ranging between 0.85 mg/L and 18.5 mg/L over the period of study. Zn levels in wetland interstitial waters were found to be highest at sites W2, W4 and W6 in the winter, autumn and spring months. In winter and spring, the concentrations of Zn in wetland interstitial water were slightly higher than measured concentrations in corresponding river water samples at all sites except site S1. (Figure 5.7(l)).

Contrastingly, Zn levels in wetland interstitial waters peak at sites W1, W3 and W5 in summer, and are also higher than river water concentrations at these sites. The deposition of substances comprising Zn from nearby human activities (refer to Table 5.5) may have resulted in the higher levels of Zn measured at these sites following rainfall events. Furthermore, higher Zn concentrations in river water samples than in wetland interstitial water samples at sites S2 and S6 in summer imply influences from upstream human activities In autumn, the concentration of Zn in river water was found to be higher than wetland interstitial water at sites S1, S3 and markedly higher at site S4. Given the lack of rainfall during this season, a likely cause of higher river water concentrations is the entrance of chemicals and waste from upstream land uses into the river system. Furthermore, the peak at R4 may have resulted from an anomalous pollution incident further upstream.

5.3.5.1.13 Lead

As portrayed in Figure 5.7(m), the concentrations of measured Pb were found to be extremely high, most especially during the months of winter. Pb concentrations ranged from a level of 0.21 mg/L to a value of 5.71 mg/L across all sites and over all seasons. It can be observed that detected levels in wetland interstitial water sampling sites and the corresponding river water sampling sites are inversely related across all seasons. Similar to the cases of Fe and Ni, it is possible that geochemical properties of wetland soils have constrained the transfer of Pb between the wetland systems and the adjacent river.

Pb concentrations in wetland interstitial water samples were highest at sites W4, W5 and W6, and were highest in river water samples at sites R3, R4 and R6 in both winter and spring. In autumn, Pb concentrations were found to peak at sites W3 and W5 in wetland interstitial water samples, and R2, R4 and R6 in river water samples. Furthermore, in summer, Pb levels peaked in wetland interstitial water samples at sites W3 and W6, and in river water samples at sites R2 and R5. The high Pb levels detected in the wetland interstitial water sample sites mentioned above (W3, W4, W5 and W6) may be attributed to human activities within the vicinity of these sites (refer to Table 5.5); whilst high concentrations of Pb in river water (R2, R3, R4, R5 and R6) may have resulted from the influence of upstream land uses.

5.3.5.1.14 Manganese

Concentrations of Mn varied between the values of 0 mg/L and 720.86 mg/L across all sites and seasons. In winter and spring the concentration of Mn in river water was higher than those of wetland interstitial water at sites S1, S3 and S5. Additionally, Mn levels peaked at sample sites R1, R3 and R5; whilst concentrations in wetland interstitial waters peaked at W3, W5 and W6. These patterns are depicted in Figure 5.7(n). Considering that winter and spring represent "dry" seasons in the area of study, high Mn concentrations at W6 may have been sourced from land use activities within the vicinity of this site. High Mn levels at sites W3 and W5, however, may have resulted from the recharge of these wetland systems by river water containing high concentrations of Mn (R3 and R5).

In summer, Mn levels in river water samples peaked at sample sites R2 and R6 and concentrations in wetland interstitial water samples were found to be higher than measured levels in river water at sites S1, S3, S4 and S5. The high levels of Mn in the river water may be attributed to the input of substances containing Mn by upstream land uses. High levels of Mn at the wetland interstitial water sites (W1, W3, W4 and W5) may be closely linked to the possible anthropogenic sources of Mn within the vicinity of these sampling sites (refer to Table 5.5). In autumn, concentrations of Mn peaked in wetland interstitial water at site W6, and peaked even more markedly at site R2 in river water samples. The former may be related to proximal anthropogenic activities, whilst the latter to a once-off pollution incident in upstream areas.

5.3.5.1.15 Copper

It can be seen by inspection of Figure 5.7(o) that Cu levels across all sites are very high, ranging between the values of 0.21 mg/L and 7.59 mg/L over all seasons. In the summer months, concentrations of Cu in wetland interstitial and river water samples share an inverse relationship. Hence, concentrations are highest at sample sites R2, R3 and R4 in river water samples, and at sample sites W3, W4 and W6 in wetland interstitial water samples. To be noted is that detected levels in both wetland interstitial water and river water are low in comparison to levels detected in the other seasons, resulting from the dilution of pollutants by rainfall.

The concentration of Cu is higher in river water than in wetland interstitial water across all sites in winter, autumn and spring. An exception is the level of Cu in river water at site S1, which is slightly lower than the detected concentration in wetland interstitial water across these seasons. The implication of the above is that the measurement of high concentrations of Cu in river water was consequential to the input of contaminants from upstream land uses as well as the concentration of Cu in the river system resulting from a lack of rain during these "dry" seasons. Furthermore, Cu levels peaked at sample site R5 during winter, autumn and spring. In this case, the inference is that upstream pollution incidents may have resulted in peaks in river water concentrations of Cu at sample site R5 throughout the year; but may be less detectable in summer due to the dilution effect.

5.3.5.2 Correlation between wetland interstitial water and river water quality

The relationship between rivers and the wetlands that delineate their floodplain has been well-documented. In light of this, a correlation analysis was performed on the mean values of parameters calculated for each season in wetland interstitial water samples and river water samples to ascertain whether a relationship exists between the two datasets, and the respective strengths and implications thereof.

The correlation co-efficient, or the *r*-value, always ranges between the values of +1 and –1 (Rumsey, 2001). Table 5.6 displays the results of the Pearson-product correlation conducted.

