A SPATIAL ANALYSIS OF THE THREATS TO WATER QUALITY OF FRESHWATER ECOSYSTEM PRIORITY AREAS IN THE UPPER CROCODILE CATCHMENT

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DECLARATION

I declare that this dissertation is my own, unaided work. It is being submitted for the Degree of Master of Science at the University of the Witwatersrand, Johannesburg. It has not been submitted before for any degree or examination at any other University.

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ABSTRACT

Freshwater ecosystems are under increasing pressure to sustain their functions on both a global and local scale. In light of growing concern regarding the pollution of South African rivers, this study focused on identifying potential threats to water quality of freshwater ecosystem priority areas (FEPAs) within the Upper Crocodile catchment. Nine sample sites located within five river FEPAs were selected for investigation and included the Skeerpoort, Sterkstroom, Buffelsfontein, Magalies and Brandvlei rivers. *In situ* measurements were taken of temperature, electrical conductivity, dissolved oxygen and pH using a YSI multi-parameter instrument. Water samples were collected at each site for the chemical analysis of nitrate, chloride, sulphate and phosphate. Biological samples were collected at four of the nine sites using SASS version 5 methodologies for biomonitoring.

Through the integration of water quality and land use data, key land use predictors of water quality within rivers FEPAs were identified. Principle component analysis revealed anthropogenic pollution sources including urban, agricultural, industrial, domestic and mining land use activities within the study area. The application of multiple linear regression showed that percentage improved grassland statistically significantly predicted autumn nitrate, winter chloride, autumn phosphate and summer dissolved oxygen across all FEPAs. Percentage urban land use was found to significantly predict autumn electrical conductivity and autumn phosphate levels within sample catchments. Percentage mines was the key predictor of sulphate concentrations. Forest plantations, cultivated and degraded thicket land use classes were not significant predictors of water quality within the study area.

The research concluded that improved grassland, urban and mining land use activities were the key predictors of water quality and represent potential threats to water quality of FEPAs within the Upper Crocodile catchment. It is recommended that water resource practitioners integrate FEPA maps and guidelines into planning processes in order to ensure sound management of freshwater ecosystems. Furthermore, the inclusion of FEPA sites into national water quality monitoring programmes and water resource management efforts will allow for river FEPAs to remain in good ecological condition.

To my mum, Sharon, the strongest woman I know. Thank you for being a constant source of encouragement and guidance throughout the process of me completing this work. You have actively supported me in my determination to realize my potential and continue to inspire me every day as I continue on my path.

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Abbreviations and Acronyms

°C	degrees Celsius
AEV	Acute Effect Value
ANOVA	Analysis of Variance
ASPT	Average Score per Taxon
CEV	Chronic Effect Value
CR	Critically Endangered
CSIR	Council for Scientific and Industrial Research
DEAT	Department of Environmental Affairs and Tourism
DO	Dissolved Oxygen
DWAF	Department of Water Affairs and Forestry
DWS	Department of Water and Sanitation
EC	Electrical Conductivity
EI & S	Ecological Importance and Sensitivity
EN	Endangered
FEPA	Freshwater Ecosystem Priority Area
FSA	Fish Support Area
GDP	Gross Domestic Product
GIS	Geographic Information Systems
LT	Least Threatened
MLR	Multiple Linear Regression
NBA	National Biodiversity Assessment
OCP	Organochlorine Pesticide
PC1	Principle Component One
PC2	Principle Component Two
PCA	Principle Component Analysis
PGM	Platinum Group Metals

pH	Potential of Hydrogen
RHP	River Health Program
SANBI	South African National Biodiversity Institute
SASS	South African Scoring System
SASS 5	SASS version 5
SLIM	Spatial and Land Information Management
SMI	SASS Macroinvertebrate Index
SPSS	Statistical Package for the Social Sciences
TWQR	Target Water Quality Range
VU	Vulnerable
WMA	Water Management Area
WRC	Water Research Commission

Chapter One: Introduction

1.1. Water Quality Crisis

The growth and survival of all living organisms is dependent on freshwater, making it an essential constituent in sustaining all life. However, freshwater management has become a concern globally (Pandey, 2015). In reviewing the water quality status of freshwater it is not only the availability but also the quality of the resource which must be taken into account with respect to anthropogenic impacts. The quantity of freshwater available for human consumption is limited since saltwater constitutes the majority of the Earths' water. Despite freshwater being an essential natural resource, it comprises only 2.5 % of global water. Of the 2.5 % of freshwater on Earth, a large portion exists in the form of ice, leaving only between 1-1.5 % of freshwater readily available for human usage (Pandey, 2015). Rivers, lakes and the atmosphere constitute a mere 0.3 % of all freshwater (Nel, 2009). Although freshwater constitutes a small fraction of global water, it is home to at least 100 000 species which represent 6 % of all known species in the world (Dudgeon, et al., 2006).

Meeting human needs for freshwater services such as irrigation, domestic water, power generation and transport has come at the expense of inland water ecosystems i.e. lakes, rivers, wetlands and estuaries (Nel, 2009). This has led to a growing need to manage freshwater sustainably. The sustainable use of freshwater resources gains further importance in light of the growing population and associated deterioration of water quality of freshwater ecosystems. Urbanisation, population growth and pollution are but few of the many threats to freshwater ecosystem health. It has been a long standing belief that surface water resources have the ability to recycle themselves; however, the rapid increase in contamination has reached levels which cannot be recycled by natural processes (Pandey, 2015).

Poor management of freshwater resources not only has direct impacts on the functioning of freshwater ecosystems but also leads to negative impacts on the livelihood of people. The use of water globally has increased six times over the last 100 years due to growth in population and demand for food (Wada, et al., 2016). Access to water is a basic human right; however, the reality is that lack of access to safe water is the cause of millions of deaths annually around the world. Many people are subjected to diseases which result from a lack of access to safe drinking water. Sound management of freshwater resources is vital due to increasing pollution and consequent changes in the way water is used (Aylward, et al., 2015). Over the last century, the demand for water has increased dramatically, to the extent that the usage per person has doubled. In addition, the rapid growth in population has raised the demand for water from the industrial and agricultural sectors (Pandey, 2015). The hazardous waste and chemicals generated from industry pose major threats to the quality of freshwater. Similarly, the agricultural sector returns large volumes of water contaminated with fertilizers into the receiving freshwater environment which adds to the deterioration of water quality (Pandey, 2015). It is therefore accepted that the protection of freshwater ecosystems is a global priority.

This study will aim to provide an insight into water quality threats to freshwater ecosystems within the context of water quality challenges in South Africa. Global water quality concerns are constantly changing with time and can differ significantly from one region to another and from one season to the next (Adeleke & Bezuidenhout, 2011). South Africa is a semi-arid country and faces several challenges in managing the quality and quantity of its water resources. The climate in the west of the country is predominantly desert and semi-desert and sub-humid conditions exist along the eastern coastal areas of the country. The average rainfall is 495 millimetres per year, which falls below half the global average of approximately 1033 millimetres per annum (King, 2016). The high evaporation loss characteristic of the South African climate contributes significantly to erratic and relatively low rainfall, creating a different and rather complex context for the investigation of freshwater threats in the country (Mantel, et al., 2010).

The growing population in South Africa and increasing economic development has led to the deterioration of freshwater ecosystems (Dekissa, et al., 2003). Freshwater ecosystems face major threats due to impacts on the quantity and quality of water which they support. Water scarcity due to high evaporation rates and low rainfall has placed South Africa as the 30th driest country in the world (Department of Water Affairs, 2012). According to the National Water Resource Strategy 2, the quantity of water that flows through South Africa's rivers amounts to roughly 49 200 million cubic meters per year. The flow in South Africa's streams can be considered to be relatively low with the occurrence of sporadic high flows (Department of Water Affairs, 2012). The significantly low flows associated with rivers can be explained by both natural processes such as droughts and human activities including upstream freshwater withdrawals and the development of dams (Mantel, et al., 2010). The impact of very low river flows can have serious consequences for the functioning of river ecosystems. In the case of fish species, their ability to reproduce naturally is disrupted and can lead to decreasing fish yields. Benthic macroinvertebrate communities are also subject to adverse impacts due to reduced river flow and subsequent water quality changes (Dekissa, et al., 2003).

Adding further complexity to the situation in South Africa is the fact that most urban and industrial developments have been situated in remote locations some distance from large water courses. As a result, the demand for water far exceeds the availability of water in several river basins. It is for this reason that widespread and large scale water transfer schemes across catchments have been implemented in South Africa (Department of Water Affairs, 2012).

In line with global trends, the expanding population and economic growth in South Africa has placed freshwater resources under a great deal of stress (Oberholster & Ashton, 2008). Water is scarce and water quality is threatened by urbanization, industry, mining activities, agriculture, afforestation and power generation (Oberholster & Ashton, 2008). The availability of freshwater resources in the country is fast reaching its limit (Department of Water Affairs, 2012). Water Demand Management (WDM) and supply side management is an area that needs further development. The protection of water resources must allow for their sustainable usage. The potential for drought and the effects of climate change should also be taken into consideration by water resource managers in order to promote the sustainable use of limited freshwater resources (Department of Water Affairs, 2012).

1.2. Freshwater Ecosystems

Freshwater ecosystems encompass a wide range of both surface and groundwater systems which include ponds, lakes, streams, rivers and wetlands (US Fish and Wildlife Service, 2012). The ecological characteristics of rivers may be influenced by many factors including topography, elevation, vegetation, soils and geology, precipitation, chemical constituents as well as human modifications which may be direct or indirect in nature (US Fish and Wildlife Service, 2012). Rivers may be impacted by land use activities in the catchment area as well as near-stream habitat including wetlands situated in the flood plain and shoreline vegetation. These effects are often challenging to manage due to the position of rivers as 'receivers' of pollution (Dudgeon, et al., 2006). Some of the key threats to freshwater ecosystems identified in South Africa and the Upper Crocodile catchment include Acid Mine Drainage (AMD), excessive phosphates and nitrates from agriculture (pesticides), industry, informal settlements (faecal pollution), failing wastewater treatment works and urban runoff. River ecosystem function is also highly sensitive to changes in temperature and the potential effects of climate change (Ormerod, et al., 2010).

Mining in South Africa has been in operation for approximately 120 years and represents a key driver for both economic growth and service delivery (Frost & Sullivan, 2011). Approximately 10 000 km^2 of interconnected mines exist in Mpumalanga and approximately

300 km² of gold mines can be found in the Witwatersrand area (Frost & Sullivan, 2011). Furthermore, the Department of Minerals and Energy has recorded approximately 770 mines that have not received rehabilitation and have been abandoned across South Africa (Frost & Sullivan, 2011). The mining industry therefore also represents a threat to the country's freshwater ecosystems due to the associated AMD produced by mining practices (Novhe, et al., 2014). AMD can originate as seepage from underground mine shafts. Other sources of mine water include run-off from open pits, tailings dams and ore stockpiles (IMIESA, 2010). Since the 1970s the large-scale closure of mines has led to serious environmental problems in South Africa (IMC, 2010). When a mine is abandoned, drainage from underground shafts enters surface waters as the shafts fill with water. In the past, the inflow of water into underground mines was identified as a setback to mining operations. As a result, water was pumped out of mines in order to allow access to gold and other reserves. However, in cases where mines were closed, the pumping of water ceased. This has led to the flooding of mining operations and contamination of water resources (IMC, 2010).

Coal and gold mines are the main generators of AMD in South Africa, decanting highly acidic water into rivers and streams (McCarthy, 2011). AMD contains high concentrations of sulphides, metals and salts as a by-product of mining activities. Research conducted by the CSIR in 2010 indicated that AMD contributes 88 % of all waste produced in South Africa. Improper mine closure therefore poses a major threat to freshwater ecosystems and the receiving aquatic environment. In Zimbabwe, a dune pyrite mine led to acidification of groundwater at a pH of 2 (Ravengai, 2005). Surface water contamination from mining operations can lead to detrimental and irreversible destruction to freshwater ecosystems. Furthermore, acidic drainage conditions and the associated water pollution can persist for several decades and result in large-scale environmental degradation (Ravengai, 2005).

The agricultural sector also plays a vital role in the growth of South Africa's economy by ensuring reliable and high agricultural yield. However, improper use of external agricultural inputs such as pesticides may pose a threat to aquatic ecosystems and affect aquatic populations and communities (Ansara-Ross, 2008). The discharge of organic pollutants into freshwater ecosystems is an issue of much concern in the Upper Crocodile catchment (van der Laan, et al., 2012). Negative impacts affect humans as well as animals, birds and aquatic organisms. In a study conducted by Sibali et al. (2008) excess concentrations of Organochlorine Pesticides (OCPs) were identified in the Jukskei River catchment in the Northwest province of South Africa. OCP levels in the Jukskei River catchment were found to be higher than the water criteria values recommended by the United States Environmental Protection Agency (USEPA). Levels also exceeded those set out by the Department of

Water and Forestry (DWAF) for the protection of the aquatic environment (Sibali, et al., 2008). The implications of the above findings are significant and can result in deteriorating water quality and elevated nutrient concentrations in the receiving rivers and streams. Agricultural practices may further contribute to streamflow reduction. The abstraction of water for irrigation reduces flow in rivers by changing their flow volumes. This was demonstrated by a study of the Duiwe River, where intense agricultural and irrigation land use practices reduced river flows (Petersen, et al., 2017). As a result, proper management of agricultural practices including agricultural runoff, unregulated abstractions and the use of pesticides are essential to prevent water quality threats to freshwater ecosystems.

Soil erosion is defined by Le Roux et al. (2008) as "a process of detachment and transportation of soil materials by wind or water", where water is the dominant cause of soil erosion in South Africa. The natural occurrence of soil erosion does not pose a threat to freshwater ecosystems and rivers but it is rather the accelerated rate of soil erosion due to human activities that is of concern (Le Roux, 2014). Human impacts include the clearing of vegetation or overgrazing. Land degradation and associated soil erosion is one of the most serious environmental problems affecting the management of rivers in South Africa (Le Roux, et al., 2008). This is largely due to the transport of sediments to rivers from other sources resulting in pollution. Increases in the levels of sediment in rivers reduce both the quality and quantity of water and the amount of habitat in freshwater ecosystems.

Eutrophication is a natural process in the ageing of lakes; however, human activities have accelerated the process resulting in undesirable nutrient concentrations in both dams and rivers in South Africa (van Ginkel, 2011). Eutrophication refers to the process of nutrient enrichment and related excessive plant growth in water bodies. Some of the negative impacts of eutrophication can be attributed to increased population growth, catchment alterations i.e. water storage facilities such as dams, wastewater discharges from inadequately treated wastewater, excessive use of fertilizers in food production and nutrient rich return flows from intensive agricultural practices (van Ginkel, 2011). Much of the research undertaken on the subject of eutrophication has its focus on lakes and reservoirs, however, there is limited understanding of impacts on rivers. Nevertheless, it is accepted that rivers situated downstream of eutrophic lakes are likely to contain higher levels of nutrients and the associated cyanobacteria (Oberholster & Ashton, 2008).

Rivers of poor quality often cannot effectively transport waste material away from catchments and these pollutants end up in reservoirs downstream of many urban areas. In addition, pollutants such as fertilizers and faecal waste often enter rivers directly and may contribute to the eutrophication of rivers. Freshwater ecosystems are being exploited to a point that has outpaced management efforts. The challenge of restoring such systems is unlikely once affected by human activities. Ultimately, deterioration in quality of the limited freshwater available has a detrimental impact on freshwater habitats and the biodiversity they support. As a result, freshwater habitats are particularly vulnerable to human influences (Dudgeon, et al., 2006)

1.3. Aquatic Biodiversity

Freshwater habitats represent a valuable natural resource having economic, cultural, aesthetic, scientific and educational benefits. However, freshwater ecosystems may very likely be the most endangered of all ecosystems in the world (Dudgeon, et al., 2006). Globally, freshwater ecosystems are facing declines in biodiversity at rates which exceed those in terrestrial habitats (Vaughn, 2010). Furthermore, the biodiversity supported by freshwaters is particularly sensitive to human activities. This is largely due to the disproportionate species richness found in freshwater habitats. Surface waters support 40 % of fish diversity and a quarter of vertebrate diversity globally (Dudgeon, et al., 2006). Biodiversity is the key factor underpinning human development. This is particularly true for developing countries such as South Africa since rural communities are most vulnerable to the effects of biodiversity loss (Vaughn, 2010). Many rural communities are directly dependent on natural ecosystems to meet basic human needs.

Aquatic biodiversity is declining at a rate which may result in irreversible consequences for all life-sustaining ecosystems. As a result, freshwater conservation in South Africa has evolved considerably over the years; consistent with trends overseas. The National Biodiversity Assessment (NBA) of 2011 identified ecosystem categories as critically endangered (CR), endangered (EN), vulnerable (VU) and least threatened (LT). Generally, the ecosystem types CR, EN and VU may be referred to as threatened. Ecosystem threat status is calculated by assessing the proportion of each ecosystem type that is still in good condition against a number of criteria based on the percentage of river length in A, B or C ecological category (DEA-SANBI, 2012).

Furthermore, CR ecosystems are defined as “ecosystem types that have very little of their original integrity (measured as area, length or volume) left in natural or near-natural condition” (DEA-SANBI, 2012). In other words, rivers considered CR within the study area, have been significantly altered from their original condition resulting in loss of some ecosystem functioning as well as loss of certain species. Critically endangered rivers therefore require more immediate conservation intervention. The 2011 National Biodiversity

Assessment indicated that 26 % of the country's rivers are critically endangered, 18 % are endangered, 11 % are vulnerable and 46 % are least threatened. According to the assessment, threat status is an important measure of the extent to which ecosystems are still untouched. It can also be a good indicator of the degree to which ecosystem function is diminishing (DEA-SANBI, 2012).

Several human influences on water quality are linked to the observed decline in aquatic biodiversity in South Africa (SANBI, 2007). Key activities underlying biodiversity loss include overutilization of water and organisms, water pollution as well as habitat destruction and degradation (Vaughn, 2010). Failing water treatment facilities, the discharge of industrial wastewater and agricultural effluent are some of the serious causes of water pollution in South Africa (Nel, et al., 2012). Other significant contributors to changes in aquatic biodiversity include environmental factors such as acid deposition and changes in precipitation patterns (Dudgeon, et al., 2006). The impact of climate change appears to be gaining much attention in water quality matters. The potential impacts of climate change on precipitation rates and patterns and temperature changes are of particular concern to the health of freshwater ecosystems (Darwall, et al., 2009). A 10-20 % reduction in precipitation due to climate change is predicted for South Africa by the end of the 21st century (Darwall, et al., 2009).

Deforestation, agricultural practices and urbanization contribute to the increase in habitat destruction and modification (WWF, 2009). These impacts may be direct as in the case of removal of river sand or they may be indirect resulting from environmental changes in the river basin (Dudgeon, et al., 2006). For example, clearing of forests leads to higher sediment loads in runoff which enters a river. This in turn leads to modification of the river habitats such as erosion of the shoreline due to an increase in river sediment (Dudgeon, et al., 2006). As such, one of the most significant threats to the aquatic biodiversity of rivers is the alteration of naturally occurring flow regimes through the construction of dams and channels (Pacific Rivers Council, 2007).

On a global scale it is estimated that approximately two thirds of fresh water flowing to the sea is diverted by approximately 40 000 large scale dams (Nilsson & Berggren, 2007). Due to the water scarcity challenges facing South Africa, main rivers are heavily utilised. Large dams can store up to two-thirds of the country's annual runoff (Nilsson & Berggren, 2007). This has led to changes in the functioning of freshwater ecosystems and is likely to have serious cumulative impacts on freshwater habitats (Nel, et al., 2012). This is particularly evident in South Africa where flow regimes are highly variable and impacts on river biota can be severe (WWF, 2013).

The introduction of alien species is widespread globally and in South Africa (Nel, et al., 2011). This has negative impacts on freshwater ecosystems; particular when viewed in combination with other existing human impacts on freshwater resources (Dudgeon, et al., 2006). A good example of the effect of alien invasive species on freshwater ecosystems is that of Lake Victoria. The Lake was considered home to one of the most diverse fish assemblages in the world prior to the 1970s (IUCN, 2013). However, the deliberate introduction of the *Lates niloticus* (Nile perch) species into the Lake resulted in the extinction of over 200 species of native fishes. This led to negative changes to traditional fishing practices and ultimately polluted the lake through increased erosion. Insect outbreaks were a further consequence (Impson, et al., 2013). In South Africa, the Nile Tilapia as well as North American species *Micropterus dolomieu* (smallmouth bass), *Micropterus salmoides* (largemouth bass) and *Micropterus punctulatas* (spotted bass) poses increasing threats to our freshwater ecosystems (IUCN, 2013).

1.4. Freshwater Ecosystem Priority Areas (FEPAs)

The statistics highlighted from the National Biodiversity Assessment of 2011 are significant given the climatic conditions in the country (DEA-SANBI, 2012). South Africa is considered to be a semi-arid area with a mean annual rainfall of 495 mm per year which is far lower than the global mean annual rainfall of 1033 mm per year (King, 2016). As a result, many of the country's main rivers are extensively utilised. Traditionally, conservation science was primarily focused on terrestrial ecosystems with little attention given to freshwater ecosystems and species. However, since the early 2000s, freshwater ecosystems have become an important part of conservation planning (Roux & Nel, 2013). The National Biodiversity Assessment of 2011 informed and supported the findings of the National Freshwater Ecosystem Priority Areas (NFEPA) project. In 2011, the Water Research Commission (WRC), the Council for Scientific and Industrial Research (CSIR), the South African Biodiversity Institute (SANBI) and the Departments of Water Affairs and Environmental Affairs collaborated in a multi-party initiative to identify spatially exclusive priority areas for conserving rivers, estuaries and wetlands for the whole of South Africa. These are referred to as Freshwater Ecosystem Priority Areas (FEPAs).

This initiative marked the first comprehensive assessment of South Africa's freshwater ecosystems (Nel & Driver, 2011). Fish sanctuaries were identified as rivers that are essential for protecting freshwater fish that are indigenous to South Africa and face threats to their ecological functioning. Most of the fish sanctuaries coincide with FEPAs. While fish sanctuaries were identified in rivers of good ecological health, Fish Support Areas (FSAs) were identified in rivers of lower ecological health. Outcomes of the project include a series of maps contained in the “Atlas of Freshwater Priority Areas of South Africa”. A map was generated for each of the nineteen Water Management Areas (WMAs) in the country. An implementation manual and technical report were also generated by the project.

An important component highlighted by the NFEPA project is that of water quality monitoring. According to the NFEPA project, water quality monitoring should be based on the location of FEPAs (Nel & Driver, 2011). The River Health Program (RHP) first established in 1994 by the Department of Water Affairs and Forestry (DWAF) was fundamental in addressing the need to monitor the ecosystem “health” of the country’s rivers. This program largely informed the development of the NFEPA project. As identified by the NFEPA project, most RHP monitoring sites were not located within FEPAs. As a result there is a requirement for monitoring to be expanded to include sampling of FEPA sites. The NFEPA manual further recognizes that the RHP program failed to maintain on-going State-of-Rivers reporting. This can be accounted for by a lack in budget and capacity. The focus of the RHP was the measuring of cumulative water quality effects for the entire catchment; however, additional monitoring of sites indicative of FEPA integrity needs to be put in place. It is suggested that monitoring FEPA integrity calls for at least one monitoring point within or upstream of a FEPA.

This study will examine the potential threats to FEPAs within the study area in order to further explore relationships between water quality and FEPAs in a highly contested context. The priority areas have been identified as valuable ecological resources that must be protected to ensure the sustainability of freshwater ecosystems and the biodiversity they support in the Upper Crocodile catchment. Through close analysis of land use and water chemistry data, the study will show the potential impacts of identified human inputs on selected river FEPAs.

1.5. Aim of Study

The aim of the study is to conduct a spatial analysis of the potential threats to water quality of Freshwater Ecosystem Priority Areas (FEPAs) in the Upper Crocodile catchment, which forms part of the Limpopo River Basin in South Africa.

1.6. Objectives

- To carry out periodic water quality sampling at selected points at selected river FEPAs within the catchment area.

- To carry out biological water quality sampling and analysis, using the SASS 5 Rapid Bio-assessment Method at a minimum of one river FEPA within the catchment area.
- To identify any potential sources of pollution across river FEPAs within the catchment through spatial and water quality analysis.
- To determine the potential threats to water quality of river FEPAs within the study area.

1.7. Research Questions

- What is the status of water quality within FEPAs in the Upper Crocodile catchment?
- What are the threats to ecosystem health of FEPAs in the Upper Crocodile catchment?
- What are the current and potential water quality impacts on FEPAs in the Upper Crocodile catchment?

The purpose of the study is to determine potential threats to water quality of river FEPAs in the study area through the collection of water chemistry and macroinvertebrate data from selected points along priority rivers and the integration of these results with land use data generated for the catchment. As such, potential sources of contamination may be determined to provide a picture of the current and potential water quality threats to the FEPAs identified for investigation.

1.8. Dissertation Outline

This chapter has provided a brief overview of the purpose of the research undertaken. The importance of protecting and maintaining ecological, biological and physical components of freshwater ecosystems cannot be emphasised enough. Given rapid declines in aquatic biodiversity exacerbated by human activities, this study will draw attention to three FEPAs within the Upper Crocodile catchment in order to shed light on the current state of water quality since their demarcation by the NFEPA project.

Chapter two will provide a survey of literature consulted in formulating the study. The criteria used in selecting FEPAs as well as the outcomes of the NFEPA project will also be discussed further. In addition, water quality parameters selected for the study will be described. National monitoring programmes have formed the basis of Integrated Water Resources Management (IWRM) in South African rivers and as such will be discussed briefly. As biomonitoring and macroinvertebrate sampling forms a small yet important component of the data collection method employed by the study, a description of the SASS (South African Scoring System) 5 Method for Rapid Biomonitoring of macroinvertebrates is also addressed.

Chapter three of the dissertation will highlight in detail the methods carried out in analysing existing water quality data as well as data collection and analysis procedures. A description of the study area and catchment will provide a context for the study. ***Chapter four*** will present the results of the field component of the study in graphical form as well as a detailed analysis of water chemistry, biological and catchment data. Statistical analyses and the findings of multiple linear regression analyses will also be described in this chapter. ***Chapter five,*** the final chapter of the research report, will provide a discussion and conclusion to the study by summarising key findings and observations and providing answers to the research questions upon which the study was based.

Chapter Two: Literature Review

2.1. Introduction

This chapter will provide a review of the literature consulted within the field of water quality and more specifically threats to water quality of freshwater ecosystems over recent years. The River Health Programme (RHP) is highlighted given the significant role it has played in addressing freshwater quality in South Africa. The importance of maintaining the ecological health of freshwater ecosystems is a key principle that is explored with respect to negative impacts on ecosystem functioning. The criteria used to develop river FEPAs are discussed in further detail as well as the outcomes of the NFEPA project and FEPA maps. Macroinvertebrate sampling is a well-accepted method for the testing of water quality and formed part of the field work component of this study. As such, the SASS 5 Method for Rapid Biomonitoring is explored briefly within this chapter in order to shed some light on the benefits of the use of macroinvertebrates as good determinants of water pollution. A description of the water quality key constituents relevant for the purposes of this study is provided, including their role and importance within freshwater ecosystems.

2.2. Water Quality in South Africa

Water quality is determined by several factors, namely, chemical, physical and biological characteristics of water (Foulds, 2014). Chemical parameters include pH, dissolved oxygen and heavy metals. Physical parameters are associated with temperature, dissolved solids, electrical conductivity, turbidity and others. Biological parameters refer to organisms such as macroinvertebrates, fish, molluscs and algae which may be present in a water sample and are good indicators of ecological integrity (Ollis, 2005). Water quality is often determined in relation to the intended use for the water as different water uses require different levels of water quality constituents (Foulds, 2014). Common uses of water include drinking, irrigation and electricity generation. The health of a water resource is affected by natural processes as well as human activities. The natural quality of water may vary from one place to the next depending on the climatic conditions, soils, rocks and topography of a region. Water pollution results when a water resource is contaminated by foreign materials such as microorganisms, industrial waste and sewage effluent (Mukhopadhyay & Mukherjee, 2013). Water pollution may also occur when anthropogenic activities result in the release of natural heavy metals into a water resource at a higher rate than the norm for that specific water body.

Water pollution can be caused by point and non-point pollution sources. Point sources of pollution are generally easier to identify and control and include industrial effluent and wastewater effluent discharges (van Vuuren, 2013). Non-point or “diffuse” pollution sources are not as easily managed and include runoff from agriculture and urban runoff among others. These sources of contamination usually enter a water resource from poorly defined areas. Van Vuuren (2013) reported on the need to improve knowledge of agricultural non-point pollution sources in South Africa. The report indicated an increase in non-point pollution in rivers in South Africa from agriculture. Agricultural runoff has several impacts on freshwater resources including salinization, eutrophication, sedimentation and increases in pathogens and heavy metals (van Vuuren, 2013).

It is therefore clear that human activity from mining, industry, agricultural practices and others affect the natural quality of a water resource (Foulds, 2014). A careful balance must be ensured between protecting and utilising water resources to achieve the principles of sustainability. Water pollution, however, is an ever-increasing cause for concern as existing water resources face serious threats to water quality. Given the scarcity of freshwater in South Africa, it has fast become a precious commodity. Its management has an immediate impact on our welfare, quality of life and economy (Strydom, 2008). Complexities in achieving sound management of freshwater resources include water availability, competencies and capacities of the institutions that manage water, as well as existing political conditions which often determine water planning, development and management processes (Adeleke & Bezuidenhout, 2011).

South Africa is currently experiencing a water crisis in the form of drought and pollution which afflicts the country. Apart from being semi-arid with an average low rainfall, water quality in the country is further threatened by high evaporation rates throughout catchments (Ringwood, 2015). It is reported that less than 9 % of South Africa’s rainfall is essentially converted to runoff. As a result, only 40mm of runoff is generated annually from rainfall which is considerably lower than the global average of 266mm (Ringwood, 2015). Much focus has been placed on the issue of water quantity over the years; however, water quality has only come to the forefront of planning in the past five or six years. The scarcity of water in the country only exacerbates water quality challenges and is one of the main concerns in water quality management within the South African context.

2.3. Threats to Water Quality of Freshwater Ecosystems

Anthropogenic activities have a profound impact on the proportion of various water quality constituents in freshwater ecosystems (Matowanyika, 2010). The provision of water for domestic, agricultural and economic use often comes at the expense of freshwater ecosystem functions. Some of the key anthropogenic aspects influencing water quality include agricultural activities and the use of fertilizers and pesticides, improper irrigation practices, industrial, domestic and mine wastewater, recreational activities and deforestation (Khatri & Tyagi, 2015). Key freshwater quality challenges for South Africa are described below.

2.3.1. Acid Mine Drainage

Many countries around the world are impacted by the environmental pollution associated with Acid Mine Drainage (AMD). These include countries with a history of mining activities as well as those with current mining operations (Johnson & Hallberg, 2005). AMD can be defined as water that is highly acidic; characterised by excess concentrations of heavy metals, sulphides and salt generated by mining activities (IMIESA, 2010). Water that is subjected to acidification has immediate detrimental effects on aquatic ecosystems (IMIESA, 2010). Furthermore; conditions of AMD and consequent metal pollution is known to prevail for extensive periods of time causing severe environmental degradation (Ravengai, 2005). The introduction of AMD into a river may hamper or in some cases destroy the functioning of aquatic organisms by reducing their ability to photosynthesise. AMD in freshwaters also reduces nutrient cycling which in severe cases can result in the death of organisms and consequent elimination of species (Ravengai, 2005).

Several studies have been conducted in South Africa surrounding the impacts of AMD on aquatic ecosystems. In 2010 the CSIR provided an insight into some of these cases. For example, in 2002, the Randfontein and Wonderfontein Spruit were reported as problematic and pollution of surface water on the West Rand was identified due to the decant of acid mine water from flooded mines in the Krugersdorp area (IMIESA, 2010). At this time, it was predicted that the problem would persist and lead to serious impacts on aquatic ecosystems.

In a further study conducted by McCarthy (2011), experiences of the impact of AMD on water quality were highlighted for both coal and gold mining in South Africa (McCarthy, 2011). Diffuse pollution from tailings dumps was identified as having a significant impact on the Blesbokspruit River located in Springs and the Klip River which drains the southern part of the Witwatersrand escarpment. Furthermore, the high levels of salinity found at the Vaal River are indicative of pollution originating from the Central and Western Basins (McCarthy, 2011). In the western basin, levels of sulphate were found to be extremely high measuring approximately 3500 mg/L and pH was between 2 and 3. The high concentration of heavy metals in the water gave rise to a bright orange trail on riverbanks caused by the formation of iron oxide which is typical of severe AMD cases. The study indicated similar findings in the case of coal mining in the Middleburg and Witbank area, with reported high salinity and sulphate concentrations in the tributaries of the Olifants River (McCarthy, 2011).

In the North West Province, mining activities have had a detrimental effect on aquatic ecosystems in the Tweelopiespruit catchment (van Eeden, et al., 2009). The Tweelopiespruit falls within the study area under investigation and flows into the Crocodile River before entering the Hartbeespoort Dam. Once a thriving farming area, the Tweelopiespruit catchment has since lost all farming land (Taylor, 2009). Having highlighted a few significant studies conducted on the topic of AMD impacts in South Africa, it is clear that the situation requires serious attention since implications on freshwater ecosystems are likely to worsen with time.

2.3.2. Invasive Alien Species

Invasive alien plant species can be defined as “alien species whose introduction does or is likely to cause economic or environmental harm or harm to human health” (NISC, 2006). The invasion of freshwater bodies by alien invasive plant species can have a profound impact on water quality and freshwater communities native to a site. Limited research has been conducted regarding the water quality state of existing FEPAs. There is an apparent need to address important conservation goals highlighted by the NFEPA project in order to ensure effective implementation. Research conducted by Chamier et al (2012) addressed some of the impacts of invasive alien species in South Africa. The study identified a few key implications of alien invasive plant species on water quality. These include reductions in streamflow and groundwater storage, eutrophication and nutrient cycling, fire incidents and soil erosion (Chamier, et al., 2012). All of these consequences contribute to a decline in water quality and pose a threat to aquatic ecosystems. It is estimated that invasive alien species (invasive riparian trees) consume approximately 7 % of South Africa’s annual runoff (Nel, et al., 2011).

A major cause of streamflow reduction is the higher evapotranspiration rates found in alien invasive plants when compared to indigenous plant species. The consequent reduction in streamflow has a direct impact on the dilution capacity of rivers. As a result, the concentration of pollutants and nutrients in a river is higher (Chamier, et al., 2012). Chamier et al. (2012) further explain the effects of an increase in biomass from alien plants due to their natural ability to grow at a faster rate than indigenous plants. The result is an escalation in eutrophication processes which can lead to toxic conditions for aquatic organisms. Furthermore, the increased biomass associated with alien species is described as having the ability to increase the intensity of fires (Chamier, et al., 2012). Research has further identified a link between aquatic weeds and increases in water-borne illnesses (Ray & Hill, 2013).

Several rivers in the Cape have undergone habitat destruction and pollution as a result of invasive alien fish. In some cases native fish species have been completely eliminated in several streams (van Vuuren, 2012). As such, rehabilitation of river ecosystems should be undertaken in order to prevent loss of habitat and native aquatic species. The Rondegat River is a tributary of the Olifants River. It is located in the Cape Floristic Region and has been classified as a river FEPA. Impson et al. (2013) highlighted a study conducted on the Rondegat River FEPA which focused on eradicating alien invasive plants and fish species along the river. The project highlighted the importance of ensuring priority areas are maintained in good condition. Alien fish species were eradicated with much success through the use of a piscicide (fish killing chemical) called Rotenone. The project was successful in restoring ecosystem function and improved biodiversity in the river (Impson, et al., 2013). This case is a great example of how good planning and coordination of management efforts can achieve success in the rehabilitation of river ecosystems under threat.

A further study conducted by Du Preez and Smit (2013) investigated non-indigenous freshwater crayfish species identified as *C. quadricarinatus* at the outlet of Lake Nyamiti in the Ndumu Game Reserve. Due to the fact that these crayfish are carriers of several pathogens, their potential impact includes destruction of aquatic organisms and disturbance of ecosystem functioning. This study emphasises the need to control freshwater crayfish imports for aquaculture (DuPreez & Smit, 2013). In the North West Province the growth of *Eichhornia crassipes* (water hyacinth) at the Hartbeespoort Dam has been ongoing for the past 30 years (Venter, 2012). The growth of invasive alien water hyacinths has placed restrictions on recreational activities at the Dam and has a relatively high surface coverage. While the majority of the invasive water plant can be found in the dam itself, hyacinths also enter downstream rivers. This causes further environmental impacts on aquatic ecosystems.

Water hyacinths are significant because they reduce dissolved oxygen and inhibit the exchange of oxygen between surface waters and the atmosphere. Furthermore, water hyacinth generates masses of organic matter. The decomposition of organic matter leads to an increase in biological oxygen demand in water bodies and consequent deterioration in water quality. In extreme cases, oxygen depletion can lead to losses in aquatic biodiversity (Osumo, 2001). A further negative impact of water hyacinth is the blockage of waterways and canals used for irrigation which poses a threat to public health. By the year 2012, the Hartebeespoort Dam Remediation Programme (Metsi a mé) had removed 44 280 m^3 of hyacinths from the Hartbeespoort Dam (Venter, 2012).

2.3.3. Agriculture

South Africa relies heavily on agriculture in order to ensure food security and support economic growth (Ansara-Ross, et al., 2008). Inefficient agricultural practices and the frequent use of a variety of available pesticides, however, are placing undue pressure on aquatic ecosystems. Van der Laan et al. (2012) conducted a scoping study that revealed high levels of nutrients as a result of irrigation practices in areas where sugar cane is produced in northern South Africa (van der Laan, et al., 2012). Water samples taken at strategic points in the Crocodile, Komati and Pongola river catchments were found to have increased salinity. High electrical conductivity and increased concentrations of both nitrate and phosphate were indicated. Runoff from fertilizers containing salts, nitrates and phosphates can contribute to increased nutrient concentrations and higher electrical conductivity in freshwater bodies. This result is likely to contribute to eutrophication of the catchment surface water (van der Laan, et al., 2012).

A further study by Sibali et al. (2008) identified significant levels of Organochloride Pesticide (OCP) compounds in the Jukskei catchment and Jukskei River. OCPs applied in agriculture were found to exceed both USEPA and the Department of Water and Sanitation (DWS) standards for the protection of aquatic environments (Sibali, et al., 2008). These results pose a serious threat to the health of aquatic biota in the Jukskei River catchment.

2.3.4. Eutrophication

A direct result of poor agricultural practices, failing wastewater treatment plants and other anthropogenic factors is the eutrophication of freshwater bodies (van Vuuren, 2008). Eutrophication is the process by which water bodies become enriched with nutrients which leads to uncontrolled plant growth (van Ginkel, 2011). One of the key consequences of eutrophication is the formation of dense algal blooms that are noxious and foul-smelling (Chislock, et al., 2013). The growth of phytoplankton affects the clarity and quality of water. Changes in the natural nitrogen and phosphorus cycles caused by human activities are a serious challenge of eutrophication, disrupting the functioning of aquatic ecosystems around the world (Oberholster & Ashton, 2008). Eutrophication can exist on levels ranging from oligotrophic (no water quality problems and low nutrient concentrations) to hypertrophic (continuous water quality problems and excessive nutrient concentrations caused by physical factors) (van Vuuren, 2012).

In South Africa the eutrophication of freshwater bodies is a serious problem that has both health and financial impacts for the country (van Ginkel, 2011). A key effect of eutrophication is the growth of cyanobacterial blooms and consequent production of cyanotoxins. Cyanobacteria (or blue-green algae) are bacteria that receive energy through photosynthesis (Frost & Sullivan, 2010). In South Africa cyanobacterial blooms occur in both dams and rivers with *Microcystis spp.* being the dominant bloom-forming cyanobacterial species (Harding, et al., 2009). The maximum total microcystin concentrations in South Africa are also reported to be higher than those of both Finland and North America (van Ginkel, 2011).