Table 5.6….\

Table 5.6: Correlation (*r*-values) between Wetland Interstitial Water Sample and River Water Sample measurements

Parameter	***r*-value per Site**					
	S1	**S2**	**S3**	**S4**	**S5**	**S6**
NH_4^+	0.948	-0.691	-0.830	-1.000	-0.889	0.999
TDS	0.885	0.342	0.234	0.950	0.800	0.811
DO	0.835	1.000	1.000	1.000	0.999	0.979
Al	0.999	0.408	0.357	0.327	0.062	0.405
V	1.000	0.943	0.994	0.980	0.997	0.993
Ba	0.989	0.873	0.995	0.999	0.996	0.994
Cr	0.130	0.902	0.981	0.997	0.992	1.000
Ni	0.912	0.944	0.517	0.963	0.986	0.981
Co	0.427	-0.008	0.996	0.994	0.998	0.997
Fe	0.999	0.256	0.745	0.974	0.961	0.910
Zn	0.997	0.986	0.985	0.883	1.000	0.998
Pb	0.955	0.573	-0.199	0.957	0.750	0.987
Mn	0.260	0.617	0.938	1.000	1.000	0.352
Cu	-0.969	0.968	-1.000	-1.000	-1.000	-1.000

From inspection of Table 5.6 above it can be asserted that a strong positive linear relationship exists between wetland interstitial water and river water for the majority of the water quality parameters at most of the selected sample sites. With regards to TDS and Mn, the relationship between wetland interstitial water and river water is shown to be perfectly positive at 50% of the sites and either weakly or strongly positive at the remaining sites. Furthermore, either a weak, moderate or perfectly positive relationship exists between Al levels in wetland interstitial water and river water at most sampling sites. A weak positive relationship between Co concentrations in wetland interstitial water and river water is also noted at site S1. Similarly, in the case of Pb, a moderate, strong or perfectly positive relationship exists between wetland interstitial water and river water at most of the sites of interest. The implication is that the concentration of these parameters in wetland interstitial water and river water increase and decrease in tandem. For instance, increases in the concentration of these elements in wetland interstitial water are likely to lead to increases in the concentration of the same elements in river water and vice versa.

On the other hand, the relationship between wetland interstitial water and river water for $NH_4^+$and Cu are shown to be either a perfectly negative linear relationship or a strong negative linear relationship at most of the sample sites. Furthermore, the relationship between wetland interstitial water and river water for Pb is shown to be weakly negative at one sampling site (S3). Therefore, as the level of these parameters increase in wetland interstitial water samples, it decreases in the corresponding river water samples.

If the concentration, in each of these instances, is higher in wetland interstitial water than in river water, the implication is that these elements were carried by overland flow from proximal anthropogenic activities into the wetland environment. In terms of these particular elements, the wetlands therefore act as “pollution sinks”, or “natural purifiers” of water entering the adjacent river system. This is accomplished as a result of geochemical properties of wetland soil which restrict the movement of these parameters, or otherwise the slow release of these parameters into fast-flowing river water. If measured concentrations of these elements are higher in river water than in wetland interstitial water, it is likely that these elements were derived from upstream activities. It is probable that upstream pollution incidents, resulting in the release of these elements, were anomalies in each instance. This postulation is based on the rationale that continuous release of these elements from upstream activities would have likely resulted in high concentrations of these parameters in the river. This in turn would have increased measured concentrations in wetland features through wetland recharge during the dry seasons, showing evidence of positive linear relationships rather than the existing negative linear relationships.

There were only four instances in which no linear relationship was shown to exist between wetland interstitial and river water, specifically at site S5 for Al, site S1 for Cr and site S2 for Co. No relationship between the river and wetland interstitial water samples for these parameters imply that detected increases in wetland interstitial water are not accompanied by corresponding increases in river water and vice versa. If detected levels of these indicators are higher in river water, it is possible that this may have been the result of pollution incidents further upstream. However, geochemical properties of wetland soils may have prevented the diffusion of these elements into wetland systems. If the concentration of these parameters is higher in wetland interstitial water, then it is likely that geochemical properties of wetland soils are such that they restrict mobility of these parameters. This has potential implications of contamination if these elements continue to accumulate in the wetland soil with limited transfer to the river water.

Overall, it is evident that a strong linear relationship exists between river water and wetland interstitial water at the selected sites of study. Hence, although the wetland systems of interest may function as natural purification systems which act to improve the quality of water being released into the adjacent river, in most cases a decline in water quality of wetland systems lead to water quality deterioration in the river. Furthermore, during the dry seasons, increased pollutant concentrations within wetland systems may be attributed to their recharge by river water, which carries contaminants from upstream activities. Once-off pollution events are a likely cause of pollution spikes noted at particular sites or at all sites within a particular time period. However, if concentrations of certain parameters in river water are consistently higher than those of wetland interstitial water throughout the year, the implication is that there is continuous influence from upstream human activities in the area of study.

5.3.5.3 Compliance with acceptable water quality limits prescribed by DWAF (1996)

The mean seasonal concentrations of water quality indicators in both wetland interstitial water and river water samples were compared to the limits prescribed by DWAF (1996) to determine the acceptability of water quality in the floodplain wetlands of interest and the adjacent river channel.

It must be noted that water quality parameters for which no specific Target Water Quality Values were stipulated by DWAF (1996a; 1996b) have not been considered.

Tables 5.7 – 5.10 show the mean seasonal concentrations of water quality parameters detected at each wetland interstitial water sampling site in relation to their respective DWAF limits.

Table 5.7: Summer concentrations of water quality parameters at each wetland interstitial water sampling site

Parameter	DWAF Limit	Mean Concentration Per Site In Summer					
		S1	**S2**	**S3**	**S4**	**S5**	**S6**
NH_4^+	0.007	ND	ND	ND	ND	ND	ND
Al	0.01	0.435	0.376	0.436	0.471	0.803	0.410
V	0.1	ND	ND	ND	ND	ND	ND
Cr	0.007	0.399	0.418	0.494	0.521	0.556	0.423
Zn	0.002	1.309	0.978	1.374	0.976	1.430	0.998
Pb	0.0002	1.138	1.123	1.475	1.330	1.189	1.533
Mn	0.18	0.033	0.035	0.035	0.032	0.018	0.017
Cu	0.0003	1.498	1.517	1.567	1.585	1.360	1.549

Table 5.8: Winter concentrations of water quality parameters at each wetland interstitial water sampling site

Parameter	DWAF Limit	Mean Concentration Per Site In Winter					
		S1	S2	S3	S4	S5	S6
NH_4^+	0.007	0.028	0.016	0.099	0.068	0.152	0.321
Al	0.01	1.740	1.620	1.600	1.610	1.640	1.640
V	0.1	1.659	1.903	5.659	4.903	7.159	6.603
Cr	0.007	0.348	1.026	4.348	4.026	5.848	5.726
Zn	0.002	5.926	13.800	9.926	16.800	11.426	18.500
Pb	0.0002	0.210	0.824	1.775	3.824	5.710	5.524
Mn	0.18	0.011	0.884	4.011	3.884	5.511	5.584
Cu	0.0003	1.636	0.221	0.625	0.623	0.234	0.205

Table 5.9: Spring concentrations of water quality parameters at each wetland interstitial water sampling site

Parameter	DWAF Limit	Mean Concentration Per Site In Spring					
		S1	S2	S3	S4	S5	S6
NH_4^+	0.007	0.514	0.508	0.550	0.534	0.576	0.661
Al	0.01	1.087	0.998	1.018	1.040	1.221	1.025
V	0.1	1.159	1.403	5.159	4.403	6.659	6.103
Cr	0.007	0.374	0.722	2.421	2.274	3.202	3.074
Zn	0.002	3.617	7.389	5.650	8.888	6.428	9.749
Pb	0.0002	0.674	0.974	1.625	2.577	3.450	3.528
Mn	0.18	0.022	0.459	2.023	1.958	2.764	2.801
Cu	0.0003	1.567	0.648	0.471	0.481	0.563	0.672

Table 5.10: Autumn concentrations of water quality parameters at each wetland interstitial water sampling site

Parameter	DWAF Limit	Mean Concentration Per Site In Autumn					
		S1	S2	S3	S4	S5	S6
NH_4^+	0.007	2.654	0.025	0.078	0.098	0.167	0.346
Al	0.01	2.180	0.540	0.533	0.537	0.547	0.547
V	0.1	1.325	1.627	2.592	2.592	3.723	3.720
Cr	0.007	0.470	0.669	9.190	9.246	10.885	11.898
Zn	0.002	6.242	8.640	5.284	9.390	7.069	9.273
Pb	0.0002	1.490	1.665	3.679	2.482	3.736	3.211
Mn	0.18	0.837	1.002	3.136	2.424	2.176	11.946
Cu	0.0003	0.398	0.221	0.625	0.623	0.234	0.205

By examining Tables 5.7 to 5.10 above, it can be observed that most water quality parameters detected across all seasons at wetland interstitial water sampling sites are above the acceptable limits prescribed by DWAF (1996a; 199b). An exception is Mn for which measured concentrations were found to be acceptable at site S1 in both winter and spring, and at all sites (S1-S6) in summer.

The seasonal variations of parameter levels at selected river water sampling sites are shown in Tables 5.11 to 5.14 below.

Table 5.11: Summer concentrations of water quality parameters at each river water sampling site

Parameter	DWAF Limit	Mean Concentration Per Site In Summer					
		S1	S2	S3	S4	S5	S6
NH_4^+	0.007	ND	ND	ND	ND	ND	ND
Al	0.01	0.370	0.382	0.529	0.536	0.523	0.362
V	0.1	ND	ND	ND	ND	ND	ND
Cr	0.007	0.418	0.492	0.483	0.512	0.502	0.406
Zn	0.002	1.277	1.379	0.967	0.984	0.852	1.141
Pb	0.0002	1.091	1.273	1.211	1.345	1.519	1.275
Mn	0.18	0.008	0.041	0.000	0.000	0.012	0.043
Cu	0.0003	1.470	1.668	1.566	1.530	1.409	1.481

Table 5.12: Winter concentrations of water quality parameters at each river water sampling site

Parameter	DWAF Limit	Mean Concentration Per Site In Winter					
		S1	S2	S3	S4	S5	S6
NH_4^+	0.007	0.015	0.018	0.040	0.047	0.075	0.019
Al	0.01	1.620	1.590	1.620	1.600	1.620	1.610
V	0.1	1.947	1.604	3.947	3.104	5.047	4.504
Cr	0.007	0.639	1.903	2.639	3.403	3.739	4.803
Zn	0.002	6.342	13.310	8.342	14.810	9.442	16.210
Pb	0.0002	0.225	2.444	4.210	3.944	2.875	5.344
Mn	0.18	2.924	0.589	4.924	2.089	6.024	3.489
Cu	0.0003	1.500	1.613	1.603	1.614	7.594	1.624

Table 5.13: Spring concentrations of water quality parameters at each river water sampling site