The Hartbeespoort Dam in the Upper Crocodile catchment has experienced prolonged periods of eutrophication. The dam is considered to be hypertrophic and is well known for excessive cyanobacterial blooms as a result of anthropogenic influences (Harding, et al., 2009). Having gained considerable attention for the poor water quality at the dam, the Hartbeespoort Dam Remediation Plan was developed in order to address the severe eutrophication impacts on the system (Harding, et al., 2009). A further effect of the presence of excess nutrients in freshwaters is the excessive growth of macrophytes. This has many negative consequences such as the blockage of waterways. Macrophyte biomass also leads to flooding and the damage of canals (van Ginkel, 2011). Impacts on aquatic ecosystems can be severe including habitat alteration and the disturbance of indigenous fauna and flora (Frost & Sullivan, 2010). In South Africa, *Eichhorina crassipes* (water hyacinth) is one of the most common invasive macrophytes found in eutrophic freshwater bodies.

A consequence of increased nutrient levels in freshwaters is the decomposition of plant species. However, the health of many aquatic animals such as fish, are also under serious threat from the impacts of excess nutrients (de Villiers & Thiart, 2007). In 2007, research carried out by de Villiers et al. (2007) provided a detailed description of the impact of excess nutrients on some of the largest rivers in South Africa. The research identified key point and non-point sources of nitrogen and phosphorus in aquatic ecosystems. Wastewater effluent from industrial, agricultural and domestic facilities were highlighted as the main point sources of nutrients while runoff from agricultural activities, urban environments, ineffective septic tanks, abandoned mines, burned forests and grasslands and un-sewered rural communities were some of the key diffuse sources of nutrients identified (de Villiers & Thiart, 2007).

A significant outcome of the research was that of the twenty largest river catchments in South Africa, only one river was found to comply with water quality guidelines for plant life. It is concluded that the nutrient concentration in South African rivers is following an upward trend and is likely to continue to impact the water quality of freshwater systems and river biodiversity (de Villiers & Thiart, 2007). Elevated nutrient levels result in the degradation of aquatic plant species by disturbing the natural competitive balance of these communities (de Villiers & Thiart, 2007). As a result, many aquatic animal species are unable to perform their functions as they rely on plant species for food, shelter, as well as breeding habitats (Chislock, et al., 2013).

In addition, the State of the Nation report of 2008, presented a perspective on the eutrophication of South Africa's' rivers located downstream of eutrophic dams in the country. Nutrient enrichment of several lakes and dams is an indication of high nutrient levels in the river system (Oberholster & Ashton, 2008). A case in point is that of the Berg River in the Western Cape. Nitrogen and phosphorus levels were found to increase by a factor of more than ten downstream of anthropogenic sources according to a study by de Villiers (2007). Fluctuations in nutrient levels due to both point and non-point sources far exceeded the maximum change as set out in the South African Water Quality Guidelines for Aquatic Ecosystems (de Villiers, 2007). Eutrophication has far reaching consequences for freshwater ecosystems. One of the environmental impacts of eutrophication is a decrease in the ecological integrity of freshwater ecosystems (Frost & Sullivan, 2010). The growth of algal blooms results in a decrease in the dissolved oxygen content of a water body resulting in the inability of most species to survive (Frost & Sullivan, 2010).

2.3.5. Climate Change and Water Temperature

Many of the threats to freshwater ecosystems that have been discussed can be viewed within the context of climate change as well as anthropogenic influences on water temperature. Climate change and global warming are now considered globally when it comes to the management of aquatic ecosystems (Pacific Rivers Council, 2007). While there are limited studies on the effects of water temperature changes on aquatic organisms in South Africa, one must consider these impacts in order to fully address the objectives of the study. Modifications of a rivers thermal characteristic can result from several anthropogenic factors. These may be direct in the form of thermal discharges or indirect as a result of changes in land use (Dallas, 2008). Indirect influences include irrigation return flows, flow modifications, alteration of riparian vegetation, inter-basin transfers and global warming.

In South Africa, changes in water temperature are mainly associated with agricultural practices, flow modifications and afforestation (Dallas, 2008). Although climate change is of significance in Southern Africa, its potential severity is unknown. However, the impacts of global warming and climate change may have important consequences for aquatic ecosystem functioning (SANBI, 2007). Increases in temperatures beyond that which is optimal for the functioning of aquatic organisms can disrupt their metabolism, growth and reproduction by limiting their supply of dissolved oxygen. This also increases their susceptibility to toxins (Dallas, 2008). Given the existing scarcity of water in South Africa, increasing water temperature and climate change may worsen the situation and threaten the health of aquatic ecosystems. Furthermore, changes in rainfall may adversely affect river flow. The inability of freshwater ecosystems to adapt to these changes may result in the deterioration of ecosystem health.

A study carried out by Rivers-Moore et al. (2008) highlighted water temperatures as one of the key abiotic drivers of species patterns within river systems. All rivers in South Africa can be classified as temperate (Rivers-Moore, et al., 2008). South Africa's rivers are highly variable and are characterized by extreme flow fluctuations. They are therefore associated with extreme temperature fluctuations, with a significant difference in thermal regime from one year to the next (Rivers-Moore, et al., 2008). Changes in land use will impact thermal patterns as a result of changes in runoff. Ultimately this will alter the biotas which have evolved under a different flow regime (Rivers-Moore, et al., 2008). The study emphasized the link between anthropogenic land use practices and river health. It was shown that low flow periods often resulted in increased water temperatures and lower levels of dissolved oxygen.

2.3.6. Flow Alteration

The flow in a river system can be altered by the removal of water or addition of water to a freshwater ecosystem (Nel, et al., 2007). The addition of water to a river is usually in the form of return flows or water transfer schemes. Discharges from wastewater treatment facilities may contribute to the alteration of river flow in some cases (SANBI, 2007). Due to the scarcity of water in South Africa, economic activities often face restrictions due to the limited availability of freshwater (SANBI, 2007). As a result, most of the large rivers in South Africa are over utilised and several hundred dams have been constructed. It is estimated that up to two thirds of the country's annual runoff is stored in large dams (Nel, et al., 2007).

These impacts result in a change in flow volume as well as the seasonality of flow into rivers. Recent studies documented by Nel (2011) have also indicated the negative cumulative impact of smaller farm dams on the quantity and quality of freshwater ecosystems in South Africa (Nel, et al., 2011). Inter-basin transfer schemes dramatically alter natural river flow regimes where water is transferred between areas to those with a higher water demand (SANBI, 2007). Species transfer may result whereby new species are introduced to a catchment through water transfer schemes. This may have severe negative implications on native species.

2.3.7. Destruction and Degradation of Natural Habitat

Loss of habitat or habitat destruction may lead to a decrease or extinction of certain species and is considered the single biggest threat to freshwater ecosystems (Pacific Rivers Council, 2007). South Africa is known as one of the most biodiverse countries in the world, with its high level of species diversity. Although it comprises only 2 % of global land surface area, it supports 10 % of all plant species and 7 % of all reptiles, birds and mammals (SANBI, 2007). However, human activities have put considerable pressure on freshwater ecosystems leading to habitat degradation and in some cases the complete destruction of natural habitat. Overexploitation, overgrazing and poor agricultural and land use practices have modified areas of natural habitat considerably (WWF, 2009). This often results in the inability of freshwater habitats to support the level of biodiversity they normally would in an intact habitat (SANBI, 2007). Habitat loss can result from human activities such as deforestation, the development of houses and the canalisation of rivers.

2.4. Land Use Predictors of Water Quality

Land use is one of the most important contributors to surface water quality within a catchment. A key global concern in environmental management is the impact of unsustainable human activities on river water quality. In order to identify threats to water quality of rivers, an understanding of the relationship between land use and water quality is important (Ding, et al., 2015). Several studies have explored relationships between land uses and water quality parameters, including some of the water quality variables selected for this particular study. The impact of land use on river nitrate and phosphate concentrations is well documented, with agricultural and urban land use being the primary anthropogenic predictors of nutrient concentration in rivers.

In a study conducted by Wang et al. (2014), urban areas were identified as the most significant predictors of water pollution in the Daliao River basin in China, while forested areas were found to have the opposite result. A further case study of the Dongjiang River in China corroborated the findings of Wang et al. (2014), having identified a strong positive relationship between urban land use and nitrate levels in river water (Ding, et al., 2015). The case study also indicated a positive correlation between forested land and dissolved oxygen and a negative correlation between forested land and temperature, electrical conductivity and nitrate. Agricultural land use, however, did not significantly influence water quality.

Van Deventer (2012) highlighted urban land use as a threat to water quality of the uMngeni catchment feeding the Midmar Dam. Tromboni et al. (2017) and Bowden et al. (2015) reported urban land use as the key driver of nutrient concentration (nitrate and phosphate) increases at streams in a largely urbanized tropical region of Brazil and the Marlatt watershed in Kansas respectively. It is therefore clear from previous studies that urban land use has been identified as a significant predictor of river water quality both internationally and in South Africa.

Agricultural (including irrigated) land and the cultivation of crops is another anthropogenic land use that has been linked closely with changes in river water quality in recent studies. In western Canada, salt concentrations in the Saskatchewan River basin were attributed to the impacts of salt affected soils or cropland (Kerr, 2017). A similar result was reported in the Upper Mngeni catchment where deterioration in water quality and ecological condition was observed in areas downstream of commercial agricultural practices. Ecological condition of the catchment was in the seriously/critically modified category (van Deventer, 2012). Similarly, agriculture contributed significantly to nitrate concentrations in the only remaining free-flowing river in the western Sierra Nevada, California (Ahearn, et al., 2005).

Castillo (2009) identified agriculture as a primary predictor of nitrate and phosphate in the Guare River catchment of northern Venezuela, where higher concentrations of nitrate were associated with larger proportions of agricultural lands. Kroll (2009) investigated the influence of land use on the water quality of rivers within the Castilla-La Mancha (Spain). The results showed that irrigated agriculture was significantly correlated with nitrate and phosphate. Grasslands, however, were identified as an important nitrate source in a river draining the Sierra Nevada in California. The study revealed a strong positive correlation between nitrate concentration and percent grassland, highlighting percent grassland as the main contributor to nitrate loading within the catchment (Kroll, et al., 2009). This is likely a result of the use of fertilizers commonly used to improve grassland for livestock or recreational purposes.

Anthropogenic land uses have also been associated with chloride, sulphate, electrical conductivity and dissolved oxygen levels within rivers. Li et al. (2013) found that urban and agricultural land uses led to significant increases in electrical conductivity, chloride and sulphate concentrations in the Han River (China) over a six year period. The result was corroborated by a principle component analysis (PCA) that identified anthropogenic inputs. Solute concentrations were found to be significantly higher where industrial activities occurred (Siyue, et al., 2013). Hua (2017) found similar results where changes in land use were investigated in relation to the water quality of the Malacca River. The study revealed that built-up or urban areas contributed significantly to electrical conductivity levels in the river while negative impacts on dissolved oxygen concentration were linked to agricultural and forested areas.

Furthermore, electrical conductivity was connected to agricultural land use activities and non-point pollution from agricultural practices through surface runoff. Other sources of electrical conductivity and changes in dissolved oxygen concentration included residential, industrial and sewage effluent. Similarly, Bowden et al. (2015) observed higher mean values for DO and EC for agricultural land use within the Marlatt watershed in Kansas. Agricultural and urban land uses are some of the main contributors to pH levels in streams. In the Marlatt watershed in Kansas, urban land use was found to lead in affecting pH levels when compared with other land use types (Bowden, et al., 2015). In the Ruo River in southern Malawi, streams downstream of agricultural and developed areas displayed elevated pH values, highlighting agricultural and urban land uses as key drivers of pH levels in the river (Kambwiri, et al., 2014).

2.5. The River Health Programme

In 1994, the Department of Water Affairs and Forestry developed the River Health Programme (RHP) in an effort to address a lack of information regarding the ecological state of rivers in South Africa (River Health Programme, 2005). This initiative represented a shift in management towards an approach which incorporated the principles of Integrated Water Resource Management (IWRM) (River Health Programme, 2005). The key objectives of the RHP were to measure, assess and report on the ecological condition of aquatic ecosystems; identify and report on trends (both spatial and temporal); identify and report on potential problems and grow awareness within communities regarding the health of the country's rivers (Strydom, 2008).

The RHP highlighted the importance of aquatic ecosystems through building capacity and awareness. It also identified some of the key pressures facing freshwater ecosystems in the country. Some of these pressures were found to occur only in certain areas while others were found to occur throughout South Africa (River Health Programme, 2005). These include urbanisation, overexploitation, modification of rivers and streams and the increasing demand on wastewater treatment works and water supplies (Strydom, 2008). Through collaboration with the CSIR, DWAF, DEAT and the WRC, the RHP developed a system of scientific reporting on the ecological state of rivers in the country known as the State of Rivers (SoR) reporting. Information was collected for ten water management areas across the country and the latest available reports were generated in 2006.

The programme used an assessment condition called the Eco Status to determine the overall health or integrity of a river. The Eco Status included a variety of river and riparian characteristics and reflected a river's ability to support biodiversity and provide goods and services (Strydom, 2008). Ultimately the Eco Status represented an integrative measure that monitors the way in which a river habitat responds to modifications to the ecosystem. The Eco Status as described in the State of Rivers Report (2004) was comprised of six indicators as listed below:

- Instream Habitat Integrity
- Riparian Zone Habitat Integrity
- Riparian Vegetation Integrity
- Fish Assemblage Integrity
- Macro-invertebrate Integrity
- Water Quality (as indicated by diatoms)

(River Health Programme, 2005)

River health categories were used to communicate the ability of a river to support a range of ecosystems and range from "Natural" to "Poor", where a "Natural" rating indicated no or minimal modification to the stream and a rating of "Poor" indicated considerable river alteration (River Health Programme, 2005). The RHP played a vital role in providing a means to develop a database of river water quality status for South African rivers. Through continued monitoring and reporting the RHP could identify trends in river health data allowing for further interpretation that can influence decision-making when it comes to water resource management. In 2016, the RHP was replaced by the River Eco-status Monitoring Programme (REMP). The new programme evolved from the RHP and is focused on monitoring the ecological condition of rivers, incorporating system drivers and biological responses reflected by river ecosystems. The REMP provides a refinement of the indicators used by the RHP as they have evolved over recent years (Cawood & Vos, 2016).

2.6. Freshwater Ecosystem Priority Areas (FEPAs)

The National Freshwater Ecosystem Priority Areas (NFEPA) project represents the first in-depth assessment of South Africa's freshwater ecosystems; identifying the most important areas within these ecosystems that must be protected in order to promote their sustainability (van Vuuren, 2012). South Africa is home to a wide range of aquatic ecosystems despite its semi-arid landscape (Foulds, 2014). The multi-party NFEPA project team aimed to meet the biodiversity goals for South Africa's freshwater ecosystems by classifying FEPAs for the entire country. A second objective of the NFEPA project was to create a basis to allow for effective implementation of measures to ensure the continued protection of FEPAs including free-flowing rivers (Nel & Driver, 2011).

2.6.1. The Importance of Freshwater Ecosystems and Sustainability

In their natural state, freshwater ecosystems are abundant in aquatic life and are richer in species than the more expansive terrestrial and marine ecosystems (Nel & Driver, 2011). Furthermore, rivers play a critical role as ecological corridors from freshwater source areas all the way to the sea.

Freshwater biodiversity and several ecosystem functions are dependent on river corridors because they rely on connectivity (Nel, et al., 2011). However, when connectivity within a catchment is disrupted, ecosystems can be severely altered and biodiversity is negatively impacted (Nel & Driver, 2011). Freshwater ecosystems are therefore of great importance in contributing to biodiversity as well as the economy. It is therefore crucial that the protection and utilisation of freshwater resources and rivers is balanced to allow for sustainable freshwater use. The NFEPA project was based on the principles of sustainable development since ecological, economic and social systems are interlinked.

2.6.2. Systematic Biodiversity Planning

The development of FEPAs was based on the principles of systematic biodiversity planning. Over the past three decades this framework has gained much credibility and is now well accepted as a strategic and scientific conservation approach for the identification and prioritization of freshwater ecosystems (Roux & Nel, 2013). Systematic conservation planning has evolved to become an accepted new applied branch of conservation biology. This approach to conservation is practiced internationally, with South Africa at the forefront of systematic biodiversity planning. It aims to reduce biodiversity loss to a minimum and maintain ecosystem services which support all ecosystems (Roux & Nel, 2013). According to Nel et al (2013), systematic biodiversity planning can be achieved by following three steps which incorporate ***representation***, ***persistence*** and setting ***quantitative biodiversity-based targets*** into spatial planning.

Representation

Representation in systematic biodiversity planning, as defined by Roux (2013), is to conserve a suitable sample of the biodiversity in an area. This is achieved by conserving a sample that is representative of all species including their habitats (Nel & Driver, 2011). As such, it supports a shift from focusing on areas known well by scientists to the inclusion and analysis of all the biodiversity of a particular area or river. In the past, areas of low economic potential were often the focus of conservation priorities, while highly productive areas were excluded (Roux & Nel, 2013). Incorporating representation into spatial planning requires the identification of biodiversity features as well as the setting of targets. Targets for biodiversity may include setting a representative percentage of a particular ecosystem type or a representative number of species of an ecosystem. Representation in systematic biodiversity planning therefore aims to retain the resilience of an ecosystem by conserving ecological and social diversity (Nel, et al., 2004).

Persistence

The NFEPA technical report defined the principle of persistence in systematic biodiversity planning as "the need to conserve the ecological and evolutionary processes that allow biodiversity to persist over time" (Nel & Driver, 2011). In earlier conservation planning, the focus was on the efficient representation of biodiversity in prioritised protected areas. However, in the 1990s it was realised that it is also important to include the natural processes that are responsible for maintaining biodiversity in conservation plans since biodiversity will not persist without these biological processes (Nel & Driver, 2011). In other words, the principle of persistence is concerned with conserving the adequate amount of an ecosystem type or ecosystem population required to ensure the sustainability of the ecosystem. These concepts should be included in targets set for ecosystem type and species (Roux & Nel, 2013). Persistence therefore falls within the requirements of sustainability as it ensures the sustainable utilisation of freshwater ecosystems by maintaining their ecological integrity (Nel, et al., 2004).

Quantitative Biodiversity-Based Targets

A core principle of systematic biodiversity planning is identifying quantitative biodiversity targets. This refers to the quantity or amount of a particular biodiversity target that should be prioritised for conservation in order to maintain ecosystem functioning (Nel & Driver, 2011). This process is important in informing future monitoring programmes since the quantitative targets can be easily measured in order to monitor conservation progress. Quantitative biodiversity targets also give resource managers a clear indication of what to plan for.

The principles of systematic conservation planning were used as a guideline in the development of FEPAs. Information gathered from previous research on conservation planning in South Africa provided a basis upon which FEPA maps were built. Furthermore, new scientific findings in systematic biodiversity planning were included in the methodology used by the NFEPA project. These included the need to manage a water resource together with the surrounding land area through the delineation of sub-catchments (Nel & Driver, 2011); the need to develop a river tree-network for the purposes of analysing upstream-downstream linkages and to incorporate longitudinal connectivity into conservation plans and the application of multiple-use zonation. Multiple-use zonation ensures that conservation recommendations and the level of protection assigned to a freshwater ecosystem are dependent on the role of the sub-catchment in achieving biodiversity targets (Nel & Driver, 2011). Threats to water quality of FEPAs are often associated with land use impacts. As such, incorporating the land area and land uses surrounding river FEPA catchments under investigation in this study, allowed for threats to aquatic ecosystems through water pollution to be identified.

2.6.3. Criteria used to Identify FEPAs

The principle underpinning the criteria used to develop FEPAs is that by conserving representative freshwater ecosystems, the species they support as well as their habitats would also be conserved to allow for sustainability (Nel, et al., 2011). As outlined in the FEPA atlas (2011), FEPAs were identified based on various criteria which incorporated the principles of systematic conservation planning. These included the representation of river, wetland and estuarine ecosystem types. These ecosystem types were used as biodiversity surrogates since they are associated with similar physical characteristics. As such, river ecosystem types identified as FEPAs are likely to exhibit similar biological responses under natural conditions. The representation of threatened freshwater species was a further criterion used; however, the NFEPA project included only fish species due to a lack of information available on other freshwater species. In addition, preference was given to ecosystem types that occurred within intact river systems (Nel, et al., 2011).

After setting criteria for the selection of priority areas, the NFEPA project carried out extensive data collection. Existing information was sourced from the River Health Programme, eco- status data for the present ecological status of rivers and the reserve determination data files of the Department of Water Affairs and Forestry (van Vuuren, 2012). The research team then developed a conservation planning tool that could achieve the criteria in the most spatially efficient manner (Nel & Driver, 2011).

2.6.4. Key Findings of the NFEPA Project

The NFEPA project found that only 35 % of the length of South Africa's main stem rivers is in good ecological condition while 57 % of tributaries are in good ecological condition (A or B ecological category). The project therefore concluded that the condition of tributaries is better than that of main rivers in South Africa. In light of this finding, it becomes important to ensure that tributaries are managed in a manner that retains their ecological and conservation value. This will allow for the conservation of biodiversity and the sustainability of more heavily utilised rivers (Nel, et al., 2012). The NFEPA project found that 57 % of river ecosystems are threatened. A large percentage of all river ecosystems are critically endangered, endangered or vulnerable, according to the National Biodiversity Assessment of 2011. It was found that the river ecosystems under threat corresponded directly with areas of intense land use (Nel, et al., 2012).

Given that FEPAs have been identified for under a quarter of the country's river length, it is important that the selected priority areas are maintained in good ecological condition (Nel & Driver, 2011). However, only 22 % of South Africa's river length is represented by FEPAs. Through sound planning and management of river ecosystems prioritised as FEPAs, freshwater ecosystems can be conserved. This can be achieved by ensuring that the priority areas are used minimally and in a sustainable manner. A further alarming finding of the project was that there are only 62 free-flowing rivers in South Africa. The World Wildlife Fund (WWF) defines a free-flowing river as "any river that flows undisturbed from its source to its mouth, at either the coast, an inland sea or at the confluence with a larger river, without encountering any dams, weirs or barrages and without being hemmed in by dykes or levees" (WWF, 2009). The NFEPA project identified 62 free-flowing or undammed rivers in the country which converts to a mere 4 % of South Africa's river length. Of the 62 identified free-flowing rivers, only 25 rivers were found to extend further than 100 km (Nel, et al., 2012).

Free-flowing rivers sustain important ecological processes; however, they are rarer than their terrestrial equivalents, namely wilderness areas (WWF, 2009). Flow modifications such as dams and channel modification pose risks to the functioning of rivers. A key impact of dam construction is the impact it has on the food supply of aquatic species i.e. fishes native to a river. Rivers play an important role in pollution control by transporting and removing pollutants and excess nutrients (WWF, 2009). These are some of the reasons why existing free-flowing rivers need to maintain their status. Priority areas selected by the NFEPA project protect more than 50 species of fish that are on the verge of extinction (Nel, et al., 2012). This is possible by conserving a portion of rivers in good ecological condition. In fact, a final key finding of the project was the that all fish close to extinction can in fact be protected by conserving just 15 % of South Africa's river length.

2.6.5. NFEPA Outcomes and Map Categories

The NFEPA project has produced useful outputs to implement the freshwater conservation goals of the project. The key outcome of the NFEPA project was the development of the ***Atlas of Freshwater Ecosystem Priority Areas of South Africa.*** The atlas represents a comprehensive assessment of freshwater ecosystems in South Africa and is the first of its kind (van Vuuren, 2012). The FEPA maps show the location of the identified FEPAs which must be protected in order to conserve freshwater ecosystems in South Africa. The atlas indicates that river FEPAs are currently in an A or B ecological category and it is recommended they remain in this condition. FEPA maps were created for each of the nineteen Water Management Areas (WMAs) in the country (Nel, et al., 2012). In identifying FEPAs, the planning units used were sub-quaternary catchments. In all, 9417 sub-quaternary catchments were delineated and fall roughly within the 1945 quaternary catchments in South Africa (Nel & Driver, 2011).

FEPA maps represent different categories which have been delineated into sub-catchments and have different management requirements. In some cases more than one category may overlap. A sub-quaternary catchment code is also provided on the maps and may be used to obtain further information on river FEPAs (Nel & Driver, 2011). This study is primarily concerned with the threats to water quality of river FEPAs within the Upper Crocodile catchment and refers only to the map category “River FEPAs and associated sub-quaternary catchments” as provided by the NFEPA technical report of 2011.

River FEPA and associated sub-quaternary catchment:
Rivers in good condition (A or B ecological category) at the time of selection were identified as river FEPAs (Nel & Driver, 2011). River FEPAs meet the criteria to achieve biodiversity goals for river ecosystems and also the fish species they support. According to the NFEPA project, this means that river FEPAs must be maintained in good condition in order for biodiversity and freshwater conservation targets to be met. River FEPAs and associated sub-quaternary catchments are indicated in a dark green colour on the map products implying the need to manage the entire sub-quaternary catchment responsibly in order to protect the river it supports. Although rivers prioritised for conservation are in a good condition ecologically, in some cases the need for rehabilitation may be necessary (Nel & Driver, 2011).

2.7. SASS Aquatic Biomonitoring

2.7.1. Benthic Macroinvertebrates and Bio-assessment of rivers

Rivers are considered one of the most threatened ecosystems worldwide and as a result the need to develop methods to assess their condition as well as to monitor rates of change has received much attention in recent years (Dickens & Graham, 2002). Several methods have been developed to assess the ecological integrity of freshwater ecosystems (Ollis, 2005). In recent years, numerous techniques for the bio-assessment of rivers have been developed with the aim to determine the health of rivers. Biomonitoring can be defined as “the systematic use of living organisms or their responses to determine the condition or changes of the environment” (Li, et al., 2010). Aquatic biomonitoring is also defined by Roux (1997) as “the gathering of biological information in both the laboratory and the field for the purpose of making some sort of assessment, decision or in determining whether quality objectives are being met regarding an aquatic environment”.

It is widely accepted that bio-assessment produces valuable data that can contribute to the management of freshwater ecosystems (Dickens & Graham, 2002). Bio-assessment now forms an important part of environmental management policies in many countries including South Africa. While there are several organisms that may be used as indicators of river health, benthic macroinvertebrates are the most commonly used (Li, et al., 2010). Aquatic communities provide a holistic view of the ecological condition of a river because they are present continuously. This allows for monitoring of biological responses over extended periods of time (Li, et al., 2010).

2.7.2. Advantages of using Benthic Macroinvertebrates as Bio-indicators

The biological condition of a river is determined by several environmental factors. Geomorphological, hydrological as well as chemical and physical characteristics of rivers all contribute to the overall health of a river ecosystem (Roux, 1997). The riparian vegetation of a river is also important when determining ecological integrity. Benthic macroinvertebrates are able to reflect all of these features of river ecosystems and are therefore good indicators of water quality (Ollis, 2005).

There are a number of authors that have documented the reasons for the use of benthic macroinvertebrates in biological monitoring of aquatic ecosystems. The advantages of this approach are numerous. Macroinvertebrates are highly diverse, constituting 54 % of all known species of organisms (Burden, et al., 2002). Including macroinvertebrates in river monitoring programs ensures a good representation of the biodiversity present at a site. Furthermore, the abundance of macroinvertebrates in rivers means they can be found in most habitats and the ubiquitous nature of macroinvertebrates allows for effective monitoring since they are always present in water ecosystems (Burden, et al., 2002).

Species sensitivity to pollution varies across macroinvertebrate communities. Coupled with their relatively quick response time to pollution, they can provide valuable data with regards to cumulative water quality effects (Ollis, 2005). In addition, macroinvertebrates are sessile in nature and therefore cannot move to less polluted waters if contaminants occur within the area they inhabit (Burden, et al., 2002). As such, they represent the site being sampled well. Because the life cycle of macroinvertebrates ranges from months to a few years, they are limited in their ability to recolonize (Burden, et al., 2002). A further advantage of macroinvertebrates as bio-indicators is that the method used is relatively inexpensive since the collection of macroinvertebrates does not rely on elaborate equipment (Ollis, 2005). Finally, the nature of macroinvertebrates allows for them to represent the history of a site, allowing for the detection of intermittent water quality changes (Li, et al., 2010). Macroinvertebrate sampling and analysis was therefore included in this study as a means to identify water quality threats at river FEPAs given the ability of macroinvertebrates to act as bio-indicators of water pollution.

2.7.3. The South African Scoring System (SASS) Rapid Bio-assessment Method for Rivers

Several methods and techniques of varying complexity have been applied for the rapid bio-assessment of overall river health. In South Africa, Chutter (1972) developed the Biotic Index nearly three decades ago. Although widely used at the time, the technique proved to be too labour-intensive (Dickens & Graham, 2002). This led to the development of SASS, a faster and more efficient index based on the Biomonitoring Working Party (BMWP) index developed in the UK (Dickens & Graham, 2002). The method evolved over several years to culminate in the SASS rapid bio-assessment method for rivers in 1998 (Dickens & Graham, 2002). Having been widely accepted in South Africa by practitioners, the SASS is now the standard for the rapid bio-assessment of rivers in South Africa. The SASS is an index based on the use of macroinvertebrates as biotic indicators of river health (Ollis, 2005). It is a relatively inexpensive technique which is used to identify declines in river water quality and also monitor changes and trends in water quality over time. The sensitivity of macroinvertebrates to pollution and habitat changes varies among different species, making them good indicators of pollution (Wolmarans, et al., 2014). In turn, biological monitoring can identify the impacts of all threats to water quality present.

The SASS method involves the identification of macroinvertebrates mostly to the family level from a list of taxa which have been pre-defined. A 'quality score' is assigned to each taxon. This score is based on the tolerances of species to water pollution and environmental disturbances. Quality scores range from 1-15 where 1 indicates a species that is highly tolerant to pollution and 15 indicates a species that is highly sensitive to pollution (Dickens & Graham, 2002). Samples are collected using a net and various kicking and sweeping techniques over specified time periods for each biotope represented. Macroinvertebrates are collected separately from three biotopes, namely, Stones (S), Vegetation (V) and Gravel, Sand and Mud (GSM) according to SASS protocol. The SASS method requires that families are identified from each biotope in the field and abundances are roughly estimated (Dickens & Graham, 2002).

The method also includes rating of the diversity of a biotope where a score of 1 would indicate low species diversity and 5 high species diversity. Three indices are generated from the SASS assessment namely the SASS score, the number of taxa (No. Taxa) and the Average Score per Taxon (ASPT). The SASS score is calculated by summing the quality scores for each taxon identified. The number of taxa is the total number of taxa identified and the ASPT is calculated by dividing the SASS score by the number of taxa. In cases where a site has low diversity, a low SASS score is reflected, however, a site may have few organisms of the appropriate sensitivity. In a study conducted by Dickens and Graham (2002), field trials were conducted in order to test the variability of the SASS method. The study found that the index that was most consistent across all biotopes was the ASPT. It was also found to be a more repeatable measure than other SASS indices when assessing the water quality of rivers (Dickens & Graham, 2002). As a result, ASPT is considered a more reliable measure of river health than the SASS score (Dickens & Graham, 2002).

2.8. Water Quality Parameters

2.8.1. Physical Parameters

2.8.1.1. Temperature

Water temperature is a physical property of water defined as “a measurement of the average thermal energy of a substance” (Department of Water Affairs and Forestry, 1996). The transfer of energy between substances creates a flow of heat. Heat transfer can change the temperature of water through the transfer of heat from the air, sunlight, a nearby water source or thermal pollution. The thermal characteristics of rivers are dependent on various characteristics of a region or catchment. These include hydrological, regional, climatological and structural characteristics of a catchment area (Dallas, 2008).

Water temperature is influenced by several natural factors including storm water runoff, groundwater inflows, air temperature, turbidity, and exposure to sunlight (Behar, 1997). As a result, water temperature must be taken into account when considering factors such as dissolved oxygen, electrical conductivity and pH of a water resource. An increase in temperature is associated with a decrease in the dissolved oxygen content and electrical conductivity of a water resource (Department of Water Affairs and Forestry, 1996). Furthermore, periods of low flow experienced in South Africa can cause water temperatures to increase and dissolved oxygen levels to decrease. (Rivers-Moore, et al., 2008).

It is important to measure water temperature when dealing with water quality due to its ability to influence other chemical and physical characteristics of water (Dallas, 2008). Within the context of this study, water temperature was selected as one of the water quality constituents measured due to the above characteristics of water temperature and more specifically, its influence on aquatic ecosystem functioning. An increase in temperature modifies several physical and chemical characteristics of water. These include chemical reaction rates and toxicity, oxygen solubility and microbial activity (Mwangi, 2014). Water temperature influences the metabolic rates and biological functioning of aquatic communities and largely determines their habitat. The South African Water Quality Guidelines, Volume 7 (1996) describe a direct relationship between temperature and the metabolic rate of aquatic organisms.

Dallas (2008) indicated that for most fish a 10 degree Celsius increase in water temperature will double physiological functioning. While some aquatic species can tolerate these changes, others may not. Temperatures exceeding 35 degrees Celsius are detrimental to aquatic life since enzymes begin to break down at these temperatures (Dallas, 2008). In addition to its potential harmful effects on aquatic ecosystem functioning, fluctuations in temperature can also increase the toxicity of water by allowing the solubility of certain water constituents (Department of Water Affairs and Forestry, 1996). These include heavy metals as well as compounds such as ammonia.

2.8.1.2. Electrical Conductivity (EC)

Electrical conductivity is a measure of the ability of water to pass an electrical current. Electrical conductivity is also directly proportional to the Total Dissolved Solids (TDS) in water (Department of Water Affairs and Forestry, 1996). The electrical conductivity of a river is dependent on the geology of the surrounding area (Edokpayi, et al., 2015). Generally rivers flowing predominantly over granite (that contains no ionised metals) display a lower electrical conductivity than those that flow over rocks with ionisable metals (Edokpayi, et al., 2015). Higher conductivity will result from the presence of various ions. Although electrical conductivity is not a measure of specific ions in water, an increase in its levels may be an indicator that polluting discharges have entered a water resource or river. The ions present in water that are able to carry an electric current include sulphate, magnesium, calcium, carbonate, bicarbonate, sodium, potassium and chloride (Behar, 1997). Conductivity is usually measured in micro or millisiemens per centimetre (μS/cm or mS/cm) or in millisiemens per meter (mS/m).

Agricultural practices can contribute to higher levels of electrical conductivity in rivers primarily through the export of salts from the source through to river systems. In a study conducted by van der Laan et al (2012), the Crocodile, Komati-Lomati and Pongola river catchments indicated an increase in electrical conductivity as a result of high salt containing agricultural return flows. Matowanyika (2010) identified high electrical conductivity levels in the Jukskei River, indicating high concentrations of ions. EC levels were higher in the dry season than the wet season within the Jukskei River (Matowanyika, 2010). In the Mvudi River, seasonal variations in EC levels followed a similar pattern with higher values recorded during the dry season (Edokpayi, et al., 2015). Higher EC levels during the dry season could be related to urban land use activities, which have been found to increase EC in surface waters (Khatri & Tyagi, 2015). Electrical conductivity is more strongly affected by urban land use during the dry season because rivers experiencing low flow rates are more greatly impacted by point-source pollution from urban areas. Furthermore, the dilution capacity of rivers is reduced during the dry season (Ding, et al., 2015).

2.8.2. Chemical Water Quality Parameters

2.8.2.1. pH

pH is defined as a measure of the hydrogen ion activity in water. It may also be considered a measure of the acidity of water. As the concentration of hydrogen ions increases, pH decreases resulting in water that is more acidic. Pure water is electrochemically neutral, having a pH of 7 (Department of Water Affairs and Forestry, 1996). In South Africa, surface waters generally have a pH ranging between 6 and 8. pH is measured on a logarithmic scale from 0 to 14 with 1 being highly acidic and 14 being highly basic in nature (Department of Water Affairs and Forestry, 1996).

Anthropogenic factors may cause the pH of rivers and streams to fall outside of the target range stipulated for the protection of aquatic ecosystems in South Africa (Department of Water Affairs and Forestry, 1996). Industrial activities are often a significant contributor to acidic waters. According to the South African Water Quality Guidelines for Aquatic Ecosystems, contributions from the industrial sector may result in three forms of pollution. The first is that of point sources of pollution through the direct discharge of effluent from various manufacturing facilities such as paper, pulp etc. The second source of acidification is a result of acid mine drainage where contaminated effluent may reduce the pH level of receiving freshwater resources to as low as 2. The third pollutant source associated with industry is that of acid rain resulting from atmospheric pollution (Department of Water Affairs and Forestry, 1996). The burning of coal is an example of a practice which significantly contributes to the acidification of freshwater through acid deposition. On the contrary, nutrient enrichment and eutrophication of streams is the main cause of increased pH values in aquatic ecosystems.

pH is an important factor in freshwater due to its implications on the ability of aquatic organisms to perform life sustaining functions, more specifically respiration and the exchange of gasses and salts in water (Robertson, 2004). The inability of freshwater organisms to perform these vital functions can result in reduced growth and may have other significant sub-lethal effects. Studies have revealed that the primary productivity of freshwater organisms is reduced significantly below a pH of 5. While diurnal fluctuations in pH in streams are a natural occurrence, research indicates that these relatively small fluctuations during the course of a day generally do not have any adverse effects on aquatic ecosystem functioning (Robertson, 2004). Edokpayi (2015) and Abowei (2010) found pH levels higher in the dry season at the Mvudi and Nkoro rivers respectively. This could be a result of the decomposition of domestic and industrial waste matter. Lalparmawii (2012) and Kambwiri (2014), however, observed higher river pH values during the wet season. In a river in Mizoram, India, higher pH values may have been a result of the leaching of rock material (Lalparmawii & Mishra, 2012) while higher pH levels in the Ruo River were associated with agricultural pollutants (Kambwiri, et al., 2014).

2.8.2.2. Dissolved Oxygen

Dissolved oxygen refers to the free (non-compounded) oxygen in water and other liquids (Canadian Water Quality Guidelines, 1999). Dissolved oxygen is responsible for moderating aerobic conditions in surface waters. It is considered a key indicator when assessing the suitability of surface waters to support freshwater organisms (Department of Water Affairs and Forestry, 1996). Oxygen in water is derived from the atmosphere as well as the process of photosynthesis of plants in aquatic ecosystems. Dissolved oxygen is essential for the survival of many aquatic organisms, namely, fish, invertebrates, plants and bacteria for the purposes of respiration (Canadian Water Quality Guidelines, 1999). Bacteria require dissolved oxygen in order to decompose organic material and contribute significantly to nutrients cycling in a water resource. Aquatic ecosystems are best suited to sustained high levels of dissolved oxygen. Dissolved oxygen levels may vary based on factors such as season, time of day, temperature, depth, altitude and the rate of flow of a river. Dissolved oxygen levels tend to decrease during periods of low flow when water temperature increases (Rivers-Moore, et al., 2008). Oxygen depletion may occur during periods of high community respiration and decomposition of organic material (Lalparmawii & Mishra, 2012)

Anthropogenic factors that may alter dissolved oxygen levels include wastewater discharges (which are oxygen consuming), nutrient enrichment, change of water temperature through thermal discharges and the addition of chemicals (Canadian Water Quality Guidelines, 1999). The decomposition of organic material such as untreated sewage or vegetation in a water body can cause a serious depletion in dissolved oxygen levels. This is the most common cause of fish deaths, particularly during summer when warm water holds less oxygen. Dissolved oxygen is usually measured in mg/L or % saturation.

2.8.2.3. Nitrogen

Nutrients, including nitrogen, are essential for the growth of aquatic plants and animals; however, the overabundance of nutrients may have severe adverse ecological effects.

Nitrogen may occur in the form of nitrate, nitrite and ammonium and is needed for the nourishment of aquatic organisms and is used by them to build protein (Camargo, 2007). Approximately 78 % of the Earth's atmosphere is nitrogen (Camargo, 2007). Due to the stability of nitrate, it is the most common form of nitrogen in aquatic ecosystems. Although nitrogen occurs abundantly in the atmosphere, it may also enter rivers through human practices including the use of fertilizers in agriculture and the discharge of partially treated wastewater (Oberholster & Ashton, 2008).