Parameter	DWAF Limit	Mean Concentration Per Site In Spring					
		S1	S2	S3	S4	S5	S6
NH_4^+	0.007	0.508	0.509	0.520	0.524	0.538	0.510
Al	0.01	0.995	0.986	1.075	1.068	1.072	0.986
V	0.1	1.447	1.104	3.447	2.604	4.547	4.004
Cr	0.007	0.528	1.197	1.561	1.957	2.121	2.604
Zn	0.002	3.809	7.344	4.655	7.897	5.147	8.675
Pb	0.0002	0.433	1.858	2.710	2.645	2.197	3.310
Mn	0.18	1.466	0.315	2.462	1.045	3.018	1.766
Cu	0.0003	1.485	1.641	1.585	1.572	4.502	1.553

Table 5.14: Autumn concentrations of water quality parameters at each river water sampling site

Parameter	DWAF Limit	Mean Concentration Per Site In Autumn					
		S1	**S2**	**S3**	**S4**	**S5**	**S6**
NH_4^+	0.007	7.873	2.744	2.745	7.779	6.963	2.690
Al	0.01	1.937	1.863	1.879	1.864	1.872	1.860
V	0.1	1.602	1.478	2.105	1.848	2.799	2.757
Cr	0.007	0.636	1.605	8.712	9.190	8.683	9.567
Zn	0.002	7.168	6.879	5.383	14.486	5.598	7.523
Pb	0.0002	2.138	3.194	1.902	3.179	1.363	2.427
Mn	0.18	2.163	720.855	2.369	1.351	2.317	1.479
Cu	0.0003	1.644	1.786	1.842	1.713	7.936	1.547

The seasonal variations in water quality parameters displayed in Tables 5.11 to 5.14 indicate that most parameters measured at river water sampling sites are higher than the DWAF standard in each case. The only parameter which was found to be compliant with acceptable limits was Mn such that compliance was noted at all sites (S1-S6) in the summer months.

The occurrence of the assessed elements in concentrations that exceed those of the respective DWAF limits imply water pollution in both the floodplain wetlands and river system. This in turn poses a number of health risks to aquatic species and humans who depend on these wetland and river systems for their livelihoods. Specifically:

- Elevated levels of Al may cause egg-shell thinning and low birth-weights in water birds, as well as respiratory problems in fish (DWAF, 1996a).
- High concentrations of Cr are likely to inhibit the growth phases in fish and aquatic invertebrates (DWAF, 1996a), and are capable of causing lung and gastrointestinal cancer in humans (DWAF, 1996b).
- The impacts of exposure to high levels of Zn in these environments may include growth deficiencies, reduced levels of reproduction and death in aquatic organisms (DWAF, 1996a), as well as gastrointestinal discomforts in humans (DWAF, 199b).
- The high Pb concentrations are highly toxic to most aquatic organisms and humans (Abed, 2006), and may cause neurological disturbances in unborn and young children (DWAF, 1996b).
- Elevated Cu concentrations may cause brain damage in mammals (DWAF, 1996a) and a number of negative impacts on human organs (DWAF, 1996b).

- Water contamination by considerable quantities of V may have adverse neurological effects on animals and result in animal paralysis (Sukdeo, 2010).
- Exposure to large amounts of NH_4^+ ions may cause reduced hatching success, growth rates and development in aquatic organisms (DWAF, 1996a) and also have a number of negative impacts on human health (DWAF, 1996b).
- Contamination of the wetland and river systems by Mn is known to cause reduced hatching success and development in aquatic organisms (Lenntech, 2009).

In light of the above, it is crucial that the necessary steps are taken to improve the quality of water in these ecosystems and minimise contamination with the aim of protecting the well-being of dependent aquatic biota and humans.

Comparison of Tables 5.7 to 5.10 and Tables 5.11 to 5.14 indicates that parameters which have been identified as the main contaminants in the wetland and river systems of interest are largely alike. To be noted is that the number of compliance cases for all parameters across all sites is notably higher in wetland interstitial water samples than in river water samples. This aligns with the inferences made in section 5.3.5.2 above, regarding the possibility that contaminants detected in the river are likely to have been sourced from upstream activities; more especially those detected during the dry seasons of winter, spring and autumn. Furthermore, careful inspection of parameter concentrations in each wetland interstitial water sample and its corresponding river water sample highlights a decrease in concentration from the former to the latter in some cases, implying the possibility that these floodplain wetlands function as natural filtration systems.

5.3.6 Conclusion

The variations in concentration of water quality parameters in both wetland interstitial water and river water samples over the selected sampling sites show a strong link between detected levels of each parameter and substances utilised in, or produced by, proximal land uses and operations in the watershed area. These include subsistence agriculture and recreational, industrial and residential activities. The detection of higher levels of certain indicators over months of particular seasons was also closely related to the rainfall regime in Durban and therefore the concentration or dilution of elements within dry and wet seasons respectively.

The correlation performed indicated that a strong positive relationship exists between wetland interstitial water and river water for the majority of water quality parameters at most of the sample sites. This implies that the water quality of floodplain wetlands have a direct impact on the water quality of the adjacent river system and vice versa.

A matter of concern is that almost all detected water quality parameters were found to be higher than the acceptable DWAF limit, indicating water pollution in both the floodplain wetlands and river system. The number of compliance cases over all seasons was found to be greater in wetland interstitial water samples than in river water samples, indicating possible influence by upstream activities. However, there are also many cases in which detected parameter concentrations in the wetland environment were higher than those measured in the river system. The implication is that the wetlands delineating the Lower uMngeni River floodplain serve as purification systems, with respect to these particular parameters, and therefore release water of a relatively higher quality into the adjacent river channel.