An excess of nitrogen in freshwaters may result in accelerated growth of aquatic plants and algae resulting in oxygen depletion as well as reduced light in deeper water (van Ginkel, 2011). Eutrophication and consequent decrease of available oxygen can hamper respiration and ultimately lead to a decline in biodiversity of a river. A further consequence of inorganic pollution in freshwaters is from nitrogen deposition and consequent acidification due to an increase in hydrogen ions present. Excess nitrate may also impair the ability of aquatic organisms to grow and reproduce as a result of direct nitrate toxicity (Camargo, 2007). Nitrate is measured in mg/L.

2.8.2.4. Chloride

Chlorides comprise approximately 0.05 % of the Earth's crust and are vital for freshwater ecosystem survival (Hunt, et al., 2012). Chlorides are chemicals found in salt compounds such as sodium chloride and potassium chloride. Seawater contains far higher levels of chloride than freshwater where levels generally range between 1 and 100 mg/L (Hunt, et al., 2012). In rivers where there are minimal anthropogenic influences, chloride is not the dominant anion present. Natural sources of chlorides include underground aquifers and some geological formations that contain groundwater (EPA, 2001). Atmospheric precipitation may also contribute to chloride levels in freshwater systems. Increases in chlorides can occur naturally during low rainfall seasons when evaporation exceeds precipitation. However, in arid environments, chlorides in freshwater ecosystems may result from the evaporation of irrigation water used in agricultural practices which leads to the leaching of salts into soils and ultimately their entry into streams and rivers (Likens, 2010). Agricultural fertilizers may be a source of chloride in rivers because they often contain chloride in the form of potassium chloride (Mullaney, et al., 2009).

In South Africa, rainfall is comparatively low due to the semi-arid nature of the country. As such, excess chloride salts or salinization is a growing concern and poses a threat to the water quality of rivers in the country. In a study carried out by Edokpayi et al (2015), chloride concentrations were found to be higher in the dry season at the Mvudi River in South Africa. This was attributed to the increased dilution of chloride that occurs during the wet season.

Land use changes, however, can lead to the long-term salinization of streams. With the continued increase in impervious surfaces, chloride levels reaching rivers can exceed the tolerance levels of many aquatic species to chloride. The widespread use of fertilizers in agricultural practices is certainly a factor which may contribute to high concentrations of chlorides in freshwater systems in South Africa. Furthermore, seepage from septic tanks and discharges from wastewater treatment works may further contribute to the chloride concentration in rivers, particularly given the lack of removal of chlorides in the wastewater treatment process (Hunt, et al., 2012). Sources of chloride contamination from sewage may include animal sewage produced by grazing cattle which may enter streams as runoff (Kumar, 2002).

Fabric softeners have also been identified as an anthropogenic source of chloride due to the release of calcium and magnesium chloride into sewage systems (Kaushal, et al., 2005). Increased chloride can have a series of ecological implications for aquatic ecosystems. These include the acidification of streams and the mobilisation of certain toxic metals from soils (IMIESA, 2010). Furthermore, excess chloride can harm aquatic plants and animals by disrupting the process of osmoregulation which allows for the balancing of salts and other solutions in their systems. This in turn may hinder the growth, reproduction and survival of aquatic organisms. In severe cases, the invasion of saltwater species may result, altering ecosystem communities (Kaushal, et al., 2005).

2.8.2.5. Sulphate

Sulphate is a salt of sulphuric acid and can be a potentially harmful substance in freshwater ecosystems (Meays & Nordin, 2013). In the industrial sector, sodium sulphate is widely used for the production of goods. Sulphate can form several different salts when it reacts with other elements. Natural sources of sulphate in river systems include the natural weathering of rocks and atmospheric deposition (EPA, 2001). As with many other chemical constituents of water, sulphate can also enter freshwater systems through anthropogenic factors (Meays & Nordin, 2013) such as non-natural acid rain. Sulphur is emitted from several industrial processes including mining operations, textile industries, pulp manufacturing and tanneries.

Agricultural runoff, wastewater discharges and the burning of fossil fuels may also affect sulphate concentrations in aquatic ecosystems (Davies, 2007). According to a study conducted by Davies (2007), sulphate is of particular significance with respect to mining activities. The high levels of sulphate released from acid mine drainage can account for increased salinity in rivers and streams. There are a series of reactions that take place during the production of acid mine water. Ultimately the oxidation of pyrite by ferric iron generates sulfuric acid which is considered highly acidic. Dabrowski et al (2013) indicated a similar result for the Olifants River where increased levels of sulphate were related to mining activities within the catchment. Urban land uses may also contribute to increases in stream sulphate concentration. Urban sources of sulphate pollution include fertilizers, sewage effluents and run-off from rainfall (Khatri & Tyagi, 2015).

2.8.2.6. Phosphate

Phosphates occur naturally in the environment in fairly small amounts. The decomposition of organic material is one of the means by which phosphate occurs naturally in freshwater ecosystems. The natural weathering of rocks may also contribute to the phosphate content of water as it is leached into soils and subsequently enters surface water bodies (van Ginkel, 2011). The key anthropogenic sources of phosphorus in aquatic ecosystems are domestic, municipal and industrial wastewater effluents, acid mine water, agricultural runoff and wastewater originating from rural communities that do not have sanitation facilities (de Villiers & Thiart, 2007). In the Berg River, phosphate concentrations were found to peak during the dry season (de Villiers, 2007). Phosphate levels were also observed to be higher during the dry season in the Likangala River as a result of farming activities close to the river banks. In the Crocodile catchment, phosphate levels are on the rise according to a study carried out by van der Laan (2012) where irrigation practices within the catchment were identified as the primary contributor to phosphate.

Although phosphorus can occur in many forms both organic and inorganic, orthophosphate is the form in which aquatic biota can immediately access phosphates for growth (Department of Water Affairs and Forestry, 1996). Plants use phosphates to develop complex cells and aquatic biotas obtain phosphate by consuming these producers. While phosphorus is considered an essential nutrient in the development of aquatic organisms, it is also the main controller of eutrophication in freshwater ecosystems (Department of Water Affairs and Forestry, 1996). Eutrophication can have severe impacts on the survival of aquatic ecosystems and therefore phosphate is an important water quality constituent for the monitoring of water quality. Ncube (2012) highlighted the severe impacts of phosphate on the Grootdraai Dam and Vaal River as a result of agricultural practices and the washing away of nutrients into these freshwater bodies. Reduced river flow during the dry season led to high evaporation rates, further increasing the concentration of pollutants in the Vaal River (Ncube, 2012). Phosphate is measured in mg/L.

2.9. Conclusion

The aim of this chapter was to provide a summary of the literature consulted in formulating this study. The numerous threats to water quality of freshwater ecosystems have been described highlighting the important role that freshwater ecosystems play in sustaining all forms of life. The increasing pressure facing freshwater resources has led to advances in conservation science and the development of priority areas for the protection of freshwater ecosystems in the form of FEPAs. The criteria used to identify priority areas as well as the map products produced were described. The NFEPA project has provided a means to manage freshwater ecosystems more sustainably through the incorporation of FEPAs in freshwater ecosystem management efforts. This study focuses on the health and potential water quality threats to FEPAs within the Upper Crocodile catchment. The chemical and physical water quality parameters measured at selected rivers were explained including a description of the SASS version 5 Method for Biomonitoring which formed the basis for the biological sampling component of the study. The following chapter will provide a detailed description of the study area as well as the methods employed in collecting and analysing water quality data.

Chapter 3: Methods

3.1. Introduction

The first part of this chapter presents a description of the Water Management Area within which the study area chosen for the purposes of the study falls. This is followed by an outline of the criteria used to select sites for further investigation and the steps taken to collect and analyse water quality data including GIS extraction.

3.2. Study Area

The Crocodile River is of great ecological significance due to the diverse river habitats it supports. The Upper Crocodile catchment lies within the Crocodile (West) and Marico Water Management Area (WMA), with the Crocodile and Marico Rivers being the two main rivers in the area (River Health Programme, 2005). Being one of the most economically developed WMAs in South Africa, surface waters are extensively utilised in this catchment which represents one of several water stressed catchments in the country (Kotze, 2012). The Crocodile (West) and Marico WMA is one of nine WMAs in South Africa and lies primarily in the North West Province of South Africa where some parts are located in the northern region of Gauteng and the south western corner of Limpopo (Amis, et al., 2007). The confluence of the Crocodile and Marico rivers form the Limpopo River that flows eastward to the Indian Ocean in Mozambique (Kotze, 2012). The total catchment area of the WMA is 47 565 km^2 (Department of Water Affairs, 2013).

There are several urban areas located in the Crocodile (West) and Marico WMA. These include the city of Johannesburg in the northern region as well as the city of Tshwane and Waterberg District Municipalities (Department of Water Affairs and Forestry, 2008). Some of the unique characteristics of the WMA include the Pilanesberg Nature Reserve, the Cradle of Humankind Heritage Site, the dolomitic wetland or eye systems, large dams such as the Hartbeespoort, Vaalkop, Roodekopjies, Klipvoor and Roodeplaat dams and the Bafokeng Tribal Area (Smith-Adao, et al., 2006).

The catchment includes the metropolitan areas of Tshwane, part of Johannesburg, part of Ekhurleni and the town of Krugersdorp in the upper reaches. Within this part of the catchment, the major contributors to South Africa’s economy are the commercial, industrial, financial and manufacturing sectors. The Upper Crocodile catchment is highly developed in the southern part. Large industries and urban developments are located in northern Johannesburg, Midrand and southern Tshwane. Large volumes of water are transferred to the catchment from the Vaal River system via the Rand Water supply system. To the north of the Magaliesberg Mountain Range, significant irrigation and mining activities are found. (Department of Water Affairs and Forestry, 2008). According to the 2004 State of Rivers report, water requirements for the catchment far exceed its natural ability to supply water.

3.2.1. Climate

The climate of the WMA as a whole is temperate ranging from semi-arid in the west to dry in the eastern parts (Smith-Adao, et al., 2006). As a result, water demand in the area is significant with irrigation accounting for 37 % (Amis, et al., 2007). Most of the rainfall in the area occurs during summer months and is strongly seasonal. The Crocodile River receives approximately 75 % of total surface runoff from the WMA (River Health Programme, 2005). Mean annual rainfall is between 400 and 800 mm. Cold winters are experienced in the upper reaches of the catchment ranging between a daily minimum of 1 ˚C and maximum of 15 ˚C. Summer temperatures range between 10 ˚C and 30 ˚C. While frost prevails in the winter season, the area north of the Magaliesberg Mountain Range experiences more moderate winter temperatures and minimal frost. In summer, daily midday temperatures range between 35 °C and 40 °C in this area (Department of Water Affairs and Forestry, 2004). Minimum temperatures typically occur in the month of June (winter) and maximum temperatures are experienced in January (summer) (River Health Programme, 2005). Evaporation rates exceed rainfall in most areas within the WMA (Smith-Adao, et al., 2006).

3.2.2. Geology and Soils

The geology of the catchment is highly diverse. The Crocodile (West) and Marico WMA comprises some of the most valuable mineral deposits in the world (River Health Programme, 2005). The geology in the north of the Magaliesberg consists predominantly of the Bushveld Igneous Complex (BIC) (Smith-Adao, et al., 2006). The Bushveld Igneous Complex consists of volcanic intrusive rock and covers a large area of the catchment located north of the Magaliesburg (Department of Water Affairs, 2004). The BIC contains the largest reserves of Platinum Group Metals (PGMs) in the country (Cawthorn, 2010). As a result, several mines have been developed in this region to exploit PGMs. Some of the large scale mines found in the area include vanadium, chrome and platinum mines. Dolomitic rock dominates in the Upper Crocodile catchment (River Health Programme, 2005). This part of the WMA incorporates portions of the Witwatersberg and the Magaliesberg mountain ranges. The soils found in the WMA have been classified into two groups. The first group consists of moderate to deep sandy loam and is situated in the southern and eastern regions. The second group dominates the rest of the catchment and consists of moderate to deep clayey loam (River Health Programme, 2005).

3.2.3. Topography

Fairly uniform terrain is found in the WMA. The Highveld plateau located in the southern region of the catchment is characterised by gentle undulating plains (Smith-Adao, et al., 2006). The central part of the WMA is characterised by plains with a low relief, while in the northern part plains and lowlands are characterised by a low to moderate relief. Altitude ranges between 1700 m.a.s.l. on the Witwatersrand to 900 m.a.s.l. at the confluence of the Limpopo and Crocodile Rivers (River Health Programme, 2005). The most significant topographical features of the area include the Magaliesberg, the Witwatersrand, the Pilanesberg and the Waterberg (Smith-Adao, et al., 2006).

3.2.4. Vegetation

The WMA is dominated by the vegetation type Mixed Bushveld (River Health Programme, 2005). This includes both open tree savanna as well as dense, short bushveld. The central part of the WMA is characterised predominantly by Mixed Bushveld while Mixed Bushveld, Mopane Bushveld and Sweet Bushveld are the dominant vegetation types in the northern parts of the WMA (Smith-Adao, et al., 2006). North-eastern Mountain Grassland and Mixed Bushveld vegetation types are found in the eastern parts. The southernmost region of the catchment is characterized by Moist Cool Highveld Grassland and Dry Sandy Highveld Grassland vegetation types (Smith-Adao, et al., 2006). In the Upper Crocodile catchment three major types can be found, namely, Clay Thorn Bushveld (in the northern areas), Mixed Bushveld (dominating the central areas including the Magaliesberg Mountains) and Rocky Highveld Grassland (in the southern areas) (Department of Water Affairs and Forestry, 2009).

3.2.5. Economic and Social Characteristics

The WMA is considered to be one of the most extensively developed and economically active catchments in South Africa, contributing 25 % of the Gross Domestic Product (GDP) to the wealth of the country (Kotze, 2012). Diverse economic activities can be found in the Gauteng, North West and Limpopo provinces in South Africa (Department of Water Affairs, 2013). The catchment is dominated by both urban and industrial developments in Johannesburg and Pretoria (River Health Programme, 2005). Agriculture also plays a key role in the development of the rural economy in the catchment.

The Upper Crocodile catchment (Tertiary catchment A21) is well developed in terms of its strong manufacturing and commercial urban economy (Department of Water Affairs, 2013). While the catchment is not as well known for its tourism features as other provinces, it does contribute significantly to the economic sector. Unique tourism attractions include the Cradle of Humankind World Heritage Site, the Pilanesberg Nature Reserve, the dolomitic eye system and the Magaliesberg Conservation Area, all of which are important for tourism and conservation activities (Kotze, 2012).

The Crocodile (West) and Marico catchment is the second most densely populated WMA in South Africa with roughly 70 % of the population residing in urban areas (Department of Water Affairs, 2013). In 2001 the population of the catchment was estimated at 6.7 million people for the WMA while 2.2 million people reside in the Upper Crocodile catchment (Department of Water Affairs, 2004). Approximately 85 % of the population reside in the Johannesburg and Pretoria areas within the Upper Crocodile and Apies/Pienaars catchments. This is largely due to many economic and employment opportunities in these areas. As a result, informal settlements have also increased in these areas along the borders of the main cities. Unemployment is rife in the area with a rate between 30 % and 40 % higher than the national average (River Health Programme, 2005).

Intensive development for economic purposes in the catchment has led to significant impacts on the natural environment (Kotze, 2012). The high level of development and extensive human activity in the catchment has caused water use to significantly exceed available water resources (Department of Water Affairs, 2013). Approximately 30 % of the North West province has been transformed to other non-natural land uses and it is rapidly approaching a critical threshold in its state of biodiversity with only 60 % of its natural habitat remaining (Kotze, 2012).

3.2.6. Land uses

Agriculture, urban development and mining are the key sectors that have contributed to land use changes in the Crocodile (West) and Marico WMA (Kotze, 2012). Agriculture contributes 73 % to the total land use in the catchment while urban development and mining contribute 24 % and 3 % respectively (Kotze, 2012). Urban areas are situated predominantly in the south-eastern parts of the WMA (River Health Programme, 2005). The southern parts of the Magaliesberg as well as the northern parts of Johannesburg comprise mainly of smallholdings and agricultural activities (River Health Programme, 2005).

Irrigation being the most prominent land use in the catchment, a wide variety of crops is farmed, including; maize, tobacco, citrus fruit, intensive vegetable production, cotton, sorghum, sunflowers and soya bean crops (River Health Programme, 2005). Dry land crops, in particular maize, are found in the south and south-eastern parts of the catchment where higher rainfall prevails (Department of Water Affairs, 2013). The Upper Crocodile catchment consists of several mining operations including large platinum and chrome mines as well as small open cast stone and sand mines. The Upper Crocodile catchment has the highest concentration of mines where limited mining activities occur in the rest of the WMA (River Health Programme, 2005). Power generation forms a smaller but important land use is the catchment. These include the Kelvin power station, the Tshwane power station and the Rooiwal power station (Department of Water Affairs, 2004).

3.3. Catchment Breakdown

DWS was responsible for dividing drainage regions in South Africa into successively smaller hydrological units from primary catchments through to secondary, tertiary and finally, quaternary catchments. This was achieved through the application of a national hierarchical drainage subdivision system (Midgley, et al., 1994). The system used to delineate catchments resembles the United States Geological Survey (USGS) hydrological units. A quaternary catchment is therefore a nested hydrological unit within the primary, secondary and tertiary catchments (van Dam, 2011). Main rivers were defined by the Department of Water Affairs as “the rivers that pass through a quaternary catchment into a neighbouring quaternary catchment” (Nel, et al., 2007) as defined on a scale of 1:500 000.

A total of 1956 quaternary catchments were identified nationally, with sizes ranging from 100 kilometres in mountainous areas with high rainfall to quaternary catchments spanning thousands of kilometres in the arid north western parts of the country (Nel, et al., 2007). More recently, the need to create finer subdivisions led to the delineation of sub-quaternary catchments (Driver, et al., 2011). Smaller sub-catchments allow for a better representation of physical and biological variability within a catchment. These include the tributaries of main stem rivers in quaternary catchments (Driver, et al., 2011).

The Upper Crocodile catchment has a catchment area of 6336 kilometres squared and includes quaternary catchments A21 A-L as delineated by the Department of Water and Forestry. The quaternary catchments which have been selected for the purposes of this study include A21 G, F and K. The NFEPA project used sub-quaternary catchments as the planning units for FEPA boundaries. Boundaries were delineated for all catchments around each river segment which resulted in 9 417 sub-quaternary catchments nationally. These sub-quaternary catchments are roughly located within the 1 945 DWAF quaternary catchments in South Africa (Nel & Driver, 2011). On average the size of a sub-quaternary catchment is approximately five times smaller than a quaternary catchment. This study is concerned with potential threats to water quality of five river FEPAs and associated sub-quaternary catchments in the Upper Crocodile catchment. The Priority Rivers investigated are the *Magalies, Brandvlei, Skeerpoort and Sterkstroom Rivers* as well as a tributary of the Sterkstroom River, namely the *Buffelsfontein* stream.

3.4. Site Selection

The analysis of existing water quality data and spatial data provided a basis upon which to identify relevant sample sites. Spatial data was important in identifying the location of existing sample sites in relation to FEPAs. In several cases DWS sample points did not cover areas of the streams which fall directly within the FEPAs identified for this study. In addition, where sample points did coincide with the relevant FEPAs, sampling frequency was found to be inconsistent. Through spatial analysis of existing DWS sample points along the main rivers, gaps in monitoring were easily identifiable. As such, some sample points were selected in order to fill in these gaps and some sample points were selected at the same points due to inconsistent available data for those points. Sites were also selected based on their accessibility. In total, nine sample sites were identified within five river FEPAs for the study area. The sites were chosen in order to determine the water quality status of the rivers at selected points within each FEPA and to identify any potential pollution sources to the FEPA catchment areas.

3.4.1. Delineation of Sample Catchments

Having chosen the sample sites for the study, it became necessary to derive smaller catchments for each sample site. This involved the use of topographical maps to manually draw a catchment from each sample point. Data for various catchment attributes were than extracted for all of the nine smaller catchments using GIS software. The catchment mapped for each sample point encompasses the area of the landscape that could have contributed to the composition of the water sample collected at the point's location. For the purposes of this study these new derived catchments are referred to as 'sample catchments' or 'sample site catchments'.

3.4.2. Criteria used in selecting River FEPAs and Sample Sites

3.4.2.1. The Skeerpoort River

The Skeerpoort River is a tributary of the Crocodile River. It is situated within a proclaimed nature reserve and has recently been classified as a river FEPA. The river is located within the A21G sub-quaternary catchment. According to the State of Rivers report of 2005, the Skeerpoort River has an Eco-Status of GOOD/NATURAL. It is also considered to be of high Ecological Importance and Sensitivity (EI & S). This is due to the unique and diverse biota supported by the river (River Health Programme, 2005). Changes in land use are largely due to farming activities in the catchment. The seven indices used by the RHP to determine Eco-Status provided the results for the overall health of the Skeerpoort River as presented below:

- ***Instream Habitat Integrity Index:*** GOOD, with dolomitic eyes at the source of the river remaining in pristine condition.
- ***Riparian Zone Habitat Integrity Index***: GOOD, showing minor impacts apart from localised bank erosion.
- ***Riparian Vegetation Integrity Index:*** FAIR due to the impact of alien invasive species and clearing of vegetation for agricultural practices.
- ***Fish Assemblage Integrity Index:*** GOOD to NATURAL. Minimal impact on fish diversity was identified as a result of farming. This was most profound at the Hartebeespoort dam where a loss of eels occurred.
- ***Macroinvertebrate Integrity Index:*** NATURAL in terms of macroinvertebrate diversity.
- ***Water Quality Index:*** NATURAL with no significant pollution.

The report further recommended that development be restricted in the Skeerpoort river catchment given its location within a proclaimed nature reserve. Two sample sites were chosen along the Skeerpoort River. The first sample point, ***Skeerpoort Outflow*** is located at the outflow of the river and corresponds with existing DWS water quality monitoring stations. The second sample site, ***SK01***, is located further upstream at a culvert. These sample points were chosen to assess water quality at specific locations of the Skeerpoort River and to identify potential threats to water quality of the priority stream.

3.4.2.2. The Sterkstroom River

The Sterkstroom River also formed part of the RHP State of Rivers reports generated in 2005. Sample points chosen for this study fall within the Upper Sterkstroom River. The RHP report revealed the overall Eco-Status of the Upper Sterkstroom River as GOOD/FAIR (River Health Programme, 2005). The Ecological Importance and Sensitivity of the river was considered MODERATE. Water removal for farming, alien invasive species and the development of resorts were found to be the main contributors to land use changes in the Sterkstroom catchment. The health of the Upper Sterkstroom River as determined by the RHP is described below:

- ***Instream Habitat Integrity Index:*** GOOD. Minor water abstraction was identified at the Buffelspoort Dam for farming activities.
- ***Riparian Zone Habitat Integrity Index:*** FAIR due to the erosion of river banks.
- ***Riparian Vegetation Integrity Index:*** FAIR due to the impact of alien invasive vegetation throughout the area.

- ***Fish Assemblage Integrity Index:*** GOOD to NATURAL, however, alteration of the migration corridors for eels led to the loss of eels.
- ***Macroinvertebrate Integrity Index:*** FAIR with some impacts due to streamflow and habitat alteration.
- ***Water Quality Index***: GOOD.

The Sterkstroom River flows through the sub-quaternary catchment A21K and like the Skeerpoort River represents a priority area for freshwater ecosystems. Access to the river made it feasible to collect samples at three points namely, ***STERK01***, ***STERK02*** and a point on a tributary of the Sterkstroom River named ***STERK03***. The first sample point is located at the inflow of the Sterkstroom River to the Buffelspoort Dam. This site was also chosen due to its suitability to collect valuable macroinvertebrate data, having a diverse river habitat. The second sample point is located near the Sparkling Waters Hotel and has previously been sampled by the DWS. The third sample point is located on a tributary of the Sterkstroom River upstream of the Buffelspoort Dam. The tributary stream, identified as a river FEPA, also falls within the A21K sub-quaternary catchment.

3.4.2.3. The Buffelsfontein Stream

The Buffelsfontein stream is a tributary of the Sterkstroom River, located upstream of the Buffelspoort Dam. The sample point ***BUFF 01*** is located approximately 1km from the STERK 01 sample point. This site was chosen because it is classified as a river FEPA and falls within the A21K sub-quaternary catchment of the Upper Crocodile catchment. Furthermore, the Buffelsfontein stream has not been reported in the RHP State of Rivers reports. As a result, limited existing water quality data was available for the site.

3.4.2.4. The Magalies River

According to the State of Rivers report of 2005, the Eco-Status of the Magalies River is POOR with a MODERATE Ecological Importance and Sensitivity (River Health Programme, 2005). Alien invasive vegetation, water abstraction and flow modification are the key factors which have resulted in land use changes in the Magalies River catchment. The associated river health indices are described for the Magalies River, according to the State of Rivers Report (2005):

- ***Instream Habitat Integrity Index:*** POOR. Large quantities of water are removed from the river for the purposes of bottling. Abstraction of water for irrigation is also high. There are 25 furrows on the Magalies River which are also used for domestic purposes.

- ***Riparian Zone Habitat Integrity Index:*** POOR. Riparian vegetation experiences flooding due to the high number of furrows on the river. Flow alteration has impacted the natural riparian habitats.
- ***Riparian Vegetation Integrity Index:*** POOR. The clearing of vegetation for agriculture and housing developments has had serious impacts. In addition, the invasion of alien vegetation is high. Common alien invasive species identified include populous (poplar), acacia (wattle) and eucalyptus (blue gum) species.
- ***Fish Assemblage Integrity Index:*** FAIR. Lower reaches of the river are impacted by flow alterations and water abstractions. Some sensitive species however still inhabit the upper reaches.
- ***Macroinvertebrate Integrity Index:*** POOR. Abstraction of water and habitat modification is responsible for poor water quality in the lower reaches; however, the upper reaches are rated as FAIR.
- ***Water Quality Index:*** GOOD with localised water quality impacts from farming and developments.

The Magalies River is a river FEPA situated in the A21F sub-quaternary catchment of the study area. The first sample point identified for the study is named ***MAGA01***. The second sample point identified, ***MAGA02***, falls within the same river FEPA. The sample point is located at a side stream to the Magalies River.

3.4.2.5. The Brandvlei River

One sample point, ***BRAND01,*** was selected along the Brandvlei River due to accessibility. This river is classified as a FEPA and is located in the sub-quaternary catchment A21F in the Randfontein area. The Brandvlei River does not form part of the RHP monitoring sites and limited data was available for the river, however, due to its status as a river FEPA it has been included for the purposes of this study. All nine sample sites are listed in the table below.

Table 1: FEPA Sample Sites Selected in the Upper Crocodile Catchment

Site Name	River name	Sub-quaternary catchment	Tributary	Latitude	Longitude
Skeerpoort Outflow	Skeerpoort River-Outflow	A21F		25.7923	27.7709
SK01	Skeerpoort River (tributary at culvert)	A21G	✓	25.86314	27.74037
BUFF 01	Buffelsfontein River (tributary of the Sterkstroom River)	A21K	✓	25.79854	27.49656
STERK 01	Sterkstroom River	A21K		25.80728	27.47829
STERK 02	Sterkstroom River	A21K		25.83368	27.38838
STERK 03	Tributary of the Sterkstroom River	A21K	✓	25.82079	27.44499
MAGA 01	Magalies River	A21F		26.00817	27.56801
MAGA 02	Magalies River	A21F	✓	26.00792	27.56876
BRAND 01	Brandvlei River	A21F		26.13493	27.58813

3.5. GIS and Data Extraction

In order to obtain spatial data for the Upper Crocodile catchment several maps were generated for the study. This involved the extraction of GIS (Geographic Information System) data for the river system details, land use, geology and soils, vegetation, water quality monitoring locations as well as natural attributes of the catchment. The primary purpose of generating spatial data was to identify the key land use contributors to water quality within river FEPA catchments.

3.5.1. Base Map

In order to generate spatial data, GIS and hydrological data were obtained from the DWS (Department of Water and Sanitation) database. The database was accessed at the DWS offices in Pretoria. The first step involved identifying the quaternary catchments which pertain to the study area. The catchment was identified as the tertiary catchment A21, within which quaternary catchments F, G, and K were selected to represent FEPAs within the catchment. The base map highlights the main tributaries of the study area and their location within river FEPA boundaries mostly coinciding with DWS sub-quaternary catchment boundaries. In other words, river FEPA boundaries were over-layed on the detailed river system layer to provide a clear spatial representation of the scope of the study area as well as the location of key rivers classified as FEPAs. A total of five river FEPAs were identified for investigation as indicated in Figure 1. FEPAs are indicated in dark green. The main rivers within the quaternary catchment F include the Brandvlei and Magalies rivers. The Skeerpoort River is located in sub-quaternary catchment A21G and the Sterkstroom and Buffelsfontein rivers are located in sub-quaternary catchment A21K.

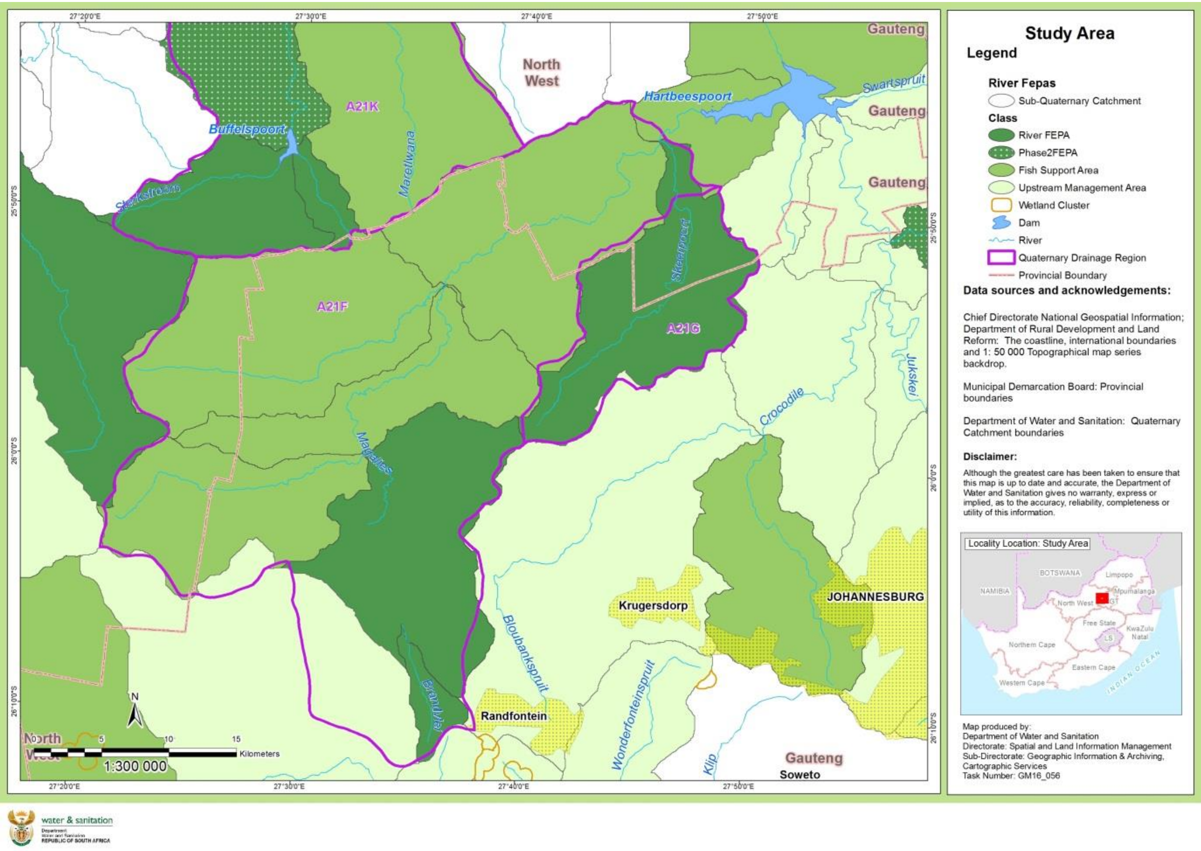

Figure 1: Study Area indicating quaternary catchment and FEPA catchment boundaries

3.5.2. Water Quality Monitoring Locations

In order to gain a clear understanding of the nature of water quality monitoring taking place within the study area, available spatial data was extracted for all existing monitoring programs in the country including the National Chemical Monitoring Programme (NCMP), the National Eutrophication Monitoring Programme (NEMP), the National Microbial Monitoring Programme (NMMP) and the River Health Programme (RHP). The River Health Program provided fairly extensive water quality data captured at several national water quality monitoring stations. The location of all RHP monitoring points was extracted and represented spatially on the base map in order to identify which monitoring stations fell within the selected FEPAs in the Upper Crocodile catchment. This was also useful in selecting the sampling sites for this study. Sample sites were selected following a review of existing DWS water quality monitoring stations. Sites selected for the purpose of this study are represented in the map below.

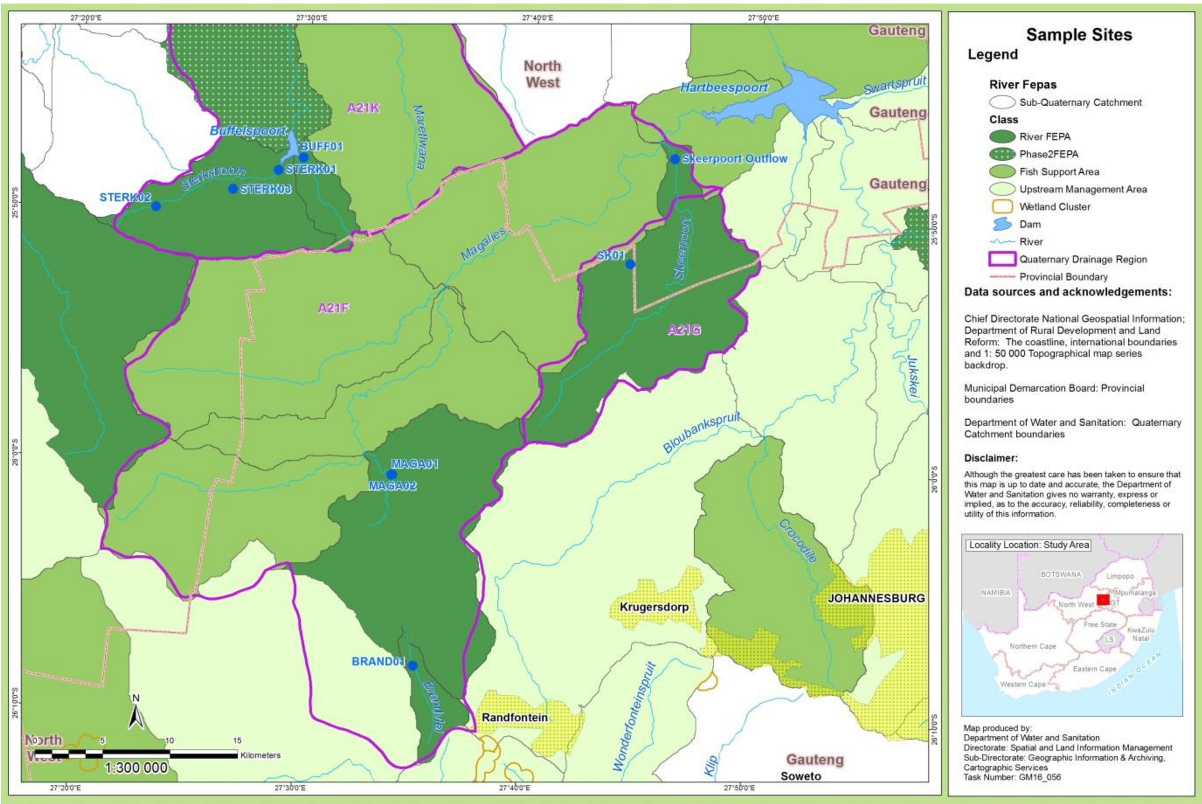

Figure 2: Water Quality Monitoring Points Selected

3.5.4. Land use

A detailed set of land use maps were generated in order to identify the various land uses which may contribute to water quality within each sample catchment. This involved extracting land use data for individual FEPA catchments and for each of the associated nested sample catchments using GIS software. Land use categories were based on land use classes as stipulated by the DWS. Furthermore, maps highlighting the percentage contribution of each land use class were generated for each sample catchment for the purposes of the study. This would prove useful in determining the main contributors to water quality within each sample catchment. In all, nine land use maps were generated for each of the nine sample site catchments.

3.6. Existing Water Quality Data

The first phase of the method employed was to collect all available water quality data for river FEPAs in the Upper Crocodile catchment pertaining to the study area. As previously mentioned, the RHP was the first national program involved in the monitoring of river health in South Africa. Therefore, all available RHP water quality monitoring data as well as any other available monitoring data for the catchment was obtained from the DWS. All information regarding the parameters being measured, the frequency of sampling and the location of sample sites were obtained from the Resource Quality Services (RQS) division of the DWS. Through extensive analysis of the historical data, all monitoring stations that were located within the rivers under investigation were identified. Water quality data was available for the Skeerpoort, Sterkstroom, Magalies, Buffelsfontein and Brandvlei Rivers. This included some random samples extending as far back as the 1970s. The RQS database included a few DWS monitoring stations that coincided with sites selected for this study. DWS monitoring station with point ID 90180 coincided with the Skeerpoort Outflow sample site. DWS monitoring station, point ID 188097 coincided with the BUFF01 sample site. Due to the inconsistency of sampling, trend analysis of historical data was not possible.

3.7. Data Collection

The data collection component of the study involved the measurement of physical and chemical water quality parameters for a sampling period of 12 months. A single set of macroinvertebrate samples were collected during the 12 month period at sample sites that were found conducive to perform the SASS 5 method for biomonitoring. Criteria used in site selection included accessibility of the site and safety when sampling as well as how well the site represented all biotopes. Data was collected over the dry and wet seasons in order to allow for analysis of seasonal variations in water quality. Nine sites were sampled at 5 river FEPAs in the Upper Crocodile catchment.

3.7.1. Physical Water Quality Data Collection

Physical water quality variables measured monthly over the 12 month period included temperature and electrical conductivity. *In situ* measurements were taken at all nine sample sites. It was important to take measurements for temperature and EC (Electrical Conductivity) immediately on-site in order to allow for the integrity of the data to be preserved and to ensure data accuracy (Wilde, 2008). Measurements were taken using an

YSI multi-parameter instrument by immersing the device into flowing water. Results were then recorded manually into a logbook and later fed into an electronic spreadsheet.

3.7.2. Chemical Water Quality Data Collection

Water samples were collected monthly at all 9 sample sites over the sampling period. Samples were prepared for chemical analysis by filtering the water immediately on-site. This was important in order to avoid changes in water quality from the time interval between sample collection and sample analysis. A plastic, pressure filtration unit was used to facilitate micropore membrane field filtration. Water samples were then labelled and stored in a cooler box until they could be transferred to a refrigerator at the WITS laboratory for further analysis.

3.7.3. Macroinvertebrate sample collection (Based on SASS 5 Rapid Biomonitoring Method)

The SASS 5 Rapid Bioassessment Method for Rivers was used to collect biological water quality samples at sites that were suitable for sample collection. The SASS 5 method is a biotic index used for the determination of river health based on aquatic macroinvertebrate communities. For the purposes of this study, macroinvertebrate data was collected once over a 12 month period in order to supplement chemical and physical water quality data for the rivers under investigation. Samples were collected at the sample sites Skeerpoort Outflow, STERK01, STERK02 and MAGA01 during spring.

Samples were collected over various river biotopes using a 1mm mesh net as specified by the SASS method. This was achieved by following a series of different techniques as outlined by Dickens and Graham (2002). The method requires that a wide area is included to ensure that the full variability of each biotope is sampled (Dickens & Graham, 2002). The method is based on assigning quality scores to taxa which have been identified to the family level. The quality score is based on the sensitivity of the macroinvertebrate to pollution from a score of 1 to 15 where a score of 1 indicates high tolerance and low sensitivity to changes in water quality and a score of 15 indicates low tolerance and a high sensitivity to pollution.