The information sources referenced in this paper are listed at the end of this thesis.

CHAPTER 6: RECOMMENDATIONS

6.1 Introduction

This chapter identifies appropriate management and monitoring measures that should ideally form part of a rehabilitation project for the Lower uMngeni River wetlands. Both strategic-level and site-specific measures have been identified in line with the main factors and activities that impact on these wetlands. A management and monitoring programme has been developed for the recommended rehabilitation project. Implementation of this plan should be driven by the need to protect the ecological integrity and continued existence of these wetland ecosystems.

6.2 Wetland Management and Rehabilitation

In order to better manage and rehabilitate the wetlands along the Lower uMngeni River, it is recommended that a rehabilitation project involving the execution of suitable management and monitoring measures be initiated. Efforts should be focused on individual wetland units rather than the network as a whole. By adopting this approach, the overall value and functioning of the system will concomitantly improve. This goal can be achieved through implementation of the following site-specific measures:

- Deactivation of the agricultural, informal residential and industrial drains;
- Re-vegetation and the control of alien vegetation;
- Enforcing restrictions on human access to wetland systems;
- Correct management of runoff and storm water entering the wetland from surrounding areas; and
- Monitoring the health status of wetlands.

Details on each of the above-mentioned measures are provided in the sub-section that follows.

6.2.1 Site-Specific Measures

The focus of wetland management must be towards both volume and quality control. A number of site-specific management measures are proposed. These should ideally feed into the relevant components of the high level management measures proposed in section 6.2.2 below, and include the following:

6.2.1.1 Installation of Gabion Weirs

A series of gabion weirs should be constructed within each of the wetland units to promote back-flooding and the re-establishment of a more natural wetness regime (Mullins, 2012). Gabion weirs are particularly suitable as they are robust (Rickard *et al.*, 2003) enough to withstand the high flow volumes in these systems. Installation is relatively easy, affordable and has a low impact on local systems (Mullins, 2012).

6.2.1.2 Re-vegetation

Prior to re-vegetation of degraded and cleared wetland areas, it is important that all solid wastes be removed from the wetland areas and their immediate surrounds. Thereafter, a mixed community of indigenous hydrophytic species should be re-established within each wetland unit. Re-vegetation will increase the habitat and biodiversity value of the wetlands (Peters *et al.*, 2012). It will also assist in slowing down the movement of water through the wetlands, thereby acting as a sediment trap, resulting in improved water quality (Day, 2009). Re-establishment of the wetness regime will also promote the return of hydrophytic species and wetland communities that characterised these areas prior to human disturbance (Mullins, 2012).

6.2.1.3 Alien Invasive Plant Control Programme

The wetlands and portions of the riparian area were noted to host a number of alien species e.g. *Ricinus communis L*. Careful control of alien plants is necessary. If managed early and effectively these species should not become a problem and minimal on-going maintenance will be required. The removal and subsequent management and monitoring of these species is essential in maintaining the biodiversity value and ecological integrity of the wetland systems (Day, 2009; Macaskill, 2010). Hence an Invasive Plant Control Programme should be developed and implemented as soon as possible.

6.2.1.4 Human Disturbance Minimisation Measures

The site currently faces a variety of pressures from direct anthropogenic disturbances including vagrants, uncontrolled access by vehicles, illegal dumping and plant harvesting. In order to mitigate and manage these threats a suitable buffer around each wetland unit should be determined and “no-go” areas, constituting the wetlands and their respective areas, should be established. Access (both pedestrian and vehicle) to these “no-go” areas should be strictly prohibited. Furthermore, human activities, apart from those already taking place (such as subsistence agricultural practices) should not be allowed to encroach upon these “no-go” areas. In line with this, the feasibility of fencing the established “no-go” areas should be considered (CSIR, 2014b). Areas of illegal dumping and soil stockpiles within “no-go” areas must be rehabilitated and re-vegetated as per the specifications listed in sub-section 6.2.1.2 above.

6.2.1.5 Stormwater Management

Management of stormwater runoff from developments and land uses within the vicinity of the wetlands features is critical to maintaining the integrity of the rehabilitated system. A stormwater management program should be developed to ensure that runoff is reticulated to approved discharge points for controlled release (beyond the demarcated “no-go” areas) (Mullins, 2012). These discharge points should have suitable scour protection (gabion or reno mattresses) and be spread along the system to prevent point-source release of runoff (CSIR, 2000). Of equal importance is the regular monitoring (particularly after large rain events) of these discharge points to ensure that no scour has occurred. Surface and groundwater monitoring should be conducted by suitably qualified specialists on a biannual basis, specifically during the summer and winter months.

6.2.1.6 Overall Monitoring

Monitoring of the site will be required to assess the restoration of the system and thereby determine the progress and ultimate success of the rehabilitation project (Mullins, 2012). It is recommended that a monitoring programme for the rehabilitation project be developed, specifying the various phases of the rehabilitation project as well as the assessment criteria, monitoring methodology, relevant legislation, responsible agent, monitoring frequency, timeframe and monitoring output relevant to each phase. The assessment criteria should include all site-specific rehabilitation and management measures recommended in sub-sections 6.2.1.1 – 6.2.1.5 above.

A proposed management and monitoring programme for the recommended rehabilitation project is provided in Table 6.1 below. It is crucial that the responsible authorities in each case are identified by the Lower uMngeni Catchment Management Forum and specified in an update of Table 6.1, prior to approval and implementation of the programme. The aim of the afore-mentioned task is to ensure that the relevant authorities promote ground-level implementation of the specified management and monitoring measures.

Table 6.1….\

Table 6.1: A proposed management and monitoring programme for the wetland rehabilitation plan (adapted from Mullins, 2012)

Rehabilitation Phase	Assessment Criteria	Management Action & Monitoring Methodology	Relevant Legislation	Frequency	Timeframe	Output
Implementation & Remediation Phase	Alien plant removal	A combination of GIS/remote sensing and wetland reconnaissance can be utilised to identify alien species occurring within the wetland environment (e.g. *Ricinus communis L.*), their respective population numbers and the extent of their occurrence. Environmentally friendly methods of alien plant removal must be employed. Weekly on-site monitoring should be conducted by a suitably qualified Environmental Officer during this phase of the rehabilitation project.	National Environmental Management: Biodiversity Act (2004) National Environmental Management Act (1998)	Weekly	Duration of remediation activities	Monthly Monitoring Reports
	Re-vegetation	A combination of GIS/remote sensing and ground-truthing can be utilised to identify areas within the wetland environment where degradation, vegetation clearing, vegetation burning, illegal dumping and soil stockpiling are evident. Areas in which illegal dumping and soil stockpiling occurs should be cleaned out before being re-vegetated. Re-vegetation methods should be employed to establish a mixed community of indigenous hydrophytic species in these areas. Weekly on-site monitoring should be performed by a suitably qualified Environmental Officer during this phase of the rehabilitation project.	National Environmental Management: Biodiversity Act (2004) National Environmental Management Act (1998)	Weekly	Duration of remediation activities	Monthly Monitoring Reports
	Fence establishment	A combination of GIS/remote sensing and ground-truthing can be used to demarcate "no-go" areas. A fence, situated between 5-10 m away from the demarcated "no-go" areas, should be erected in an environmentally responsible manner It should also be ensured, however, that the fence type enables the movement and migration of smaller fauna to and from the river and wetland systems; thereby minimizing potential negative impacts on ecological corridors and related processes. Once established, weekly on-site monitoring of fence integrity should be conducted by a suitably qualified Environmental Officer during this phase of the rehabilitation project.	National Water Act (1998) National Environmental Management Act (1998)	Weekly	Duration of remediation activities	Monthly Monitoring Reports
	Installation of gabion weirs	A combination of GIS/remote sensing and ground-truthing can be used to determine suitable locations for the installation of gabion weirs within each wetland unit of interest. Installation of the weirs should be carried out in an environmentally responsible manner. The necessary authorisations and permits should be obtained prior to initiation of construction activities. It is recommended that a suitably qualified Environmental Officer conduct weekly on-site inspections during this phase of the rehabilitation project.	National Water Act (1998) National Environmental Management Act (1998)	Weekly	Duration of remediation activities	Monthly Monitoring Reports
	Reticulation of stormwater runoff to approved discharge points	A combination of GIS/remote sensing and ground-truthing can be used to determine appropriate locations for discharge points, beyond the established "no-go" areas. It must be ensured that these discharge points have suitable scour protection (gabion or reno mattresses) and be spread along the system to prevent point-source release of runoff. The necessary permits should be obtained to undertake this activity. It is recommended that a suitably qualified Environmental Officer conduct weekly on-site inspections during this phase of the rehabilitation project.	National Water Act (1998) National Environmental Management Act (1998)	Weekly	Duration of remediation activities	Monthly Monitoring Reports
Recovery Phase	Alien plant re-emergence monitoring and removal	A combination of GIS/remote sensing and ground-truthing can be utilised to identify and monitor areas where alien plants have re-emerged subsequent to removal in the implementation and remediation phase of the rehabilitation project. Alien plants within identified areas of re-emergence should be removed by employing environmentally friendly methods of removal. Monthly on-site inspections should be conducted a suitably qualified Environmental Officer during this phase of the rehabilitation plan.	National Environmental Management: Biodiversity Act (2004) National Environmental Management Act (1998)	Monthly	6 months	Monthly Monitoring Reports
	Re-vegetation success monitoring	GIS/remote sensing and ground-truthing can be used to determine the success of re-vegetation initiatives carried out in the implementation and remediation phase of the rehabilitation project. Additional areas within the wetland	National Environmental Management: Biodiversity	Monthly	6 months	Monthly Monitoring