Following sample collection, identified taxa were scored on the SASS 5 scoring sheet by ticking off identified taxa under the corresponding biotope. Abundances of taxa were also roughly estimated according to SASS 5 protocol. A score of 1 was assigned if one macroinvertebrate was observed, “A” for 2-10, B for 10-100, C for 100-1000 and a D for > 1000 taxa. According to the SASS 5 method SASS scores are calculated from data identified on-site. However, for the purposes of this study, samples were chemically preserved as specified by SASS protocol. Individual organisms were picked out of the sample and preserved in vials containing 70 % ethanol for further investigation in a laboratory. The results obtained were therefore referred to as SASS Macroinvertebrate Index Scores (SMIs) derived from SASS scores since results generated from preserved samples are not considered SASS scores.

Equipment used to carry out the collection of macroinvertebrate samples included a 1mm mesh net on a 30cm square frame as specified by SASS 5 protocol. White, flat-bottomed trays were used for sample identification (one tray for each biotope). Forceps, pipettes and a magnifying lens were used for sample collection and identification. Waders were also used by the operator to ensure safety during field work. The table below provides a description of the biotopes sampled using the SASS method.

Table 2: SASS 5 Biotopes and Description

Biotope	Abbreviation	Description
Stones in current, bedrock or any solid object in current	SIC	▪ Loose stones (pebbles and cobbles) located where flowing water prevents settling of silt ▪ Stones > 2cm < 25cm in diameter ▪ Bedrock: Boulders > 25cm in diameter, large sheets of rock, waterfalls, chutes
Stones out of current, bedrock or any solid object out of current	SOOC	▪ Loose stones (pebbles and cobbles) out of current where fine sediment settles on surfaces ▪ Stones > 2cm < 25cm in diameter ▪ Bedrock: Boulders >25cm in diameter, large sheets of rock
Marginal vegetation in current	MvegIC	▪ Vegetation hanging into or growing at the edge of the river in current
Marginal vegetation out of current	MvegOOC	▪ Vegetation hanging into or growing at the edge of the river out of current
Aquatic vegetation	None	▪ Largely submerged vegetation not confined to river banks
Gravel, sand and mud	GSM	▪ Gravel: stones < 2cm in diameter ▪ Sand: sand grains < 2mm in diameter ▪ Mud: mud, silt and clay particles < 0.06mm in diameter

(Dickens & Graham, 2002)

The SIC biotope sample was collected using the SASS 5 protocol which involved a series of kicking techniques. The net was placed downstream of the stones to be kicked in a position which allowed for dislodged biota to be swept into the net by the current. Where possible, stones were turned over to further dislodge any clinging biota. Kicking was carried out for two minutes. The SOOC sample was collected by kicking and scraping stones and bedrock whilst sweeping the net over the disturbed area to contain biota in the net. Samples collected for the SIC and SOOC biotope were combined into one sample. Following collection of the sample in the net, the samples were washed down to the bottom of the net and transferred to vials containing 70 % ethanol for sample preservation.

Marginal vegetation both in and out of current were sampled over a length of 2m. As far as possible, more than one location and varying flow velocities were sampled. The technique involved vigorously pushing the net back and forth through a specific area to dislodge biota. According to SASS 5 protocol, the net was kept below the surface to avoid collecting terrestrial biota. The same technique was applied to sample aquatic vegetation over an area of approximately one square meter. Marginal and aquatic vegetation samples were combined and specimens collected were preserved in vials containing 70 % ethanol.

Gravel, Sand and Mud (GSM) was sampled for approximately one minute by stirring and shuffling feet whilst continuously sweeping the net over the disturbed area to collect macroinvertebrates present. Finally, the SASS 5 method requires that hand-picking and visual observation be included as part of the sample collection process. This was carried out for approximately one minute and involved noting organisms which may have been missed during sampling such as snails or pond skaters. Extra taxa were recorded on the SASS score sheet under the appropriate biota.

Calculating the SASS score is based on three indices, namely, the SASS Score, Number of Taxa (No. Taxa) and the Average Score per Taxon (ASPT). The first step in determining the SASS score is to tick taxa seen in the “Total” column of the score sheet. Quality scores assigned to each taxon are then summed to provide the SASS Score. The total number of taxa found = No. Taxa. Dividing the SASS Score by the No. Taxa provides the ASPT. The SMI scores produced by this study were based on the above method used to derive SASS scores.

3.8. Data Analysis

Spatial data and water quality data were analysed to determine potential threats to water quality of river FEPAs for each sample catchment. Raw data collected was sorted and consolidated into an excel document.

3.8.1. Laboratory Analysis of Water Samples

Chemical data was analysed using standard laboratory techniques at the WITS laboratory. Analyses were performed for phosphate, nitrate, chloride and sulphate. A Hach DR600 was used to perform chemical analysis of filtered samples. The procedure used is equivalent to the USEPA and Standard Method 4500-P-E for wastewater and adapted from the *Standard Methods for the Examination of Water and Wastewater.*

3.8.2. SASS 5 Data Analysis and Interpretation Guidelines

The *SASS Data Interpretation Guidelines* as outlined by Dallas (2007) were developed in an effort to standardise the method used for the interpretation of SASS data. The guidelines were developed using available data from the Rivers Database and within a spatial framework (Ecoregion Level 1 and geomorphological/longitudinal zones) (Dallas, 2007). Reference sites were identified from the Rivers Database for comparison of data. Biological Bands or Ecological categories were then developed for the interpretation of SASS data.

Table 3: Biological Bands/Ecological Categories for Interpreting SASS Data

Biological Band/Ecological Category	Ecological Category Name	Description	Colour
A	Natural	Unmodified natural	Blue
B	Good	Largely natural with few modifications	Green
C	Fair	Moderately modified	Yellow
D	Poor	Largely modified	Red
E	Seriously modified	Seriously modified	Purple
F	Critically modified	Critically or extremely modified	Black

Dallas (2007)

The ecological bands identified for FEPAs were not based on strict SASS methodology. However, the laboratory based scores referred to as SASS Macroinvertebrate Index (SMI) scores for the purposes of this study, were a reflection of the result that may have been found for ecological bands if proper field based SASS was carried out. SMI data were analysed by comparing data to reference conditions in the relevant ecoregions as outlined by Dallas (2007) in the SASS interpretation guidelines. All sample sites, apart from BRAND01, fell within the Western Bankenveld. BRAND01 fell within the Highveld Ecoregion Interpretation was based on the notion that if either SASS Score or ASPT is above the Band value it will fall into that Band (Dallas, 2007). For the purposes of this study, SMI scores were interpreted based on SASS score.

3.8.3. Statistical Analysis

3.8.3.1. Catchment Data

Land use data was compared with water quality results to identify relationships between anthropogenic land uses and water quality parameters at the sample sites. This was carried out in an effort to investigate the hypothesis that anthropogenic land uses produce water pollution. Land use percentages were tabulated for all nine sample sites and included a total of twelve land use types. The land use classes highlighted below were identified across all sample catchments. Land use classes were based on the recommended minimum legend structure for preferred stratification level (1:50 000 scale) of the DWS.

Table 4: Land Use Classes

Land Use Class	Definition
Thicket, Bushland, Bush Clumps, High Fynbos	Communities generally comprising tall, woody, plants which may be have single or multiple stems and with no obviously distinguishable structure. Canopy cover (total) is higher than 10 % and canopy heights range from 2 m to 5 m. Predominantly indigenous plants, growing within natural or semi-natural environments. Includes dense bushes. Canopy cover density classes may be mapped if desired, based on sparse (< 40 %), open (40 – 70 %), and closed (> 70 %).
Woodland	All wooded areas with a tree canopy between 10 – 70 %. A broad sparse - open – closed canopy community, typically consisting of a single tree canopy layer and a herb (grass) layer. The canopy is composed mainly of self-supporting, single stemmed, woody plants greater than 5 metres in height. Predominantly indigenous species, growing under natural or semi-natural conditions (although some areas of exotic species may be included). Does not include planted forests.
Forest (indigenous)	All wooded areas with a tree canopy > 70 %. A multi-strata community, with interlocking canopies, composed of canopy, sub-canopy, shrub and herb layers. The canopy is composed mainly of self-supporting, single stemmed, woody plants > 5 metres in height.
Forest (plantations)	All areas of systematically planted, man-managed tree resources composed of primarily exotic species (including hybrids).

	Category includes both young and mature plantations that have been established for commercial timber production, seedling trials and woodlot / windbreaks of sufficient size to be identifiable on satellite imagery. Excludes all non-timber based plantations such as tea, sisal, citrus, nut crops etc.
Unimproved Grassland	All areas of grassland with <10 % tree and/or shrub canopy cover, and >0.1 % total vegetation cover Dominated by grass like non woody rooted herbaceous plants *Essentially indigenous species growing under natural or semi-natural conditions.*
Improved Grassland	As above, except …. *Planted grassland, containing either indigenous or exotic species, growing under man-managed (including irrigated) conditions for grazing, hay or turf production, recreation (i.e. golf) etc.*
Wetlands	Natural or artificial areas where the water level is permanently or temporarily near the land surface. Typically covered in either herbaceous or woody vegetation cover. The category includes fresh, brackish and salt water conditions. Examples include pans (with non-permanent water cover), and reed-marsh or papyrus-swamp.
Bare Rock and Soil	Natural areas of exposed sand, soil or rock with no or very little vegetation cover during anytime of the year, (excluding agricultural fields with no crop cover, and open cast mines and quarries). Examples would include rock outcrops, beach sand, and dry river bed material.
Degraded Thicket	Permanent or near-permanent, man-induced areas of very low vegetation cover (i.e. removal of tree, bush, or herbaceous cover) *in comparison to the surrounding natural vegetation cover*. Typically associated with subsistence level agriculture and rural population centres, where overgrazing of livestock and / or wood-resource removal has been locally excessive. Often associated with severe soil erosion problems.
Cultivated	Areas of land that are ploughed and / or prepared for raising crops (excluding timber production). Unless otherwise stated, includes areas currently under crop, fallow land, and land being prepared for planting. Class boundaries are broadly defined to encompass the main areas of agricultural activity, and are not

	defined on exact field boundaries. As such all sub-classes may include small inter-field cover types (e.g. hedges, grass strips, small windbreaks), as well as farm infrastructure. Several sub-classes are defined, based on the following parameters : <u>Commercial</u>: characterised by large, uniform, well managed field units (i.e. ± 50 ha), with the aim of supplying both regional, national and export markets. Often highly mechanised. <u>Semi-Commercial</u>: characterised by small – medium sized field units (i.e. ± 10 ha), within an intensively cultivated site, often in close proximity to rural population centres. Typically based on multi-cropping activities where annual (i.e. temporary crops) are produced for local markets. Can be irrigated by either mechanical means or gravity-fed channels and furrows. Medium - low levels of mechanisation. <u>Subsistence</u>: characterised by numerous small field units (less than ± 10 ha) in close proximity to rural population centres. Field units can either be grouped either intensively or widely spaced, depending on the extent of the area under cultivation and the proximity to rural dwellings and grazing areas. Includes both rain fed and irrigated (i.e. mechanical or gravity fed), multi-cropping of annuals, for either individual or local (i.e. village) markets. May include fallow and 'old fields', and some inter-field grazing areas (which are often classified as degraded). <u>Permanent Crops</u>: lands cultivated with crops that occupy the area for long periods and are not re-planted after harvest. Examples would include sugar cane and citrus orchards. Note in the case of sugar cane, the growing season is typically 15 – 18 months per ratoon (i.e. harvest), with 2 – 3 ratoons possible before re-planting. Sugar cane is mapped as a separate crop type, and includes both large and small scale commercial activities, as well as fallow (i.e. burnt /cleared) areas. <u>Temporary Crops</u>: land under temporary crops (i.e. annuals) that are harvested at the completion of the growing season, and that will remain idle until re-planted. In general this refers to maize and soya bean cultivation within the Pongola catchment,

	although cotton is locally dominant amongst the larger commercial sugar cane plantation areas. Irrigated / Non-Irrigated: major irrigation schemes (i.e. areas supplied with water for agricultural purposes by means of pipes, overhead sprinklers, ditches or streams), and are often characterised.
Urban/Built – up (Residential)	Formal built-up areas, in which people reside on a permanent or near-permanent basis, identifiable by the high density of residential and associated infrastructure. Includes both towns, villages, and where applicable, the central nucleus of more open, rural clusters. Unless otherwise specified (and or mapped), will include both residential, commercial, industrial and transportation land-uses as well. Low density smallholdings frequently located on the urban fringe are mapped as a separate sub-classes, subdivided by the appropriate (level I) background vegetation type. If visible, individual farm units are also mapped as isolated smallholding units.
Urban/Built-up (Commercial)	Mercantile: Non-residential areas used primarily for the conduct of commerce and other mercantile business, typically located in the central business district (CBD). Often consisting of a concentration of multi-level buildings, but also includes small commercial zones (i.e. spaza shops) within former black townships. Education, Health, IT: Non-residential, non-industrial sites or complexes associated with educational (i.e. schools, universities), business development centres such as industrial 'techno-parks', and / or social services (i.e. hospitals), often consisting of a concentration of multi-level buildings (Note: only mapped if clearly identifiable, otherwise included within '*commercial* / *mercantile*' or '*suburban* 'categories).
Urban/Built-up (Industrial)	Industrial Transport (heavy): Non-residential areas with major industrial (i.e. manufacture and/or processing of goods and products) or transport related infrastructure. Examples would include power stations, steel

	mills, dockyards, train stations and airports (i.e. Johannesburg). Industrial Transport (light): Non-residential areas with major technology, manufacturing or transport related infrastructure. Examples would include light manufacturing units, warehouse dominated business development centres, and small airports (i.e. Lanseria). Also includes similar structures such as pig and battery hen breeding units.
Mines and Quarries	Underground/Subsurface Mining: Active or non-active underground or sub-surface based mining activities. Category includes all associated surface infrastructure etc. Surface-based Mining: Active or non-active surface-based mining activities. Includes both hard rock or sand quarry extraction sites, and opencast mining sites i.e. coal. Category includes all associated surface infrastructure. Mine tailing/Waste Dumps: Non-vegetated, exposed mining (and heavy industry) extraction or waste material

Minimum legend structure for preferred stratification level (NLC, 2000)

Exploratory data analysis was conducted to identify patterns in the catchment data collected. Bivariate scatterplots are an effective tool for the examination of deterministic patterns in data as opposed to those patterns associated with time i.e. temporal trends and seasonality. A series of scatterplots were created in order to investigate the effects of land use on water chemistry. The mean value for chemical parameters was used since there were several observations for each sample site over the sample period. Land use percentages represented the independent (predictor) variables and annual mean values for water chemistry represented the dependent variables. A series of graphs are presented in the results section.

Seasonal mean chemistry was also plotted against percentage land use for all nine data points. Seasonal mean chemistry was analysed because land use impacts can differ from one season to the next. All scatter plots were fitted with a trend line using Microsoft Excel. The plots were used to indicate which land uses appeared to influence water chemistry. Time-plots were generated in order to identify seasonal variations in water chemistry over the course of sampling. The exploratory data analysis component therefore formed the basis for selection of predictor variables for further statistical analysis using multiple linear regression analysis.

3.8.3.2. Water Chemistry

Raw data for chemical and physical water quality was tabulated for each site in an excel spreadsheet. Data cleaning and checking was then carried out in order to ensure accuracy of the data entered before compiling a clean data file for water chemistry. This was followed by the generation of summary/descriptive statistics for water chemistry. Certain specific features of a dataset are characterised using descriptive statistics. In order to avoid misrepresentation of the original data, as much information as possible has been summarized by the descriptive statistics.

The summary statistics included the minimum, maximum, mean and standard deviation for all water quality parameters across all nine sample sites. The central tendency of a set of data is one of the most important measures in descriptive statistics. As a result, the mean of the chemical data was calculated for each sample site. The standard deviation was calculated in order to indicate the dispersion of the dataset, another important measure of any dataset. pH values were transformed to H^+ in order to calculate descriptive statistics for pH and then converted back to pH values. Bivariate plots of water chemistry concentrations versus time were generated in order to give a visual perspective of data patterns. Time plots were created to identify seasonal variation in water chemistry over the sampling period.

3.8.3.3. Principle Component Analysis (PCA)

PCA is a powerful multivariate statistical technique which transforms a group of potentially correlated variables into a smaller number of new variables. It is considered a data - reduction tool and shares many similarities with exploratory factor analysis. The primary aim of PCA, therefore, is to reduce a relatively larger number of variables into a smaller dataset of ‘artificial’ variables called ‘principle components’ where the new set of variables still account for most of the variance in the original dataset. New variables or principle components are orthogonal and uncorrelated to each other and combined; explain the total variance of the dataset. The first principle component retains the majority of the variance such that as the component number increases, each subsequent component accounts for less of the total variance. Ultimately, many variables can be summarized by a few components. PCA has been well used in the field of environmental science to assess the impact of anthropogenic activities on water quality (Kebede & Kebedee, 2012).

A PCA was run using annual mean concentrations of water quality parameters to identify the main axes of chemical variation in the dataset. The software used for the analysis was SPSS version 24. The choice of which principle components to retain was based on linearity between variables (including variables with at least one correlation greater than 0.3), the KMO measure for the overall dataset, the KMO measure for individual variables, Bartlett's test of sphericity and the results of a scree plot (Laerd Statistics, 2015). The first two components were retained since they explained most of the variability in the dataset.

3.8.3.4. Multiple Linear Regression

Multiple linear regression analysis was performed for each independent variable using SPSS statistics version 24 in order to identify catchment predictors of water quality and determine potential threats to water quality of the priority freshwater ecosystems under investigation.

3.8.4. South African Water Quality Guidelines

Department of Water Affairs and Forestry is the custodian of South Africa's water resources. In 1994, the department brought about the development of a set of guidelines for various water uses in South Africa. The seventh volume in the series of water quality guidelines, namely, *The South African Water Quality Guidelines for Aquatic Ecosystems* is used as the basis for the evaluation of water quality data for the purposes of this study. The South African Water Quality Guidelines provide information for specific water quality constituents where the term constituent refers to any property of water as well as any substances dissolved (or suspended) in it (Department of Water Affairs and Forestry, 1996).

The guideline includes a description of the Target Water Quality Range (TWQR), the Chronic Effect Value (CEV), the Acute Effect Value (AEV) and supporting information. According to the guidelines, the TWQR is not considered a water quality standard but rather a management objective. It describes the range of concentrations or levels of water quality parameters within which aquatic ecosystems remain safe from adverse effects. The CEV is defined by the guideline as "that concentration or level of a constituent at which there is expected to be a significant probability of measurable chronic effects to up to 5 % of the species in the aquatic community". The AEV is defined as 'that concentration or level of a constituent above which there is expected to be a significant probability of acute toxic effects to up to 5 % of the species in the aquatic community' (Department of Water Affairs and Forestry, 1996).

3.9. Conclusion

The study area has been described with respect to overall catchment characteristics for the Crocodile (West) and Marico WMA and where possible for the Upper Crocodile catchment specifically. Key criteria used in selecting sample sites included selecting points within sub-quaternary catchments classified as river FEPAs and selecting sites where sampling proved feasible. Data collection comprised three components, namely, physical, chemical and macroinvertebrate water quality data collection. The generation of land use maps comprised the spatial data component and were compiled in order to determine catchment predictors of water quality. The primary concepts used to analyse water quality data included the generation of a series of scatter plots comparing land use percentage's with water chemistry for all sites and the application of multiple linear regression to determine catchment predictors of water quality. Chapter four will present the key results of the study.

Chapter Four: Results

4.1. Introduction

The following chapter highlights the key findings of the study. Descriptive statistics were calculated for water chemistry in order to investigate patterns in water quality data. Mean values were calculated for annual and seasonal data. These statistics are described in Tables 7 through 15 for each sample site. In the case of seasonal mean values, summer mean values were based on the months of December 2014 and January 2015. Autumn mean values included data for March and April 2015. Winter mean values were based on data collected during June and July 2015 and spring mean values included the months of November 2014, September 2015 and November 2015.

Exploratory data analysis was conducted to identify linear associations in the dataset and more specifically to explore relationships between the dependent variables (water chemistry) and the independent or predictor variables (anthropogenic land use). Time plots for water chemistry were useful in identifying patterns in water quality over time and seasonal variation in water quality across sample sites. Scatter plots were produced using Microsoft Excel by plotting mean values for water chemistry against percentage land use for all twelve land use classes identified within the sample catchments. Since anthropogenic land use is of particular relevance to the study, significant correlations between anthropogenic land use and water chemistry formed the basis for statistical analysis.

Principle component analysis (PCA) of the mean concentrations of water quality variables emphasized the variation in the dataset. PCA further highlighted the most meaningful water quality parameters in the dataset and allowed for the identification of potential pollution sources of FEPA sample catchments. The results of exploratory data analysis are presented graphically in the form of time plots, scatter plots and tables. Multiple linear regression analysis was applied to further investigate potential land use predictors of water quality at river FEPA sample catchments. A model was developed for each dependent variable. The land use variables selected for inclusion in each model were based on results obtained through analysis of the scatterplots and PCA. Checking the dataset for the assumptions of MLR analysis played a role in the exclusion or inclusion of predictor variables from MLR models. Multiple regression analysis was performed using SPSS Statistics version 24. The results are presented in the form of graphs, charts and tables. This chapter is focused primarily on describing the data; however, more in depth interpretation of the results of the study and water quality implications for FEPA sample catchments is included in the discussion chapter to follow.

4.2. GIS and Land Use Data

The primary spatial data generated for the purpose of the study were land use maps. Working closely with the Spatial and Land Information Management (SLIM) section of the DWS, land use maps were created for the overall tertiary catchment (A21), for individual river FEPA catchments (as delineated by the NFEPA project) and finally for FEPA sample catchments pertaining to each sample point. Sample catchments were delineated from topographic maps of FEPA catchments.

Sample catchments were physically delineated by the area upstream of each sample point. The methods were carried out manually using topographic maps, based on the primary contour lines and paying careful attention to the direction of water flow within each catchment. In all, nine sample catchments were delineated for each of the nine sample points. The next step in the analysis of catchment data was to extract land use data for each sample catchment. This was followed by the extraction of land use percentages for each sample catchment. The extraction of land use percentages was carried out in order to determine the relative proportion of each land use type within sample catchments. Land use percentages were tabulated for all nine sites under investigation (Table 4.1 and Table 4.2). Where appropriate, land use classes with several categories were aggregated for the purposes of data analysis and to allow for easier interpretation. As such, the land use classes depicted on the catchment maps have been aggregated for statistical analysis.

The land use categories for "cultivated, temporary, commercial, dry land" and "cultivated, temporary, commercial, irrigated" were combined to form one land use category called "cultivated". Similarly, "Forest plantations (Acacia spp)" was changed to the "Forest plantations" land use category. The "Urban" land use category referred to in this chapter includes several categories for urban land use, namely, "Urban/Built-up (smallholdings, grassland)", "Urban/Built-up (industrial/transport: light)", "Urban/built-up (industrial/transport: heavy)", "Urban/Built-up (residential, formal suburbs)" "Urban/Built-up (commercial, education, health, IT)" and "Urban/Built-up (commercial, mercantile)". The land use class "Thicket, Bushland, Bush clumps, High Fynbos" was abbreviated to "Thicket"; and the land use class "Mines and Quarries" to "Mines". Finally, the land use class "Bare Rock and Soil" is referred to as "Barerock". All nine FEPA sample catchments are presented below.

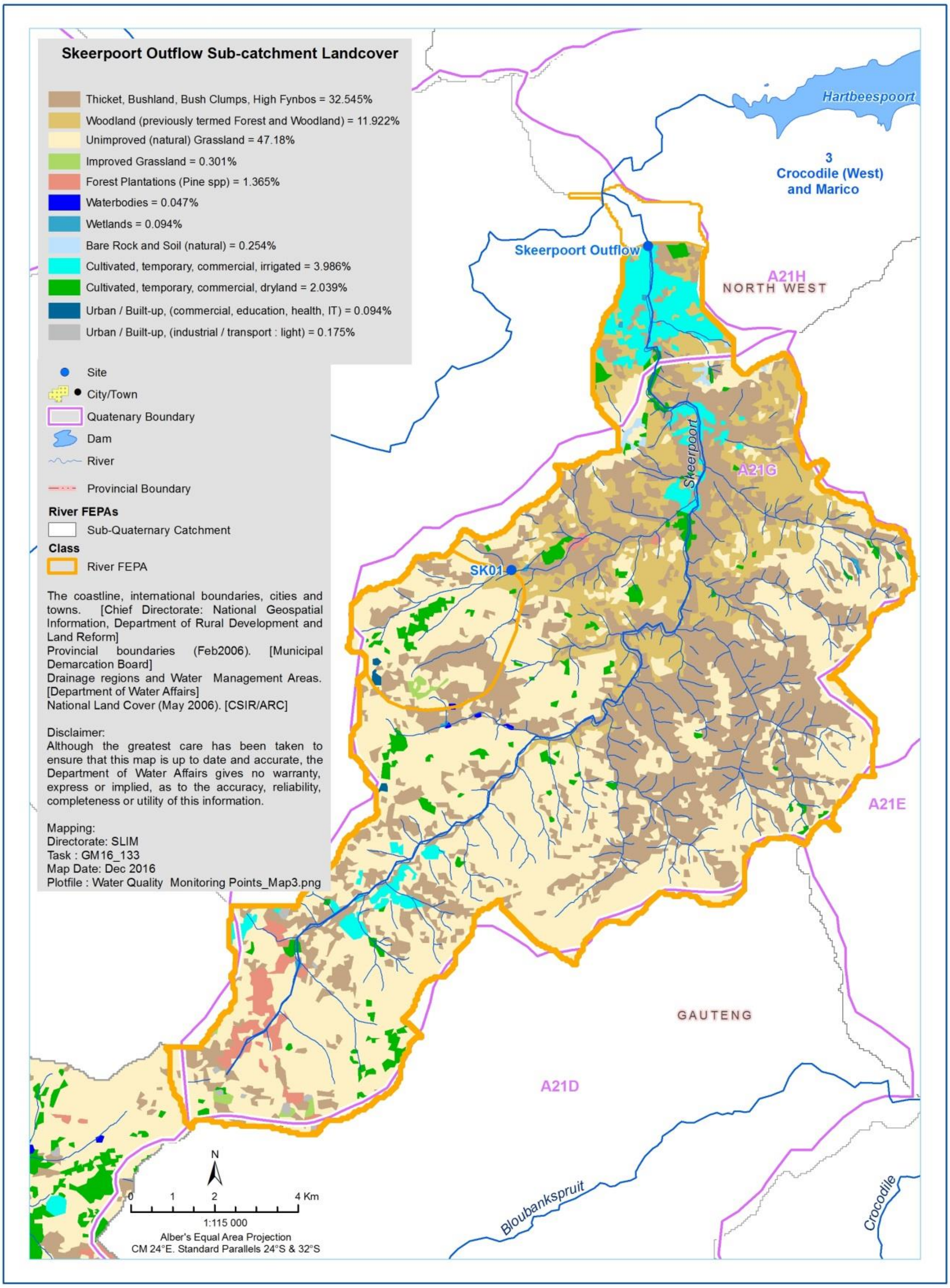

Figure 3: Percentage Land Use – Skeerpoort Outflow FEPA Sample Catchment

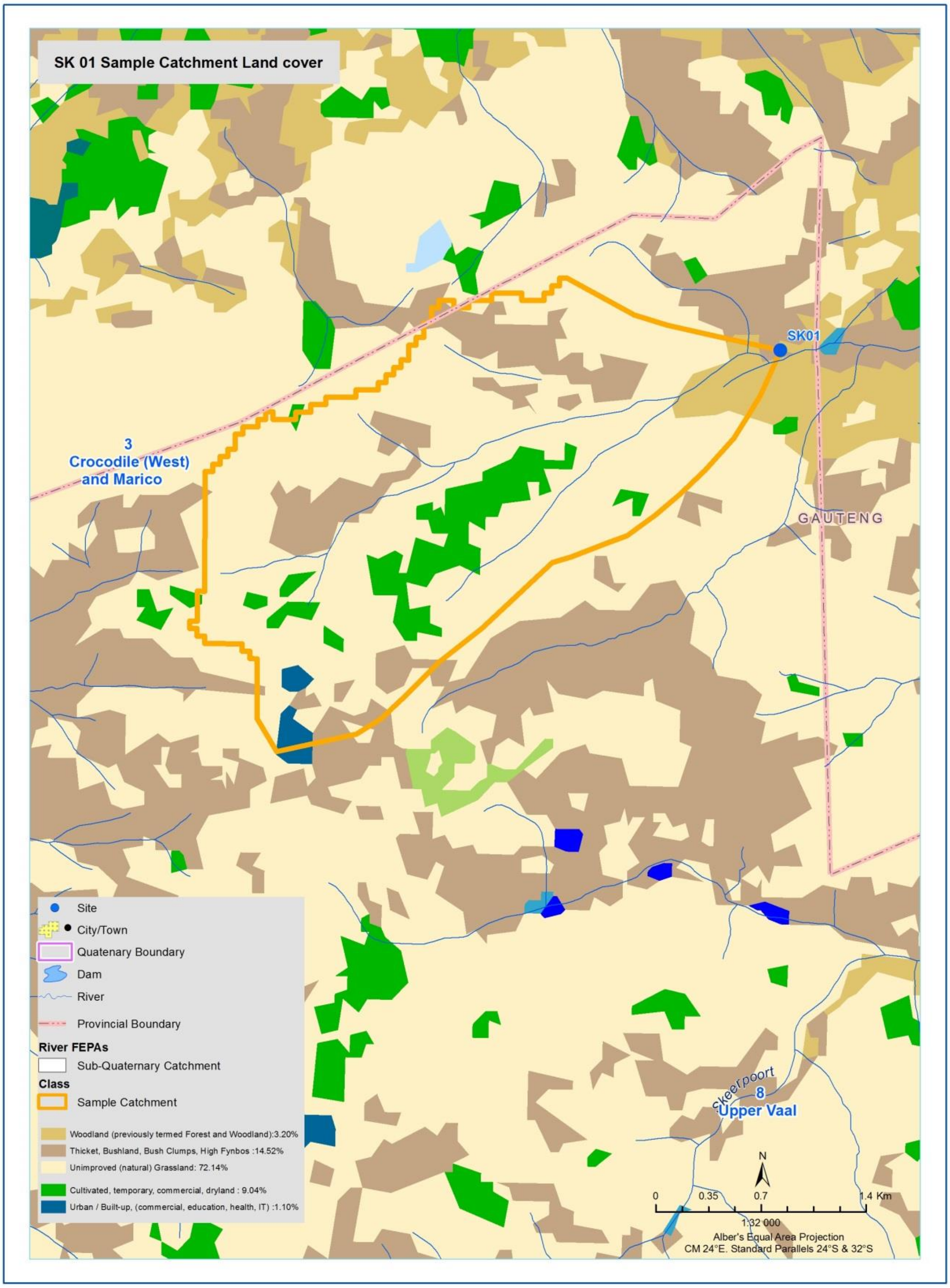

Figure 4: Percentage Land Use – SK01 FEPA Sample Catchment

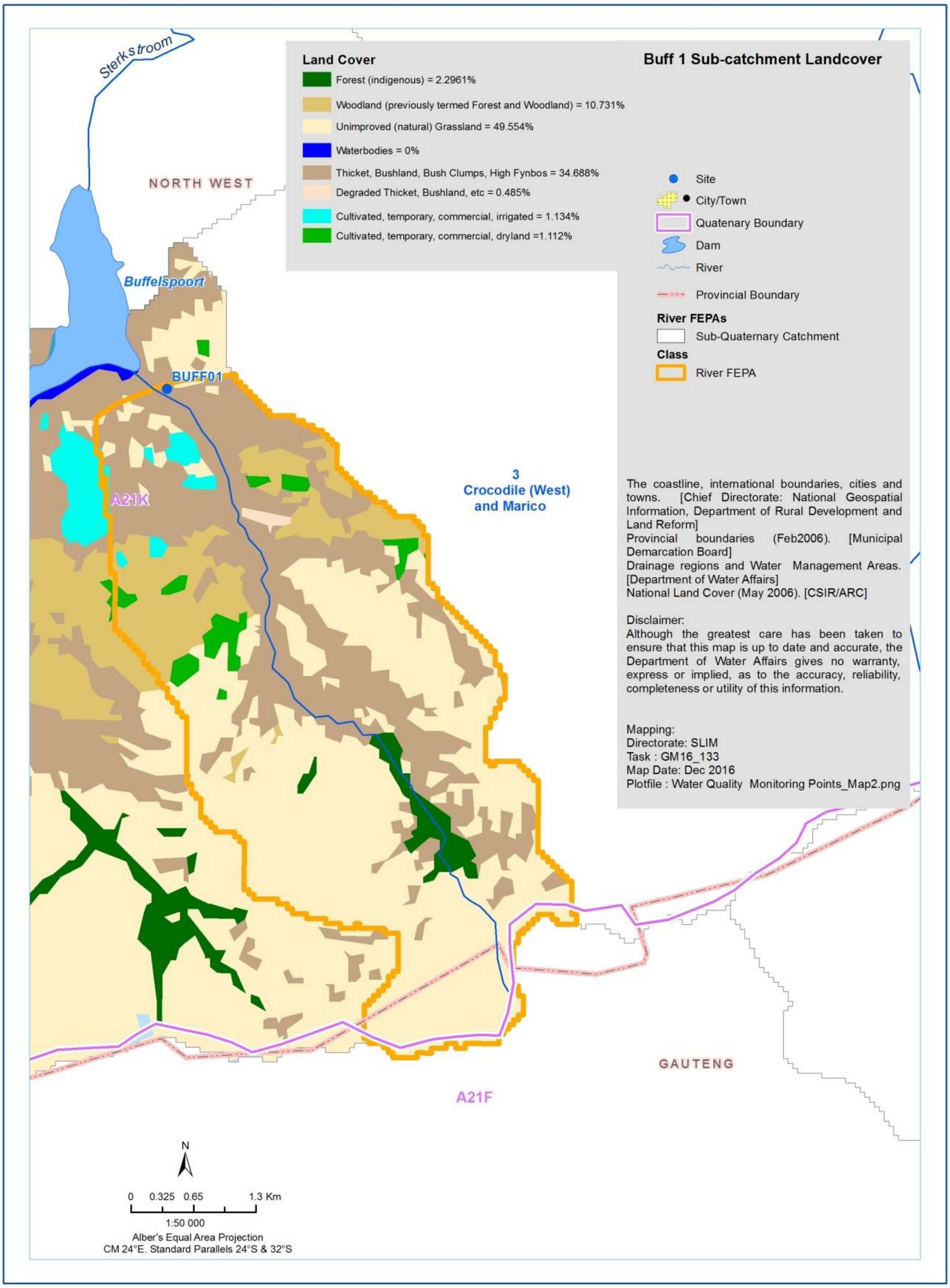

Figure 5: Percentage Land Use – BUFF01 FEPA Sample Catchment

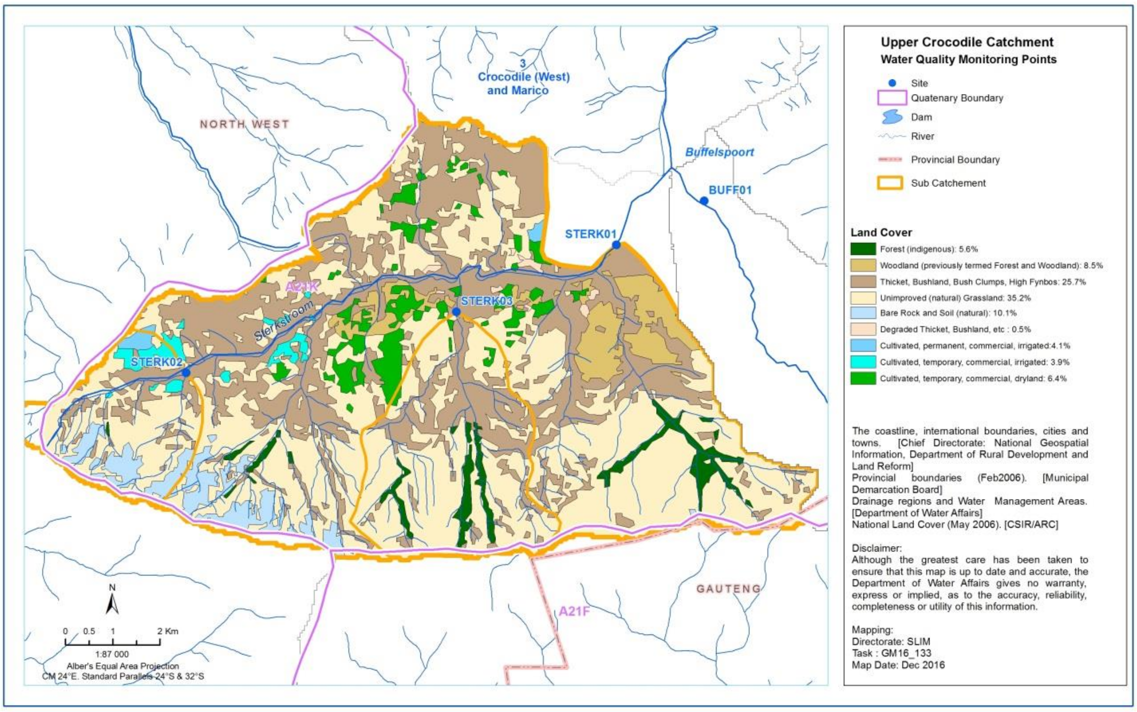

Figure 6: Percentage Land Use – STERK01 FEPA Sample Catchment

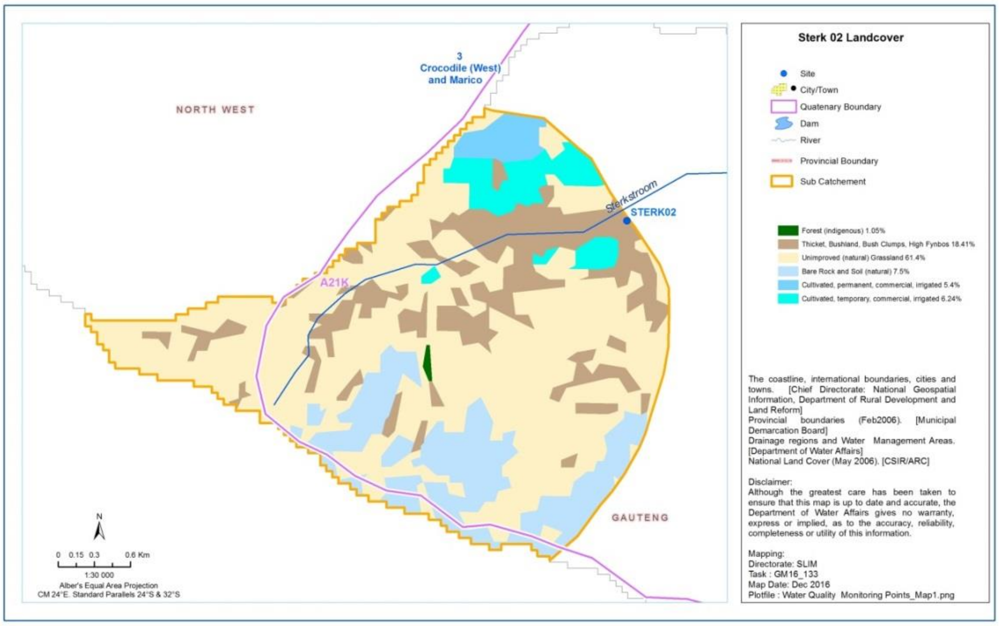

Figure 7: Percentage Land Use – STERK02 FEPA Sample Catchment

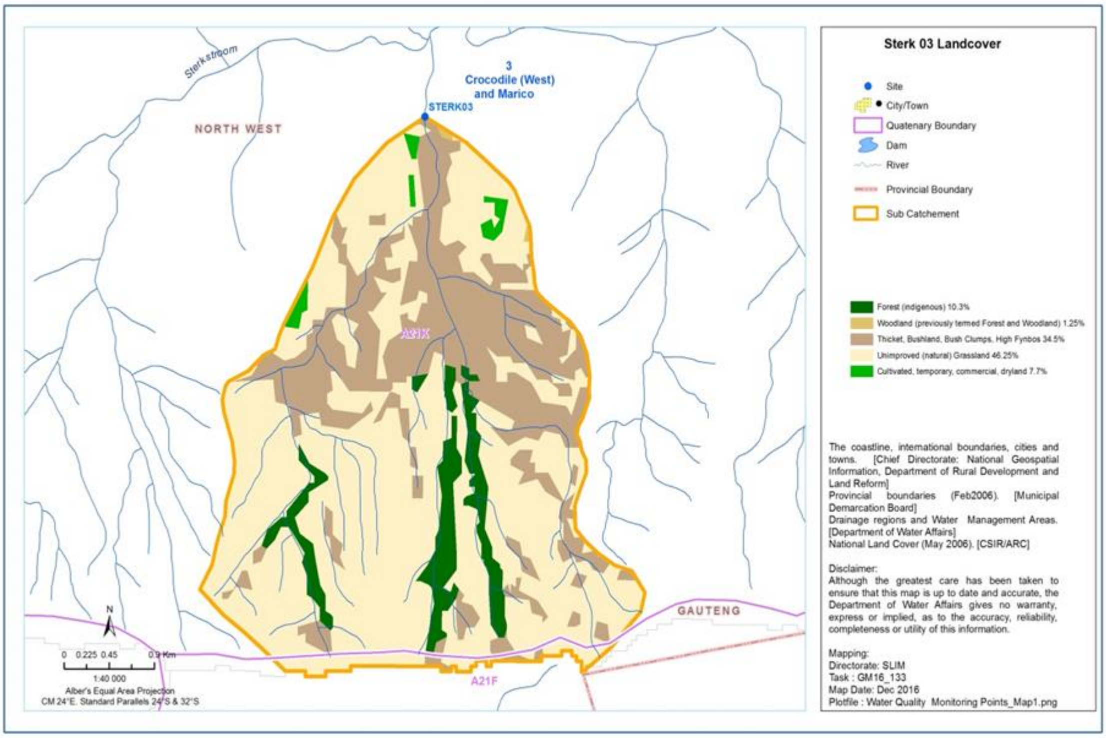

Figure 8: Percentage Land Use – STERK03 FEPA Sample Catchment

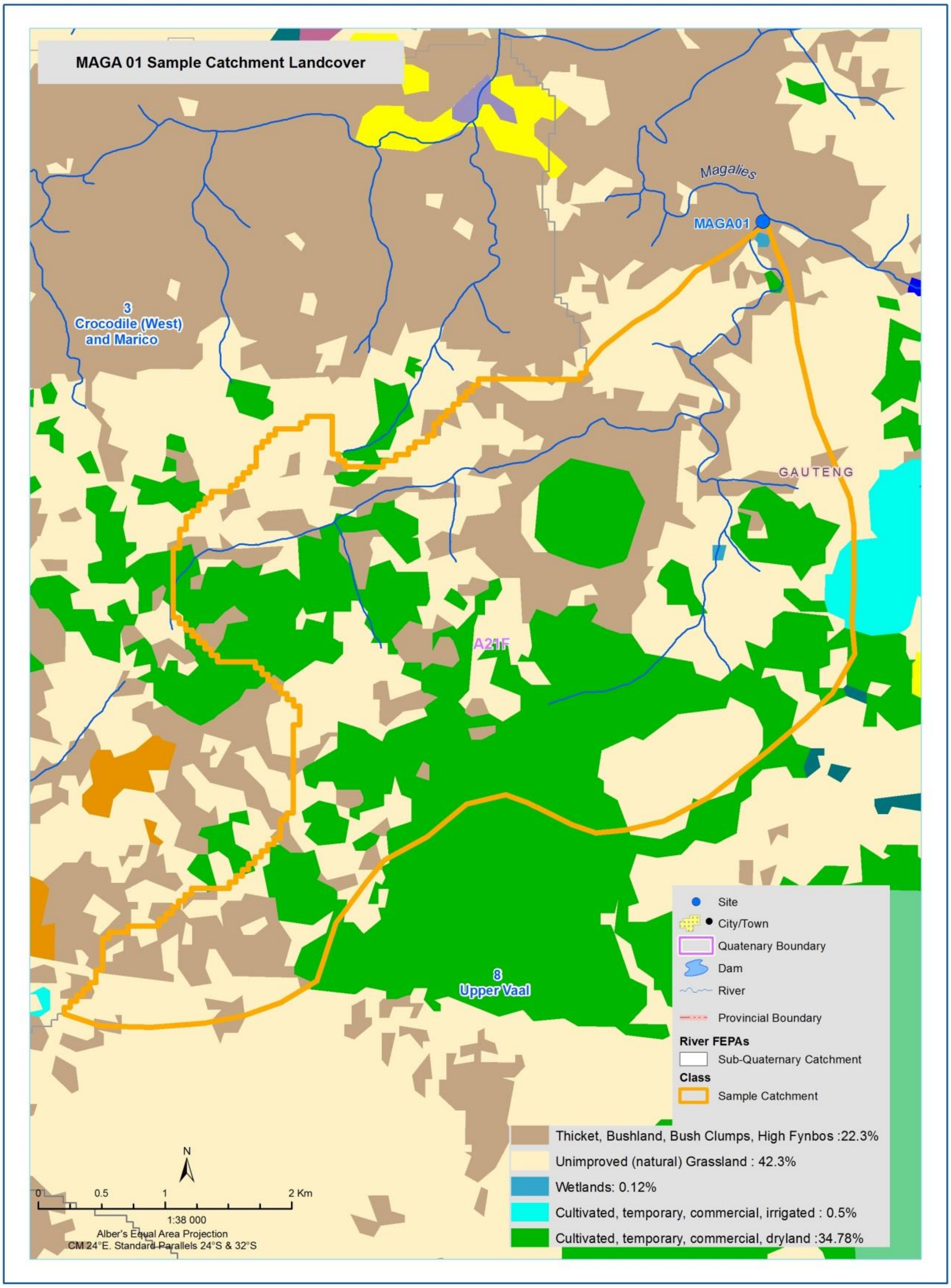

Figure 9: Percentage Land Use – MAGA01 FEPA Sample Catchment

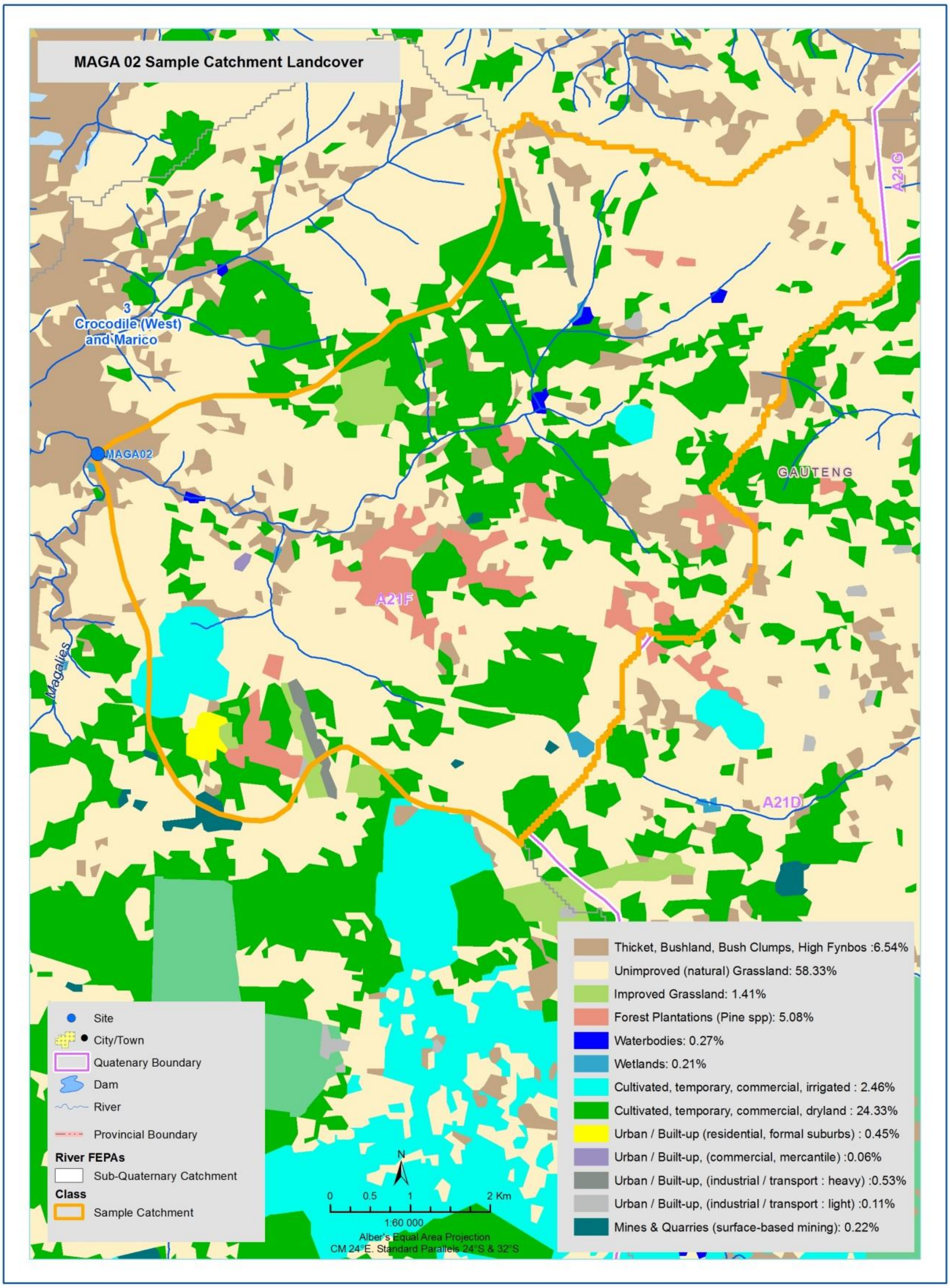

Figure 10: Percentage Land Use – MAGA02 FEPA Sample Catchment

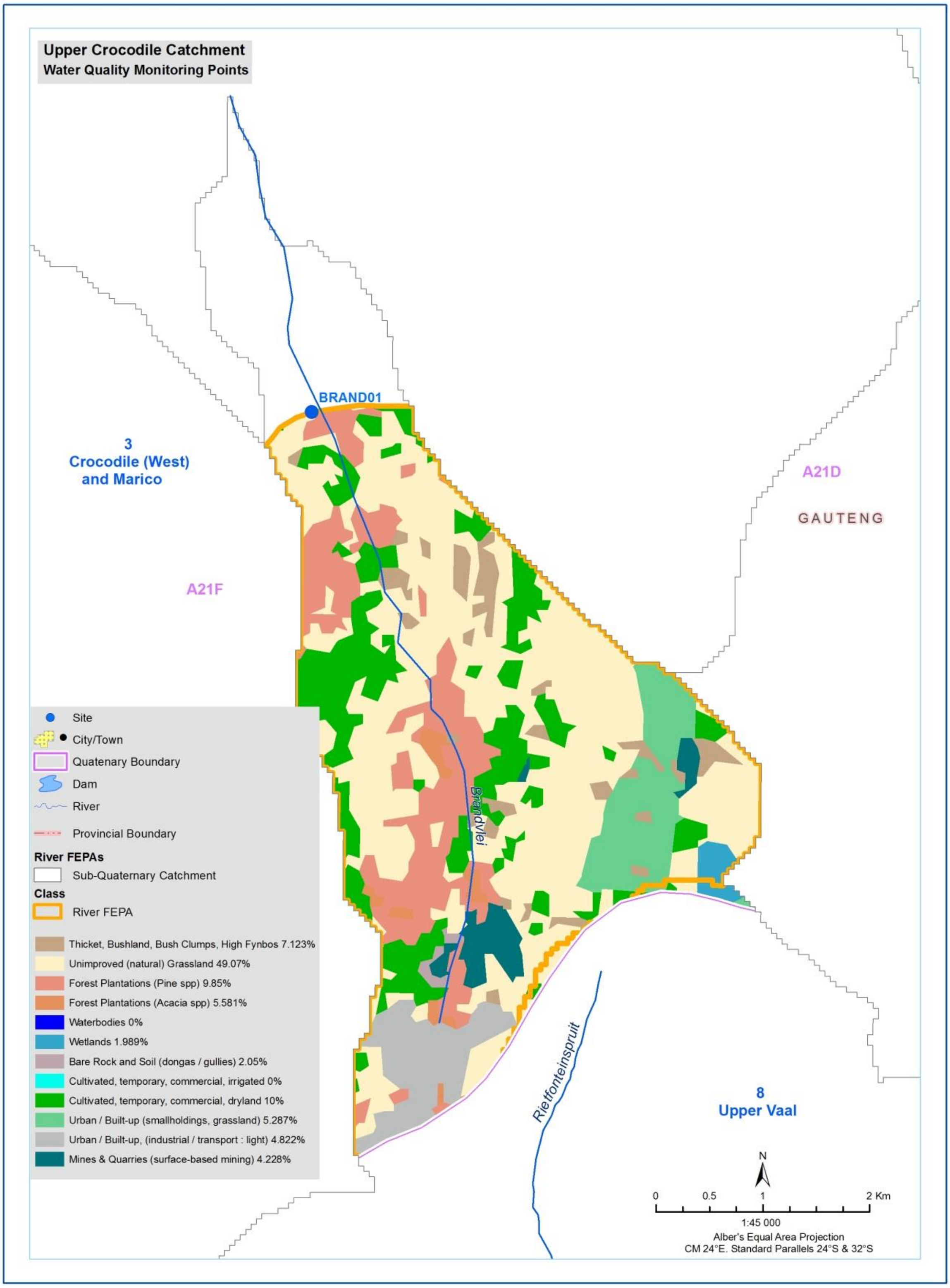

Figure 11: Percentage Land Use – BRAND01 FEPA Sample Catchment

The sample site Skeerpoort Outflow is located on the Skeerpoort River FEPA, located within the A21G quaternary catchment as delineated by the Department of Water and Sanitation. The sample catchment also coincides with the FEPA catchment boundary defined by the NFEPA project. Two sample sites were selected, the first of which is situated at the outflow of the Skeerpoort River while the second sample point (SK01) is situated along a tributary of the main river. While the FEPA sample catchment is dominated by unimproved grassland (47.2 %), anthropogenic land uses did contribute to a small proportion of land area within the catchment. Cultivated land comprised 6 %, forest plantations 1.4 % and urban land area 0.3 % of the catchment land area.

While sample site SK01 falls within the same river FEPA catchment as the Skeerpoort Outflow sample site, the sample catchment boundary delineated for this study provides detailed land cover data specific to the area that was sampled. Cultivated land comprised a higher 9 % of the catchment while urban land use accounted for 1 % of the catchment land area. The BUFF01 sample catchment is located downstream of the Buffelspoort Dam and falls within the A21K quaternary catchment as defined by the DWS. Degraded thicket comprised 0.5 % of the land area within the catchment while cultivated land contributed to 2.3 % of the catchment.

The STERK01 sample catchment coincides with the river FEPA catchment boundary defined by the NFEPA project and is situated within the A21K quaternary catchment. Sample catchments STERK02 and STERK03 are nested within the overall STERK01 catchment boundary. Sample points are located at various sites along the Sterkstroom River FEPA. Anthropogenic land uses within the catchment consisted of degraded thicket (0.5 %) and a relatively high proportion of cultivated land (14.4 %). Sample catchments STERK02 and STERK03 were influenced by cultivated land use. The STERK02 sample site is comprised of 11.6 % cultivated land area while STERK03 sample site is comprised of 7.7 % cultivated land area. Degraded thicket did not contribute to these catchments.

The Magalies River FEPA was sampled at two points, the first of which is located on the Magalies main river, namely MAGA01. Sample catchment MAGA02 is located at the confluence of the Magalies River and a tributary. Both sample catchments are situated within the A21F quaternary catchment and reflected a higher level of anthropogenic land use activities than for all other sample catchments. For both catchments, cultivated land (both dry and irrigated) comprised 26.8 %, forest plantations 5.1 %, urban land 1.2 % and mines 0.22 % of the total catchment land area. The Brandvlei River is represented by sample catchment BRAND01, located within the quaternary catchment A21F. The highest percentage of mines (4.2 %) and urban land (10.11 %) can be found in this catchment. Cultivated land comprises 15.43 % of the catchment land area.

The table below provides a summary of land use percentages comprising each sample catchment.

Table 5: Land Use Percentages for FEPA Sample Catchments

LAND USE CLASS (%)						
SITE	**Thicket**	**Degraded Thicket**	**Unimproved Grassland**	**Improved Grassland**	**Forest (plantations)**	**Forest (indigenous)**
Skeerpoort Outflow	32.55	0	47.18	0.3	1.37	0
SK01	14.52	0	72.14	0	0	0
BUFF01	34.69	0.49	49.55	0	0	2.3
STERK01	25.7	0.5	35.2	0	5.6	0
STERK02	18.41	0	61.4	0	0	1.05
STERK03	34.5	0	46.25	0	0	10.3
MAGA01	22.3	0	42.3	0	0	0
MAGA02	6.54	0	58.33	1.41	5.08	0
BRAND01	7.12	0	49.07	0	0	0

Table 6: Land Use Percentages for All Sample Catchments (continued.)

Land Use (%)						
SITE	**Wetlands**	**Barerock**	**Woodland**	**Cultivated**	**Urban**	**Mines**
Skeerpoort Outflow	0.09	0.25	11.92	6.03	0.27	0
SK01	0	0	3.2	9.04	1.1	0
BUFF 01	0	0	10.73	2.25	0	0
STERK 01	0	10.1	8.5	14.4	0	0
STERK 02	0	7.5	0.0	11.64	0	0
STERK 03	0	0	1.25	7.7	0	0
MAGA 01	0.12	0	0	35.28	0	0
MAGA 02	0.21	0	0	26.79	1.15	0.22
BRAND 01	1.99	2.05	0	10	10.11	4.23

There was a variety of land use classes across a range of sample catchments within the study area, as depicted above. Across all sites, percent thicket varied between 6.54 % and 34.69 %. This indicated significant variability, with BUFF01 showing the highest percentage of thicket and MAGA02 the lowest. Degraded thicket values, on the other hand, ranged between 0.49 % and 0.5 % across all sites. Sample site SK01 contained the highest percentage of unimproved grassland (72.14 %), while sample catchment STERK01 contained the lowest percentage of unimproved grassland (35.2 %). Improved grassland varied between 0.3 % at Skeerpoort Outflow and 1.41 % at MAGA02. Forest plantations ranged between 1.37 % at Skeerpoort Outflow and 5.6 % at STERK01. Indigenous forest ranged between 1.05 % at STERK02 and 10.3 % at STERK03.

BRAND01 was found to have the highest percentage of wetlands (1.99 %), while Skeerpoort Outflow showed the lowest percentage of wetlands (0.09 %). Barerock varied between 0.25 % at Skeerpoort Outflow and 10.1 % at STERK01. The sample catchment with the highest proportion of woodland was Skeerpoort Outflow (11.92 %) and the lowest proportion of woodland was STERK03 (1.25 %). Percentage of cultivated land showed significant variation across all nine sample catchments. The sample catchment with the highest percentage of cultivated land was MAGA01 (35.28 %). The sample catchment with the lowest proportion of cultivated land was BUFF01 (2.25 %). Urban land use varied from 0.27 % at Skeerpoort Outflow to 10.11 % at BRAND01. Finally, mines were present in two of the sample catchments. Mining was highest at sample catchment BRAND01 which contained 4.23 % mines and lowest at MAGA01 which contained 0.22 % mines. Analysis of land use maps revealed degraded forest, improved grassland, forest plantations, cultivated, mines and urban land use as the anthropogenic land uses which may contribute to the water quality of FEPA sample catchments.

4.3. Exploratory Data Analysis - Water Chemistry

4.3.1. Water Chemistry: Descriptive Statistics

Descriptive statistics were calculated for all water quality parameters measured at each sample catchment in order to summarize the key features of the dataset. The calculation of descriptive statistics for the dataset was important in order to understand the data and present it in a meaningful way. In doing so, patterns in the dataset could be identified. The tables below describe the minimum, maximum, mean (or average) and standard deviation values that were calculated for all water quality parameters. For pH, the geometric mean was calculated. The anti-log of pH was calculated prior to calculation of mean and standard deviation values for pH.

The mean value can be described as the most common measure of central tendency. The standard deviation can be explained as the positive square root of the variance. It is the average distance between the observed values and the mean. Annual and seasonal mean values were also calculated for water chemistry in order to investigate seasonal patterns in the dataset.

Table 7: Skeerpoort Outflow - Descriptive Statistics

Skeerpoort Outflow										
	Unit	**n**	**Min**	**Max**	**Mean**	**Mean (S)**	**Mean (A)**	**Mean (W)**	**Mean (Spr.)**	**Standard Deviation**
Nitrate	mg/L	8	0.03	1	0.55	0.37	0.71	0.8	0.7	0.31
Chloride	mg/L	8	0.7	3.5	1.96	2.4	1.6	1.7	3.5	0.83
Sulphate	mg/L	8	0	2	1	0.5	1.5	1	1	0.76
Phosphate	mg/L	8	0	0.5	0.08	0.02	0.01	0.03	0.5	0.17
Temperature	°C	9	10	21.8	17.67	20.5	19.55	10	18.33	3.6
DO	mg/L	9	7.27	13.45	8.65	7.59	8.28	13.45	7.86	1.89
EC	µS/cm	9	284.1	553	414.44	304.8	512	450.9	381.83	111.96
pH	pH units	9	7.68	8.22	7.91	7.8	8.05	8.15	7.86	0.22

S = summer; A = autumn; W = winter; spr. = spring; n = number of samples

Table 8: SK01 - Descriptive Statistics

SK01										
	Unit	**n**	**Min**	**Max**	**Mean**	**Mean (S)**	**Mean (A)**	**Mean (W)**	**Mean (Spr.)**	**Standard Deviation**
Nitrate	mg/L	9	0	0.02	0.01	0.01	0.02	0	0.01	0.01
Chloride	mg/L	9	0.7	15	3.61	2.15	3.45	1.5	6.10	4.42
Sulphate	mg/L	9	0	60	10.44	0	0	0	31.33	21.72
Phosphate	mg/L	9	0	0.13	0.04	0.04	0.75	0.02	0.02	0.04
Temperature	°C	9	7.1	25.6	17.39	20.8	17.45	7.70	21.53	6.39
DO	mg/L	9	5.01	9.12	6.45	5.72	5.75	8.54	6.03	1.33
EC	µS/cm	9	404.4	953	608.87	455.7	770.5	568.5	630.13	184.71
pH	pH units	9	7.56	8.21	7.77	7.63	7.77	7.91	7.88	0.20

S = summer; A = autumn; W = winter; spr. = spring; n = number of samples

Table 9: BUFF01 - Descriptive Statistics

BUFF01										
	Unit	**n**	**Min**	**Max**	**Mean**	**Mean (S)**	**Mean (A)**	**Mean (W)**	**Mean (Spr.)**	**Standard Deviation**
Nitrate	mg/L	9	0	0.01	0.01	0.01	0.01	0.01	0.01	0.01
Chloride	mg/L	9	1.1	3.3	1.79	1.5	2.5	1.2	1.9	0.68
Sulphate	mg/L	9	0	1	0.11	0	0	0	0.33	0.33
Phosphate	mg/L	9	0	0.05	0.03	0.02	0.04	0.02	0.03	0.02
Temperature	°C	9	9	23.7	18.16	22.45	20	9.85	19.6	5.37
DO	mg/L	9	6.5	10.44	8.24	7.88	7.91	9.99	7.55	1.20
EC	µS/cm	9	46.1	112.9	73.6	47.75	95.95	68.7	79.2	24.86
pH	pH units	9	6.7	7.32	7.07	7.01	7.06	7.22	7.16	0.19

S = summer; A = autumn; W = winter; spr. = spring; n = number of samples

Table 10: STERK01 - Descriptive Statistics

STERK01										
	Unit	**n**	**Min**	**Max**	**Mean**	**Mean (S)**	**Mean (A)**	**Mean (W)**	**Mean (Spr.)**	**Standard Deviation**
Nitrate	mg/L	9	0.04	0.1	0.07	0.05	0.07	0.08	0.07	0.02
Chloride	mg/L	9	1.3	4.7	2.03	1.6	3.05	1.6	1.93	1.03
Sulphate	mg/L	9	0	1	0.33	0.5	0	0.5	0.33	0.5
Phosphate	mg/L	9	0	0.04	0.03	0.02	0.04	0.03	0.03	0.01
Temperature	°C	9	11.7	23.4	18.86	21.9	20.5	12.15	20.2	4.22
DO	mg/L	9	5.57	8.49	7.20	7.2	7.34	7.77	6.72	0.84
EC	µS/cm	9	41.7	85	60.24	43.80	82.45	61.65	56.47	17.29
pH	pH units	9	6.71	7.43	7.09	6.99	7.15	7.39	7.11	0.24

S = summer; A = autumn; W = winter; spr. = spring; n = number of samples

Table 11: STERK02 - Descriptive Statistics

STERK02										
	Unit	**n**	**Min**	**Max**	**Mean**	**Mean (S)**	**Mean (A)**	**Mean (W)**	**Mean (Spr.)**	**Standard Deviation**
Nitrate	mg/L	8	0	0.06	0.02	0.02	0.02	0.01	0.03	0.02
Chloride	mg/L	8	0.6	2.3	1.44	0.9	1.2	1.55	1.7	0.52
Sulphate	mg/L	8	0	1	0.13	0	0.5	0	0	0.35
Phosphate	mg/L	8	0	0.14	0.04	0	0.03	0.04	0.07	0.01
Temperature	°C	9	12	22.1	17.57	19.6	19.05	12.2	18.8	3.53
DO	mg/L	9	3.89	7.56	6.35	7.51	6.17	6.55	5.58	1.19
EC	µS/cm	9	17.2	47.4	34.14	18.3	43.25	38.4	35.8	11.12
pH	pH units	9	6.29	7.16	6.63	6.66	6.65	6.93	6.63	0.29

S = summer; A = autumn; W = winter; spr. = spring; n = number of samples

Table 12: STERK03 - Descriptive Statistics

STERK03										
	Unit	**n**	**Min**	**Max**	**Mean**	**Mean (S)**	**Mean (A)**	**Mean (W)**	**Mean (Spr.)**	**Standard Deviation**
Nitrate	mg/L	6	0.01	0.03	0.01	0.01	0.01	_	0.02	0.01
Chloride	mg/L	5	0.6	1.6	1.02	0.9	0.6	_	1.35	0.38
Sulphate	mg/L	5	0	0	0	0	0	_	0	0
Phosphate	mg/L	5	0.02	0.04	0.03	0.03	0.04	_	0.04	0.01
Temperature	°C	6	19.2	25.4	22.53	23.1	19.7	13.4	21.95	3.00
DO	mg/L	6	5.63	7.95	6.88	6.96	7.54	8.73	6.79	1.11
EC	µS/cm	6	31.5	65	41.05	33.65	55.4	382.8	48.45	16.08
pH	pH units	6	6.46	8.66	6.92	6.97	7.41	8.26	7.83	0.94

S = summer; A = autumn; W = winter; spr. = spring; n = number of samples

Table 13: MAGA01 - Descriptive Statistics

MAGA01										
	Unit	**n**	**Min**	**Max**	**Mean**	**Mean (S)**	**Mean (A)**	**Mean (W)**	**Mean (Spr.)**	**Standard Deviation**
Nitrate	mg/L	7	0.01	0.19	0.09	0.03	_	0.15	0.09	0.06
Chloride	mg/L	6	1.5	2.8	2.08	1.9	_	2	2.35	0.44
Sulphate	mg/L	7	0	0	0	0	_	0	0	0
Phosphate	mg/L	7	0	0.06	0.03	0.01	_	0.03	0.04	0.02
Temperature	°C	7	13.4	20.09	16.99	20.15	_	10.6	16.97	2.63
DO	mg/L	7	6.84	10.11	8.24	7.05	_	7.28	8.77	1.07
EC	µS/cm	7	218.9	426.7	302.27	228.95	_	412.2	292.27	94.84
pH	pH units	7	7.56	8.31	7.84	7.64	_	8	7.9	0.33

S = summer; A = autumn; W = winter; spr. = spring; n = number of samples

Table 14: MAGA02 - Descriptive Statistics

MAGA02										
	Unit	**n**	**Min**	**Max**	**Mean**	**Mean (S)**	**Mean (A)**	**Mean (W)**	**Mean (Spr.)**	**Standard Deviation**
Nitrate	mg/L	6	0	0.01	0.03	0.1	_	0.01	0.01	0.04
Chloride	mg/L	6	11.6	26	17.8	13.35	_	18.3	20.6	4.76
Sulphate	mg/L	7	0	5	2	0	_	4.5	1.67	2.52
Phosphate	mg/L	7	0.02	0.05	0.03	0.04	_	0.04	0.03	0.01
Temperature	°C	6	6.9	21.5	15.09	20.55	_	8.4	16.4	5.46
DO	mg/L	6	1.64	6.29	4.73	4.63	_	6.29	3.86	1.68
EC	µS/cm	6	246.6	579	361.44	274.45	_	433.2	374	122.33
pH	pH units	6	6.98	7.84	7.29	7.16	_	7.84	7.33	0.34

S = summer; A = autumn; W = winter; spr. = spring; n = number of samples

Table 15: BRAND01 - Descriptive Statistics

BRAND01										
	Unit	n	Min	Max	Mean	Mean (S)	Mean (A)	Mean (W)	Mean (Spr.)	Standard Deviation
Nitrate	mg/L	2	0.01	0.01	0.01	0.01	_	_	0.01	0
Chloride	mg/L	2	0.3	0.6	0.45	0.3	_	_	0.6	0.21
Sulphate	mg/L	2	1	4	2.5	1	_	_	4	2.12
Phosphate	mg/L	2	0.02	0.03	0.03	0.02	_	_	0.03	0.01
Temperature	°C	2	16	21.4	18.7	21.4	_	_	16	3.82
DO	mg/L	2	5.54	5.72	5.63	5.72	_	_	5.54	0.13
EC	µS/cm	2	31.4	46.2	38.8	46.2	_	_	31.4	10.47
pH	pH units	2	6.32	6.97	6.53	6.97	_	_	6.32	0.46

S = summer; A = autumn; W = winter; spr. = spring; n = number of samples

4.3.2. Variation in Water Quality

Time plots were created for each water quality parameter in order to identify seasonal variation in data over the sampling period. This also provided information regarding seasonal variations in water quality between FEPA sample sites. The time plots below indicate the findings for water chemistry at each of the nine sample sites. BRAND01 was not included since data was collected for only two months at the site due to the drying up of the stream.

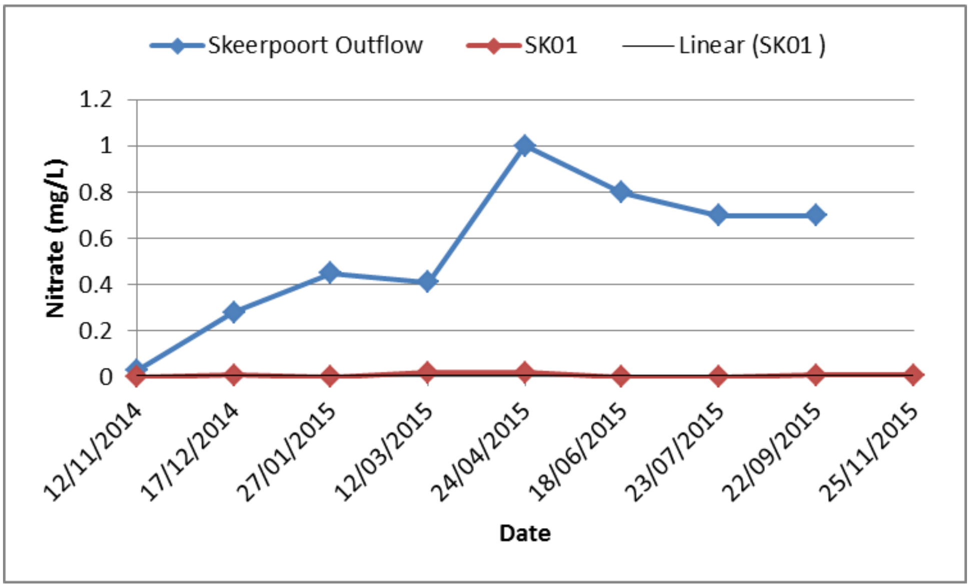

Figure 12: Nitrate versus time at Skeerpoort Outflow and SK01

The above time plot indicates the variation in nitrate concentration exhibited at Skeerpoort Outflow and SK01 on the Skeerpoort river. At Skeerpoort Outflow, nitrate concentration peaked in autumn at 1 mg/L and was lowest in spring at 0.03 mg/L. SK01, however, did not exhibit significant seasonal variation in nitrate levels, ranging between 0 mg/L in spring, summer and winter and reaching a maximum concentration of 0.02 mg/L in autumn. Overall, the result revealed significantly higher nitrate concentrations observed at Skeerpoort Outflow compared to SK01.

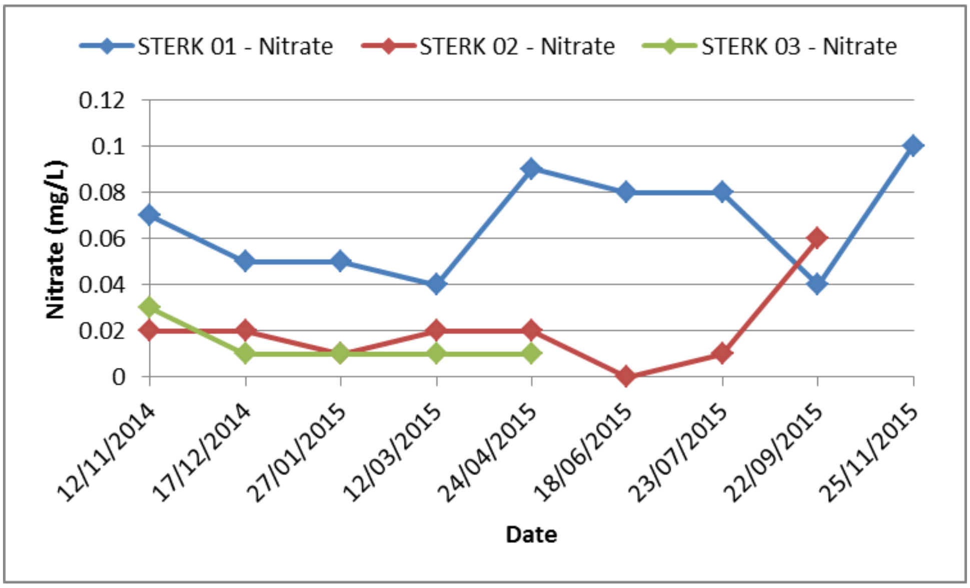

Figure 13: Nitrate versus time at STERK01, STERK02 and STERK03

STERK01 and STERK02 indicated a peak in nitrate levels during spring 2015 at 0.1 mg/L and 0.06 mg/L respectively. Nitrate concentration was highest at STERK01 throughout the sampling period. Increased levels during spring could be attributed to agricultural activities at the sites, since cultivated land comprised a significant proportion of land use at the catchments. Nitrate levels were lowest in spring and autumn at STERK01 at 0.04 mg/L and lowest in winter at STERK02 at 0 mg/L. STERK03 also exhibited the highest nitrate concentration during spring 2014 at 0.03 mg/L, after which nitrate concentration remained constant at 0.01 mg/L during summer, autumn and spring of 2015. The result is confirmed by the comparatively lower percentage of cultivated land at STERK03 sample catchment.

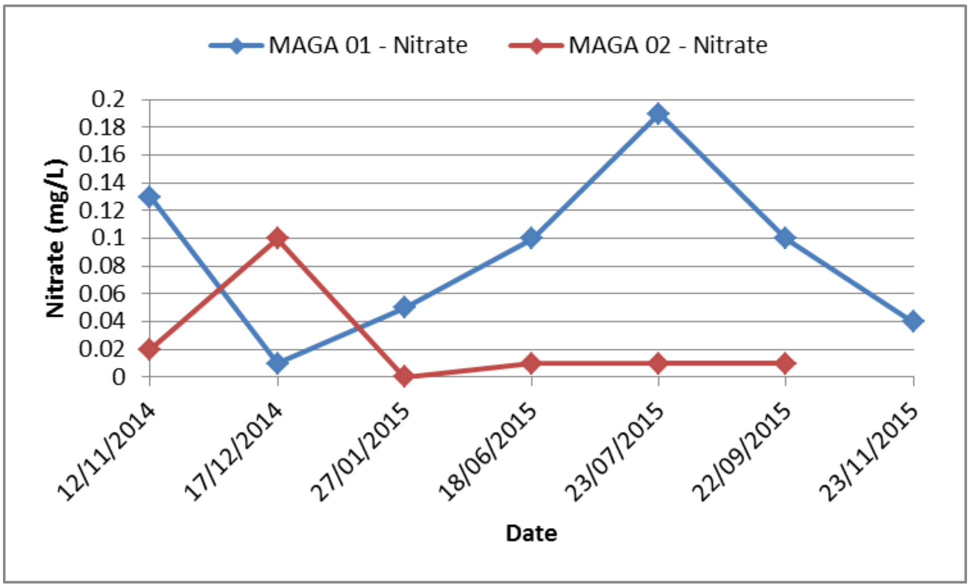

Figure 14: Nitrate versus time at MAGA01 and MAGA02

At MAGA01, nitrate levels were highest during winter (0.19 mg/L) and lowest during summer (0.01 mg/L) while MAGA02 showed peak nitrate levels during summer (0.1 mg/L). Higher nitrate levels were therefore observed at MAGA01 compared with MAGA02.

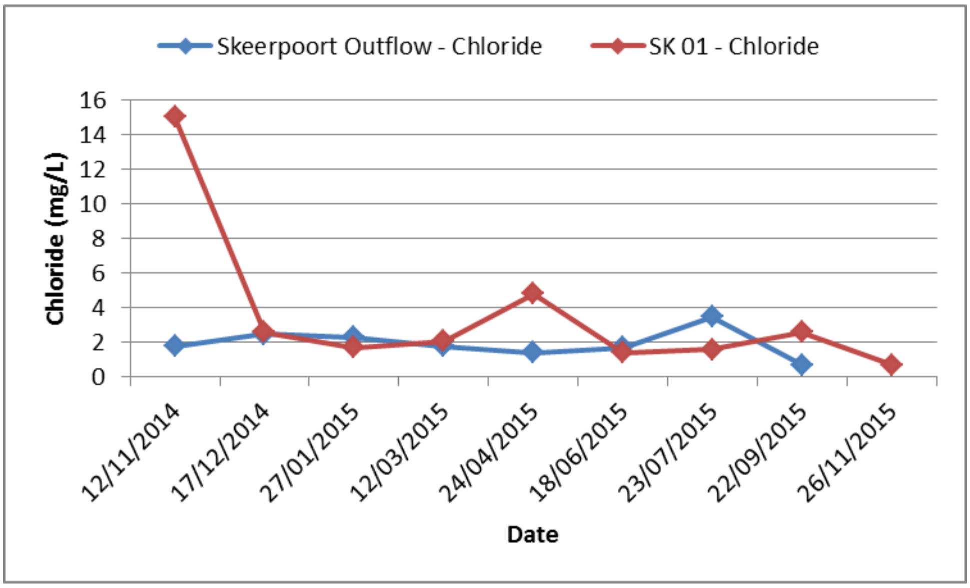

Figure 15: Chloride versus time at Skeerpoort Outflow and SK01

Skeerpoort Outflow did not indicate any significant variation in chloride concentration between seasons, with concentrations ranging between 0.7 mg/L in spring and 3.5 mg/L in winter. At SK01, a high reading of 15 mg/L was observed for chloride during the first month of sampling (spring 2014); however, no clear seasonal pattern was observed over the following seasons. The initial high reading for chloride concentration could be linked to sewage contamination or agricultural runoff. When fertilizers are applied in a single large dose the result is often soil with a high salt content. As such, runoff from agricultural practices in the area may have led to high chloride levels in the receiving stream during spring of the sampling period.