Rehabilitation Phase	Assessment Criteria	Management Action & Monitoring Methodology	Relevant Legislation	Frequency	Timeframe	Output
		environment where degradation, vegetation clearing, vegetation burning, illegal dumping and soil stockpiling are evident should be identified, cleaned out where necessary, and thereafter re-vegetated. It is recommended that monthly on-site inspections be performed by a suitably qualified Environmental Officer during this phase of the rehabilitation project.	Act (2004) National Environmental Management Act (1998)			Reports
	Surface and groundwater sampling and monitoring at approved discharge points	Discharge points need to be monitored to ensure that no scour has occurred. Surface and groundwater sampling and monitoring should be conducted by suitably qualified specialists on a monthly basis during this phase of the rehabilitation project.	National Water Act (1998) National Environmental Management Act (1998)	Monthly	6 months	Monthly Monitoring Reports
	River water, wetland interstitial water and wetland sediment sampling and monitoring	The findings of this study can be used as baseline data for monitoring purposes. For continuity, all samples, namely wetland interstitial water, river water and wetland sediment samples, should be collected from the same sampling sites used in this study (Refer to Appendix B for GPS co-ordinates of each sampling site) and possibly additional sampling sites. The aim is to determine the level of wetland and river contamination. This will in turn be used to inform management measures. Sampling and monitoring should be conducted on a monthly basis by a suitably qualified specialist during this phase of the rehabilitation project.	National Water Act (1998) National Environmental Management Act (1998)	Monthly	6 months	Monthly Monitoring Reports
	Fence integrity monitoring and repair	The condition of the fence erected in the implementation and remediation phase of the rehabilitation project should be thoroughly checked to ensure that it hasn't been damaged or removed. Sections of the fence that have been damaged or destroyed should be repaired within no more than 14 days from the time of inspection. Furthermore, it should also be ensured that erection of the fence has not resulted in adverse impacts on fauna, which may include movement and migration restriction/death. Monthly on-site monitoring should be performed by a suitably qualified Environmental Officer during this phase of the rehabilitation project.	National Water Act (1998) National Environmental Management Act (1998) National Environmental Management: Biodiversity Act (2004)	Monthly	6 months	Monthly Monitoring Reports
Operational Phase	Alien plant re-emergence monitoring and removal	A combination of GIS/remote sensing and ground-truthing can be utilised to identify and monitor areas where alien plants have re-emerged subsequent to removal undertaken in the recovery phase of the rehabilitation plan. Alien plants within these identified areas of re-emergence should be removed by employing environmentally friendly methods of removal. It is recommended that annual on-site monitoring be undertaken by a suitably qualified Environmental Officer during this phase of the rehabilitation project.	National Environmental Management: Biodiversity Act (2004) National Environmental Management Act (1998)	Annually	3 years	Annual Monitoring Reports
	Re-vegetation success monitoring	GIS/remote sensing and ground-truthing can be used to determine the success of re-vegetation initiatives carried out in the implementation and remediation, and recovery phases of the rehabilitation plan. Additional areas within the wetland environment where degradation, vegetation clearing, vegetation burning, illegal dumping and soil stockpiles are evident should be identified, cleaned out where necessary, and thereafter re-vegetated. Annual on-site monitoring should be performed by a suitably qualified Environmental Officer during this phase of the rehabilitation project.	National Environmental Management: Biodiversity Act (2004) National Environmental Management Act (1998)	Annually	3 years	Annual Monitoring Reports
	Fence integrity monitoring and repair	The condition of the fence erected in the implementation and remediation phase of the rehabilitation plan should be thoroughly checked to ensure that it hasn't been damaged or removed. Sections of the fence that have been damaged or destroyed should be repaired within no more than 14 days from the time of inspection. It should also be ensured that	National Water Act (1998) National Environmental	Annually	3 years	Annual Monitoring Reports

Rehabilitation Phase	Assessment Criteria	Management Action & Monitoring Methodology	Relevant Legislation	Frequency	Timeframe	Output
		erection of the fence has not resulted in adverse impacts on fauna, such as movement and migration restriction/death. It is recommended that annual on-site monitoring be conducted by a suitably qualified Environmental Officer during this phase of the rehabilitation project.	Management Act (1998) National Environmental Management: Biodiversity Act (2004)			
	Surface and groundwater sampling and monitoring at approved discharge points	Discharge points need to be monitored to ensure that no scour has occurred. Surface and groundwater sampling and monitoring should be conducted by suitably qualified specialists on a quarterly (seasonal) basis during this phase of the rehabilitation project.	National Water Act (1998) National Environmental Management Act (1998)	Quarterly (Seasonally)	3 years	Annual Monitoring Reports
	River water, wetland interstitial water and wetland sediment sampling and monitoring	For continuity, wetland interstitial water, river water and wetland sediment samples should be collected from the same sampling sites used in this study (Refer to Appendix B for GPS co-ordinates of each sampling site) and possibly additional sampling sites. The aim is to determine the level of wetland and river contamination. This will in turn be used to inform management measures. Sampling and monitoring should be conducted by suitably qualified specialists on a quarterly (seasonal) basis during this phase of the rehabilitation project.	National Water Act (1998) National Environmental Management Act (1998)	Quarterly (Seasonally)	3 years	Annual Monitoring Reports

Note: Following the 3 year period allocated to the operational phase of the rehabilitation project, findings of the Annual Monitoring Reports can be used to inform decisions regarding whether the duration of this phase should be extended, and to determine suitable monitoring frequencies and outputs for the assessment criteria to be considered during the extended timeframe.

6.2.2 High-Level Measures

Broad measures that should be implemented to improve the condition of the wetland systems include:

- Ensuring legislative compliance and adherence to policies that promote wetland protection.
- Controlling and monitoring effluent releases from land use activities within the area;
- Developing and implementing a Wetland Management Plan for the Lower uMngeni sub-catchment by using the WET-Manage Tool established by the Water Research Commission (WRC); and
- Ensuring that the proposed management and monitoring programme (refer to Table 6.1 above) is incorporated into other existing and future management plans for the uMngeni catchment and Lower uMngeni sub-catchment (such as the existing MCMP and a proposed Wetland Management Plan for the Lower uMngeni sub-catchment).

6.3 Conclusion

In order to improve the health status of the floodplain wetlands of interest in this study, factors that act to deteriorate water and sediment quality, and thereby degrade these wetland systems, should be properly managed and monitored as part of the recommended rehabilitation project. This proposed project should be integrated into higher-level strategic plans and programmes being developed for the uMngeni system, with aim of ensuring longevity of the wetland systems through maintenance of optimum wetland conditions.