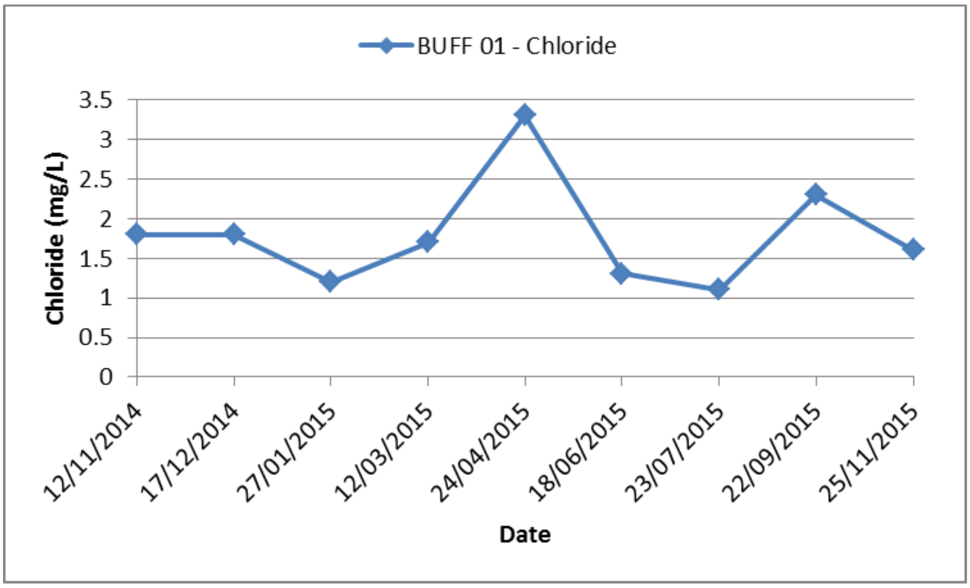

Figure 16: Chloride versus time at BUFF01

Chloride concentration showed an increase during autumn (April) of the sampling period, reaching a maximum concentration of 3.3 mg/L. Chloride levels were lowest in summer (January) and winter (June-July).

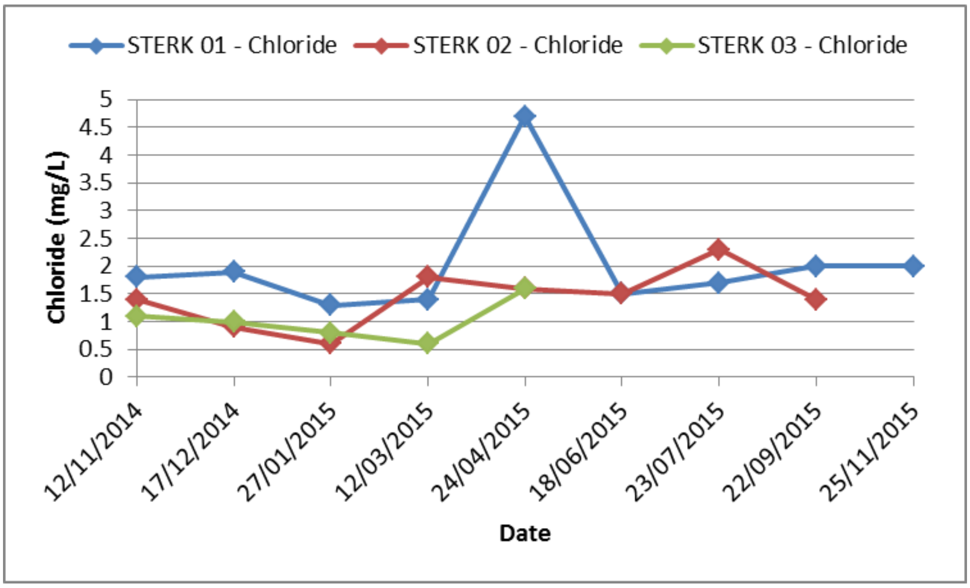

Figure 17: Chloride versus time at STERK01, STERK02 and STERK03

Chloride concentration peaked in autumn at STERK01 and STERK03 and in winter at STERK02, however, STERK01 indicated a significantly higher peak chloride concentration, reaching a maximum of 4.7 mg/L in autumn. At STERK01, chloride concentration was lowest in summer at 1.3 mg/L. STERK02 and STERK03 indicated minimum chloride concentrations in autumn of 0.6 mg/L. Higher concentrations of chloride were therefore observed during the dry season at all three sites. This could be a result of lower dilution during the dry season.

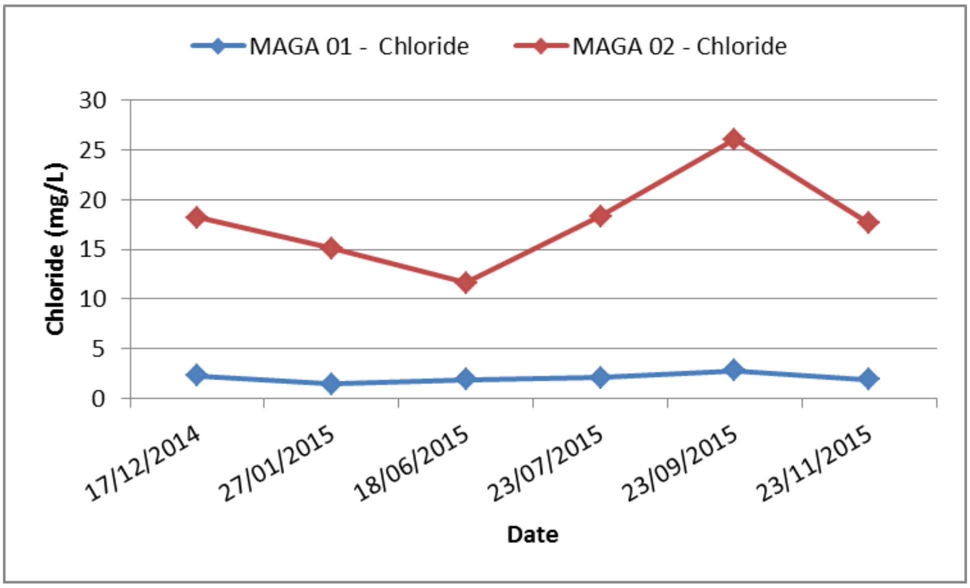

Figure 18: Chloride versus time at MAGA01 and MAGA02

MAGA01 did not display seasonal variation in chloride concentration. At MAGA02, chloride levels were high during spring (26 mg/L) and lowest in winter. MAGA02 clearly exhibited significantly higher chloride concentrations than MAGA01 over the period of sampling. This could be a result of agricultural runoff or sewage pollution within the MAGA02 FEPA catchment.

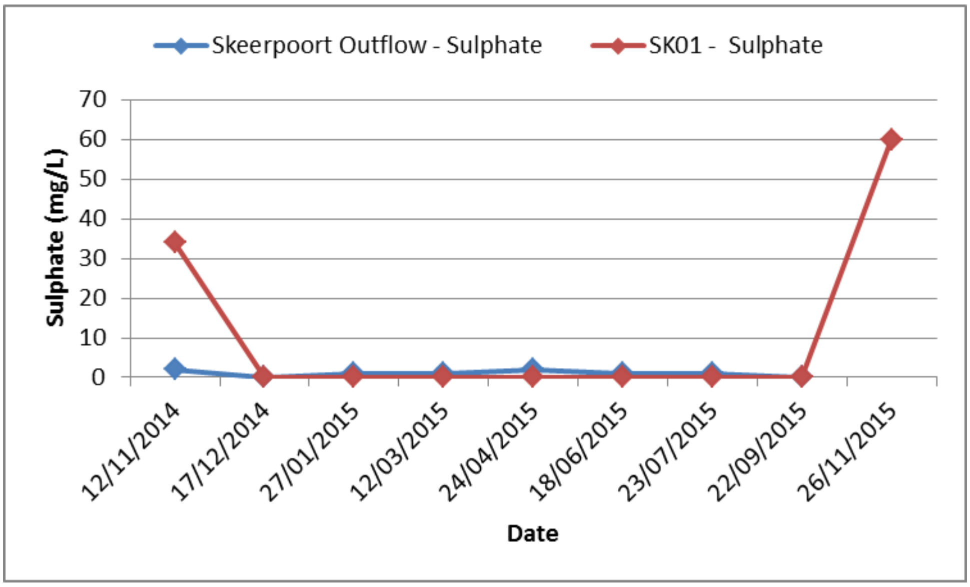

Figure 19: Sulphate versus time at Skeerpoort Outflow and SK01

At Skeerpoort Outflow, sulphate concentration did not exhibit any significant variation between seasons, measuring a maximum concentration 2 mg/L in spring 2014 and autumn and measuring 0 mg/L in summer and spring. SK01, however, exhibited unexpected results with a high sulphate concentration of 60 mg/L in spring 2015. All other measurements revealed a sulphate concentration of 0 mg/L, with the exception of spring 2014 when sulphate concentration was 34 mg/L. The observed spikes in chloride concentration could be attributed to unexpected large discharges of agricultural, mine drainage, urban or industrial effluent into the stream.

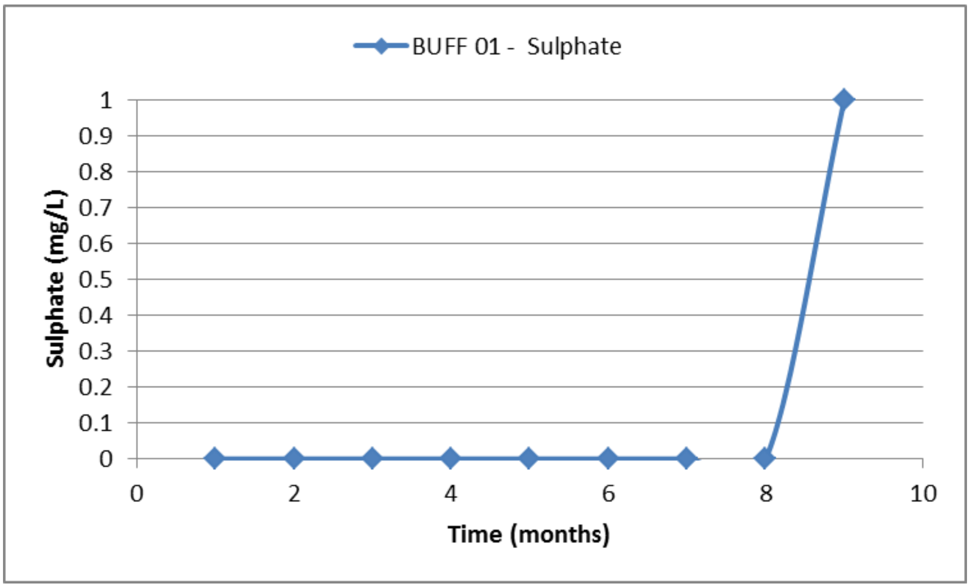

Figure 20: Sulphate versus time at BUFF01

Sulphate clearly did not vary seasonally at BUFF01. The only reading above 0 mg/L was observed during spring when levels reached 1 mg/L. This could be attributed to reduced stream flow which was observed at the time of sampling, as well as inputs from agricultural practices.

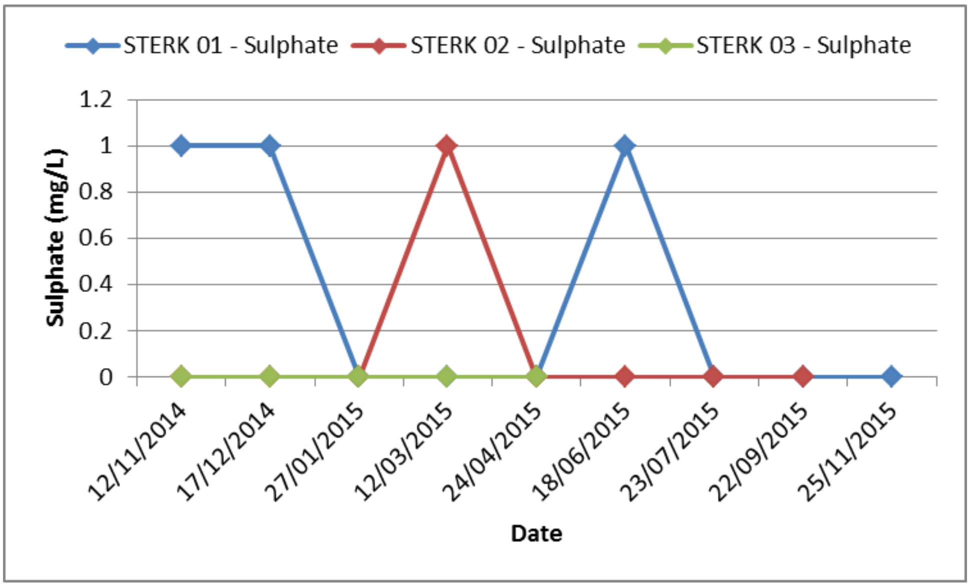

Figure 21: Sulphate versus time at STERK01, STERK02 and STERK03

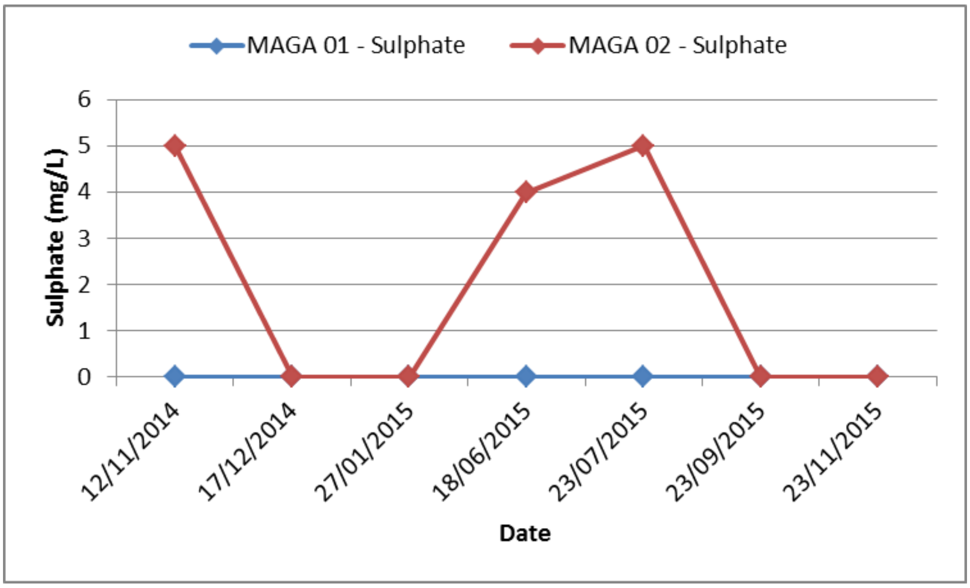

Figure 22: Sulphate versus time at MAGA01 and MAGA02

Sulphate concentration did not sure any clear seasonal patterns at sample sites along the Sterkstroom and Magalies rivers. Sulphate was detectable at some sites and not detectable at others.

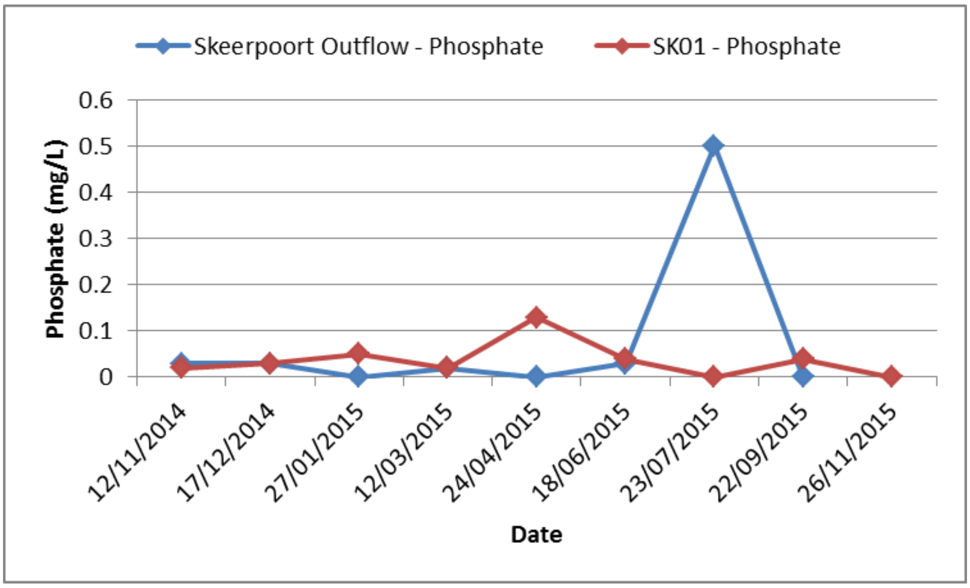

Figure 23: Phosphate versus time at Skeerpoort Outflow and SK01

Phosphate level at Skeerpoort Outflow was 0 mg/L in summer, autumn and spring 2014 and peaked at 0.5 mg/L in spring 2015. SK01 did not show any significant variation in phosphate concentrations over the seasons, however, levels peaked in autumn at 0.13 mg/L.

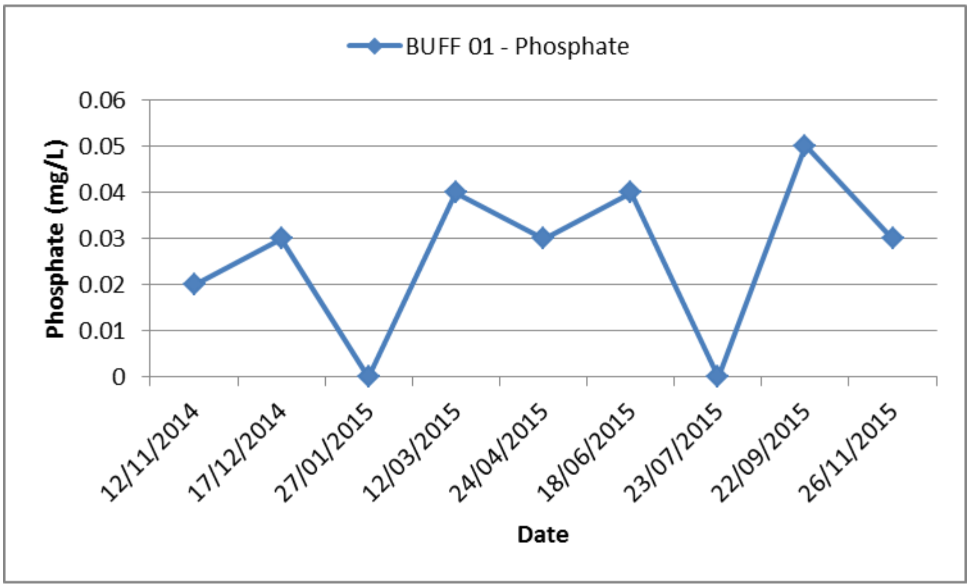

Figure 24: Phosphate versus time at BUFF01

Phosphate concentration showed some fluctuation over the seasons although levels were low, peaking in spring at 0.05 mg/L and measuring 0 mg/L in summer (January) and winter (July).

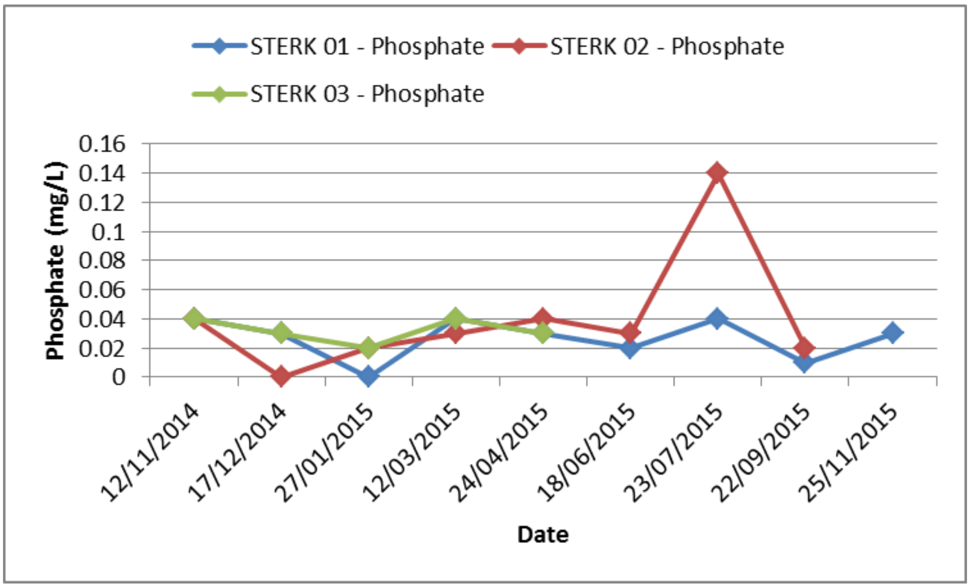

Figure 25: Phosphate versus time at STERK01, STERK02 and STERK03

Maximum phophate concentrations were significantly higher at STERK02, peaking at 0.14 mg/L during winter, while STERK01 and STERK03 did not vary significantly across seasons.

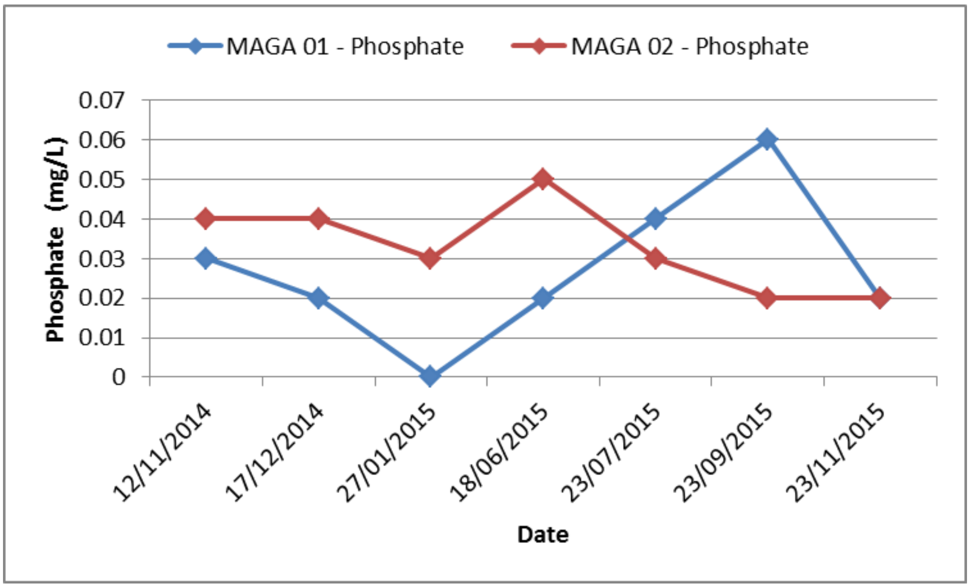

Figure 26: Phosphate versus time at MAGA01 and MAGA02

MAGA01 indicated a maximum phosphate concentration of 0.06 mg/L in spring 2015 while MAGA02 showed a maximum phosphate concentration of 0.05 mg/L in winter. At both sites, phosphate concentration was lowest during summer.

The time plots that were created for temperature can be found in Appendix A. Since temperature followed the climatic conditions expected for the study area, temperature variations over the sampling period were consistent across all sites with minor variations in maximum and minimum temperatures between sites. In summer, temperature was highest at BUFF01 at 23.7 °C and lowest at MAGA01 at 20.9 °C. The highest temperature recorded in autumn was 22.9 °C at STERK01, while the lowest temperature recorded in autumn was 19.8 °C at SK01. Winter temperatures were lowest at MAGA02 at 6.9 °C and highest at MAGA01 at 14.3 °C. Spring temperature was highest at SK01 at 25.6 °C and lowest at MAGA01 at 20.9 °C.

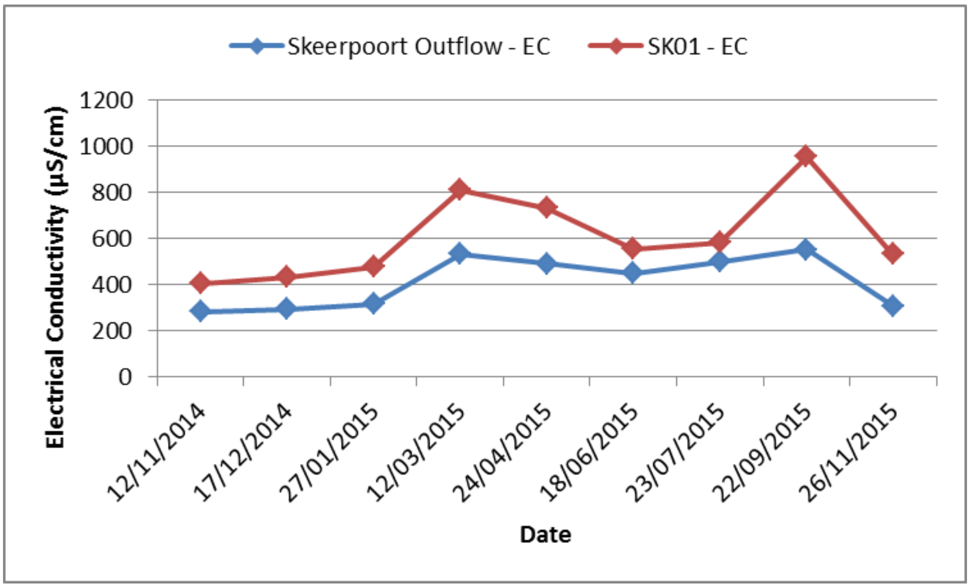

Figure 27: EC versus time at Skeerpoort Outflow and SK01

Variation in EC followed a similar pattern at Skeerpoort Outflow and SK01, with EC levels peaking in spring 2015 and lowest in spring 2014. SK01, however, exhibited higher EC levels throughout the sampling period, with a maximum value of 953 µS/cm and a minimum value of 404.4 µS/cm during the first month of sampling. At Skeerpoort Outflow, EC level was highest at 553 µS/cm and lowest at 284.2 µS/cm. Discharges from industrial or agricultural practices could be a potential source of increasing EC levels during autumn and spring at the sample sites.

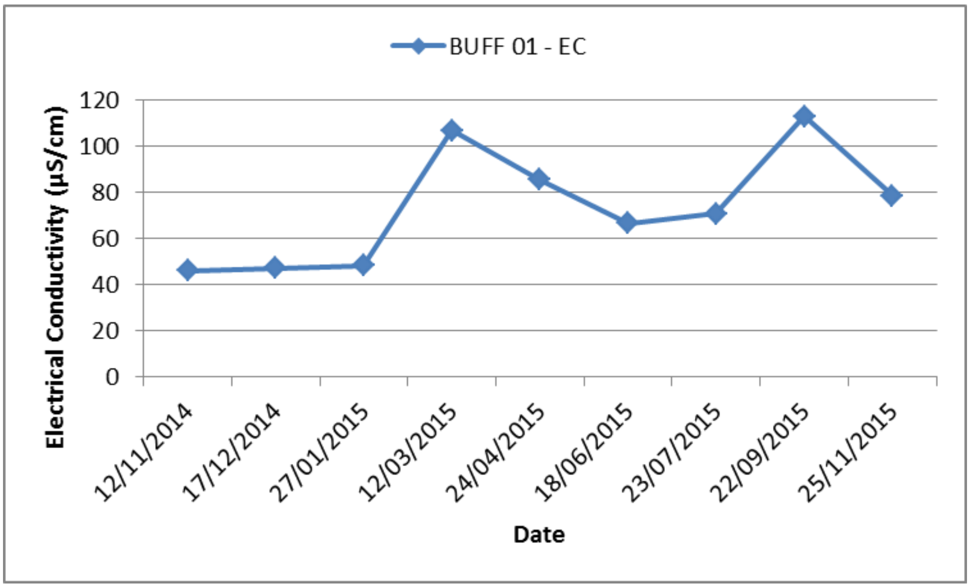

Figure 28: Electrical Conductivity versus time at BUFF01

BUFF01 appeared to show a maximum EC level of 112.9 µS/cm in spring 2015 (September) and a minimum EC level of 47.1 µS/cm in spring 2014 (November). Overall, EC levels were higher during autumn and spring and lower during summer and winter. The pattern observed is similar to that observed at Skeerpoort Outflow and SK01, however, overall EC levels were significantly lower.

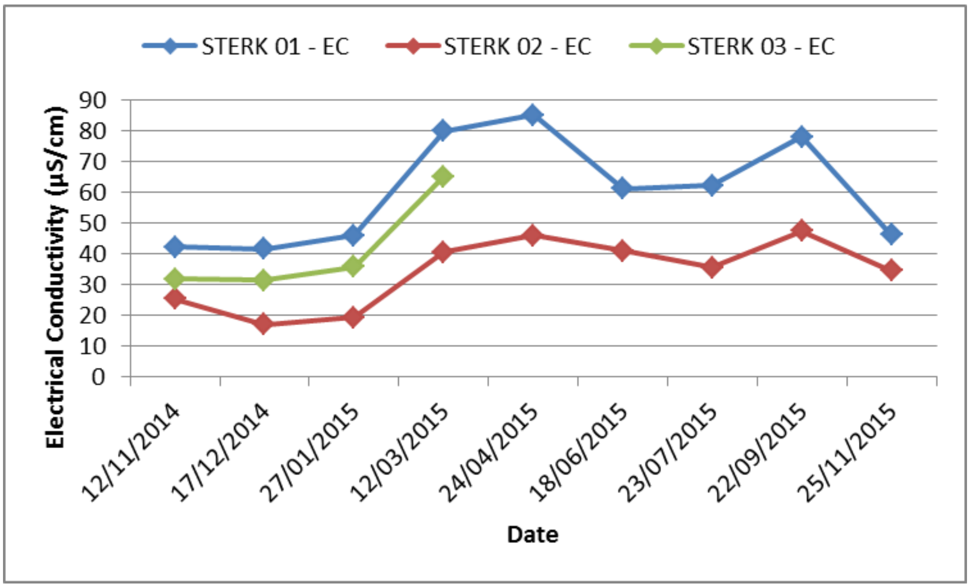

Figure 29: Electrical Conductivity versus time at STERK01, STERK02 and STERK03

Overall, STERK01, STERK02 and STERK03 also showed significantly lower EC levels compared with that of Skeerpoort Outflow and SK01. The pattern of EC levels showed increases during autumn and spring 2015, with STERK01 indicating higher EC levels and

STERK03 indicating the lowest levels of EC (31.5 µS/cm) across all three sites. Only four measurements were recorded for STERK03 due to the site drying up during the remainder of the sampling campaigns. The mean value for EC is therefore not strictly comparable since it represents different periods. STERK02 clearly displayed lower EC levels for the common sampling period.

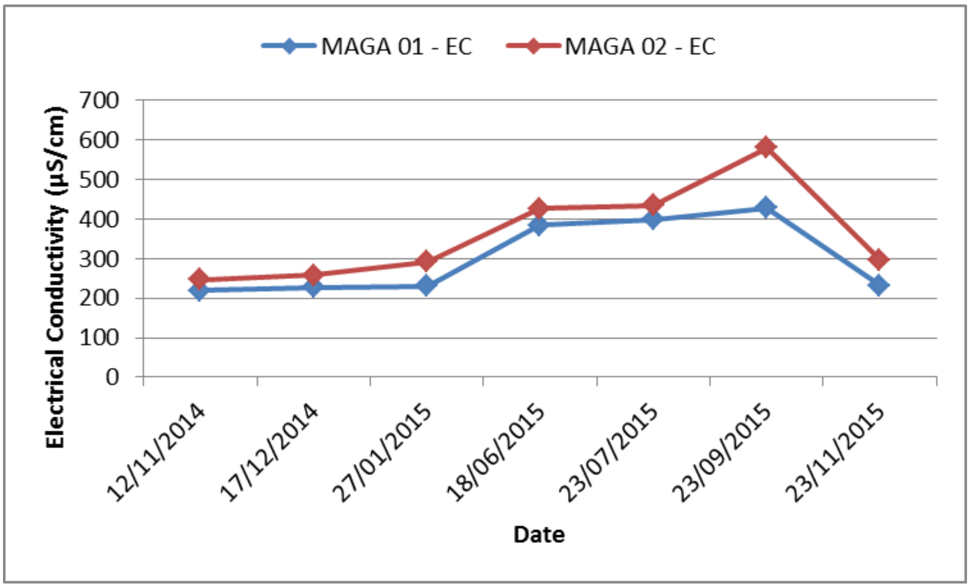

Figure 30: Electrical Conductivity versus time at MAGA01 and MAGA02

EC level reached a high level at MAGA02 during spring 2015 at 579 µS/cm and showed higher EC levels than MAGA01 across all sampling seasons. Although Skeerpoort Outflow exhibited the highest EC levels, EC levels at MAGA02 were also notably higher than that of BUFF01, STERK01, STERK02 and STERK03. EC was lowest in spring 2014 at MAGA01 and MAGA02.

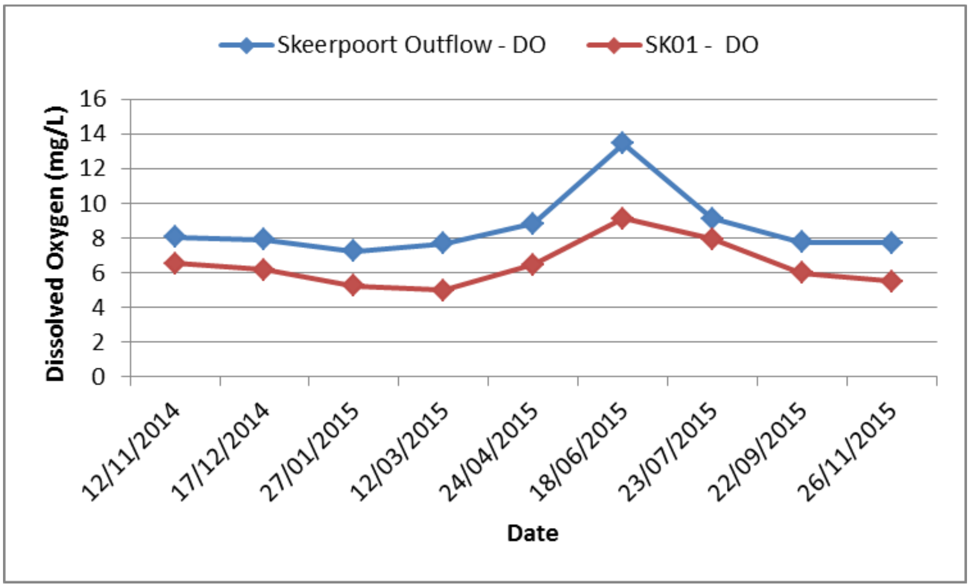

Figure 31: Dissolved Oxygen versus time at Skeerpoort Outflow and SK01

Dissolved oxygen was generally higher at Skeerpoort Outflow than SK01 across all seasons. At both sites, DO peaked during winter with a maximum concentration of 13.45 mg/L at Skeerpoort Outflow and 9.12 mg/L at SK01. DO concentration was lowest during summer at Skeerpoort Outflow (7.27 mg/L) and lowest during autumn at SK01 (5.01 mg/L).

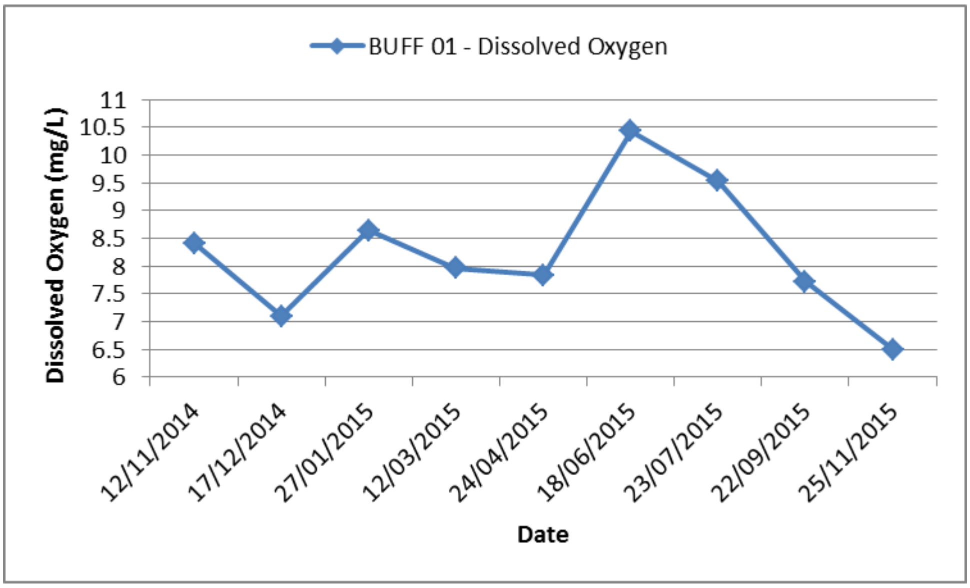

Figure 32: Dissolved Oxygen versus time at BUFF01

Dissolved oxygen varied between 6.5 mg/L in spring 2015 and 10.44 mg/L in winter. Overall, DO levels were observed to be higher during the dry season.

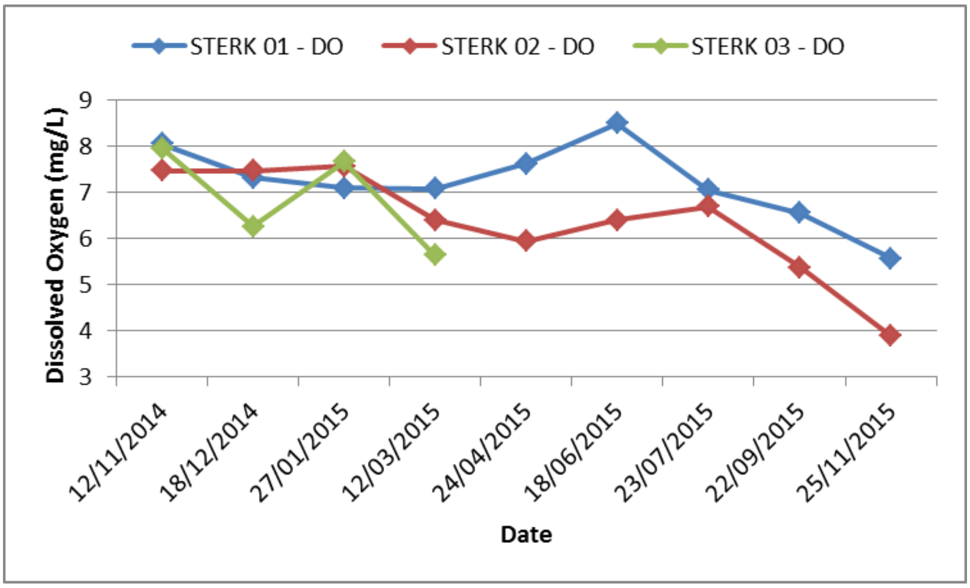

Figure 33: Dissolved Oxygen at STERK01, STERK02 and STERK03

The highest concentration of DO was observed at STERK01 during winter at 8.49 mg/L. DO was lowest at STERK01 during spring 2015 at 5.57 mg/L. STERK02 experienced maximum DO concentration during summer at 7.56 mg/L and DO concentration decreased significantly to 3.89 mg/L in spring 2015. Fewer measurements were available for STERK03 due to the stream drying up, however, from the observations recorded, DO was highest during spring 2014 at 7.95 mg/L and lowest in autumn at 5.63 mg/L.

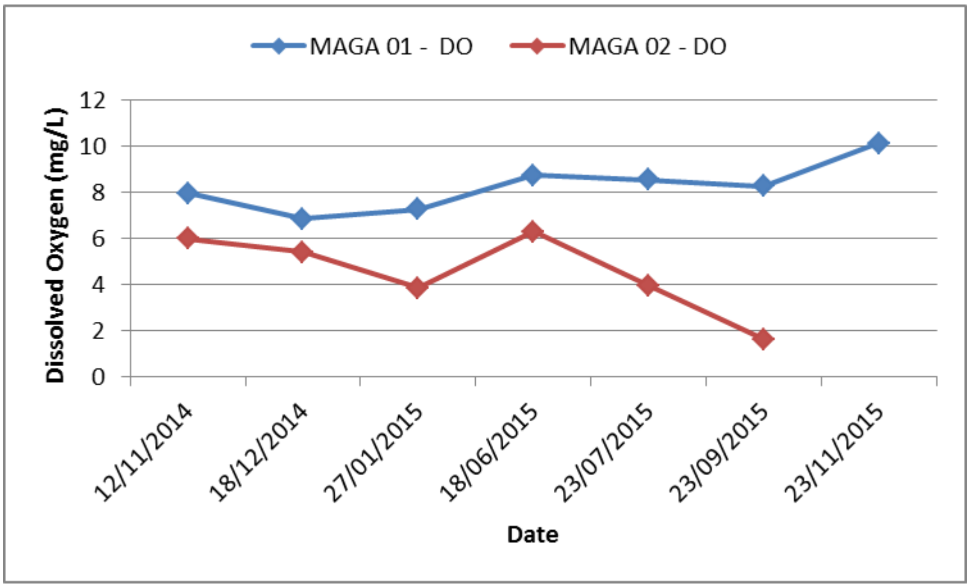

Figure 34: Dissolved Oxygen versus time at MAGA01 and MAGA02

DO levels were higher at MAGA01, peaking at 10.11 mg/L in spring 2015 and lowest in summer at 6.84 mg/L. DO concentration peaked in winter at MAGA02 at 6.29 mg/L and decreased significantly to 1.64 mg/L in spring 2015. The result shows an interesting difference in DO concentration between the two sites, indicating potential pollution sources at MAGA02 during spring. The very low DO concentration could be due to the impacts of storm water runoff from agricultural fields and nutrient enrichment at the sample site.

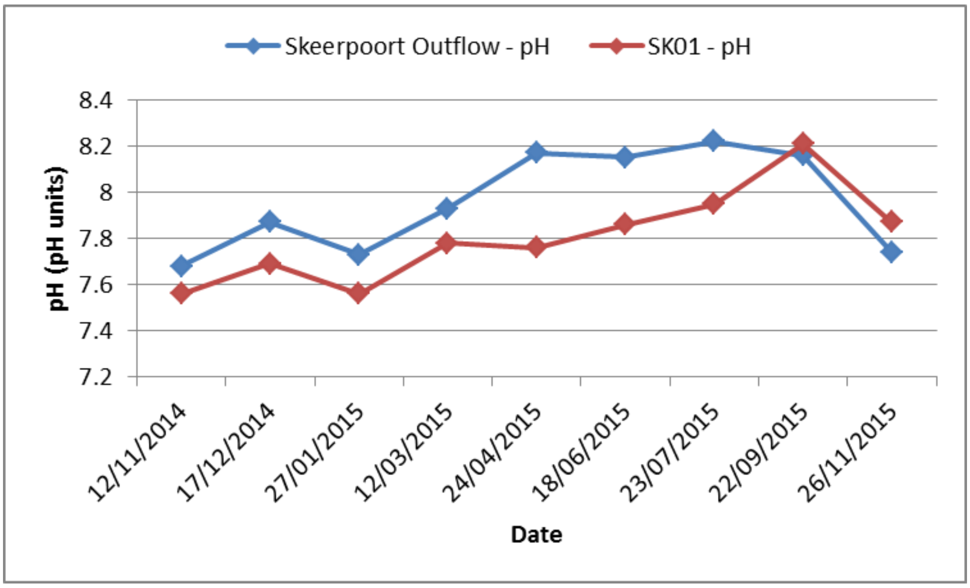

Figure 35: pH over time at Skeerpoort Outflow and SK01

At Skeerpoort Outflow, pH increased from 7.69 pH units in spring 2014 to 8.22 pH units in winter while SK01 showed an increase in pH from 7.56 pH units in spring 2014 to 8.21 pH units in spring 2015. Increasing pH levels can be attributed to increases in temperature and nutrients observed at the sample sites.

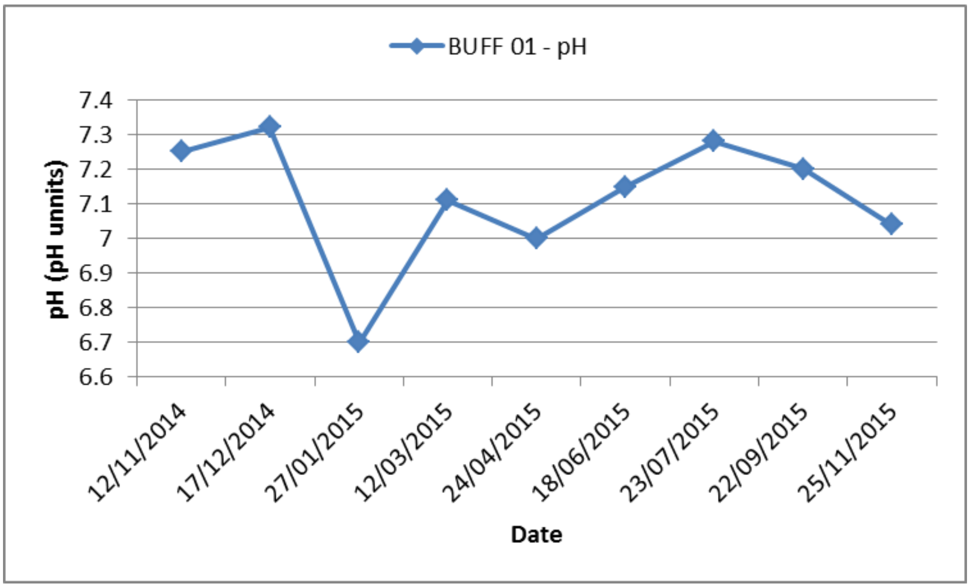

Figure 36: pH versus time at BUFF01

At BUFF01, pH was highest during summer (December 2014) at 7.32 pH units and also lowest in summer (January 2015) at 6.7 pH units. The decrease in pH may be associated with higher temperatures during the month of January.

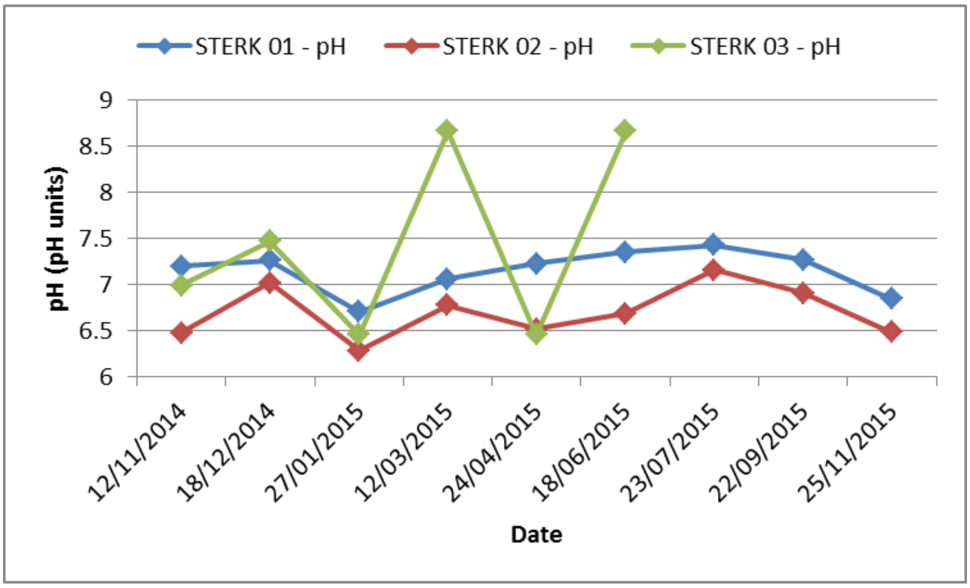

Figure 37: pH versus time at STERK01, STERK02 and STERK03

pH levels were lowest during summer (January) at all three sites, with the lowest pH level of 6.29 pH units measured at STERK02. Overall, pH was higher at STERK01 than STERK02. A significant increase in pH was observed at STERK03 during winter, with levels peaking at 8.66 pH units. This could be a result of the lower water temperatures at the site during winter.

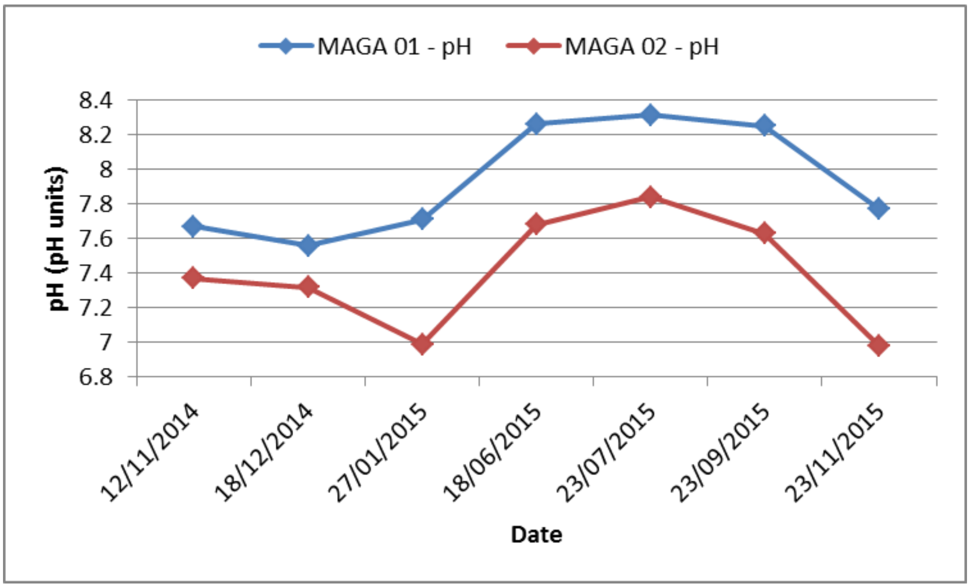

Figure 38: pH versus time at MAGA01 and MAGA02

pH levels at MAGA01 and MAGA02 displayed a similar pattern, with pH levels increasing over time. Maximum pH values for both sites were recorded in winter at 8.31 pH units and

7.84 pH units respectively. At MAGA01, pH was lowest in summer at 7.56 pH units. At MAGA02, pH was lowest in spring 2015 at 6.98 pH units.

4.4. Exploratory Data Analysis: Water Chemistry and Land Use

Scatter plots were generated in order to explore linear relationships between water quality parameters and anthropogenic land use. Mean values for water chemistry were plotted against land use percentages, resulting in a series of scatterplots. While both annual and seasonal mean values were created for water chemistry, only data which indicated the strongest relationships were included for further analysis. Seasonal mean values were used for analysis of linear associations between water chemistry and land use. The specific season selected for each water quality parameter was based on the strength of linear associations identified. As such, the data used for analysis may differ from one parameter to the next, as illustrated below (e.g. mean autumn nitrate showed the strongest linear association with land use while for chloride, mean winter values indicated the strongest linear associations with land use).

Scatter plots were used to identify linear relationships between variables for inclusion in multiple linear regression analysis, discussed later in this chapter. Some of the variables considered for multiple regression analysis are presented below. Water quality variables were plotted as the dependent variables and land uses were plotted as the independent variables. R-square values were generated in Microsoft excel, but only for the purposes of exploratory data analysis. The table below highlights the R-square and p values obtained for the linear correlations explored. Analysis of the scatter plots revealed urban, improved grassland, degraded thicket and forest plantations as key anthropogenic land uses that showed linear associations with mean water chemistry.

The results below indicate the strongest linear relationships observed between mean water chemistry and land use. As a result, the data represented below may differ in season between water quality parameters. For example, nitrate displayed the strongest linear relationship with improved grassland during autumn, therefore the results highlight data for mean autumn nitrate and percentage improved grassland. Mean autumn electrical conductivity showed a statistically significant relationship with percentage urban land use. This is presented graphically in figure 39.

Table 16: Correlations between Water Chemistry and Land Use

Mean Water Chemistry	R	P value
Autumn Nitrate vs % Improved Grassland	0.98	p <.01
Winter Chloride vs % Improved Grassland	0.98	p <.01
Winter Chloride vs % Urban	0.65	p <.113
Winter Sulphate vs % Forest plantations	0.66	p <.102
Autumn Electrical Conductivity vs % Urban	**0.93**	**p <.007**
Autumn Phosphate vs % Improved Grassland	0.65	p <.167
Autumn phosphate vs % Urban	0.71	p <.110
Summer Dissolved Oxygen vs % Improved Grassland	0.65	p <.059
Autumn pH vs % Urban	0.58	p <.222

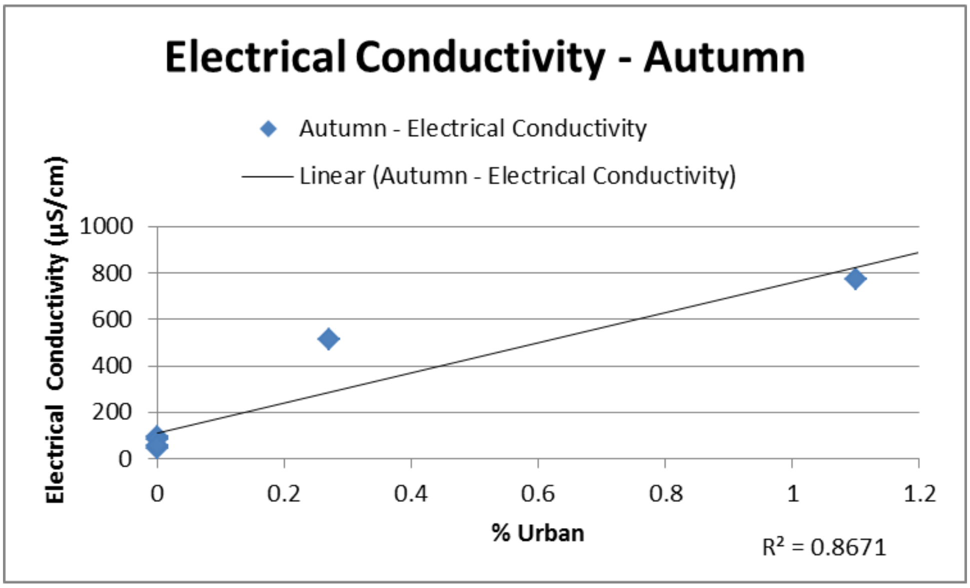

Figure 39: Mean Autumn Electrical Conductivity versus % Urban

Mean autumn EC displayed a moderately strong positive associated with urban land use, p<.007.

4.5. Exploratory Data Analysis: Principal Component Analysis (PCA)

A PCA was run in SPSS statistics version 24 in order to identify the primary axes of chemical variation in the dataset. The analysis was based on annual mean concentrations of water quality parameters, namely, nitrate, chloride, sulphate, phosphate, temperature, electrical conductivity, dissolved oxygen and pH. The assumptions of linearity and sampling adequacy were checked prior to running the analysis to ensure that the dataset was suitable for the application of PCA.

4.5.1. Linearity

The correlations matrix was examined in order to check the assumption of linearity between variables. The purpose of testing for linearity was to determine whether there were any strongly correlated variables in the dataset. The level of correlation considered acceptable for inclusion in PCA is generally $r \geq 0.3$ (Laerd Statistics, 2015). The assumption of linearity was met for the dataset. All variables indicated at least one correlation greater than 0.3 with another variable as indicated in table 17.

Table 17: Correlations Matrix for PCA

Correlation Matrix								
		Nitrate	Chloride	Sulphate	Phosphate	EC	DO	pH
Correlation	Nitrate	1.000	-.108	-.149	.957	.357	.547	.560
	Chloride	-.108	1.000	.127	-.112	.382	-.593	.163
	Sulphate	-.149	.127	1.000	.100	.713	-.308	.332
	Phosphate	.957	-.112	.100	1.000	.493	.470	.579
	EC	.357	.382	.713	.493	1.000	.028	.845
	DO	.547	-.593	-.308	.470	.028	1.000	.485
	pH	.560	.163	.332	.579	.845	.485	1.000

4.5.2. Sampling Adequacy

The Kaiser-Meyer-Olkin (KMO) index and Bartlett's test of sphericity test whether the initial variables can be summarized into a reduced number of factors. The KMO measure for the overall dataset was 0.4. According to Kaiser (1974) classification of measure values, values > 0.6 are suggested as the minimum requirement to meet the assumption of sampling adequacy. A result of 0.4 therefore indicated a relatively low sampling adequacy for the dataset. The KMO measures for individual variables showed similar results, with KMO measures ranging between 0.3 and 0.5. Given the low values obtained using the KMO measure, Bartlett's test of sphericity was applied to determine whether the dataset was suitable for PCA. Bartlett's test of sphericity was statistically significant for the dataset, *p*<.000. This result confirmed that PCA was an appropriate statistical technique for the dataset.

4.5.3. Communalities

Communality can be described as the proportion of the variance of each variable that is accounted for by the principle components analysis. It can also be expressed as a percentage. Table18 includes the communality values obtained for each water quality variable.

Table 18: PCA Communalities

Communalities		
	Initial	Extraction
Nitrate	1.000	.829
Chloride	1.000	.523
Sulphate	1.000	.624
Phosphate	1.000	.824
EC	1.000	.976
DO	1.000	.802
pH	1.000	.831
Extraction Method: Principal Component Analysis.		

4.5.4. Extracting Components

The number of components produced by a PCA is equal to the number of variables present in the original dataset. PCA therefore produced seven components. However, the purpose of using PCA for exploratory data analysis was to explain as much of the variability in the dataset using as few components as possible. Two criteria were used to identify the main axes of variation in the dataset and retain the most important components for further analysis and interpretation, namely, the cumulative percentage of variance explained by a prescribed number of components and inspection of the scree plot. The proportion of variance explained by each component as well as its contribution to the total variance is presented in table 19 under the ‘Initial Eigenvalues’ column below.

Table 19: Total Variance Explained

Total Variance Explained									
Component	Initial Eigenvalues			Extraction Sums of Squared Loadings			Rotation Sums of Squared Loadings		
	Total	% of Variance	Cumulative %	Total	% of Variance	Cumulative %	Total	% of Variance	Cumulative %
1	3.238	46.257	46.257	3.238	46.257	46.257	3.147	44.957	44.957
2	2.171	31.015	77.273	2.171	31.015	77.273	2.262	32.316	77.273
3	.945	13.499	90.772						
4	.582	8.319	99.091						
5	.051	.732	99.823						
6	.009	.135	99.958						
7	.003	.042	100.000						
Extraction Method: Principal Component Analysis.									

An eigenvalue is a measure of the variance explained by each component. The first component explained 3.24 eigenvalues of variance, which was 46.26 % of the total variance explained. The first criterion was based on the retention of all components that could explain at least 60 % or 70 % of the total variance i.e. the cumulative percentage explained by a specific number of components. Using the higher percentage of 70 % led to the retention of the first two components which explained 77.27 % of total variance cumulatively. A scree plot is a line graph which plots the total variance explained by each component (i.e. its eigenvalue) against its respective component. Since there are as many components as there are variables, the scree plot below contained eight components. Generally, components before the last inflection point on the graph are selected for retention. By visual inspection, it was decided that the first two components would be retained. Taking into account the cumulative percentage of variance explained and the result of the scree plot, it made theoretical sense to extract the first two components for further analysis.

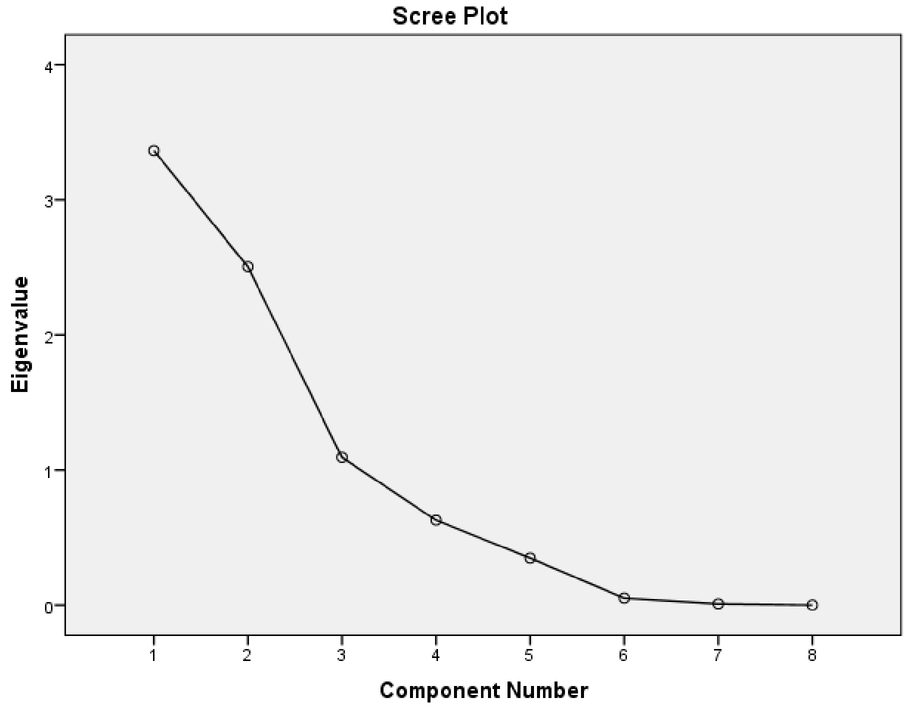

Figure 40: Scree Plot for PCA

A Varimax rotation with Kaiser Normalization was applied to the data. Application of the Varimax rotation yielded an increase in the weighting of higher loadings and a decrease in the weighting of lower loadings, allowing for the results to be more easily interpreted (Gordo, et al., 2014). Below is the rotated component matrix that was produced by the PCA.

Table 20: Rotated Component Matrix for PCA using Varimax Rotation

Rotated Component Matrix[a]		
	Component	
	1	2
Nitrate	**.906**	
Phosphate	**.905**	
pH	**.783**	.466
DO	**.741**	-.503
EC	.535	**.830**
Sulphate		**.788**
Chloride		**.684**

Extraction Method: Principal Component Analysis.
Rotation Method: Varimax with Kaiser Normalization.

a. Rotation converged in 3 iterations.

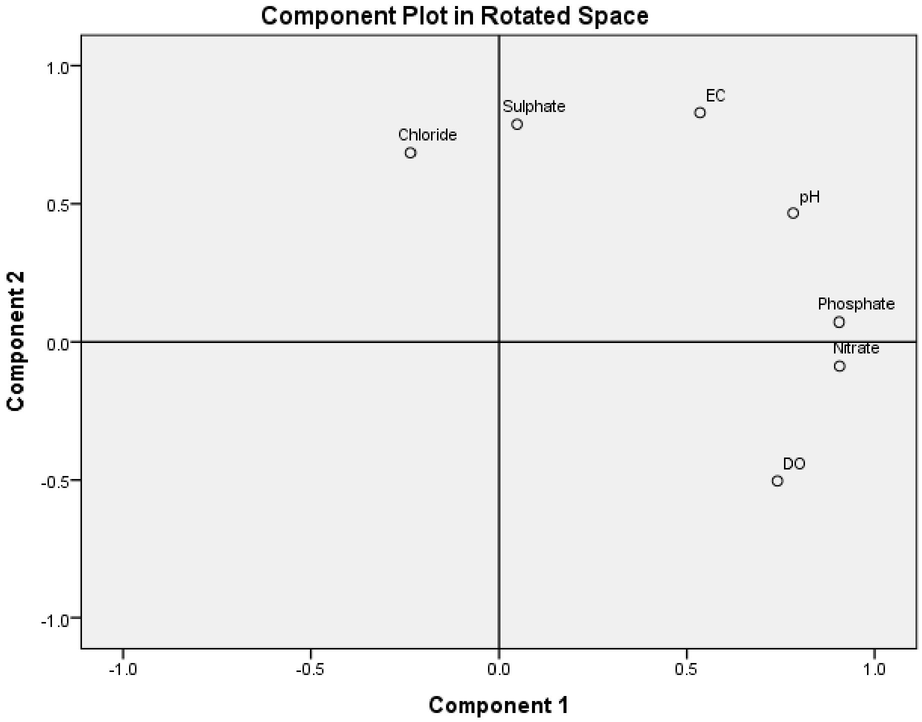

Figure 41: Biplot for PCA

The rotated components table and Biplot describe how the retained, rotated components 'load' on each variable and represents the key output of the PCA. In other words, the components comprise the data and the loadings report how each variable is related to the components. Principle Component one (PC1) accounted for 46.26 % of the total variance explained. It comprised nitrate, phosphate, pH, DO, and EC. EC loaded on component one and component two, however, loadings were significantly higher on the second component. Nitrate and phosphate displayed the highest factor loadings of 0.906 and 0.905 respectively. High loadings of nitrate and phosphate were indicative of anthropogenic sources of water quality such as industrial, domestic and cultivated land use. PC1 further indicated urban sources which are often associated with an increase in nitrate and phosphate in surface waters. High loadings of pH and DO could be associated with urban, farming and mine runoff. PC1 was therefore indicative of the land use classes' urban, cultivated, improved grassland and forest plantations.

Principle component two (PC2) accounted for 31.02 % of the total variance explained and comprised EC, sulphate and chloride, with EC displaying the highest loading of 0.830. High loadings of EC may be indicative of saline industrial effluent and domestic effluents. Surface runoff (urban, industrial or agricultural) as well as sewage effluent could be associated with high EC levels. Significant loadings of sulphate (0.788) and chloride (0.684) can be associated with anthropogenic pollution drivers such as industrial and domestic effluents and domestic fertilizers and sewage effluent often associated with urban land use activities. Furthermore, sulphate is a key indicator of acid mine drainage. PC2 was associated with urban, improved grassland and cultivated land uses. The results of the PCA clearly confirmed anthropogenic land uses as potential sources of pollution within river FEPA sample catchments. As a result, the potential land use sources of water quality revealed by the PCA included cultivated, improved grassland, forest plantations, degraded thicket, mines and urban land use.

4.6. Multiple Linear Regression

Multiple linear regression (MLR) was carried out in order to predict a continuous dependent variable through the analysis of multiple independent variables. A multiple linear regression was therefore used to determine whether anthropogenic land uses predicted water quality at river FEPA sample catchments. Furthermore, multiple regression analysis was applied to determine the overall fit (variance explained) of the model and the relative contribution of the predictor (land use) variables to the total variance explained.

A MLR model was generated for the water quality parameters nitrate, chloride, sulphate, phosphate, electrical conductivity, and dissolved oxygen using the relevant seasonal mean values for water chemistry. Temperature was not included in MLR analysis since it was determined not to be relevant to the study. Furthermore, regression analysis did not yield significant results for pH. It was therefore assumed that land use did not contribute significantly to pH levels at FEPA sample catchments. Based on the results of the scatterplots and the PCA, the anthropogenic land use predictor variables selected for MLR analysis were improved grassland, forest plantations, degraded thicket, cultivated, urban and mines for all models. Water quality parameters were the dependent variables and land use classes were the independent variables or predictor variables.

The dataset was tested for the assumptions of independence of observations, linearity, multicollinearity, unusual points and normality prior to analysis by multiple regressions. ***Independence of observations*** in a regression analysis refers to the presence of first-order autocorrelation in a dataset. In order for a multiple regression analysis to proceed, it was necessary to ensure that observations were not strongly correlated to each other. The Durbin-Watson statistic was run as part of the MLR procedure to test for the assumption of independence of observations. ***Linearity*** was tested in order to establish whether there was a linear relationship between the dependent variable and each independent variable.

This was achieved through the analysis of partial regression plots that were generated as part of the multiple regression procedure. ***Multicollinearity*** may occur when there is a high correlation between two or more independent variables. This can lead to difficulties when interpreting which independent variable contributed to the variance in the dependent variable. To ensure there were no problems of collinearity in the datasets, Tolerance and Variance Inflation Factor (VIF) values were generated and inspected to ensure that all independent variables exhibited Tolerance values greater than 0.1 (or VIF values less than 10).

Unusual points or outliers can be described as data points that do not fit the general pattern of the dataset. As part of the regression analysis, casewise diagnostics statistics were requested for the purpose of identifying potential outliers in the dataset. SPSS statistics was instructed to treat cases or observations with standardized residuals greater than ± 3 standard deviations as outliers. Casewise diagnostics was therefore used to highlight any cases where that cases' standardized residual was greater than ± 3 standard deviations. SPSS statistics produces a 'casewise diagnostics' table indicating any observations with standardized residuals greater than the requested ±3 standard deviations only. Since all cases contained in the dataset met the requirements of having standardized residuals less than ±3 standard deviations, no statistics were produced in SPSS output. This meant that there were no significant outliers present in the dataset. In order to run inferential statistics, the residuals (errors in prediction) of the dataset need to be normally distributed. For the purposes of this study, a partial regression plot (P-P plot) was used to test for ***normality*** of the dataset.

Following the testing of assumptions, the goodness of fit of each model was interpreted. The multiple correlation coefficient, R, represents the Pearson correlation coefficient between the values predicted by the regression model (i.e. predicted values) and the actual values for mean water chemistry (the dependent variable). The multiple correlation coefficient can provide an indication of the model fit, with a value ranging between -1 and +1 where the higher the values the stronger the linear association. An R value of zero indicates no linear relationship between the dependent variable and the independent variable. The total variation explained by the model can be determined by the R square and adjusted R square values. Adjusted R-square adjusts the statistic by taking into account the number of independent variables in the model. It is therefore the more appropriate value used to explain the fit of the model since more than one predictor was used in the analysis. A 'Model Summary' table was generated for each MLR model for interpretation of the model fit.

The F-ratio, a ratio of the explained variance to the unexplained variance, was used to test the overall significance of the fitted models. The F-statistic was applied to test whether the regression model for a particular dependent variable was significant or not through the Analysis of Variance (ANOVA). Where a model was significant, it was assumed that there was a linear relationship between the dependent variable and the predictor variables. The final step in the MLR analysis involved interpretation of the slope coefficients and relative contribution of each land use variable to the overall prediction of water chemistry within river FEPA sample catchments.

4.6.1. MLR Model 1: Autumn Nitrate

A multiple regression analysis was carried out to further explore linear relationships identified between mean autumn nitrate levels and land use. The data used for nitrate included mean values for the autumn months March and April 2015. The criteria used in selecting autumn values for regression analysis was based on the fact that exploratory data analysis identified the most significant relationships between nitrate and land use during the autumn months of sampling. Multiple regression analysis including all six of the selected anthropogenic land uses indicated multicollinearity between degraded thicket, forest plantations and cultivated land use. Furthermore, the regression analysis excluded mines based on the fact that no correlations were identified between mines and nitrate concentration. As a result, the regression analysis was re-applied to include improved grassland and urban land use as the predictor variables. MLR output for the final model is presented below in the form of tables and charts. The data was screened for violation of assumptions prior to analysis.

4.6.1.1. Autumn Nitrate: Assumptions

The dataset met the assumption of independence of observations, as assessed by a Durbin-Watson statistic of 2.629. Furthermore, analysis of partial regression plots revealed approximately linear relationships between nitrate and improved grassland and urban land use. While mean autumn nitrate was positively associated with urban land use, the correlation was less strong. Collinearity statistics confirmed that there were no problems with collinearity in the dataset since all tolerance values were greater than 0.1 (VIF less than 10). With respect to normality, analysis of a partial regression plot of standardised residuals for mean autumn nitrate showed that residuals were close enough to normal to proceed with multiple linear regression analysis.

4.6.1.2. Autumn Nitrate: Descriptive Statistics

Descriptive statistics were generated in SPSS in order to explain the basic features of the data, as presented below. There were six observations in the sample. The spread of data appeared approximately normal, with urban land use showing a higher mean value (0.23 %) compared to improved grassland (0.05 %). Urban land use also showed greater variability in data with a higher standard deviation.

Table 21: Autumn Nitrate - Descriptive statistics

Descriptive Statistics			
	Mean	Std. Deviation	N
Nitrate	.1400	.28014	6
Urban	.2283	.44047	6
Improved_Grass	.0500	.12247	6

4.6.1.3. Autumn Nitrate: MLR Model Fit

The model summary table below provides a summary of the overall regression model fit. An R-square value of 0.997, indicated that improved grassland and urban land use explained 99.7 % of the variability in mean autumn nitrate concentrations, compared to the mean model. Adjusted R-square was 0.989. Therefore, R square for the overall model was 99.7 % with an adjusted R-square of 98.9 %, a large effect size according to Cohen (1988).

Table 22: Autumn Nitrate - MLR Model Summary

Model Summary[b]					
Model	R	R Square	Adjusted R Square	Std. Error of the Estimate	Durbin-Watson
1	.997[a]	.994	.989	.02872	2.629
a. Predictors: (Constant), Improved_Grass, Urban					
b. Dependent Variable: Nitrate					

4.6.1.4. Statistical Significance of the MLR Model for Mean Autumn Nitrate

At the $p<.05$ significance level, improved grassland and urban land use statistically significantly predicted mean autumn nitrate concentration, F (2, 3) = 236.318, $p<.001$. The significance of the overall model is presented in the ANOVA table below.

Table 23: Autumn Nitrate - Analysis Of Variance for MLR

ANOVA[a]						
Model		Sum of Squares	df	Mean Square	F	Sig.
1	Regression	.390	2	.195	236.318	.001[b]
	Residual	.002	3	.001		
	Total	.392	5			
a. Dependent Variable: Nitrate						
b. Predictors: (Constant), Improved_Grass, Urban						

4.6.1.5. Land Use Predictors of Mean Autumn Nitrate Concentration

The regression equation for the model can be expressed in the following form:

Predicted Nitrate = b_0 + (b_1 x Improved grassland) + (b_2 x Urban)

Where, b_0 is the intercept (or constant) and b_1 and b_2 are the slope coefficients.

Table 24: Autumn Nitrate Regression Coefficients

Coefficients						
Model		Unstandardized Coefficients		Standardized Coefficients	t	Sig.
		B	Std. Error	Beta		
1	(Constant)	.028	.014		1.915	.151
	Urban	-.007	.029	-.011	-.234	.830
	Improved_Grass	2.281	.105	.997	21.726	.000

Improved grassland added statistically significantly to the prediction, $p<.000$. An increase in improved grassland of one percent is associated with an increase in nitrate concentration of 2.281 mg/L. The result therefore suggests that improved grassland was the key anthropogenic land use class that contributed to mean autumn nitrate concentrations across FEPA sample catchments.

4.6.2. MLR Model 2: Winter Chloride

The MLR model for mean winter chloride and anthropogenic land use identified multicollinearity between mines and improved grassland. The MLR analysis was then applied again excluding mines; however, the result indicated a statistically insignificant prediction. Analysis of the partial regression plots generated revealed linear associations between chloride concentration and improved grassland and chloride concentration and cultivated land. In order to further explore these linear associations, the final MLR model was based on the independent variables improved grassland and cultivated land use.

4.6.2.1. Winter Chloride: Assumptions

The dataset that was used to develop an MLR model for winter chloride met all four of the required parametric assumptions. There was independence of residuals (or errors) for the dataset, as assessed by a Durbin-Watson statistic of 1.340. Furthermore, improved grassland and cultivated land use were found to be approximately linearly related to mean winter chloride. Examination of partial regression plots indicated that improved grassland exhibited a stronger positive association with chloride concentration than cultivated land use. The dataset displayed no issues with collinearity since all tolerance values were greater than 0.1. A test for normality of the dataset indicated deviations from the norm; however, MLR analysis proceeded due to the robust nature of the method.

4.6.2.2. Winter Chloride: Descriptive Statistics

The table below highlights the summary statistics that describe the dataset for mean winter chloride and land use. There were nine observations in the sample. The average percentage of cultivated land was 13.68 (SD = 10.62) which was significantly higher than the average percentage of improved grassland which was 0.19 (SD = 0.47).

Table 25: Winter Chloride - Descriptive Statistics

Descriptive Statistics			
	Mean	Std. Deviation	N
Chloride	3.9789	5.47299	9
Improved_Grass	.1900	.46813	9
Cultivated	13.6811	10.62492	9

4.6.2.3. Winter Chloride: MLR Model Fit

The model summary table below provides a summary of the overall regression model fit for mean winter chloride. An R-square value of 0.911 meant that improved grassland and cultivated land use explained 91.1 % of the variability in chloride concentration, compared to the mean model. Adjusted R-square was 88.1 %, a large effect size according to Cohen (1988).

Table 26: Winter Chloride - MLR Model Summary

Model Summary[b]					
Model	R	R Square	Adjusted R Square	Std. Error of the Estimate	Durbin-Watson
1	.954[a]	.911	.881	1.89021	1.340
a. Predictors: (Constant), Cultivated, Improved_Grass					
b. Dependent Variable: Chloride					

4.6.2.4. Statistical Significance of the MLR Model for Mean Winter Chloride

At the *p*<.05 significance level, improved grassland and cultivated land use statistically significantly predicted mean winter chloride, *F* (2, 6) = 30.534, *p*<.001.

Table 27: Winter Chloride - Analysis Of Variance for MLR

ANOVA[a]						
Model		Sum of Squares	df	Mean Square	F	Sig.
1	Regression	218.192	2	109.096	30.534	.001[b]
	Residual	21.437	6	3.573		
	Total	239.629	8			
a. Dependent Variable: Chloride						
b. Predictors: (Constant), Cultivated, Improved_Grass						

4.6.2.5. Land Use Predictors Contributing to Mean Winter Chloride Concentration

The regression equation for the model can be expressed in the following form:

Predicted Chloride = b_0 + (b_1 x Cultivated) + (b_2 x Improved Grassland)

Where, b_0 is the intercept (or constant) and b_1 and b_2 are the slope coefficients.

Table 28: Winter Chloride Regression Coefficients

Coefficients[a]						
Model		Unstandardized Coefficients		Standardized Coefficients	t	Sig.
		B	Std. Error	Beta		
1	(Constant)	1.409	1.070		1.317	.236
	Improved_Grass	10.775	1.563	.922	6.895	.000
	Cultivated	.038	.069	.074	.555	.599

Improved grassland added statistically significantly to the prediction, $p<.000$. An increase in improved grassland of one percent is associated with an increase in mean winter chloride concentration of 10.78 mg/L.

4.6.3. MLR Model 3: Winter Sulphate

Multiple regression analysis of mean winter sulphate and anthropogenic land use identified multicollinearity between mines, urban and improved grassland. Furthermore, urban land use did not indicate a linear association with mean winter sulphate concentration. As a result, urban and improved grassland were removed from the analysis. Cultivated land use and degraded thicket were further removed from the analysis because they did not meet the assumption of linearity, as indicated by partial regression plots generated in SPSS statistics. Forest plantations and mines were chosen for multiple linear regression analysis.

4.6.3.1. Winter Sulphate: Assumptions

The assumption of independence of residuals was met for the dataset, as assessed by a Durbin-Watson statistic of 1.210. It was therefore accepted that there were no correlations between adjacent observations (i.e. their errors). Partial regression plots indicated approximately linear positive associations between forest plantations and mines with mean winter sulphate concentration. The dataset further met the assumptions of multicollinearity (all tolerance values were greater than 0.1) and normality (as assessed by a partial regression plot of standardised residuals for mean winter sulphate.

4.6.3.2. Winter Sulphate: Descriptive Statistics

Descriptive statistics explained the basic features of the dataset for sulphate. There were seven observations in the sample. Forest plantations had a higher mean value (1.72 %) and variability in data (SD = 2.52) compared to the mean value (0.03 %) and standard deviation (0.08) observed for mines.