CHAPTER 7: CONCLUSIONS

7.1 Conclusions and Key Findings

The aim of this study was to assess the health of the wetlands delineating the Lower uMngeni River floodplain. This was achieved through carrying out physico-chemical assessments of wetland sediment, wetland interstitial water and river water samples. The key findings of this study include the following:

- Primary impacts to the wetland systems include notable surface and sub-surface agricultural, informal residential and industrial drainage; removal of indigenous vegetation; and alien plant encroachment.
- Concern is incited by the high concentrations of heavy metals, particularly Pb, within the uppermost laminae of wetland soil cores. It can be concluded that wetland sediment are anthropogenically enriched by all detected metals, except Ni and Cr.
- The generally moderate to high metal concentrations within the first three laminae of wetland soils are indicative of the recency of pollution events. This together with the predominance of fine-grained sediment and high organic matter content within wetlands are suggestive that the sediment component of the system is potentially a 'contaminant sink' rather than a 'contaminant source'.
- Although pollution loading is generally low in wetlands delineating the Lower uMngeni River, as supported by the low PLI values calculated, monitoring is required to ensure that these conditions are at least maintained if not improved.
- A strong positive relationship exists between wetland interstitial water and river water at most sample sites. This is such that parameter concentrations in wetland interstitial water and river water are likely to increase and decrease together. The health of the respective ecosystems is therefore intrinsically linked.
- Excessive contaminant levels in wetland sediment, wetland interstitial water and river water are typically related to dominant land use activities situated within the vicinity of these sample sites. Elemental distribution therefore varies across all sample sites. Generally, higher concentrations of certain parameters were detected in samples situated within a short distance from informal settlements, recreational areas and light and heavy industrial zones.

- Detection of higher levels of certain indicators within particular seasons is also closely related to the rainfall regime in Durban which in turn determines the direction of water flow and the concentration or dilution of elements within the riverine and wetland systems.
- A number of water quality indicators measured in wetland interstitial and river water samples were found to be non-compliant with the acceptable DWAF limits, as set out in the South African Water Quality Guideline Series (DWAF, 1996a; 1996b),. This implies that waters constituting the Lower uMngeni River and wetland features of interest are contaminated, which in turn compromises the health and well-being of dependent humans and biota.
- In line with the above findings, and considering that the Lower uMngeni sub-catchment is identified as an “endangered” ecosystem (DEA, 2011), it is imperative that the management and monitoring measures proposed as part of the recommended rehabilitation project, be considered.

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APPENDIX A: GIS PIXEL COUNT

IMAGE PIXEL COUNT	VALUE	PERCENTAGE (%)	DESCRIPTION
2958	1	0.56	Natural Fresh Water
10	2	0.00	Plantation
2649	4	0.50	Wetlands
809	5	0.15	Wetlands-mangrove
9374	9	1.79	Sugarcane - commercial
810	10	0.15	Sugarcane - emerging farmer
1435	11	0.27	Mines and quarries
279952	12	53.34	Urban
6495	13	1.24	Golf courses
6746	14	1.29	Rural dwellings
9286	18	1.77	Forest
33248	19	6.33	Dense bush (70-100 cc)
3181	10	0.61	Bushland (< 70 cc)
10066	22	1.92	Grassland / bush clumps mix
107157	23	20.42	Grassland
153	24	0.03	Bare sand
1590	26	0.30	Degraded bushland (all types)
2770	27	0.53	Degraded grassland
6280	34	1.20	KZN national roads
3173	35	0.60	KZN main & district roads
673	36	0.13	Dams
2437	37	0.46	Estuarine Water
30345	38	5.78	Marine Water
3263	39	0.62	Coastal Sand and Rock
TOTAL			
524860		**100.00**	

APPENDIX B: GPS CO-ORDINATES OF SAMPLE SITES

SAMPLE SITE	SITE LANDMARKS	GPS CO-ORDINATES
Wetland Interstitial Water Sample Sites		
W1	Four-Ways Manufacturers	30^{0} 57.194' E 29^{0} 47.821' S
W2	Recreational Field	30^{0} 57.513' E 29^{0} 47.854' S
W3	Papwa Sewgolum Golf Course	30^{0} 57.971' E 29^{0} 47.840' S
W4	N2 Bridge	30^{0} 58.798' E 29^{0} 48.213' S
W5	Springfield Industrial Area	31^{0} 0.325' E 29^{0} 48.642' S
W6	Springfield Industrial Area	31^{0} 0.436' E 29^{0} 48.684' S
River Water Sample Sites		
R1	Four-Ways Manufacturers	30^{0} 57.194' E 29^{0} 47.819' S
R2	Recreational Field	30^{0} 57.516' E 29^{0} 47.852' S
R3	Papwa Sewgolum Golf Course	30^{0} 57.974' E 29^{0} 47.837' S
R4	N2 Bridge	30^{0} 58.799' E 29^{0} 48.202' S
R5	Springfield Industrial Area	31^{0} 0.327' E 29^{0} 48.648' S
R6	Springfield Industrial Area	31^{0} 0.435' E 29^{0} 48.689' S
Wetland Soil Samples Sites		
S1	Four-Ways Manufacturers	30^{0} 57.196' E 29^{0} 47.822' S
S2	Recreational Field	30^{0} 57.516' E 29^{0} 47.857' S
S3A	Papwa Sewgolum Golf Course	30^{0} 57.967' E 29^{0} 47.841' S
S3B	Papwa Sewgolum Golf Course	30^{0} 57.966' E 29^{0} 47.844' S
S4	N2 Bridge	30^{0} 58.779' E 29^{0} 48.201' S
S5A	Springfield Industrial Area	31^{0} 0.328' E 29^{0} 48.644' S
S5B	Springfield Industrial Area	31^{0} 0.330' E 29^{0} 48.640' S
S6	Springfield Industrial Area	31^{0} 0.433' E 29^{0} 48.684' S