Table 29: Winter Sulphate - Descriptive Statistics

Descriptive Statistics			
	Mean	Std. Deviation	N
Sulphate	.8571	1.65112	7
Forest_pts	1.7214	2.52653	7
Mines	.0314	.08315	7

4.6.3.3. Winter Sulphate: MLR Model Fit

The model summary below describes the overall fit of the regression model for mean winter sulphate and land use. An R-square value of 0.961 indicated that mines and forest plantations accounted for 96.1 % of the variability in mean winter sulphate concentration compared to a simple model based on mean values. Adjusted R-square was 94.1 % which is considered to be a large effect size according to Cohen (1988).

Table 30: Winter Sulphate - MLR Model summary

Model Summary[b]					
Model	R	R Square	Adjusted R Square	Std. Error of the Estimate	Durbin-Watson
1	.980[a]	.961	.941	.40019	1.210
a. Predictors: (Constant), Mines, Forest_pts					
b. Dependent Variable: Sulphate					

4.6.3.4. Statistical Significance of the MLR Model for Mean Winter Sulphate

At the *p*<.05 significance level, mines and forest plantations statistically significantly predicted mean winter sulphate concentration, $F\ (2, 4) = 49.068$, $p<.002$.

Table 31: Winter Sulphate - Analysis of Variance for MLR

ANOVA[a]						
Model		Sum of Squares	df	Mean Square	F	Sig.
1	Regression	15.717	2	7.858	49.068	.002[b]
	Residual	.641	4	.160		
	Total	16.357	6			
a. Dependent Variable: Sulphate						
b. Predictors: (Constant), Mines, Forest_pts						

4.6.3.5. Land Use Predictors Contributing to Mean Winter Sulphate Concentration

The regression equation for the model can be expressed in the following form:

Predicted Sulphate = $b_0 + (b_1 \times$ Mines) + $(b_2 \times$ Forest Plantations)

Where, b_0 is the intercept (or constant) and b_1 and b_2 are the slope coefficients.

Table 32: Winter Sulphate Regression Coefficients

Coefficients[a]						
Model		Unstandardized Coefficients		Standardized Coefficients	T	Sig.
		B	Std. Error	Beta		
1	(Constant)	.138	.188		.734	.504
	Forest_pts	.097	.080	.148	1.210	.293
	Mines	17.598	2.425	.886	7.257	.002

Mines contributed statistically significantly to the prediction of mean winter sulphate concentration, $p<.002$. A one percent increase in mines can be associated with an increase in sulphate concentration of 17.6 mg/L. Forest plantations, however, did not contribute significantly to the prediction of mean winter sulphate concentration.

4.6.4. MLR Model 4: Autumn Phosphate

A multiple linear regression analysis was carried out to determine whether anthropogenic land use contributed to mean autumn phosphate concentrations. As in the case of chloride, sulphate and nitrate regression analysis, inclusion of all anthropogenic land uses did not yield significant results for phosphate. Mines were excluded from the analysis due to a lack of correlation with other variables. Forest plantations indicated a very low tolerance value and were therefore also excluded from further analysis due to issues of multicollinearity. The final MLR analysis included cultivated, improved grassland and urban land use as the predictor variables for phosphate concentration.

4.6.4.1. Autumn Phosphate: Assumptions

The dataset revealed a Durbin-Watson statistic of 2.702 which was considered acceptable to meet the assumption of independence of observations. There was therefore no first-order autocorrelation between variables. Examination of partial regression (P-P) plots showed a negative linear association between improved grassland and phosphate concentration. Urban and cultivated land, however, were positively linearly associated with mean autumn phosphate concentration. Improved grassland and urban land use showed stronger linear associations with phosphate concentration than cultivated land use. Furthermore, the dataset showed no problems of collinearity since all tolerance values were greater than 0.1. The assumption of normality was also met, as indicated by a partial regression plot which confirmed that residuals close enough to normal to proceed with MLR analysis.

4.6.4.2. Autumn Phosphate: Descriptive Statistics

There were six observations in the sample for phosphate. Cultivated land use had the highest average (7.86 %) and standard deviation (3.98) while improved grassland had the lowest average (0.05 %) and standard deviation (0.12).

Table 33: Autumn Phosphate - Descriptive Statistics

Descriptive Statistics			
	Mean	Std. Deviation	N
Phosphate	.0400	.02280	6
Improved_Grass	.0500	.12247	6
Cultivated	7.8550	3.97575	6
Urban	.2283	.44047	6

4.6.4.3. Autumn Phosphate: MLR Model Fit

With an R-square of 92.8, urban, improved grassland and cultivated land use collectively accounted for 92.8 % of the variation in phosphate concentration, a large effect size according to Cohen (1988).

Table 34: Autumn Phosphate - MLR Model Summary

Model Summary[b]					
Model	R	R Square	Adjusted R Square	Std. Error of the Estimate	Durbin-Watson
1	.985[a]	.971	.928	.00612	2.702
a. Predictors: (Constant), Urban, Improved_Grass, Cultivated					
b. Dependent Variable: Phosphate					

4.6.4.4. Statistical Significance of the MLR Model for Mean Autumn Phosphate

At the *p*<.05 significance level, urban, improved grassland and cultivated land use statistically significantly predicted phosphate concentration, F (3, 2) = 22.483, *p*<.043.

Table 35: Autumn Phosphate - Analysis of Variance for MLR

ANOVA[a]						
Model		Sum of Squares	df	Mean Square	F	Sig.
1	Regression	.003	3	.001	22.483	.043[b]
	Residual	.000	2	.000		
	Total	.003	5			
a. Dependent Variable: Phosphate						
b. Predictors: (Constant), Urban, Improved_Grass, Cultivated						

4.6.4.5. Land Use Predictors Contributing to Mean Autumn Phosphate Concentration

The regression equation for the model can be expressed in the following form:

Predicted Phosphate = b_0 + (b_1 x Urban) + (b_2 x Improved grassland) + (b_3 x Cultivated)

Where, b_0 is the intercept (or constant) and b_1, b_2 and b_3 are the slope coefficients.

Table 36: Autumn Phosphate Regression Coefficients

Coefficients[a]						
Model		Unstandardized Coefficients		Standardized Coefficients	t	Sig.
		B	Std. Error	Beta		
1	(Constant)	.037	.006		5.751	.029
	Improved_Grass	-.126	.023	-.677	-5.488	.032
	Cultivated	4.111E-5	.001	.007	.058	.959
	Urban	.039	.006	.746	6.172	.025

Improved grassland added statistically significantly to the prediction, p<.032. An increase in improved grassland of one percent is associated with a decrease in phosphate of 0.13 mg/L. Urban land use also contributed statistically significantly to the prediction, p<.025. An increase in urban land use of one percent is associated with an increase in phosphate concentration of 0.04 mg/L.

4.6.5. MLR Model 5: Autumn Electrical Conductivity

Autumn electrical conductivity displayed the strongest correlations with urban and cultivated land use, according to multiple regression analysis. Multicollinearity was detected between degraded thicket, forest plantations and cultivated land use during testing of the assumptions for multiple regression analysis. There were also no correlations observed for mines. Improved grassland was excluded from the analysis due to the non-linear association it exhibited with electrical conductivity values.

4.6.5.1. Autumn EC: Assumptions

No correlations were identified between adjacent observations for the dataset. This was confirmed by a Durbin-Watson statistic of 1.372. The assumption of linearity was met for the dataset. Cultivated and urban land use both exhibited positive linear associations with mean autumn electrical conductivity, as assessed by partial regression plots. A stronger association was observed between EC and urban land use than EC and cultivated land use. All tolerance values in the dataset were greater than 0.1 (VIF less than 10), ensuring that collinearity was not a problem with the dataset. Finally, the dataset met the assumption of normality, as indicated by a P-P plot of standardized residuals for mean autumn EC. Standardized residuals appeared normally distributed.

4.6.5.2. Mean Autumn Electrical Conductivity: Descriptive Statistics

The descriptive statistics below indicated that the average value for cultivated land use (12.33 %) was significantly higher than that of urban land use (0.20 %). Similarly, cultivated land use showed greater variability in data with a standard deviation of 10.84 compared to a standard deviation of 0.41 for urban land use.

Table 37: Autumn Electrical Conductivity - Descriptive Statistics

Descriptive Statistics			
	Mean	Std. Deviation	N
EC	259.9214	280.29519	7
Cultivated	12.3343	10.83990	7
Urban	.1957	.41125	7

4.6.5.3. Autumn Electrical Conductivity: MLR Model Fit

The model summary indicated an adjusted R-square value of 0.750. As such, cultivated and urban land use explained 75 % of the variability in EC levels, a medium to strong effect size according to Cohen (1988).

Table 38: Autumn Electrical Conductivity - MLR Model summary

Model Summary[b]					
Model	R	R Square	Adjusted R Square	Std. Error of the Estimate	Durbin-Watson
1	.926[a]	.857	.750	105.31211	1.712
a. Predictors: (Constant), Urban, Forest_pts, Unimpr_Grass					
b. Dependent Variable: EC					

4.6.5.4. Statistical Significance of the MLR Model for mean Autumn Electrical Conductivity

At the $p<.05$ significance level, urban and cultivated land use statistically significantly predicted mean autumn EC levels for the overall regression model, $F\ (2,\ 4) = 11.093$, $p<.023$.

Table 39: Autumn Electrical Conductivity - Analysis Of Variance for MLR

ANOVA[a]						
Model		Sum of Squares	df	Mean Square	F	Sig.
1	Regression	399385.320	2	199692.660	11.093	.023[b]
	Residual	72007.024	4	18001.756		
	Total	471392.344	6			
a. Dependent Variable: EC						
b. Predictors: (Constant), Urban, Cultivated						

4.6.5.5. Land Use Predictors Contributing to Mean Autumn EC Level

The regression equation for the model can be expressed in the following form:

Predicted EC = b_0 + (b_1 x Urban) + (b_2 x Cultivated)

Where, b_0 is the intercept (or constant) and b_1 and b_2 are the slope coefficients.

Table 40: Autumn Electrical Conductivity Regression Coefficients

Coefficients[a]						
Model		Unstandardized Coefficients		Standardized Coefficients	t	Sig.
		B	Std. Error	Beta		
1	(Constant)	90.749	89.431		1.015	.368
	Cultivated	3.572	5.156	.138	.693	.527
	Urban	639.278	135.912	.938	4.704	.009

From the coefficients table above, urban land use statistically significantly predicted EC levels across sample catchments, $p<.009$.

4.6.6. MLR Model 6: Summer Dissolved Oxygen

Prior to running the multiple linear regression analysis to include anthropogenic land uses, the assumption of multicollinearity was checked. The result indicated collinearity between urban and mines. In addition forest plantations, cultivated and degraded thicket violated the assumption of linearity. Urban and improved grassland showed the strongest linear associations with mean summer DO and were selected for further analysis.

4.6.6.1. Summer Dissolved Oxygen: Assumptions

The dataset displayed independence of observations, as assessed by a Durbin-Watson statistic of 3.053. With respect to linearity, partial regression plots revealed approximately linear negative correlations between mean summer DO concentrations and improved grassland and urban land uses. Furthermore, the assumption of multicollinearity was met, where tolerance values for all cases were greater than 0.1. A P-P plot of standardised residuals for mean summer DO confirmed that the dataset was approximately normally distributed, meeting the final assumption of normality and allowing for MLR analysis to continue.

4.6.6.2. Summer Dissolved Oxygen: Descriptive Statistics

There were nine observations within the sample. Urban land use showed more variability in the data, having a higher standard deviation than that of percent improved grassland. Urban land use also exhibited a higher mean value of 1.4 % compared to 0.19 % for improved grassland.

Table 41: Summer Dissolved Oxygen - Descriptive Statistics

Descriptive Statistics			
	Mean	Std. Deviation	N
DO	6.6956	1.08893	9
Improved_Grass	.1900	.46813	9
Urban	1.4033	3.29950	9

4.6.6.3. Summer Dissolved Oxygen: MLR Model Fit

The regression model fit for DO and land use indicated an R-square value of 0.550. Improved grassland and urban land use therefore explained 55 % of the variability in mean summer dissolved oxygen, a medium effect size according to Cohen (1988).

Table 42: Summer Dissolved Oxygen - MLR Model Summary

Model Summary[b]					
Model	R	R Square	Adjusted R Square	Std. Error of the Estimate	Durbin-Watson
1	.814[a]	.663	.550	.73047	3.053
a. Predictors: (Constant), Urban, Improved_Grass					
b. Dependent Variable: DO					

4.6.6.4. Statistical Significance of the MLR Model for Mean Summer Dissolved Oxygen

At the $p<.05$ significance level, improved grassland and urban land use statistically significantly predicted mean summer DO, $F\ (2, 6) = 5.889$, $p<.038$.

Table 43: Summer Dissolved Oxygen - Analysis Of Variance for MLR

ANOVA[a]						
Model		Sum of Squares	df	Mean Square	F	Sig.
1	Regression	6.285	2	3.142	5.889	.038[b]
	Residual	3.202	6	.534		
	Total	9.486	8			
a. Dependent Variable: DO						
b. Predictors: (Constant), Urban, Improved_Grass						

4.6.6.5. Predictor Variables Contributing to Mean Summer Dissolved Oxygen Levels

The regression equation for the model can be expressed in the following form:

Predicted EC = b_0 + (b_1 x Improved grassland) + (b_2 x Urban)

Where, b_0 is the intercept (or constant) and b_1 and b_2 are the slope coefficients.

Table 44: Summer Dissolved Oxygen regression Coefficients

Coefficients[a]						
		Unstandardized Coefficients		Standardized Coefficients		
Model		B	Std. Error	Beta	t	Sig.
1	(Constant)	7.223	.289		24.963	.000
	Improved_Grass	-1.573	.553	-.676	-2.846	.029
	Urban	-.163	.078	-.493	-2.075	.083

Improved grassland added statistically significantly to the prediction, $p<.029$. A one percent increase in improved grassland is associated with a 1.57 mg/L decrease in dissolved oxygen concentration.

4.7. SASS Biomonitoring

SASS sampling methodology was carried out at four of the nine sample sites, namely, Skeerpoort Outflow, STERK01, STERK02 and MAGA01. All sites were sampled for macroinvertebrates once over the 12 month sampling period during spring. Spring is considered the ideal season to perform aquatic biomonitoring due to the abundance of macroinvertebrate species present during the season. Spring was therefore chosen in order to allow for good representation of water quality at the chosen sites.

Several samples were collected over the sampling period, however, these were considered practice runs and formed part of the process of learning the SASS methodology. Four sites were sampled for macroinvertebrates due to the fact that some of the rivers had dried up during the sample period and also due to the fact that some sites were inaccessible. In some cases, sites were not suitable to perform the macroinvertebrate sampling due to the poor range and extent of biotopes. SMI scores based on SASS score are presented in Table 50.

Table 45: Aquatic Biomonitoring Results

SMI Scores				
SITE	**SMI Score**	**No. of Taxa**	**ASPT**	**ECOLOGICAL BAND**
Skeerpoort Outflow	72	14	5.14	D
STERK 01	60	12	5	D
STERK 02	12	3	4	E/F
MAGA 01	54	10	5.4	C

From the above table it is clear that all sites have deteriorated from a good ecological condition. STERK02 had the lowest ASPT across all sites, with a score of 4. In addition, STERK02 fell into the ecological band E/F, meaning that the site has been seriously or critically modified from its natural state. Skeerpoort Outflow and STERK01 also showed a considerable decline in ecological condition, both falling within the ecological band D, meaning that these sample sites have been largely modified from their natural state. MAGA01 had the highest average score per taxon (ASPT) with a score of 5.4. MAGA01 fell within the ecological band C. The site has been moderately modified yet is no longer in a good ecological condition. Plotting SMI Score against mean annual water chemistry highlighted the relationship between the biological results and that of water quality across sample catchments. Macroinvertebrate sampling was performed at four of the nine sample sites. The results indicated that there were no significant correlations between SMI score and water chemistry. This is likely due to the very small number of samples taken (n=4).

4.8. Conclusion

Key results obtained from the study have been presented in this chapter. Land use maps revealed the anthropogenic land uses associated with each FEPA sample catchment. These included urban, improved grassland, forest plantations and degraded thicket. Principle component analysis confirmed anthropogenic land use as a potential source of water pollution at sample catchments. Principle components one and two suggested urban, improved grassland, forest plantations, degraded thicket, mines and urban land use as potential pollution sources. The component matrix and Biplot indicated that phosphate and nitrate concentrations were associated with industrial, domestic and cultivated land use practices. DO and pH were associated with urban, farming and mining activities, according to principle component one. Principle component two revealed a relationship between EC and saline industrial effluent and wastewater discharges. Industrial and domestic effluents such as domestic fertilizer and wastewater effluent were further associated with chloride concentration while sulphate concentration was associated with acid mine drainage.

Multiple linear regression analysis therefore included all identified land uses, based on exploratory data analysis, as the predictor variables for each independent variable or water quality parameter. In cases where the assumptions of MLR analysis were not met, specific land use predictors were removed from the analysis. Multiple linear regression analysis highlighted improved grassland, forest plantations and urban land use as the primary land use predictors of water quality of FEPA sample catchments. Biological monitoring indicated that correlations between SMI scores and water chemistry were not significant. The next chapter will provide an interpretation of the results presented in this chapter. Chapter five will discuss the implications of the results presented in the context of threats to water quality of river FEPAs within the study area. Recommendations for future research and the limitations of the study are also addressed.

Chapter Five: Discussion, Conclusions and Recommendations

5.1. Introduction

FEPAs play a vital role in ensuring the sustainability of freshwater ecosystems in South Africa, yet they represent only twenty two percent of South Africa's river length (Nel & Driver, 2011). As such, the need to maintain these priority areas in good ecological condition plays an important role in the management of freshwater resources in South Africa.

The purpose of this study was to investigate the status of water quality at five river FEPAs in the upper Crocodile catchment and to identify potential threats to water quality of the selected priority areas. Water quality monitoring at FEPA sites is an important way to ensure that priority rivers maintain their ecological integrity; however, regular monitoring of FEPA health has not been undertaken in recent years. Through analysis of both water chemistry and land use data, potential threats to water quality of FEPAs within the study area were investigated, the results of which were presented in chapter four. This chapter will focus on interpretation of key findings of the study in order to answer the questions posed. Interpretation of the results will reveal whether anthropogenic land uses predicted water quality at river FEPAs within the Upper Crocodile catchment. This chapter will attempt to answer the research questions posed by the study. Following a discussion of the results, conclusions will be drawn in order to summarize the main findings of the study. The limitations of the study and recommendations for further research are also addressed in this chapter.

5.2. GIS and Land Use Data

Land use maps generated using GIS software revealed the proportion of anthropogenic land uses that comprised FEPA sample catchments. Spatial analysis allowed for existing anthropogenic land uses to be identified within each FEPA catchment. MAGA02 sample catchment contained the highest proportion of improved grassland (1.41 %) and cultivated land (35.3 %). STERK01 displayed the highest percentage of degraded thicket (0.5 %) and forest plantations (5.6 %). BRAND01 comprised the highest percentage of urban land (10.11 %) and mines (4.23 %). Overall, STERK01, MAGA01 and BRAND01 were identified as the FEPA catchments with the highest proportion of anthropogenic land uses within the study area. The land uses improved grassland, degraded thicket, forest plantations, cultivated, mines and urban land use therefore represented potential water quality threats to FEPA sample catchments and were investigated further. Improved grassland was considered anthropogenic since it includes the growing of grassland under man-managed (including irrigated) conditions for grazing, the production of hay and recreation. Improved grassland is therefore often associated with the use of fertilizers.

5.3. Variation in Water Chemistry

Nitrate concentration was highest at Skeerpoort Outflow compared with other sites, peaking during autumn at 1 mg/L, with a minimum value of 0.03 mg/L during spring. Along the Sterkstroom River, STERK01 showed higher average nitrate concentrations than STERK02 and STERK03 for the duration of sampling, peaking at 0.1 mg/L during spring. MAGA01 displayed higher nitrate concentrations than MAGA02, peaking in winter at 0.19 mg/L. Although nitrate levels remained low throughout the duration of sampling, increases during the dry season may be associated with anthropogenic inputs. Key sources of nutrients include non-point sources such as agricultural runoff, urban runoff, sewage runoff from septic tanks and un-sewered rural communities (de Villiers & Thiart, 2007). Furthermore, accumulation of pollutants during the dry season may be a result of the reduced dilution capacity of the rivers during the low rainfall period.

Chloride concentrations were highest during the dry season (autumn/spring) across all sample sites. The highest value for chloride concentration was recorded at MAGA02 (26 mg/L) during spring. The high chloride concentration indicated pollution from agricultural runoff, road runoff or sewage effluent which was likely prevalent at the time of sampling. Edokpayi et al. (2015) found similar results in the Mvudi River, South Africa, where chloride concentrations were found to be higher in the dry season. This was attributed to the increased dilution of chloride that occurs during the wet season. Increased chloride in streams can lead to several ecological implications for aquatic ecosystems including acidification (Kaushal, et al., 2005).

Sulphate concentrations peaked during the dry season (spring) at all sample sites. SK01 displayed an unusually high sulphate concentration of 60 mg/L during spring 2015 and also showed high sulphate in spring 2014 (34 mg/L). High sulphate concentrations in surface waters is a key indicator of acid mine drainage (AMD). Dabrowski et al (2013) and McCarthy (2011) attributed increases in sulphate concentrations in the upper Olifants river catchment to mining activities and the impacts of AMD. Mine water could be the source of sulphate pollution within the study area. Other possible sources of pollution include the burning of fossil fuels and runoff from agricultural, urban and industrial practices. Urban sources of sulphate pollution include fertilizers, sewage effluents and rainfall runoff (Khatri & Tyagi, 2015). Sulphate concentration was not influenced by seasonal variation at Skeerpoort Outflow, BUFF01, STERK03 and MAGA01.

Phosphate concentrations were highest during the dry season (winter/spring) at all FEPA sample catchments. The highest value was recorded at Skeerpoort Outflow at 0.5 mg/L. STERK02 showed significantly higher phosphate concentrations compared to STERK01 and STERK03, although levels remained within acceptable limits for aquatic ecosystems. SK01, STERK01 and STERK03 did not exhibit significant seasonal variations in phosphate concentration. De Villiers (2007), Chimwanza et al (2006) and van der Laan (2012) reflected similar outcomes having reported higher phosphate concentrations during the dry season at the Berg River, the Likangla River and within streams in the Crocodile catchment respectively. Van der Laan (2012) identified irrigation practices as the primary contributor to phosphate concentrations within the Crocodile catchment. Furthermore, reduced river flow during the dry season can lead to higher evaporation rates and elevated phosphate concentrations (Ncube, 2012).

Electrical conductivity levels were highest during the dry season (spring/autumn) at all sample sites, with the highest value observed during the first month of sampling at Skeerpoort Outflow (953 µS/cm). EC level was lowest at STERK03 during summer, reaching a minimum value of 31.5 µS/cm. High EC levels could be attributed to industrial, agricultural and domestic discharges. Furthermore, urban land uses have been found to increase EC levels in surface waters (Khatri & Tyagi, 2015). The results were corroborated by previous research. Matowanyika (2010) identified higher EC levels in the Jukskei River in the dry season than the wet season. Similarly, seasonal variation in EC showed elevated concentrations during the dry season at the Mvudi River (Edokpayi, et al., 2015).

Dissolved oxygen levels were also higher during the dry season (winter/spring) and lower during the wet season at most sites. At STERK02, however, DO was highest during summer. Furthermore, DO was lowest at MAGA02 during spring, measuring as low as 1.64 mg/L. Lower DO concentrations during the wet season may have been attributed to higher water temperatures (Rivers-Moore, et al., 2008). Average pH values were highest during winter months at all sample sites except BUFF01 where pH values peaked during summer. Minimum pH values occurred at STERK02 (6.29 pH units) during summer. Maximum values occurred at STERK03 (8.66 pH units) during winter. Higher pH values may have been a result of lower water temperatures during winter while lower pH values may have been a result of higher water temperatures and nutrients during summer. However, it is more likely that diffuse pollution sources were the main contributor to pH fluctuations between sites. Nutrient enrichment is one of the main causes of increased pH in streams while a decrease in pH may be a result of acid mine drainage or discharges from industrial land use activities. Studies conducted by Edokpayi (2015) and Abowei (2010) reported outcomes of a similar nature, finding higher pH values in the dry season at the Mvudi and Nkoro rivers respectively.

5.4. Exploratory Data Analysis – The Relationship between Water Quality and Land Use

Exploratory data analysis was the first step in identifying potential links between water chemistry and land use data. Analysis of scatterplots revealed linear associations between nitrate, chloride, phosphate, dissolved oxygen and percentage improved grassland. Phosphate concentration and dissolved oxygen levels were negatively correlated with percentage improved grassland. While nitrate and chloride displayed positive correlations with percentage improved grassland, the result was not statistically significant. Chloride, phosphate, EC and pH values were found to be positively correlated with percentage urban land use with only autumn EC indicating a statistically significant relation with percentage urban land use. Sulphate concentrations were positively correlated with percentage mines. Percentage improved grassland, mines and urban land use therefore exhibited the strongest linear associations with water chemistry, based on exploratory data analysis using scatterplots.

The links between water chemistry and land use identified by this study are corroborated by previous research. In the Crocodile, Komati and Pongola catchments, increased levels of phosphates were found to be caused by irrigation practices in the sugarcane producing regions of South Africa (van der Laan, et al., 2012). The study corroborated the link between phosphate and the use of fertilizers identified for this study. Furthermore, Kroll (2009) identified positive correlations between urban land use and nitrate, phosphate and EC in the Castillo-La Mancha (Spain). The study further observed positive correlations between agriculture and nitrate and EC.

Principle component analysis was carried out in order to further investigate polluting effects from anthropogenic land use sources within FEPA sample catchments. Principle component one (PC1) accounted for 46.26 % of the total variance explained and comprised nitrate, phosphate, pH, DO and EC. Nitrate and phosphate exhibited the highest factor loadings of 0.906 and 0.905 respectively. PC1 therefore suggested urban, agricultural (fertilizers), industrial and domestic sources of water pollution. Principle component two (PC2) accounted for 31.02 % of the total variance explained and included loadings of EC, sulphate and chloride. Electrical conductivity had the highest factor loading on PC2 (0.830), indicative of saline industrial effluent and domestic effluents as well as surface runoff from urban, industrial and agricultural sources. High loadings of chloride and sulphate indicated possible pollution from acid mine drainage, domestic fertilizers, industrial and sewage effluents. Principle component analysis therefore provided a basis to further investigate land use predictors of water chemistry through the application of multiple linear regression analysis, having confirmed the effects of anthropogenic pollution sources within FEPA sample catchments.

5.5. Multiple Regression Analysis

The primary land use predictors of water chemistry within river FEPA sample catchments were identified through multiple regression analysis. A model was developed for nitrate, chloride, sulphate, phosphate, EC and DO. Land use did not contribute significantly to pH levels across FEPA sample catchments. The analyses revealed the most significant correlations between water quality parameters and anthropogenic land use and provided the key conclusions to the study:

- Percentage improved grassland was found to statistically significantly predict autumn nitrate, winter chloride, autumn phosphate and summer dissolved oxygen across FEPA catchments.
- Percentage urban land use statistically significantly predicted autumn EC and phosphate levels. Percentage mines significantly predicted winter sulphate concentrations at FEPA sample catchments.
- Improved grassland, mines and urban land use were therefore the key land use predictors of water quality within the study area and represent potential threats to water quality of FEPA sample catchments.
- Forest plantations, cultivated and degraded thicket land use types did not significantly predict water chemistry at FEPA sample catchments and are not considered a threat to FEPA health currently.

Several studies were found to agree with the findings, documenting anthropogenic land uses as significant contributors to the pollution of surface waters. Van Deventer (2012) reported deterioration in water quality of the uMngeni River as a result of upstream agricultural practices. In the Guare River in northern Venezuela, agriculture and the use of fertilizers were the primary predictors of nitrate and phosphate concentrations. In a further study, Castillo (2009) investigated the impact of land use on water quality of rivers in the Castilla-La Mancha (Spain). The results showed that irrigated agriculture was significantly correlated with nitrate and phosphate concentrations. Grasslands were also identified as an important nitrate source in a river draining the Sierra Nevada in California. The study revealed a strong positive correlation between nitrate concentration and percent grassland, highlighting percent grassland as the main contributor to nitrate loading within the catchment (Kroll, et al., 2009).

Li et al (2013) found that urban and agricultural land uses led to significant increases in electrical conductivity, chloride and sulphate concentrations in the Han River (China). Hua (2017) found similar results where changes in land use were investigated in relation to the water quality of the Malacca River. The study revealed that built-up or urban areas contributed significantly to electrical conductivity of the river while dissolved oxygen depletion was linked to agricultural and forested areas. Furthermore, electrical conductivity was connected to agricultural land use activities and non-point pollution from agricultural practices through surface runoff. Other sources of electrical conductivity and dissolved oxygen pollution included residential, industrial and sewage effluent. Similarly, Bowden et al (2015) observed higher mean values for EC for agricultural land use within the Marlatt watershed in Kansas.

Acid mine drainage has been extensively linked to higher concentrations of sulphates in receiving surface waters. Mine effluents containing acids and sulphates may enter streams as run-off. Van Eeden et al (2009) reported on the impact of mining activities on the Tweelopiespruit and Wonderfonteinspruit catchments on the West Rand. The report linked high sulphate concentrations in the Tweelopiespruit catchment to the dischargeof AMD from abandoned mines in the area. AMD effluent containing high levels of sulphates can lead to the deterioration of aquatic ecosystems and the loss of species diversity as was the case in the Tweelopiespruit catchment (van Eeden, et al., 2009). For this study, the sample catchment most affected by mining activities and the impacts of AMD was the BRAND01 sample catchment, representing the Brandvlei River in the Randfontein area. Sample catchment BRAND01 comprised the highest percentage of mines (4.2 %). The closure of several mines on the Witswatersrand (including the Randfontein area) is the most likely source of AMD within the Brandvlei River and the BRAND01 sample catchment. This may have adverse impacts on water quality of the river and could contribute to irreversible destruction to the aquatic ecosystems it supports.

5.6. Aquatic Biomonitoring (based on SASS version 5)

Biological water quality analyses were performed at four FEPA sites within the upper Crocodile catchment. The NFEPA project classified all river FEPAs within a good ecological category (ecological category A or B). However, this study has shown the deterioration of all river FEPA sites within the study area, namely Skeerpoort Outflow, STERK01, STERK02 and MAGA01. MAGA01 was found to have been moderately modified from its natural state (ecological category C). This is likely a result of agricultural impacts within the catchment. Cultivated land (both irrigated and dryland) accounted for 35.3 % of the catchment area. Skeerpoort outflow and STERK01 fell into ecological category D, indicating that the sites were largely modified. Deterioration of the Skeerpoort River may be attributed to the impacts of agricultural activities within the Skeerpoort Outflow sample catchment. Both irrigated and dryland cultivation comprised 6 % of the catchment land area. Forest plantations and urban land use activities may have further contributed to the decline in health of the river. The urban centre likely to have impacted the site is the village of Skeerpoort situated at the Hartebeespoort Dam in the North West Province. Within the STERK01 sample catchment, cultivated land comprised 14.4 % of all land used and likely contributed to deteriorating health of the Sterkstroom River FEPA and STERK01 sample site. Of greater concern was the seriously to critically modified status of the STERK02 river FEPA sample catchment, which fell into the ecological category E/F. Inefficient agricultural practices may have led to a decline in river health at the STERK02 sample site since irrigated cultivated land contributed to 11.6 % of the catchment land area.

5.7. Conclusions

This study investigated five river FEPAs within the upper Crocodile catchment in order to determine whether FEPAs have remained in good ecological condition or whether they are under threat to water quality from land use activities within the catchment. Freshwater ecosystems are especially susceptible to changes in water quality from land use practices since pollutants accumulate in rivers which act as sinks within the landscape. Water quality monitoring of river FEPAs is a means to ensure that the important ecosystem services they provide are sustained (Driver, et al., 2011). This study revealed that river FEPAs in the upper Crocodile catchment face water quality threats from anthropogenic land use practices.

Higher concentrations of chemicals during the dry season were indicative of the influence of anthropogenic land use practices on water quality. Nitrate, chloride, sulphate, phosphate, EC, DO and pH were influenced by changes in season. DO level; however, peaked during the wet season at STERK02. According to the South African Water Quality Guidelines for Aquatic Ecosystems (volume 7), the Target Water Quality Range (TWQR) for nitrate is < 0.5 mg/L and the TWQR for phosphate is 0.5 µg/L or 0.005 mg/L. Sites along the Skeerpoort River were found to exceed the TWQR for nitrate. Mean annual nitrate concentration was 0.6 mg/L for the Skeerpoort Outflow sample site. However, mean winter nitrate was 0.8 mg/L and mean nitrate during autumn and spring was 0.7 mg/L. In terms of phosphate concentration, all the rivers sampled exceeded the TWQR. Mean phosphate concentration at the Skeerpoort Outflow sample site was 0.08 mg/L while the highest mean phosphate concentration at the site occurred during spring, reaching a level of 0.5 mg/L and exceeding the TWQR by 0.495 mg/L. Furthermore, SK01 and STERK02 sample sites had a mean phosphate concentration of 0.04 mg/L, 0.035 mg/L higher than the TWQR. BUFF01, STERK01, STERK02, MAGA01, MAGA02 and BRAND01 exceeded the TWQR for phosphate by 0.003 mg/L, with mean phosphate concentrations of 0.03 mg/L. Increases in nutrients during the dry season may be associated with non-point water quality threats from agricultural and urban runoff due to the application of fertilizers. The impact of such land use practices may contribute to future eutrophication within the catchments in the absence of regular water quality monitoring. The sudden spikes in sulphate concentration observed at SK01 during the dry season are a cause for concern. The result was unexpected particularly given the small percentage of urban and agricultural land present within the catchment.

Investigation of scatterplots led to the inclusion of percentage improved grassland, percentage mines and percentage urban land use in further statistical analyses. PCA further emphasised the impact of anthropogenic land use on FEPA catchments, allowing for key land uses to be selected for MLR analysis. The results of exploratory data analysis therefore revealed the relationship between certain pollutants, land use activities and water quality across FEPA catchments and highlighted key potential threats. MLR analysis allowed for conclusions to be drawn with regards to land use predictors of FEPA water quality. The expectation was that MLR would corroborate the findings of exploratory data analysis by revealing anthropogenic land uses as key predictors of water quality. These expectations were met, highlighting percentage improved grassland, percentage urban and percentage mines as the primary predictors of water chemistry.

The link between percentage improved grassland and nitrate, chloride, phosphate and DO is indicative of the impact of agricultural inputs on FEPA health. Improved grassland refers to planted grasslands growing under man-managed conditions. In South Africa, the irrigation of pastures occurs throughout the year. The impact of the application of fertilizers has proven to be significant for this study. Excess nutrients can lead to eutrophication of streams and consequent degradation of water quality. A one percent increase in improved grassland is associated with a 2.38 mg/L increase in nitrate concentration and a 0.13 mg/L increase in phosphate concentration. The result is interesting considering that the percentage cover for improved grassland was very low across catchments. The study therefore indicates extreme sensitivity to this particular land use class and warrants further investigation.

Chloride from animal sewage is the most likely reason for the correlation identified between improved grassland and chloride concentration (Kumar, 2002). High levels of chloride can result in a number of implications for river FEPAs, including river salinization. High levels of chloride may even hinder growth, reproduction and the survival of aquatic organisms. A one percent increase in improved grassland within FEPA catchments is associated with a significant increase in chloride concentration of 10.78 mg/L. Improved grassland is considered a potential threat to water quality of FEPAs within the catchment and may further contribute to eutrophication of FEPA streams.

Percentage urban land use statistically significantly predicted EC and phosphate at FEPA catchments. Key urban related sources of phosphate and EC in aquatic ecosystems include domestic, municipal and industrial wastewater effluents. Although EC does not measure specific ions, the prediction indicates that increases in solute concentrations are likely a result of urban activities within the catchments. A one percent increase in urban land use is associated with a 0.04 mg/L increase in phosphate concentration and a 639.3 μS/cm increase in EC. The significant impact of urban land use on EC and to a lesser extent phosphate concentration is a clear potential threat to water quality of FEPA catchments. Increases in urban land use activities will place significant strain on river FEPAs to retain their integrity.

South Africa relies heavily on mining activities in order to meet its development needs. Well-managed mining operations and proper mine closures are not generally associated with water pollution. However, the improper closure of mines associated with acid mine drainage has contributed significantly to the pollution of streams in several areas of the country. The FEPA catchments investigated for this study represent an area which may be under threat from AMD impacts. The percentage of mines within the catchments was statistically significantly correlated with sulphate concentration within FEPA catchments. The most likely cause is AMD run-off. High sulphate in aquatic ecosystems can be potentially harmful to river ecosystems and in some cases can lead to stream acidification from run-off during rainfall events. Higher concentrations during the dry season at FEPA catchments may have been associated with the reduced dilution capacity of the streams. A one percent increase in mines is associated with a 17.6 mg/L increase in sulphate concentration, a significant impact. Increases in mining activities therefore represent a potential threat to water quality within FEPA sample catchments. In the context of this study, the sample catchment with the highest percentage of mines is the BRAND01 sample catchment located in the Randfontein area which forms part of the Witswatersrand mining belt. Randfontein is considered a mining town situated in the West Rand in Gauteng. Potential sources of AMD across FEPA sample catchments may include the Randfontein gold mine which is operational as well as acid mine water originating from mine voids in the Witswatersrand.

The conclusions drawn in this chapter highlight the importance of regular water quality monitoring at river FEPAs in order to identify potential threats to FEPA health from anthropogenic land use activities. The study was able to address this need by providing an insight into the status of water quality at five river FEPAs within the Upper Crocodile catchment as well as the potential threats to water quality identified within the catchment. It is clear that improved grassland, urban and mining land use activities have contributed significantly to water chemistry within river FEPA catchments over the course of this study. One can therefore conclude that careful management of these land uses is important if river FEPAs are to continue to function sustainably. Urbanisation occurs more rapidly in developing countries like South Africa. Furthermore, the rate of infrastructure development is outpaced by urban expansion resulting in inadequate provision of drinking water and wastewater treatment. In order to achieve sustainable management of urban catchments, the re-use of wastewater provides an opportunity to support freshwater ecosystem services. However, more detailed research is required to explore the impacts of anthropogenic pressures on freshwater ecosystems in South Africa and the study area (Capps, et al., 2016). Agricultural practices including cultivation and livestock farming can seriously impact surface water quality. Within the study area, irrigated and dryland cultivation may impact river FEPAs due to runoff from fertilizers and the deposition of sediment eroded from fields. More efficient use of fertilizers and management of soil erosion are needed to prevent degradation of river water quality within the study area.

5.8. Limitations to the Study

- A longer duration of sampling would have yielded more reliable data and may have revealed more valuable patterns in data.
- The sample size was restricted to five river FEPAs in the upper Crocodile catchment. A larger sample size may have allowed for better representation of the overall health of river FEPAs within the sub-management area.
- Historical water quality data was not available for all river FEPAs selected for this study. As a result, trend analysis was not possible for all sites.

5.9. Recommendations

Based on the findings of this study, it is apparent that land use activities are not being managed adequately within river FEPA catchments within the upper Crocodile catchment. It is therefore recommended that:

- Water resource practitioners and the relevant water sector institutions integrate FEPA maps and management guidelines into water resource planning and management.
- National monitoring programs incorporate river FEPAs into routine monitoring of water quality in order to ensure they remain in good ecological condition.
- Authorities need to manage basic infrastructure to ensure proper treatment of wastewater within urban centres. Water resource managers should focus on developing new ways to re-use wastewater to allow for FEPA catchments to function sustainably.
- Farmers should be encouraged through education to follow agricultural practices that are less polluting to river FEPAs. Cutting back on fertilizer use would prove economically beneficial and reduce deteriorating water quality of river FEPAs within the Upper Crocodile catchment. Controlling soil erosion and using animal waste to produce fertilizer are further examples of ways in which farmers can better manage agricultural practices to ensure the health of FEPAs is maintained.

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